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WMO/IGAC Impacts of Megacities on Air Pollution and Climate

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WORLD METEOROLOGICAL ORGANIZATION
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IMPACTS OF MEGACITIES ON AIR POLLUTION AND CLIMATE

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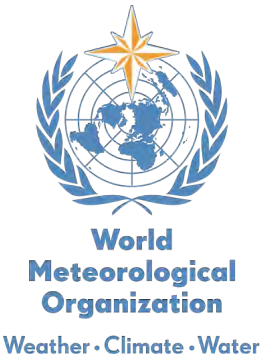
Michael Gauss, Laura Gallardo Klenner, Mark Lawrence,
Abdourahamane Konare and Cathy Liousse



September 2012
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Preface

With increasing global population and extensive, ongoing urbanization, over half the world's population resides now in urban areas and this number is projected to nearly double by 2050, from 3.5 billion to 6.3 by 2050. The number of megacities (cities with population over 10 million) currently is 23 and is expected to reach 37 in 2025.

Megacities are areas with the most intensive human activities, including economic and social activities, together with tremendous energy consumption. These activities lead to concentrated emissions of air pollutants, greenhouse gases, and waste heat, which impact terrestrial and aquatic ecosystems as well as air quality and climate.

The negative impact of megacities on local air quality has long been recognized. In recent years, the impact of anthropogenic emissions from megacities on regional and global climate has also received increasing attention. Both of these impacts are linked through energy consumption derived from fossil fuel combustion with emissions that change the atmospheric concentrations of short-lived species that impact both human health and climate, such as aerosols and ozone. Because of these linkages, it has been argued that megacities are the best places to realize the co-benefits of simultaneously controlling air pollution and reducing climate change.

With the growing trend towards urbanization, understanding the role of megacities in local to global atmospheric chemistry is critical to effectively realize the co-benefits of controlling air pollution and reducing climate change. The scientific and engineering knowledge that has been accumulated when developed megacities dealt with their air quality problems in earlier years is a significant resource for current and future megacities. Experiences of developed countries show that the pronounced air quality degradation that accompanied past development can be avoided. In recent years, there have been a growing number of internationally coordinated integrated studies and collaborative projects examining the impacts of megacities on air pollution and climate change. An assessment of these research results and general information about megacities are important for both scientific communities and policy makers dealing with urbanization, air quality management, and climate change.

At the IGAC (International Global Atmospheric Chemistry) Scientific Steering Committee (SSC) meeting in September 2007inSeattle, USA, a possible initiative focused on an Integrated Megacities Assessment was discussed. In May 2008, the plan for an IGAC Assessment on Impacts of Megacities on Air Quality and Climate was formally proposed and approved by the IGAC SSC in Cape Town, South Africa. By June 30, 2008, an outline was sent out for feedback and to identify potential contributors. On September 8, 2008, at the IGAC SSC meeting in Annecy France, we discussed the outline, identified the lead authors and contributing authors, formed an IGAC Megacity Report Working Group, and decided on the lead authors meeting schedule.

The first lead authors meeting was held at Peking University in Beijing on May 22-24, 2009, with the presence of Sarah Doherty, Michael Gauss, Laura Gallardo Klenner, Abdourahamane Konare, Mark G. Lawrence, David Parrish, Min Shao, Min Hu, and Tong Zhu, supported by the IGAC Project Office and CAREBeijing project. The outline and lead authors of each chapter were decided at the meeting.

At the 24th IGAC SSC meeting in October 2009 held in Kyoto, Japan, it was decided that the World Meteorological Organization (WMO) and IGAC would publish the report jointly. WMO is placing a growing interest and focus on megacities, especially through its Commission on Atmospheric Sciences (CAS) and GAW Urban Research Meteorology and Environment (GURME). Urban areas as environments pose unique challenges to atmospheric modelling and monitoring and create a multi-disciplinary spectrum of potential threats, including air pollution, which needs to be addressed in an integrated way.

The second lead authors meeting was held on November 16-18, 2009 to compile and work on detail contents of the report, at Peking University in Beijing. The attendees included Sarah Doherty, Michael Gauss, Maria Kanakidou, Laura Gallardo Klenner, David Parrish, and Tong Zhu, supported by the IGAC International Project Office and the State Key Joint Laboratory for Environmental Simulation and Pollution Control, Peking University. The lead authors met again in July 2010 during the 11th IGAC Science Conference in Halifax, Canada, to discuss the integrated and outlook topics of the report.

The IGAC Megacity Report Working Group includes Sarah Doherty, Laura Gallardo, Michael Gauss, Maria Kanakidou, Abdourahamane Konare, Mark Lawrence, Cathy Liousse, Megan Melamed, David Parrish, and Tong Zhu (Group Leader). In early 2011, Megan L. Melamed replaced Sarah Doherty as the IGAC Executive Officer; she also took over Sarah Doherty's role as a lead author of Chapter 1, and contributed greatly to editing and finalizing the report.

Given the current state of available information about many megacities around the world, a truly comprehensive, integrated assessment of the impact of megacities on air pollution is not possible at this time. However, an initial assessment of what information is available on air pollution in megacities across Africa, Asia, South America, North America, and Europe was deemed to be valuable and worth pursuing. Therefore, a large portion of the report is devoted to summarizing the current situations of megacities on different continents. Exceptions are Chapter 1, which gives the introduction of this report and Chapters 7 and 8 that provide an overview of international collaborative research activities and key issues and outlook. Due to decades of air pollution measurements and studies in North America, Chapter 5does provide an integrated analysis about megacities in North America.

Many issues and scientific questions remain to be addressed and discussed, such as tropical/sub-tropical cities versus mid-latitude cities, direct and indirect radiative forcing of aerosols in megacities and the surrounding regions. We plan to address these issues and scientific questions, along with more integrated analysis, in a future updated version of this report.

Tong Zhu
On behalf of
IGAC Megacity Report Working Group

Liisa Jalkanen
World Meteorological Organization

Acknowledgements

The lead authors meetings of the report were financially supported by the IGAC International Project Office, the U.S. National Oceanic and Atmospheric Administration (NOAA), CAREBeijing project and the State Key Joint Laboratory for Environmental Simulation and Pollution Control, Peking University.

We would like to acknowledge the great efforts of all the contributing authors of this report. Special appreciation is extended to Beth Tully at University of Washington, Edit-Design Center, for her graphic design work on all the figures throughout the report, June Landenburger and Steven Brey, University of Washington undergraduate students, for their work on the references, figures, and editing of the report, and Pauline Mooney of World Meteorological Organization for making the final editing of this report.

CHAPTER 1 - INTRODUCTION

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As of 2008, for the first time, the majority of the world's population is living in urban areas [http://www.unfpa.org/pds/urbanization.htm], many in megacities (with populations over 10 million). Megacities are not only the centre of growing economies, but are also large sources of air pollutants and climate-forcing agents [*Parrish and Zhu*, 2009]. This chapter first provides an introduction to world urbanization trends (Section 1.1.) followed by an introduction to the science behind how air pollution is formed (Section 1.2) and scientific methods to study air pollution (Section 1.3). The chapter concludes with an overview of the health impacts caused by air pollution (Section 1.4) and air quality regulations (Section 1.5).

1.1 URBANIZATION

It has long been recognized that air pollution has negative impacts on human health and ecosystems. More recently, the role of anthropogenic emissions in altering climate has come to the forefront. While the former has typically been considered a local issue and the latter a global issue, air pollution and climate change are inexorably linked. Intense source regions in particular can have local to regional to hemisphere-scale impacts on both air quality and climate. With the growing trend towards urbanization, understanding the role of large urban agglomerates in local to global atmospheric chemistry is critical to effectively addressing both air pollution and climate change.

Between 1950 and 2005 the world's population increased from 2.5 billion to 6.5 billion and is expected to increase to 9.1 billion in 2050 [*http://esa.un.org/unpd/wup/index.htm*]. The increase in the world population is paralleled by an increase in the percentage of the world's citizens living in urban areas. As of 2008, for the first time in history, over half the world's population (50.1%) resides in urban areas (Figure 1), a dramatic increase from 30% in 1950. By 2030, the proportion of the population living in urban areas is expected to be over 50% in all major geographical areas of the world (Figure 2). Currently, urban population fractions are higher in developed than developing countries, with the exception of South America where the urban population can reach as high as 83% of the population. However, with most of the world population growth expected to occur in developing countries, urbanization is increasing at a higher rate in developing countries. The ongoing trend in urbanization means that over 85% of the growth in world population between 2000 and 2024 is expected to occur in urban areas of developing countries [*Montgomery*, 2008]. These dramatic increases in population and urbanization, especially in the developing world, have been accompanied by technological and economic growth and development, yielding changes in land use, energy use, and transportation. The resulting changes due to urbanization have dramatic impacts on anthropogenic and biogenic emissions and have notably altered local to global-scale

atmospheric composition, increasing the importance of understanding the impacts of urbanization on atmospheric chemistry.

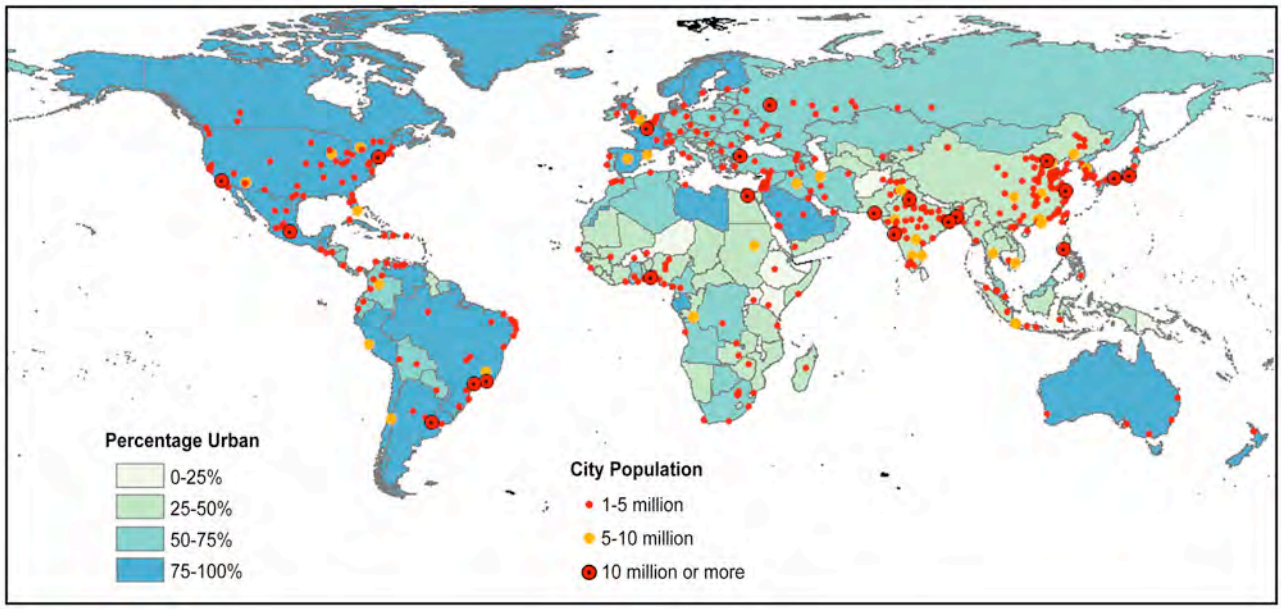

Figure 1 - Urban Agglomeration in 2009 (urban proportion of the world population: 50.1%)
Source: [*UN Department of Economic and Social Affairs,* 2010]

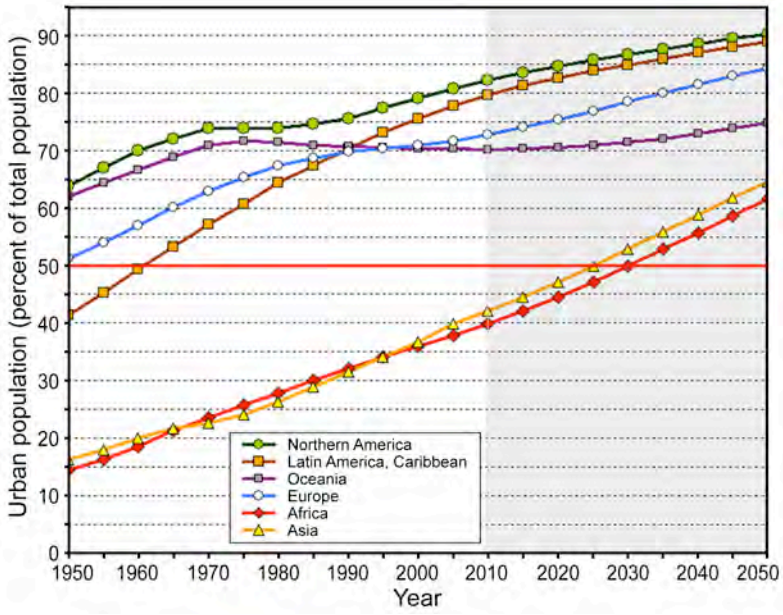

Figure 2 - Urban Population by major geographical area (in percent of total population)
[*UN Department of Economic and Social Affairs,* 2010]

This report focuses on the atmospheric chemistry of megacities, typically defined as urban agglomerates with ≥10 million inhabitants. The definition of a megacity is an imperfect and inexact metric, however, as it does not account for other important factors such as population density (Figure 3) or criteria for defining its boundaries (e.g. administrative boundaries vs. county boundaries vs. the "true" urban area, all of which can change with time [*Montgomery*, 2008], nor

does it account for highly urbanized regions where multiple smaller cities essentially constitute one large urban area. In the context of atmospheric chemistry, one of the most important characteristics of a "megacity" is an area of highly concentrated anthropogenic emissions.

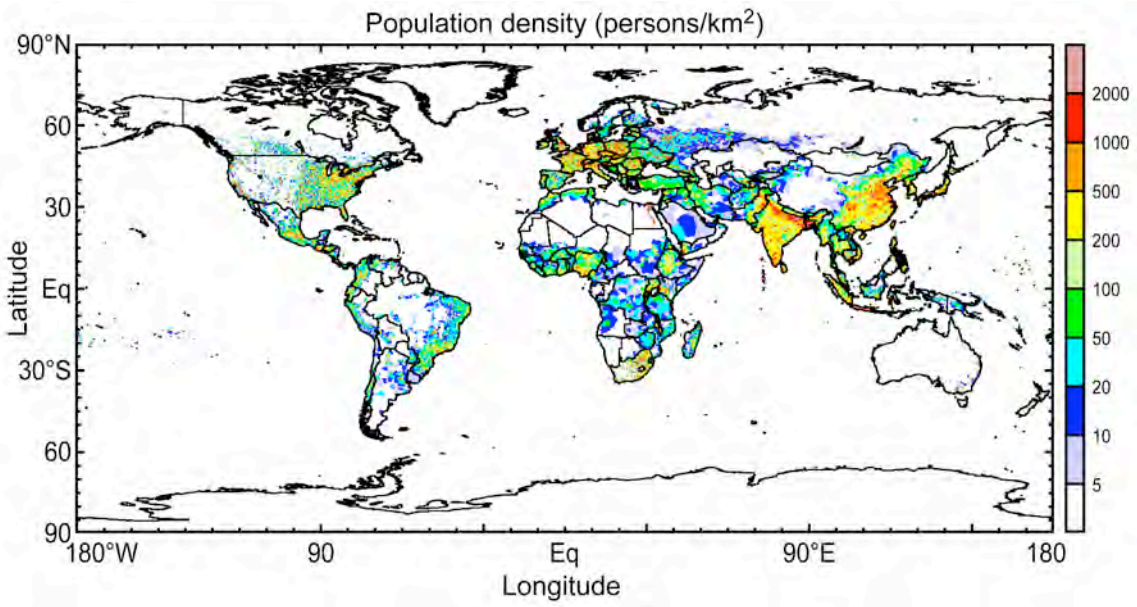

Figure 3 - Map of the world population density (persons per km^2), based on 0.25^o gridded data for 2000 from the Center for International Earth Science Information Network (CIESIN) at Columbia University [*http://sedac.ciesin.columbia.edu/gpw/*]

Therefore, this report also addresses "major population centers (MPCs)" and "major industrial centers (MICs) [*Lawrence et al.*, 2007]. MPCs are areas where several cities are situated such that they effectively constitute an important regional source of anthropogenic emissions. The northeast US urban corridor of New York City, Newark, Washington D.C. and Boston, the Po Valley in Europe, or the Pearl River Delta in Asia are all considered to be MPCs. MICs are areas with concentrated industrial emissions in proximity to urban areas, such as the Highveld near Johannesburg, South Africa or the Houston, Texas industrial area in the US MPCs and MICs are not only complex in terms of their atmospheric chemistry, but they pose a special difficulty in the context of emissions regulation since they generally encompass more than one regulatory area.

The geographic scale and intensity of megacities' impact on air pollution and climate is determined not only by the extent of their emissions but also by their regional geography and meteorology. These factors affect the degree to which the emissions are trapped, such as by valley or basin walls or by persistent atmospheric inversions, and the degree to which the emissions are transported to the regional to global scale, such as via uplift in deep convective systems. These factors also combine to determine how megacity emissions interact with emissions from surrounding areas. Notably, most of the world's megacities are situated in coastal zones. In these areas, the mixture of urban and marine air masses result in a unique set of chemistry. For instance, meso-scale circulations export sulphur and nitrogen compounds out to sea, which can trigger emissions of halogens that are subsequently transported back into urban areas by the sea breeze giving rise to ozone production [*Von Glasow,* 2008*; Lawler et al.,* 2009]. Continental emissions, e.g., biomass burning, can also interact with urban emissions, altering how the urban emissions affect air quality and climate. Clearly, each urban area's unique set of characteristics must be considered in the study of the impacts of urbanization on atmospheric chemistry.

The impact of megacities on air pollution and climate must also be considered in the context of the world's total population. Emissions associated with the economic activity required to feed, house, clothe and otherwise provide for the needs of the 9.1 billion people expected in 2050 will inevitably impact the atmosphere. When addressing the fraction of emissions that originate from megacities, it is important to consider how the magnitude and the impact of these emissions would be different if the population and associated economic activity of the megacities were dispersed in a less concentrated population distribution. Given the concentration of wealth and intellectual resources in megacities, it may well be that the overall impact on the atmosphere is less if the world's population is indeed concentrated in urban areas as current forecasts predict.

This assessment seeks to summarize the geography, meteorology, emissions, atmospheric chemistry, and climate of megacities in North America, South America, Europe, Asia, and Africa and to provide a summary of the research done and still needed regarding megacities and atmospheric chemistry.

1.2 ATMOSPHERIC CHEMISTRY OF AIR POLLUTION

The negative impacts of air pollution in urban areas have a long history dating back to the 13th century when coal began to replace wood as a heating source. For many centuries, "dilution is the solution", i.e. putting taller and taller smoke stacks on emission sources, was used to curb the human health impacts in urban areas. Then, after a series of "minor" air pollution episodes, over 4000 people died in the London Killer Smog Episode of 1952 when a dense fog containing sulphuric acid particles persisted for days. That same year a landmark paper was published by A.J. Haagen-Smit on the "Chemistry and Physiology of Los Angeles Smog", a city that also continually experienced air pollution episodes. Haagen-Smit showed that the cause of air pollution in Los Angeles was the release of large quantities of hydrocarbons and nitrogen oxides to the atmosphere that through a complex series of photochemical reactions create ozone and other harmful secondary pollutants. Although different emission sources and atmospheric chemistry caused the London Killer Smog episode and Los Angeles smog, both of these historical events motivated the scientific and policy communities to begin unravelling the atmospheric chemistry involved in air pollution and implementing regulations to improve air quality.

Today, air pollution typically refers to a set of criteria pollutants typically referred to as lead (Pb), carbon monoxide (CO), sulphur dioxide (SO_2), nitrogen dioxide (NO_2), ozone (O_3), and particulate matter (PM). These criteria pollutants were historically designated as such due to their impacts on human health and later to their impacts on ecosystems. The proceeding sections summarize the environmental impacts and basic atmospheric chemistry of four criteria pollutants: SO_2, NO_2, O_3, and PM. These four criteria pollutants all have an impact on air pollution and climate and are key active species in atmospheric chemistry.

1.2.1 Acid rain formation

Both SO_2 and NO_2 were named criteria pollutants for their role in the formation of acid rain, a general term for wet and dry deposition of sulphuric and nitric acids (H_2SO_4 and HNO_3, respectively) and for their direct and indirect human health impacts (see following sections on tropospheric ozone and particulate matter). Acidic deposition lowers the pH of lakes, rivers, and stream causing harm to entire watersheds and ecosystems and also erosion of building and automotive parts. The formation of acid rain occurs through a series of chemical reactions that are central to the relationship between the precursors and the sulphuric and nitric acid deposition [*Seinfeld and Pandis*, 1998]. In industrialized regions, anthropogenicsources, i.e. energy production, industry and transportation, typically dominate emissions of SO_2 and NO_2 and their precursors.

Sulphur dioxide is converted to sulphuric acid via both aqueous phase (eq. 1.1) and gas-phase oxidation (eq. 1.2). The aqueous phase pathway is believed to be responsible for more than 50% of the ambient sulphate concentration [*Seinfeld and Pandis*, 1998]. The lifetime of SO_2 in the aqueous phase pathway is very short whereas the lifetime of SO_2 in the gas-phase oxidation

pathway is on the order of 1-2 weeks, which explains why high concentrations of sulphate (SO_4^{2-}) can be found both near and far downwind from SO_2 sources.

$$SO_2(g) \Leftrightarrow SO_2 \cdot H_2O$$
$$SO_2 \cdot H_2O \Leftrightarrow HSO_3^- + H^+$$
$$HSO_3^- + H_2O_2(aq) \Leftrightarrow SO_2OOH^- + H_2O \quad \text{(eq. 1.1)}$$
$$SO_2OOH^- + H^+ \rightarrow H_2SO_4$$

$$SO_2 + OH + M \rightarrow HSO_3 + M$$
$$HSO_3 + O_2 \rightarrow SO_3 + HO_2 \quad \text{(eq. 1.2)}$$
$$SO_3 + H_2O + M \rightarrow H_2SO_4$$

Nitrogen oxides are converted to nitric acid primarily by gas-phase oxidation during the daytime (eq. 1.3) and by a nitrate radical (NO_3) heterogeneous pathway during nighttime when NO_3 cannot be photolyzed (eq. 1.4).

$$\mathbf{OH + NO_2 + M \rightarrow HNO_3 + M} \quad \text{(eq. 1.3)}$$

$$NO_2 + O_3 \rightarrow NO_3 + O_2$$
$$NO_3 + NO_2 \Leftrightarrow N_2O_5 \quad \text{(eq. 1.4)}$$
$$N_2O_5 + H_2O(aq) \rightarrow 2HNO_3(aq)$$

Although the gas-phase oxidation rate of NO_X to nitric acid is approximately 10 times faster than the gas-phase oxidation rate of SO_2 to sulphuric acid, high concentrations of nitric acid in the aqueous form are not generally found in most source regions. This is because sulphuric acid formed in the gas-phase immediately associates with water molecules to form sulphuric acid aerosol, which excludes nitric acid. Nitric acid will remain in the gas phase until deposited to surfaces or absorbed by a cloud or rain droplet with lower concentrations of sulphuric acid, which can occur downwind from the emission sources. It is also important to note that both sulphuric and nitric acid formation is dependent on OH formation or in more general terms, the oxidation capacity of the atmosphere [*Jacob*, 1999].

1.2.2 Tropospheric ozone

Ozone near the surface in the troposphere is harmful to human health and ecosystems due to its ability to oxidize biological tissue. A common human health impact of tropospheric ozone is respiratory illnesses such as asthma in children. Tropospheric ozone is formed when volatile organic compounds (VOC) are oxidized in the atmosphere in the presence of nitrogen oxides (NO + NO_2 = NO_X) and sunlight. VOC comprise a very large (many 100s of individual species) family of organic compounds, including both non-methane hydrocarbons (NMHC) and oxidized organic species. In addition to chemical formation, ozone is transported from the stratosphere to the troposphere through stratospheric/tropospheric exchange. Figure 4 provides a simplified schematic of the physical and chemical processes involved in the tropospheric ozone budget.

The major emission source of NO_X in urban areas is fossil fuel combustion, typically utilized for transportation, electrical power generation and industrial processes. However, the major urban emission sources of VOCs are both anthropogenic and natural. Anthropogenic VOC emission sources are combustion, fuel evaporation, solvent use, and chemical manufacturing. The primary natural VOC source is emissions from terrestrial vegetation, such as from forests [*Jacob*, 1999]. Therefore, the natural emissions of VOCs have an impact on the effectiveness of either NO_X or VOC controls to reduce O_3 concentrations.

Tropospheric ozone production occurs when the hydroxyl radical oxidizes VOCs, carbon monoxide (CO), and methane (CH_4) in the presence of nitrogen oxides (NO_X) [*Penkett et al.*, 1999]. Due to the large number of different VOC species, their complex oxidation pathways and their numerous emission sources, the formation of tropospheric O_3 is extremely complex.

Equations 1.5 show the simplest pathway, the formation of O_3 during the oxidation of CO. In addition, ozone and its precursors from large urban areas can be transported on hemispheric scales in the free troposphere and increase background levels of surface ozone such that the hemispheric transport may offset local mitigation strategies to reduce ozone levels [*Jacob and Winner*, 2009]. To fully understand the tropospheric O_3 budget, a quantitative understanding of the identity and sources of its precursors, the numerous chemical reactions that constitute the atmospheric VOC oxidation processes, and transport processes that control background ozone and its precursor levels as well as stratospheric/tropospheric exchange must be incorporated into atmospheric models. Finally, tropospheric ozone is also a radiatively active trace gas, and thus impacts climate at regional and global scales.

$$OH + CO \rightarrow CO_2 + H$$
$$H + O_2 + M \rightarrow HO_2 + M$$
$$HO_2 + NO \rightarrow OH + NO_2 \qquad \text{(eq. 1.5)}$$
$$NO_2 + h\upsilon \rightarrow NO + O$$
$$O + O_2 + M \rightarrow O_3 + M$$
$$Net: CO + 2O_2 \rightarrow CO_2 + O_3$$

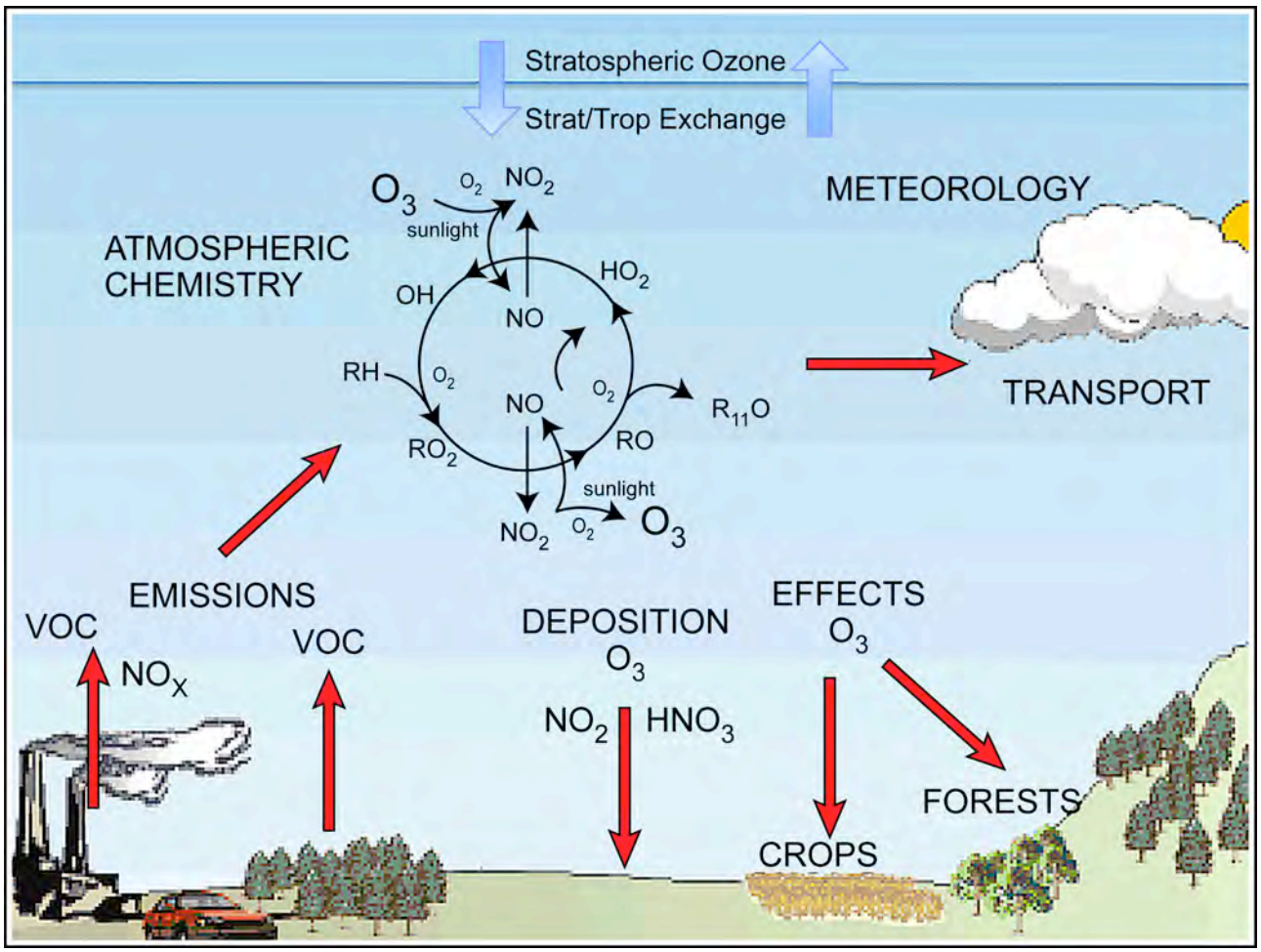

Figure 4 - Physical and chemical processes affecting tropospheric ozone
[*http://www.globalchange.umich.edu/gctext/Inquiries/Inquiries_by_Unit/Unit_9.htm*]

1.2.3 Particulate matter

An aerosol is technically defined as a suspension of particulate matter (fine solid or liquid particles) in gas. Aerosols and particulate matter (PM), which are often used synonymously, are referred to in a variety of different terminologies as shown in Table 1. Aerosols and PM have a variety of impacts including causing respiratory illnesses, decreasing visibility, and impacting climate through both direct (the particles themselves absorbing or scattering radiation) and indirect (the particles serving as a cloud condensation nuclei) effects.

Table 1 - Terminology Related to Atmospheric Particles
Source: *Seinfeld and Pandi*s, 1998 (pg. 97)

Aerosols, aerocolloids, Aerodisperse systems	Tiny particles dispersed in gases
Dust	Suspension of solid particles produced by mechanical disintegration of material such as crunching, grinding, and blasting.
Fog	A loose term applied to visible aerosols in which the dispersed phase is liquid. Usually a dispersion of water or ice close to the ground.
Fume	The solid particles generated by condensation from the vapour state, generally after volatilization from melted substances, and often accompanied by a chemical reaction such as oxidation.
Haze	An aerosol that impedes vision and may consist of a combination of water droplets, pollutants, and dust.
Mist	Liquid, usually water in the form of particles suspended in the atmosphere at or near the surface of the Earth; small water droplets floating or falling, approaching the form of rain, and sometimes distinguished from fog as being more transparent or as having particles perceptibility moving downward.
Particle	An aerosol particle may consist of a single continuous unit of solid or liquid containing many molecules held together by intermolecular forces. A particle may also be considered to consist of two or more such unit structures held together by interparticle adhesive forces such that it behaves as a single unit in suspension or upon deposit.
Smog	A term derived from smoke and fog, applied to extensive contamination by aerosols. Now sometimes used loosely for any contamination of the air.
Smoke	Small gas-borne particles resulting from incomplete combustion, consisting predominately of carbon and other combustible material, and present in sufficient quantity to be observable independently of the presence of other solids.
Soot	Agglomerations of particles of carbon impregnated with “tar”, formed in the incomplete combustion of carbonaceous material.

Figure 5 summarizes the main sources and sinks of atmospheric aerosols. Particles may be directly emitted from a source (primary aerosol) or formed in the atmosphere through a gas-to-particle conversion process (secondary aerosol). Aerosols have both natural and anthropogenic sources. Natural sources include dust, sea spray, forest fires, volcanoes and vegetation. Anthropogenic sources include transportation, industry, fires, mechanical sources, and human induced changes in vegetation. Particles change their size and composition in the time between emission and their removal from the atmosphere by dry or wet deposition. The chemical and physical mechanisms of particle formation and transformation are not completely understood and remain a topic of continual research.

The composition of aerosols varies greatly across urban areas due to different emission sources and meteorological conditions. Figure 6 shows aerosol mass spectroscopy (AMS) measurements of aerosol taken at a variety of locations in the Northern Hemisphere [*Jimenez et al*., 2009]. Organic aerosols (OA) are of particular importance since they make up between 20 to 90% of the submicron particle mass. Organic aerosols exist in the atmosphere as both primary and secondary aerosol. Primary organic aerosol (POA) is directly emitted from emission sources such as fossil fuel combustion and biomass burning. However, the evolution of POA in the atmosphere is not well understood. Secondary organic aerosols (SOA) result from the oxidation of gas-phase species. It is believed that SOA accounts for a large portion of total OA and hence aerosol in general. The evolution of the gas-phase species to SOA is also still poorly understood.

There is a need for further research on OA in order to reduce uncertainty of the role of aerosol in human health and climate.

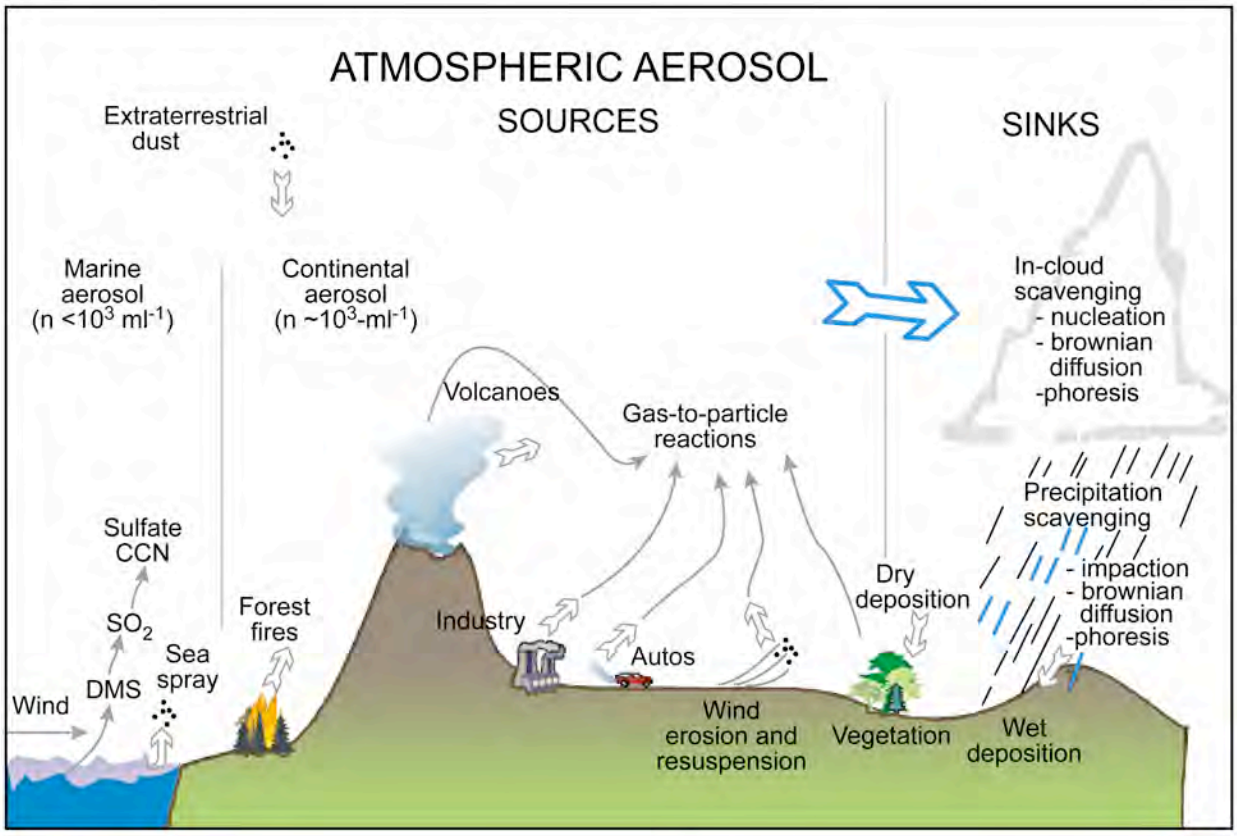

Figure 5 - Sources of atmospheric aerosol [*http://www.ems.psu.edu/~lno/Meteo437/Figures437.html*]

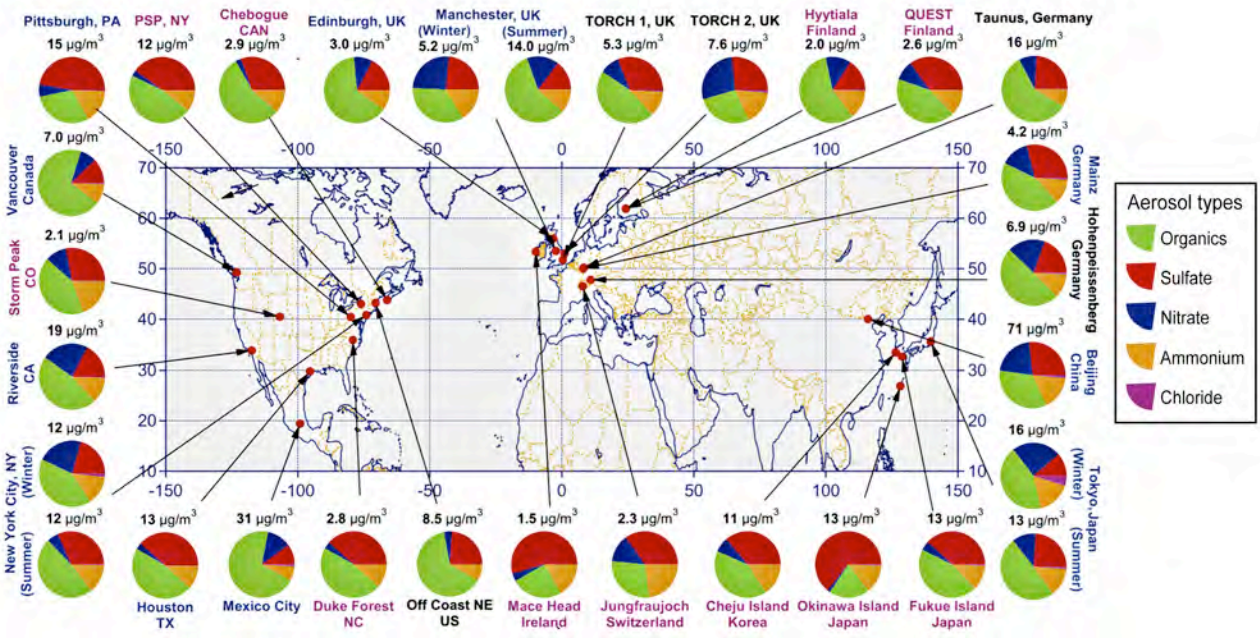

Figure 6 - Average aerosol mass concentration and chemical of composition measured with an AMS at multiple locations in the Northern Hemisphere. [*Zhang et al.*, 2007]

1.2.4 Climate change

Air pollution and climate change have largely been kept separate in both the scientific and policy communities. This was primarily due to the temporal and spatial differences between air pollution and climate with air pollutants being short-lived reactive species that have impacts on the local and regional scale and climate forcing agents being longer-lived radiatively active species that have impacts on the global scale. However, more recently and into the future, it is clear that air pollution and climate change are inexorably linked (Figure 7). Air pollutants that typically were researched and regulated for their air quality impacts are now being recognized as important drivers of climate change. In addition, the scientific community has begun to provide information on the regional impacts of climate change and how these changes may impact air pollution. It is clear that air pollution and climate change are issues that now have overlapping temporal and spatial scales and should be addressed in an integrated manner.

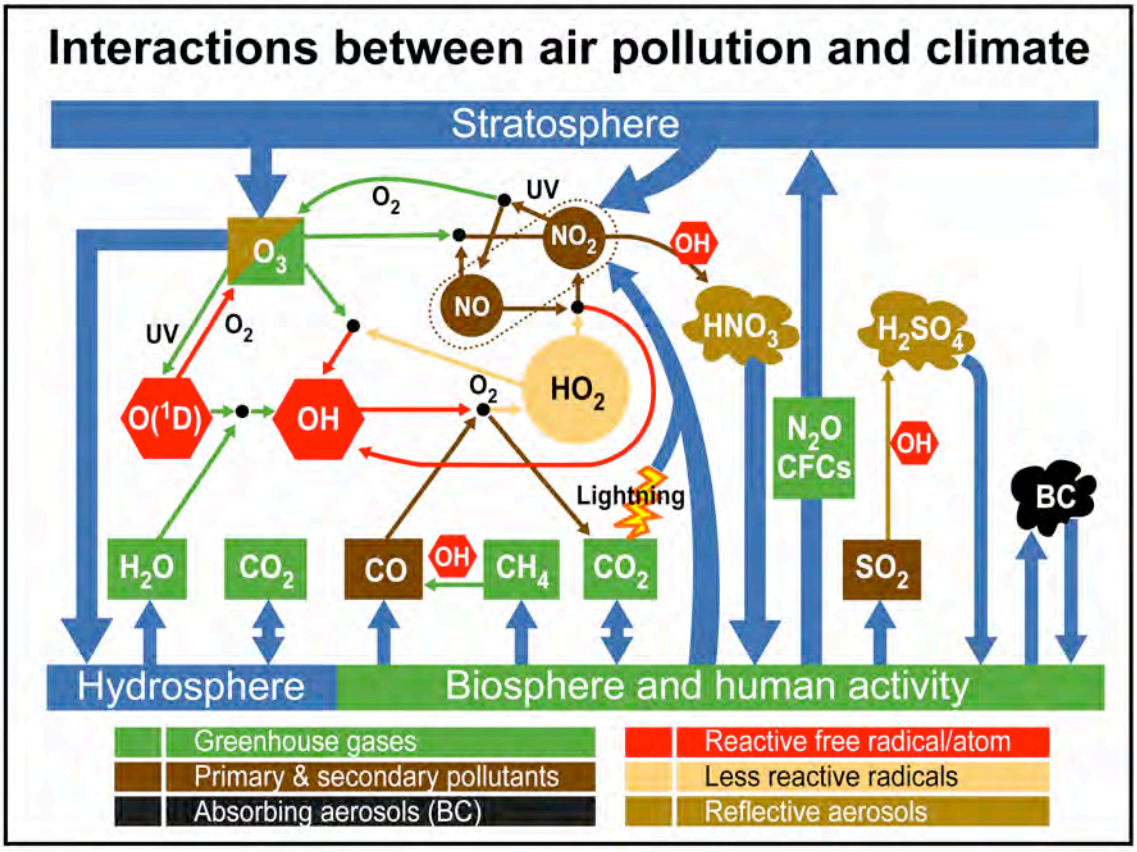

Figure 7 - Air pollution and climate are inexorably linked
[*http://www.ocw.cn/OcwWeb/Chemical-Engineering/10-571JSpring-2006/CourseHome/index.htm*]

Figure 8 shows the global average estimates and ranges for human-caused radiative forcing in 2005 through the principal climate forcing agents [*IPCC*, 2007]. Species that typically were considered solely air pollutants in the past include tropospheric ozone, black carbon, and other aerosols. The positive radiative forcing due to tropospheric ozone and black carbon has led to much interest in mitigating both of these air pollutants in order to both improve air quality and reduce radiative forcing, a win:win strategy. The linkage between air pollution and climate for aerosols is more complicated. Mitigation of aerosols from an air quality standpoint is clearly a win strategy for human health and wellbeing. However, from a climate standpoint, mitigating aerosols would eliminate a cooling effect, thus increasing the overall net global radiative forcing, a net loss for climate [*Ramanathan and Feng,* 2008]. Figure 9 depicts a diagram showing that trade-off that policy makers must consider when improving air quality or mitigating climate change.

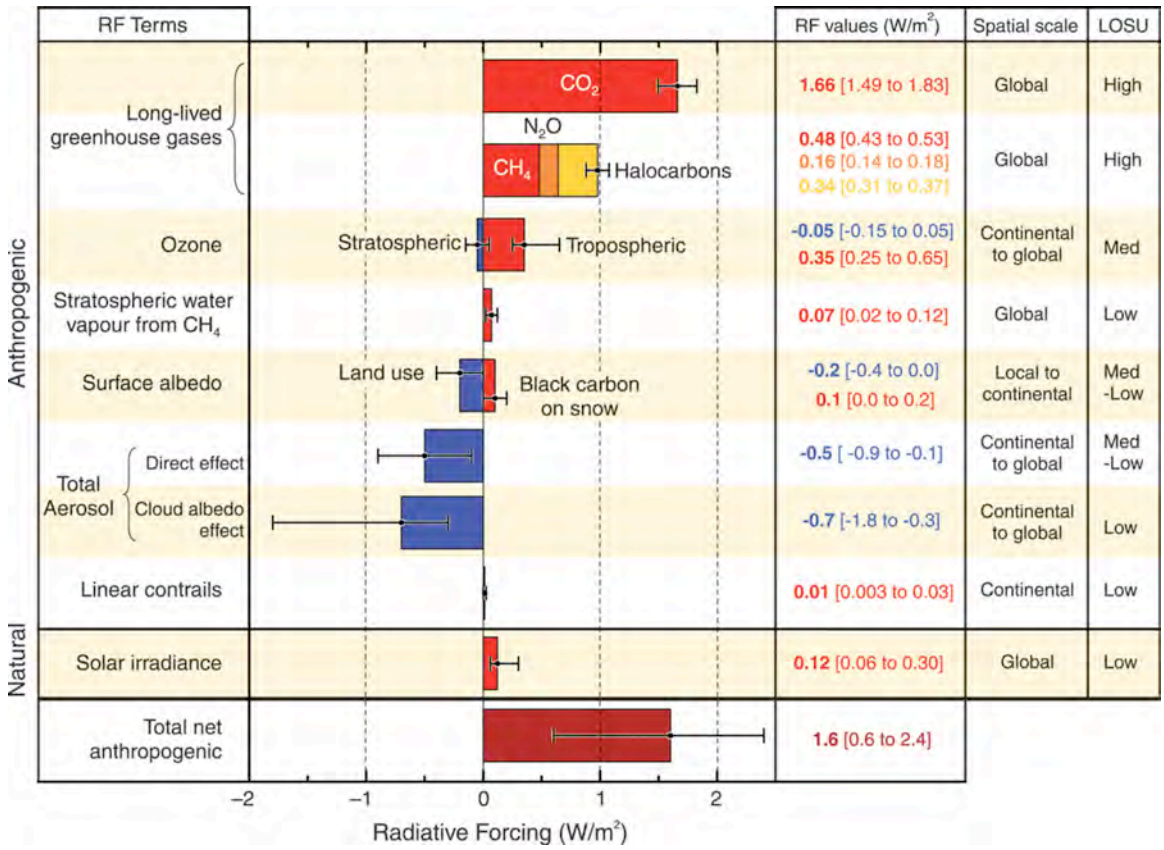

Figure 8 - Global average radiative forcing estimates and ranges in 2005 [*IPCC, 2007*]

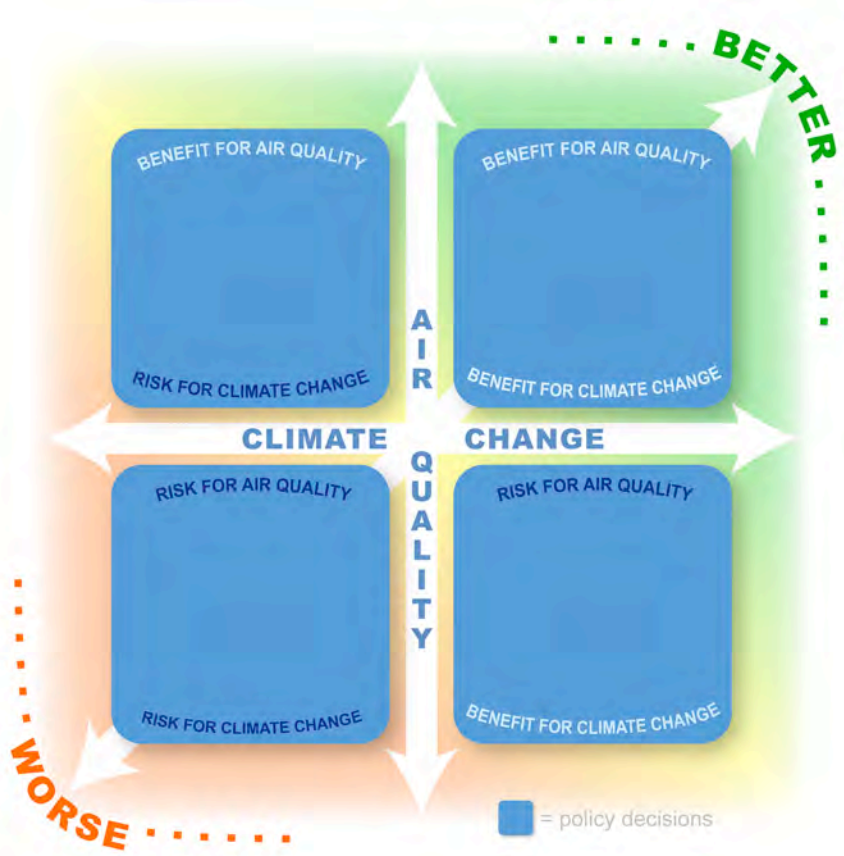

Figure 9 - Schematic of the trade-offs between the implications for regional air quality and climate change of new policies for management of the atmosphere [adapted from *Williams, 2012*]

In addition to air pollution having an impact on radiative forcing, climate change has an impact on air pollution meteorology and chemical process. Projected changes in surface temperature and precipitation (Figure 10) due to radiative forcing caused by both long and short-lived climate forcers will impact regional air pollution. A strong warming occurs over the northern mid-latitude continents and no area shows cooling [*Jacob and Winner*, 2009]. The increase in temperature leads to an increased frequency of heat waves, which are strongly associated with high pollution episodes, e.g., the 2003 heat wave in Europe. Precipitation is expected to increase due to increased evaporation from the oceans. However, the frequency and intensity of the increased precipitation varies considerably at a regional scale. Less frequent but heavier precipitation events could lead to more pollution episodes by reducing wet deposition of aerosol and other pollutants. Models also indicate a warming climate could impact large-scale atmospheric dynamic patterns. For example, *Leibensperger et al.* [2008] show a significant long-term decline in the number of summertime mid-latitude cyclones across the northeastern United States, which in turn strongly correlates with the number of high ozone episodes due to decreased pollutant ventilation. It is now widely recognized that air pollution and climate change can no longer be considered as separate issues in the scientific and policy communities [*Tai et al.,*2010].

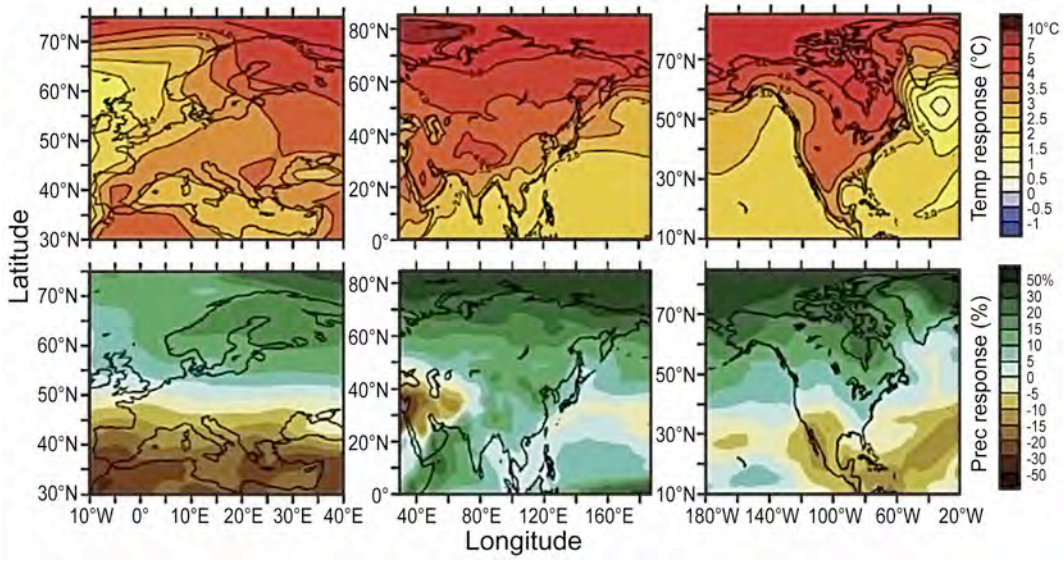

Figure 10 - Differences in annual mean surface temperatures and precipitation in Europe, Asia, and North America for 2080-2099 vs. 1980-1999 averaged over an ensemble of about 20 GCMs contributing to the IPCC 4th Assessment [*Jacob and Winner*, 2009]

1.3 HEALTH IMPACTS OF AIR POLLUTION

There is great evidence linking air pollution with mortality and morbidity in the general population [*Brunekreef and Forsberg*, 2005; *Zanobetti and Schwartz,* 2005; *Pope and Dockery*, 1996; *Nawrot et al*., 2006; *Dominici et al.*, 2006; *Barnett et al.*, 2006; *Guaderman et al.,* 2007]. Several physiopatological mechanisms of injury are described [*Osornio-Vargas et al.*, 2003; *Inoue et al.*, 2006;*Yeatts et al.,*2007; *Sakamoto et al.*, 2007; *Barlowet al.,* 2007]. Public health damage is consistently found with adverse effects concentrated in urban areas both in developed and developing countries [*WHO*, 2005]. The range of adverse health effects is broad, affecting both the respiratory and the cardiovascular system. Children, women, and older adults are the most susceptible to these adverse health effects in the general population [*WHO*, 2005; *Peel et al.*, 2007; *Miller et al.,*2007; *Tecer et al*., 2008]. The risk increases with intensity of exposure. Little information supports the presence of a threshold level for these effects. In fact, effects were found at very low levels of particulate matter, e.g. 3 to 5 µg/m3. The adverse health effects of air pollution are observed both in short-term exposures and for long-term exposures [*WHO*, 2005]. These problems are exacerbated by indoor air pollution typically caused by the burning of solid fuels.

Low-income population in general, and women and young children in particular are subject to a higher exposure since they spend the most time near the domestic hearth [*WHO*, 2009]. One of the main sources of pollution and health damage is traffic [*Kim et al.,* 2008].

The World Health Organization conducted a study of the burden of disease caused by environmental problems attributed to air pollution effects on respiratory diseases, perinatal conditions and birth defects, cancer, cardiovascular diseases, bronchial obstructive disease, and asthma. The study estimated that in developing countries 42% of all respiratory diseases are attributable to air pollution [*Prúss-Utsun*and *Corvalán*, 2006]. Exposure to complex mixtures of air pollutants, mainly particulate matter and ozone, causes structural lung changes that are induced by sustained inflammation, leading to vascular reconstruction of the lung airways and impairing the repair process. Children are particularly vulnerable to respiratory problems because of their physical characteristics and behaviour. In children under 5 years of age it has been estimated globally that acute lower respiratory infections (pneumonia, bronchiolitis and bronchitis) are responsible for about 20% of the 10.6 million deaths annually worldwide. About 90% of these deaths are due to pneumonia [*Rudan et al.,* 2004].

Over the last decade research shows an increased risk of cardiovascular disease due to both particulate matter and ozone exposure [*Tsai et al.*, 2003; *Kan and Chen*, 2003; *Hong et al.*, 2002; *Tamagawa and Van Eaden*, 2006; *Maheswaran et al.*, 2005; *Higgs*, 2011]. Many cardiovascular indexes show cardiovascular injury induced by increased levels of ambient particles (changes in the heart rate, or heart rate variability, blood pressure, vascular tone, and blood coagulability). In addition, chronic exposure to increased concentrations of particulate air pollutants accelerates the progression of atherosclerosis [*Simkhovich et al.*, 2008]. The evidence suggests that stroke mortality and hospital admissions are higher in areas with elevated levels of outdoor air pollution because of the combined acute and chronic effects of air pollution on stroke risk [*Maheswaran et al.*, 2005]. The impacts of air pollution on human health are the main driver toward implementing air quality regulations.

Vector-borne diseases, impacts on maternal and newborn health, nutrition of general public and heat related stress are among the health problems that have been linked with climate change [*Aklesso et al.,* 2011; *Rylander et al.,* 2011; *Myers and Bernstein,* 2011; *Toutan et al.,* 2011].

1.4 AIR QUALITY REGULATION

Following the major pollution episodes in Los Angeles USA and London UK in the 1950s, governments began to implement legislation to first fund research on air pollution and then to control air pollution. This effort culminated in the US with enaction of the 1970 Clean Air Act, which authorized the development of comprehensive federal and state regulations to limit emissions from both stationary and mobile sources and initiated the National Ambient Air Quality Standards (NAAQS) amongst other regulatory programmes. The Clean Air Act was amended in 1977 and again in 1990, with the latter implementing the Acid Rain Programme and significantly increasing the authority and responsibility of the federal government to protect human health and the environment from the impacts of air pollution (http://www.epa.gov/air/caa/caa_history.html). Table 2 shows the current NAAQS for the US

Table 2 - US National Ambient Air Quality Standards

Pollutant	Concentration	Averaging Period
Particulate Matter ($PM_{2.5}$)	15 µg/m3 35 µg/m3	1 year 24 hour
Particulate Matter (PM_{10})	150 µg/m3	24 hour
Ozone	0.075 ppm	8 hour
Nitrogen Dioxide	100 ppb 53 ppb	1 hour 1 year
Sulphur Dioxide	75 ppb 0.5 ppm	1 hour 3 hour

In Europe, air quality management began with the signing of the United Nations Economic Commission for Europe (UNECE) 1979 Geneva Convention on Long-Range Transboundary Air Pollution (LRTAP), which has been extended by eight protocols, the 1999 Gothenburg Protocol being the last one. LRTAP aims to limit and gradually reduce and prevent air pollution by developing policies and strategies across its 51 parties. In parallel to LRTAP, in 2001 the European Commission established National Emissions Ceilings (NEC), which set national emissions limits for four pollutants that are responsible for acidification, eutrophication, and ground-level ozone pollution. The NECs are largely based on the Gothenburg Protocol. In 2005, the European Commission also launched the Thematic Strategy on Air Pollution (TSAP), the first of its seven Thematic Strategies in the European Union's Sixth Environment Action Programme (EAP). TSAP established interim objectives for air pollution in the EU and proposes appropriate measures for achieving them, e.g. setting air quality standards and rules for monitoring. Air quality management in the European Union is thus an interplay between the Air Quality Directives defined under TSAP, the NEC directives, and LRTAP. Table 3 shows the current European Commission Air Quality Standards.

Table 3 - European Commission Air Quality Standards

Pollutant	Concentration	Averaging Period
Particulate Matter ($PM_{2.5}$)	25 µg/m3	1 year
Particulate Matter (PM_{10})	50 µg/m3	24 hours
Ozone	120 µg/m3	8-hr
Nitrogen Dioxide	40 µg/m3 200 µg/m3	1 year 24 hour
Sulphur Dioxide	125 µg/m3 350 µg/m3	24 hours 1 hour

In 1987, the World Health Organization (WHO) published *Air Quality Guidelines for Europe*. The aim of the guidelines was to provide a basis for protecting public health from adverse effects of air pollutants, to eliminate or reduce exposure to those pollutants, and to guide national and local authorities in risk management decisions. In 2005, following important new research from low- and middle-income countries, the WHO released new *Air Quality Guidelines* for four common pollutants (PM, O_3, NO_2, and SO_2) that are intended to inform policy-makers from different parts of the world on appropriate targets for policy related to air quality management. Table 4 shows the 2005 WHO air quality guidelines.

Table 4 - World Health Organization Air Quality Guidelines

Pollutant	Concentration	Averaging Period
Particulate Matter ($PM_{2.5}$)	10 µg/m^3 25 µg/m^3	1 year 24 hour
Particulate Matter (PM_{10})	20 µg/m^3 50 µg/m^3	1 year 24 hour
Ozone	100 µg/m^3	8 hour
Nitrogen Dioxide	40 µg/m^3 200 µg/m^3	1 year 1 hour
Sulphur Dioxide	20 µg/m^3 500 µg/m^3	24 hour 10 minute

It must be pointed out that air quality improvements throughout the world do not merely follow from the dictation of air quality standards similar to those designed by US EPA, the European Commission, or the WHO. Each country's national air quality standards should and will likely vary according to the approach adopted for balancing health risks, technological feasibility, economic considerations, and various other political and social factors. Determining national air quality standards and enforcing them depends greatly on the level of development and national capability in air quality management.

1.5 SCIENTIFIC TOOLS FOR STUDYING AIR POLLUTION IN MEGACITIES

One of the main challenges in addressing the impacts of megacities on the environment and human health is the interaction of different spatial scales. Traditional problems of urbanization include local to regional air quality and health issues. Scientific tools have been developed to cover local to regional scales at appropriate spatial and temporal resolutions. However, given their large and increasing emission strength, megacities can also have effects on the global scale, which can only be assessed by tools with global coverage. Global coverage, however, usually implies a loss of detail. The study of megacity impacts, both in terms of air quality and climate, thus necessitates the development and use of scale-bridging observations, emission inventories, and modelling.

1.5.1 Ground-based, ship, and aircraft observations

Ground-based observation networks are crucial for studying atmospheric chemistry in megacities. In particular, continuous ground-based meteorological networks are essential in order to characterize meteorological processes that control air pollution transport and stagnation events. In addition, year-round ground-based observation networks of atmospheric pollutants can detect exceedances of air quality standards, identify trends, detect or quantify emission sources, and determine the effects of air pollution control measures. Several examples of long-term ground-based observation databases of different spatial coverage are the AIRBASE database through EIONET (http://air-climate.eionet.europa.eu/databases/airbase), the EBAS database hosted by NILU Norway (http://ebas.nilu.no/), and the AirParifdatabase for Paris (http://www.airparif.asso.fr/). In addition, many regulatory agencies in developed and developing countries have air pollution monitoring networks that provide hourly concentrations of important air pollutants such as O_3, SO_2, NO_X, PM_{10}, etc. The number of air pollution monitors within a city and the air pollutants measured vary greatly. A more globally consistent air pollution monitoring network would provide a robust dataset to study atmospheric chemistry in megacities and the environmental and human health impacts of air pollutants.

More recently, surface “super-sites” have been incorporated into megacity field campaigns to complement and extend observations from aircraft and satellites. Super-sites provide simultaneous measurements of a variety of chemical and meteorological parameters and tend to have more specific scientific purposes than long-term ground-based observation networks. For example, a series of super-sites may be used to study the chemical composition of air parcels or of particles as a function of time and location. Some examples of data from super-sites are the recent MEGAPOLI [*http://megapoli.dmi.dk/*] and MILAGRO [*http://www.eol.ucar.edu/projects/milagro/*] campaigns that focused on Paris, France and Mexico City, Mexico respectively.

In addition to surface super-sites, field campaigns in megacities often include aircraft and ships that provide highly sophisticated platforms for studying atmospheric chemistry. Such platforms include in-situ and/or remote sensing instrumentation that provides observations of primary pollutants, secondary species, meteorological conditions, vertical profiles, and a wide range of other parameters. An airborne research platform provides a unique way to study the source region, vertical and horizontal dispersion, and chemical and physical transformation of atmospheric pollutants. Meanwhile, ship platforms provide the means to study meteorological and chemical processes at the surface ocean-atmosphere interface and in the marine boundary layer, which are regions difficult to measure using ground-based and aircraft observations, see Chapter 7 for further details.

The type of platform and instrumentation used to study atmospheric chemistry is dependent on the unique characteristics of the megacity of interest. Ground based, aircraft, and ship observations often provide a very detailed local to regional view of air pollution. Therefore, there is a need to integrate observations from field campaigns and monitoring networks in order to bridge scales from a local to regional to global level. Such integration would enhance the global perspective of the impacts of air pollution from megacities while still maintaining the critical detailed local and regional information.

1.5.2 Satellite observations

Ground, ship, and aircraft observations provide a detailed snapshot of atmospheric composition. Satellite-based observations provide a complementary global, continuous perspective, overcoming some of the temporal and spatial limitations of surface and aircraft measurements. Historically, satellites were most readily used to determine stratospheric atmospheric composition, largely because the presence of clouds and the overhead stratosphere make tropospheric measurement challenging. However, in recent decades, new instruments targeting tropospheric composition have been developed and deployed on satellites, rapidly enhancing our ability to track global tropospheric composition. These observations can provide critical information for monitoring and forecasting of air quality, studying long-range transport of pollution, and monitoring emissions of air pollutants and climate forcers. The majority of tropospheric measurements from space have employed nadir (downward looking) geometry. Occultation measurements offer much better sensitivity to trace species in the atmosphere; however detection is limited to the upper troposphere, which is generally less informative for air quality applications [*Burrows et al.,* 2011].

Solar backscatter measurements observe reflected and backscattered solar radiation in the ultraviolet (UV), visible (VIS), and near infrared (NIR) spectral region and are typically sensitive down to the lowest layers of the atmosphere, except in the case of significant cloud cover. This near surface sensitivity and the narrow field of view obtained with nadir observations are important pre-requisites for obtaining information about air pollutants. A number of key atmospheric trace gases absorb in the UV-visible range: O_3, NO_2, HCHO, SO_2, BrO, and Glyoxal (CHOCHO). Retrievals of trace gas concentrations are a two-step process. First the slant column densities (SCD) are estimated along the radiation path using absorption spectroscopy and second this quantity must be converted to the vertical column density (VCD) via the application of an air mass factor (AMF) (Figure 11). An AMF is the enhancement of the optical path length from the VCD due to the viewing geometry of the satellite, the scattering properties of the atmosphere and surface, and the vertical distribution of the absorber. AMFs are commonly determined by combining trace gas profiles estimated from a chemical transport model with a radiative transfer model. Accounting for clouds, aerosols and stratospheric contributions are non-trivial challenges associated with these retrievals. Figure 12 illustrates recent satellite instruments that apply solar backscatter techniques to measure global tropospheric composition.

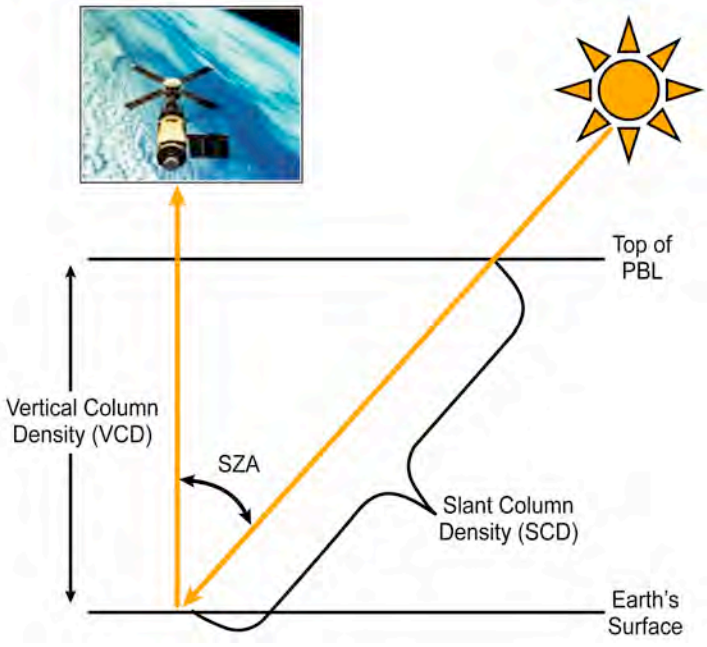

Figure 11 - Simplified schematic of solar backscattering measurements

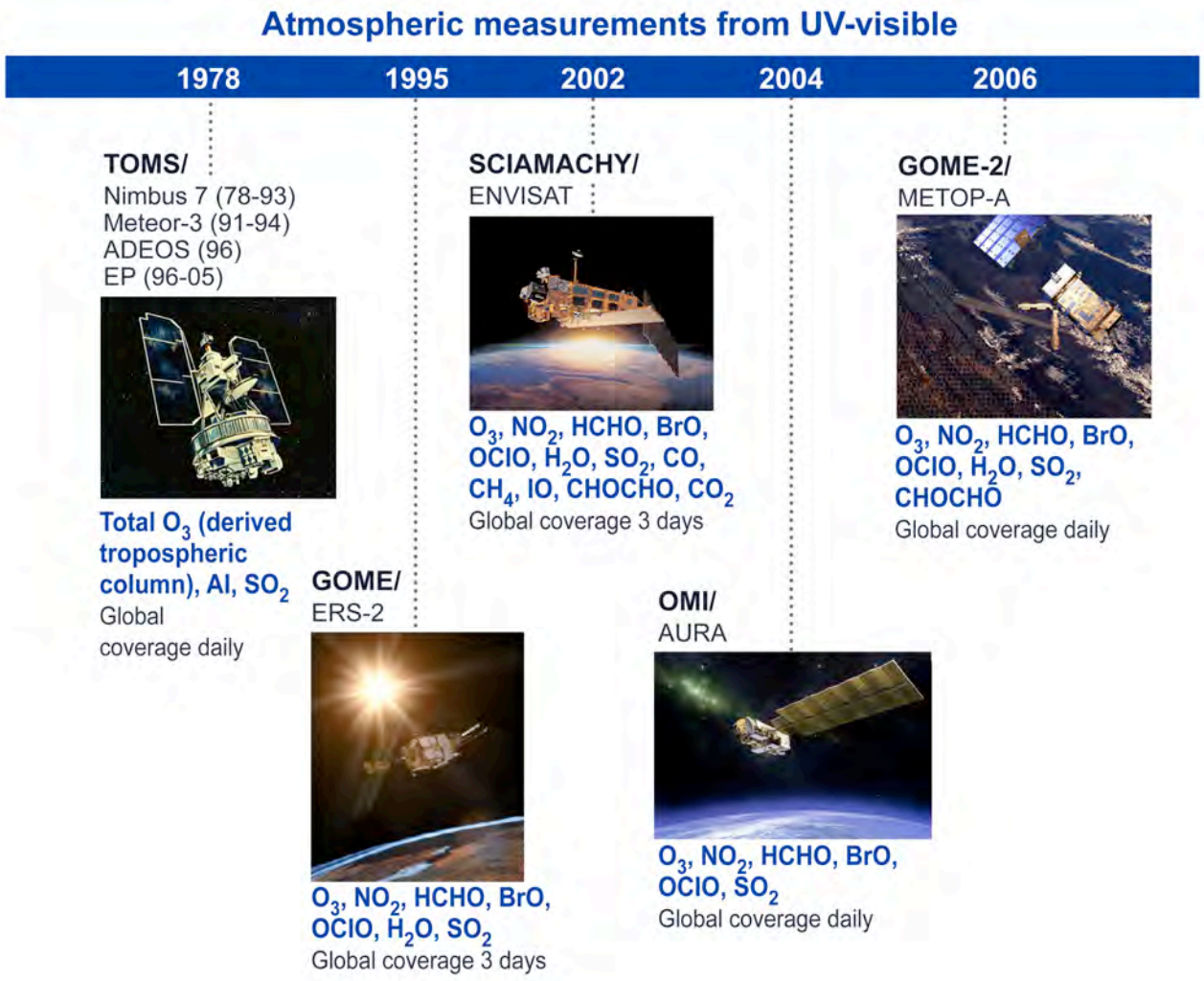

Figure 12 - Atmospheric composition measurements from UV-VIS-NIR backscatter instruments

A disadvantage of solar backscatter sounders such as GOME and SCIAMACHY is that the measurements require solar light, i.e., they can only be performed on the Earth's sunlit side. In contrast, satellite observations using thermal emission permit measurements during night as well as day. Thermal emission measurements use the thermal infrared (TIR) spectral region to measure tropospheric trace gases (Figure 13), including tropospheric O_3, CO, CH_4, H_2O, HDO and volcanic SO_2. Several experimental retrievals for species with weaker emission features, such as NH_3 and CH_3OH, have been recently demonstrated with the TES and IASI instruments [*Beer et al*., 2008; *Clarisse et al*., 2009]. Retrieval of these species and others may become routine in the future with higher spectral resolution instruments. Measurement sensitivity in the TIR depends primarily on the thermal contrast between the Earth's surface and the lowest layer of the atmosphere, which is typically highest in the mid to upper troposphere [*Deeter et al.,* 2007]. Thus, these instruments provide a characterization of the total column concentration, but with less sensitivity to boundary layer concentrations. Developing techniques to enhance sensitivity to the boundary layer, for example by integrating UV/visible and IR measurements is a major research challenge. Figure 14 shows the history of thermal emission instruments used to measure atmospheric composition.

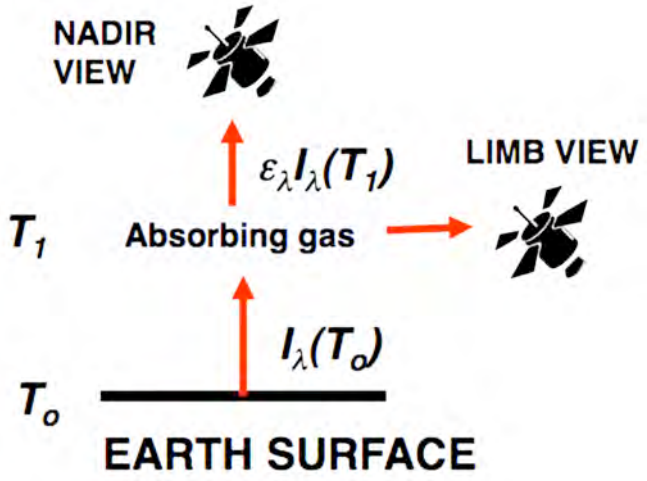

Figure 13 - Schematic of thermal emission measurements

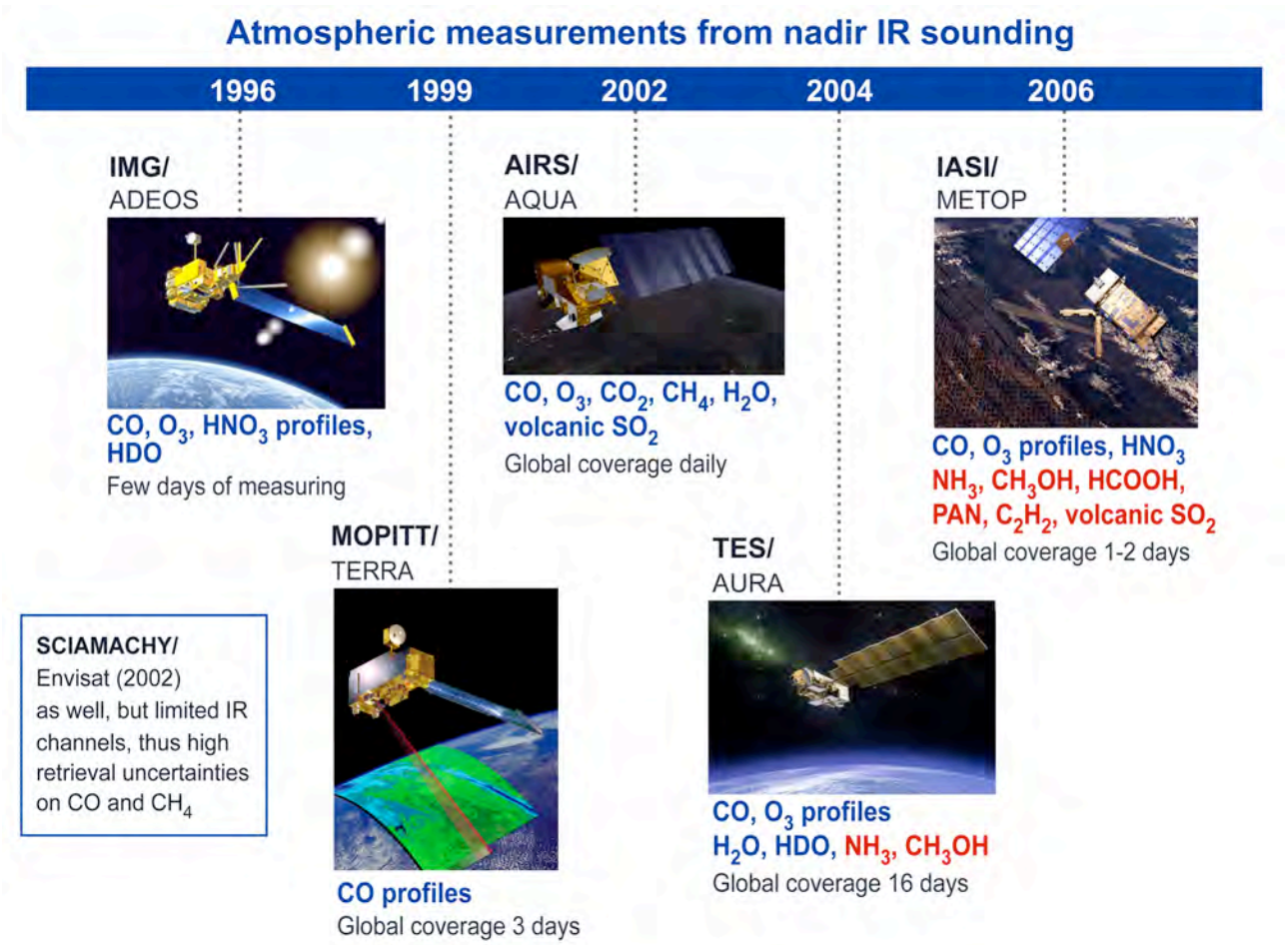

Figure 14 - Atmospheric composition measurements from thermal emission instruments

Tropospheric aerosols are challenging to characterize due to the variety and variability of aerosol sources as well as the short residence time of aerosols in the atmosphere. From space, the total radiative extinction resulting from aerosols can be measured. When summed vertically, this is the aerosol optical depth (AOD). Further details on the composition, size and properties of

these aerosols are difficult to extract. AOD is typically measured at visible wavelengths and is thus most sensitive to fine-mode aerosol. The information is therefore optimum for air quality applications focused on $PM_{2.5}$. Efforts to convert AOD observations to surface $PM_{2.5}$ concentrations require additional information on the vertical distributions and speciation of total aerosol, often obtained from chemical transport models [*Liuet al*., 2004]. AOD in the mid-visible is also very sensitive to the surface reflectivity. Historically, the first AOD retrievals were performed solely over oceans, where dark surfaces were more easily characterized. More recently, both multi-spectral (e.g. MODIS) and multi-angle (e.g. MISR) approaches have been applied to account for the impact of surface reflectivity and measure AOD over land as well as ocean [*Levy et al.,* 2010]. The presence of clouds is an additional challenge for passive remote sensing observations of aerosols. Active sensors, such as lidars, can profile aerosol and cloud extinction at high vertical resolution. In recent years, CALIOP observations have provided detailed observations of aerosol plumes, and overlaying vertical layers. The drawback of these observations for air quality applications is their limited coverage and long repeat times. However, detailed snap shots may provide important insight into aerosol distributions and export mechanisms over urban centres (Figure 15).

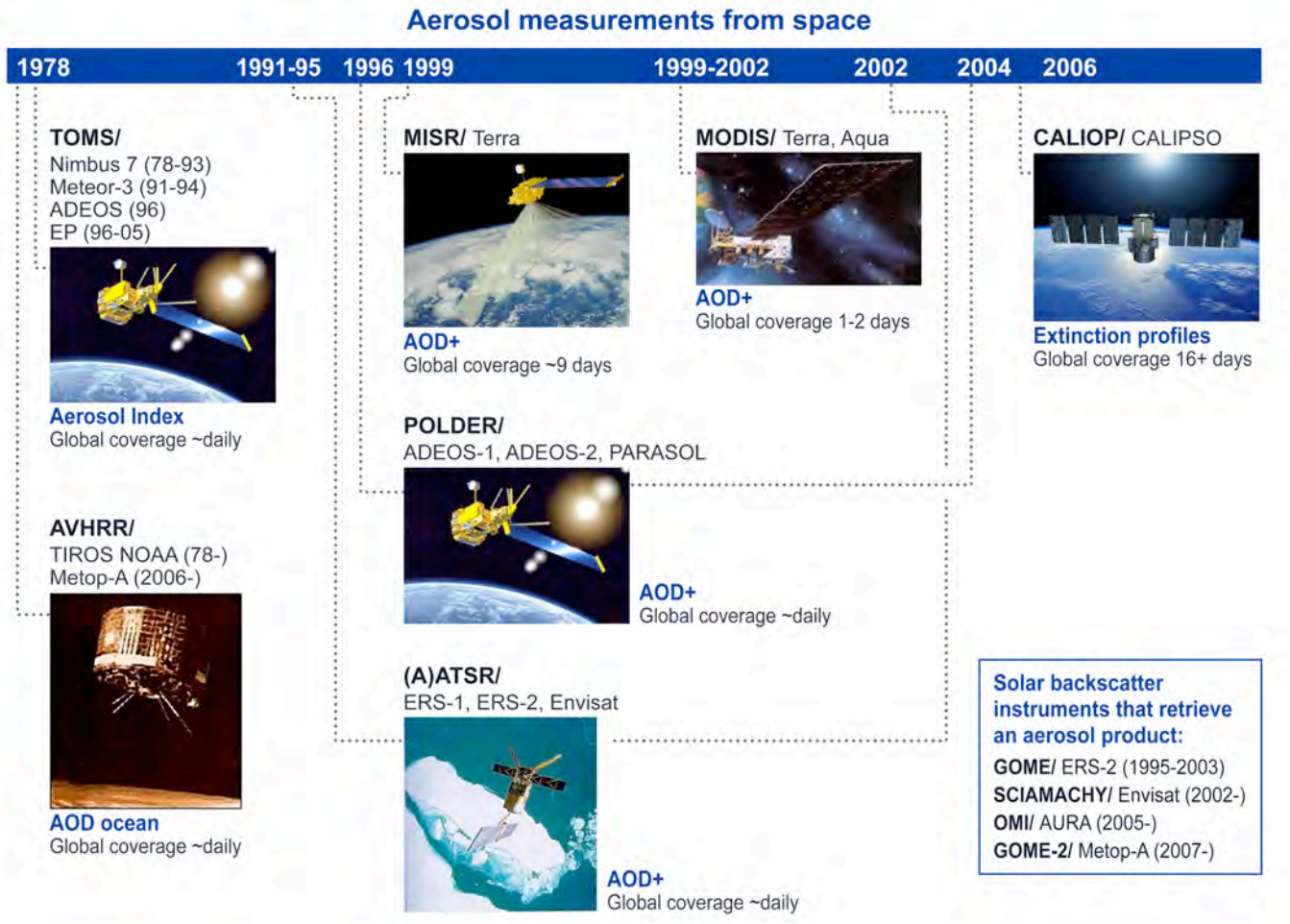

Figure 15 - Aerosol measurements from space

All current satellites relevant for air pollution research are in polar low Earth orbit (LEO). This limits the number of passes over a given area to twice per day, but observations can be limited further by cloud cover and by the swath width of satellites dictating a longer time between repeat visits. For air quality monitoring, more frequent observations would be highly desirable. This can be achieved with several LEO satellites or, for the tropics and mid-latitudes, with a geostationary satellite such as the proposed GeoTROPE mission [*Burrows et al.,* 2004]. At present such an instrument is not in orbit, but in a few years the launch of the geostationary Sentinel 4

UVN instrument is planned, which will deliver information on air pollutants at much higher temporal resolution than currently available. Similar instruments are planned for Asia and North America.

1.5.3 Emission inventories

A critical step in improving our understanding of the impact of megacitieson air quality, atmospheric composition, and climate is the development of high-quality emission inventories of relevant gases, aerosols, and their precursors. An emissions inventory is a current, comprehensive listing, by source, of air pollutant emissions associated with a specific geographical area for a specific time interval *[http://epa.gov/air/oaqps/eog/course419a/index.html].* Therefore, emissions inventories are developed for local, regional, and global applications as well as scientific and policy applications (chemical transport models, global climate models, trend analysis, regional and local scale air quality modelling, regulatory impact assessments, and human exposure modelling). Global emission inventories typically have resolutions of about half a degree or very recently one tenth of a degree (EDGAR V4). Regional inventories often have scales from 5 to 100 km whereas local inventories have resolutions of 1 to 10 km.

Two fundamental tools for developing emission estimates of air pollutants are emission factors and emission estimation models. Emission factors relate the quantity of a pollutant released to the atmosphere as a function of activity level for a given source by

$$E = A \; x \; EF \; x \left(1 - \frac{ER}{100}\right) \qquad \text{(eq. 1.6)}$$

where *E* is the emissions, *A* is the activity rate, *EF* is the emissions factor, and *ER* is the overall emission reduction efficiency using capture or control techniques (given in %). Emission factor uncertainty is dependent on the kind of emissions released, the number of tests used to determine the emissions factor, using the appropriate percentile within the distribution range, and the number of similar emissions units within a specific area *[http://www.epa.gov/ttn/chief/efpac/abefpac.html]*. Determining emission factors for every source and every pollutant under a variety of operating conditions throughout the world is a daunting task. Therefore, most emission factors are developed from only a limited sampling of the emissions source population for any given category, i.e. source population average. The limited number of samples likely results in emission factors not statistically representative of the actual emissions of a source category [*Seinfeld and Pandis*, 1998].

Emission estimation models use empirically developed process equations to estimate emissions from a given source. In general, an emission estimation model is used rather than an emission factor when a large number of equations, interactions, and parameters impact the emissions estimate. In many cases, emission estimation models are used to develop inventories for mobile and non-road source categories since it is difficult to directly measure activity for numerous individual sources operating under a wide variety of conditions.

Emission estimates derived using emission factors or emission estimation models are then developed into an emission inventory. There are two approaches to emission inventory development: down-scaled and bottom-up. The down-scaled approach develops emission inventories based on global, national, or regional data by downscaling the larger scale data using some measure of activity data or proxy (e.g. population density) related to the emissions in the area of study. The bottom-up approach estimates emissions for individual sources and then aggregates all sources in the area of study to derive regional, national, or global emission estimates. The down-scaled approach is typically used when local emissions are unknown, cost prohibitive to obtain, or not publically available. Figure 16 shows how a bottom-up emissions inventory can be merged into a regional down-scaled emissions inventory. In addition, aircraft and satellite observations can be used to derive top-down emission estimates using a-priori information from a bottom-up emissions inventory. The result is an optimized a posteriori estimate of emissions [*Martin et al.,* 2003].

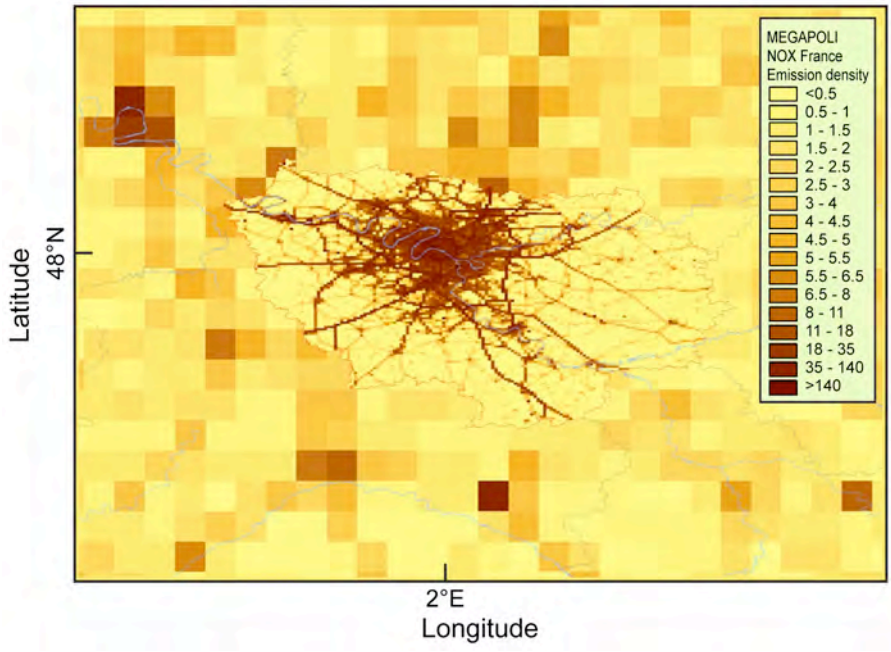

Figure 16: Merging a local bottom-up emisison inventory for Paris [*Airparif*, 2010] into a regional down-scaled Europan emisison inventory [*Denier van der Gon et al.*, 2009; 2011].

However, discrepancies often exist between emission inventories developed by down-scaled versus bottom-up approaches as illustrated in four European megacities or urban agglomerations; London, Paris, Rhine-Ruhr area (Germany) and the Po Valley (Italy). The local inventories use local statistics and activity data to estimate the emissions within their domain using a bottom-up approach. The European emission inventory is a down-scaled inventory based on national totals by source sector and then distributed over a grid by using source sector specific spatial distribution proxies [*Denier van der Gon et al.,* 2009]. A ratio comparison of down-scaled versus bottom-up emission inventories for these four cities is shown in Figure 17. A value of 1 in Figure 17 indicates that the local megacity inventory and the regional scale down-scaled inventory have the same estimate for the pollutants from the megacity domain. The differences can be quite dramatic. For example, the PM_{10} emission allocated to London and Paris by the regional scale inventory is a factor of 3-4 higher than the local inventories for London and Paris. The discrepancy for NO_X is limited but for other pollutants like PM, NMVOC, and CO the discrepancies are significant. SO_2 stands out in Figure 17 but this is not very relevant because European cities are small sources of SO_2, hence even a small mis-allocation of emissions using the down-scaling approach results in several factors overestimation. In general the smaller the domain, the larger the discrepancy between down-scaled and bottom-up emission inventories.

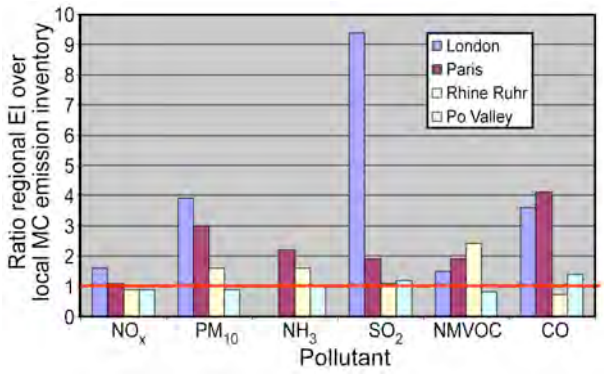

Figure 17 - Ratio of megacity emissions derived from the Regional European scale inventory compared to the local megacity emission inventory [*Denier van der Gon et al.*, 2011]

Emission inventories are critical in order to understand atmospheric chemistry in general, but especially for megacities. It is clear that discrepancies exist between down-scaled versus bottom-up methodologies as well as between national and regional emission inventory development. In order to understand atmospheric chemistry in megacities, an integrated emissions approach that transcends methodologies and political boundaries is needed. Such an approach would allow for scale-bridging of emission inventories that would provide consistent local, regional, and global emissions inventories.

1.5.4 Modelling

Along with laboratory studies and field measurements, modelling represents one of the main pillars of atmospheric chemistry research. Atmospheric models can be thought of as mathematical representations of the chemical and physical behaviour of the atmosphere. Most commonly, models are used to simulate the state of the atmosphere on computers. For this purpose the atmosphere is divided into a large number of small compartments, so-called grid boxes, for each of which the chemical and physical parameters are calculated based on initial conditions, internal tendencies, and external forcings. For instance, the concentration of a chemical component after some short amount of time, usually referred to as *time step*, can be calculated from its initial value and the rate of change during the time step. The rate of change will be a result of chemical reactions, emissions, transport through winds, wet removal by precipitation, and many other physico-chemical processes. The new calculated concentration will then serve as initial value for the calculation of the next time step, and so forth.

In such calculations, atmospheric models have to rely on various kinds of input, for instance values for chemical reaction rates, the radiation from the sun, and spatially and temporally resolved emission data. By using all available input information as well as our basic understanding of the processes occurring in the atmosphere, a model calculates the chemical state for each of its grid boxes and for each time step, thus yielding the distribution and evolution of chemical components in space and time.

An atmospheric model can be used in two fundamental ways, either as a *diagnostic* tool, or as a *prognostic* tool. In the former case the model aims to answer the question why things are the way they are. For example, what are the physical and chemical processes that contribute to observed air pollution? In contrast to the real atmosphere, processes can be switched on and off in a model in order to assess their importance. The impact of different types of emissions, e.g. anthropogenic vs. natural sources, can thus be investigated. Concentrations of species that cannot easily be measured can be calculated even for remote or inaccessible regions of the atmosphere, thus allowing gap-free distributions for all chemical components included in the model. As prognostic tools, models are used to calculate future distributions of chemical components. In this way, the future evolution of atmospheric composition responding to natural and anthropogenic influences can be predicted. Different emission scenarios can be simulated, and the effectiveness of different emission reduction scenarios can be assessed.

However, in order to gain confidence in a model it is important to carefully evaluate it against measurements. Only when it is established that the model succeeds in simulating past and present states of the atmosphere with reasonable accuracy, can it be used to predict future behaviour.

Figure 18 shows the basic components of atmospheric models, summarizing the basic input and output components of atmospheric models. There are various types of atmospheric models: Chemical transport models take meteorological parameters (temperature, wind, precipitation, etc.) as input and calculate the chemical composition of the atmosphere. Other models, e.g. numerical weather prediction models and climate models, calculate meteorological parameters themselves and take chemical distributions (e.g. ozone and aerosols) as input. A more advanced group of models calculates both meteorology and chemistry in a coupled way, allowing for interactions between chemistry and climate, but are computationally expensive so that simplifications have to be made, or the *model resolution* has to be coarse.

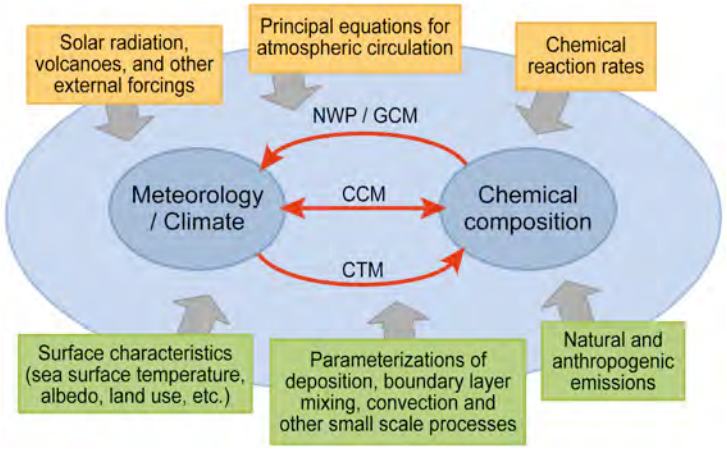

Figure 18 - Concept of atmospheric modelling. Yellow boxes: A priori knowledge that is not calculated by the model itself. Green boxes: Forcings that may or may not be calculated by the model itself but are needed for accurate model output. Blue ovals: Main purpose (or output) of the model. Numerical Weather Prediction models (NWP) and General Circulation models (GCMs) use prescribed fields of chemical species (e.g. aerosols) and calculate meteorology, while chemical transport models (CTMs) take meteorology as input and calculate the chemical composition of the atmosphere. Coupled climate-chemistry models (CCMs) calculate both meteorology and chemical composition and allow for couplings between the two. This figure is not exhaustive but is meant to illustrate the most basic components only

Resolution is one of the most fundamental features of a model. As a model usually calculates only one value per parameter for each grid box, assuming that the value is constant all over the grid box, many patterns that vary on finer spatial scales than the grid box dimension cannot be resolved. For example, some air pollutants have very short lifetimes compared to atmospheric transport timescales, and thus have rather uneven distributions, which reflect point sources of emissions. Also, topography and land use vary, in general, on smaller scales than can be resolved in a regional, let alone, global model. In order to resolve fine scale patterns in a model it is necessary to run the model on a higher resolution (i.e. to use a smaller grid box size). When a large domain has to be investigated this necessarily leads to a large number of grid boxes in the model, and often exceeds the limitations with respect to computer power.

Megacities affect their environment on very local scales, down to street level, but as strong emission sources they also have global environmental impacts. Conversely, global change will influence megacities related to both climate change and long-range transport of air pollution. Given the multi-scale character of the effects *of* and *upon* megacities, trade-offs have to be made in modelling. Models having a sufficiently fine resolution for local air pollution studies cannot be run globally, but only for a confined region of the atmosphere (so-called regional or local models). The provision of boundary conditions for these models is not trivial, and effects from the model domain on the areas beyond cannot be taken into account. On the other hand, global models that address changes in large-scale meteorological parameters and changes in the background concentrations of long-lived air pollutants have too coarse a resolution to be applied to local studies. Although computer power is increasing, this problem cannot be easily solved because the ongoing (and desirable) inclusion of ever more complex physical and chemical processes largely compensates for the increase in available computing power.

Baklanov and Nuterman [2009] therefore note the importance of building a chain of models of different spatial scales with nesting of high-resolution/small-domain models into lower-resolution/larger-domain models. Their example describes the bridging from regional to local scales, using coupled systems of obstacle-resolved urban models and coarser-scale local models. It is shown how features from outside the obstacle-resolved model influence its results, thus justifying a proper inclusion of information from the coarser scale. In general, scale interactions can play an important role in both directions, i.e. not only from the larger scale to the smaller scale, but

also from the urban/microscale to larger scale processes (e.g. transport of atmospheric pollutants, initially released and dispersed in a street canyon, urban climate and wind climatology, etc.).

Several scale-bridging methods can be distinguished. The following main strategies have been considered:

a) Online coupling of two versions of the same model (course resolution/large domain and fine resolution/small domain).
b) Zoomed grid realizing high resolution in a region of special interest, with a continuous transition towards coarser resolutions outside that region.
c) Offline coupling of two models of different scales, e.g. a global model delivers 3-hourly boundary conditions to a regional model, or the (fine resolution) output from a regional model is used to improve (coarse resolution) results of the larger scale model.
d) Online coupling of two different models of different scales, e.g. one regional model and one global model exchange information at each model time step, within one combined model system.

The techniques can also be grouped in terms of information flow only: 1) one-way nesting, where effects of the local/micro-scale on the larger scale are not considered, and 2) two-way nesting, where the scale effects in both directions are considered. In the last case both domains are run simultaneously to enable feedbacks (corresponding to item (d) in the list above), and the terrain in the overlapping areas must be comparative to avoid mass imbalances and numerical noise.

Observations of the atmosphere are typically unevenly distributed in space and time and with differing precision and accuracy. Models, on the other hand, provide a self-consistent framework but are subject to multiple errors due to uncertain estimates of parameters, inadequate representation of processes, and lacking of process understanding. However, the combination of measurements and models provides complementary information that leads to a better description of the system and its evolution. To achieve this, the combination must be made in an optimum sense, which either minimizes the distance between model and observations (variational approach) or minimizes the error variance of the system's predictor (statistical approach). Both variational (e.g., 4-D Var) and statistical (e.g., Kalman filter) methods have been widely used by the meteorological community over the last decades for dealing with weather forecasting as a way to avoid the uncontrolled propagation of errors due to the uncertain and incomplete description of initial conditions [*Kalnay*, 2003]. In atmospheric chemistry these methods are nowadays being increasingly used as both in-situ and remote observations become more readily available. These methods provide a way to improve emissions at different scales [*Pétron et al*., 2002; *Chai et al*., 2009; *Saide et al*., 2011], to limit the propagation of model errors [*Elbern et al*., 2007; *Carmichael et al*., 2008], and to evaluate and optimize monitoring networks [*Hoelzemann et al*., 2009; *Abida et al*., 2008].

1.6 SUMMARY

This chapter summarizes the world trend in urbanization, the atmospheric chemistry of air pollution and scientific tools to investigate air pollution, the impacts of air pollution on human health, and air quality regulations. The following chapters describe the current scientific knowledge of atmospheric chemistry in megacities in Africa, Asia, South America, North America, and Europe. A summary of megacity field campaigns is presented in Chapter 7 and the final chapter presents key issues that still need to be addressed in the study of atmospheric chemistry of megacities.

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CHAPTER 2 - AFRICA

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2.1 INTRODUCTION

Megacities are generally defined as cities with at least 10 millions inhabitants. In Africa, Johannesburg in South Africa, Cairo in Northern Africa and Lagos (Nigeria) are such megacities. Figure 1 shows the population density of Africa.

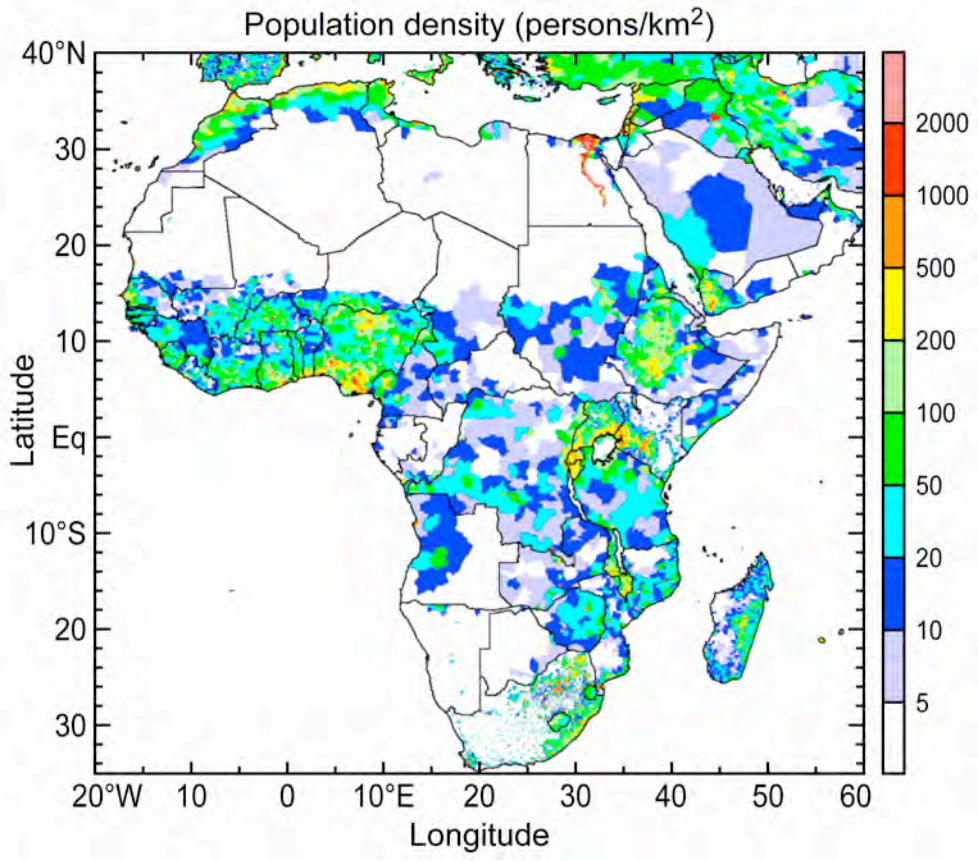

Figure 1 - Map of the population density in Africa (persons per km²), based on 0.25° gridded data for 2000 from the Center for International Earth Science Information Network (CIESIN) at Columbia University *(http://sedac.ciesin.columbia.edu/gpw/)*

As clearly seen from emissions in Figure 2, in these cities, population is exposed to high pollution levels. Nevertheless, in Lagos, Nigeria, Abidjan, Accra and other capitals, there is an explosive growth of population, partly due to high levels of rural migration towards coastal cities. Consequently, by 2020 population is expected to reach above 10 millions inhabitants in Abidjan and between 5-10 millions in Dakar, Bamako, Accra, Lomé and Cotonou. With such projections, a number of megacities will emerge in West Africa. This population pressure greatly increases pollution levels from traffic, burning of household wastes and charcoal and wood as domestic

energy sources. In addition, refineries and other industries, thermal and cement plants, power plants, roads and building construction are strong sources of pollutants, which also significantly impact air quality in African urban areas.

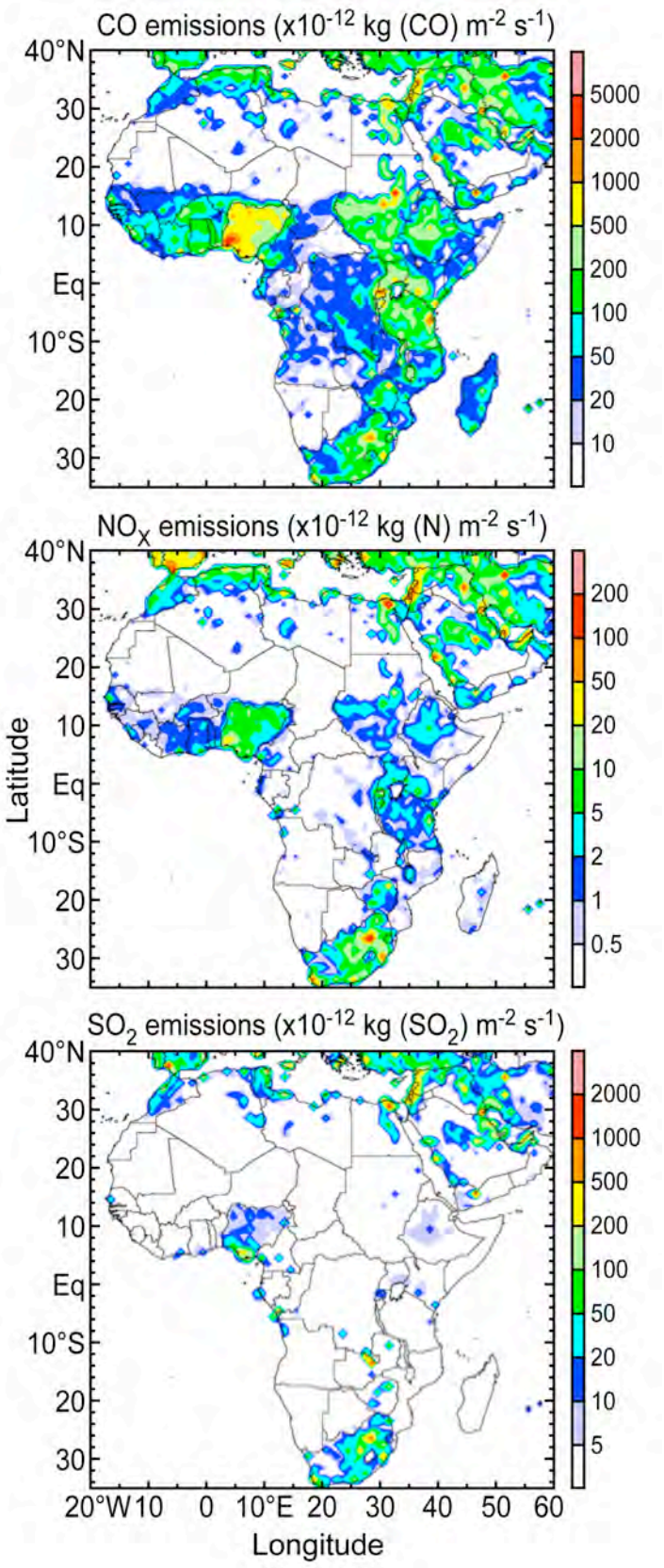

Figure 2 – (a) CO emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (b) NO_X emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (c) SO_2 emissions for the year 2000 based on the EDGARv3.2 FT2000 database

Figure 3 displays black carbon anthropogenic emissions from a regional emission inventory in different sectors (traffic, biofuel, power plant, industries) and different African regions. Anthropogenic Black carbon (BC) emissions in Africa are of the order of 0.68 TgC (about 13% of global emissions), whereas African anthropogenic organic carbon (OC) emissions are above 40%. The same pattern occurs in West, Central and South Africa, with predominance of domestic fire emissions and a moderate traffic contribution. It is also important to note the contribution of industrial activities and power plant emissions in South Africa. In Northern Africa, traffic is the predominant emission source with a smaller contribution of domestic fires and industrial activities. Total budgets for every region are of the same order, with enhanced emissions in West Africa as compared to other areas. With such differences in mind between African areas, this chapter will comprise three parts, respectively relating to megacities in (1) West Africa, (2) South Africa and (3) Northern Africa. In each part, specific features of some cities will be presented, followed by descriptions of on emission sources, atmospheric pollution and the impacts on health.

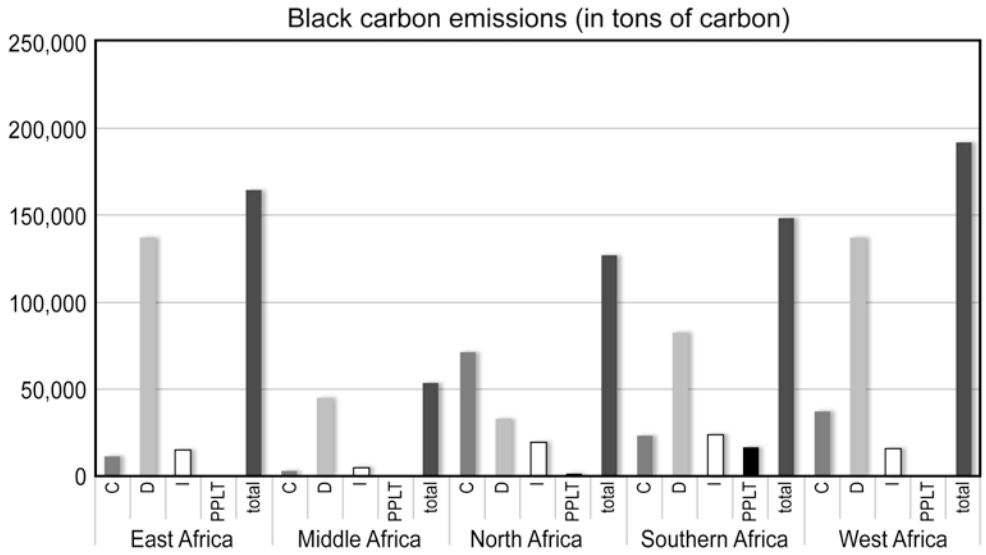

Figure 3 - Black carbon emission (in tons of carbon) in different sectors (traffic (C), biofuel (D), power plant (PPLT), industries (I)) and different African regions [*Liousse et al.*, 2012; *Assamoi*, PhD 2011]

2.2 WEST AFRICA

2.2.1 Specific features of some West African cities

In West Africa, climatic domains range from wet Tropics with heavy rainfall to arid deserts. Climate is wetter and humid in the South, getting drier and arid when moving north. So, there are two categories of cities: “wetter” ones along the Gulf of Guinea and others, in drier sahelian areas.

After the independencies in the 1960’s, large cities have been very attractive for neighbouring and rural population, leading to a net demographic densification, especially in coastal cities. Such unplanned, spontaneous city sprawl has led to socio-economical difficulties (lack of jobs, violence, etc.) with accompanying extreme poverty and the formation of shantytowns at the periphery of many cities. Consequently, there is a lack of basic amenities such as drinkable water, sewage systems, waste treatment, electricity, roads, urban transportation and even schools.

This also causes an intense degradation of suburban environments. One of the main reasons for the degradation, noticeable in Figure 2 and 3 and stressed in Table 1 for CO and volatile organic compound (VOC) emissions in Ouagadougou, is the increase in biofuel use, the main source of energy for cooking and heating. Gas energy is still a luxury in the absence of generalized wind or solar energy use. Initiatives already exist to test cleaner fuels (such as gas fuels) in new cook stoves in Ghana [*Bawakyillenuo*, 2012]. Another pollution source is related to

vehicles, such as second hand engines from Europe (France-Aurevoir vehicles) and from generalized 2-wheels individual transportation. The relative importance of traffic sources on NO_2, CO and VOC emissions is highlighted in Table 1 for Ouagadougou.

Table 1 - Gas pollutants and their origin in Ouagadougou (in tons/year)

Pollutants / Emission type	NO_2	CO	VOC
Wood combustion	98	17.127	1.346
Charcoal combustion	40	5.64	443
Automobile	846	3.766	479
Motorbikes	12	1.980	1.188
Total	996	28.517	3.456

Moreover, urban environment is degraded by external factors. For example, dust transport from northern deserts causes visibility reductions during Harmattan periods in late fall and winter (late November to mid-March). This pollution, reaching the surface in winter, is found aloft (900-1800 m) in summer. A second example as shown in Dakar [*Doumbia et al.,* 2012], is generalized biomass burning that occurs during winter (dry season) [*Liousse et al.,* 2010]. Finally, climatic factors also play an important role: high temperatures and humidity in coastal cities, drought, winds, etc., cause conditions for intense photochemistry, which results in strong pollution formation enhancement.

As a result, these cities and megacities are increasingly facing very acute public health problems due to air pollution at the origin of inhalation, ingestion and dermal contacts with pollutant gases and particles of all sizes (from nanometers to micrometers) and compositions.

Seven representative capitals are selected in West Africa, Abidjan (Ivory Coast), Cotonou (Benin), Bamako (Mali), Dakar (Senegal), Ouagadougou (Burkina Faso), Lagos (Nigeria) and Accra (Ghana). For each of them, environmental features will be highlighted in terms of geographical location, population and other pertinent economical parameters. This will allow a more detailed examination of the impact of those parameters upon air quality in West African sub regions.

Ouagadougou in Burkina Faso (12° 22' 20" N, 1° 31' 15" W, 300 m a.s.l.) is the country's largest city, with a population of 1.5 million inhabitants (2006). Ouagadougou, has a flat topography, with very poor peripheral environment, and is regularly swept by harmattan winds. Based on the Köppen climate classification, this city features a typical tropical savanna climate. The city is in the Soudano-Sahelian area, with rainfall of about 900 mm per year. The rainy season stretches from May to October, with 30°C average temperature. The dry season runs from December to May with a minimum temperature (19°) in December-January and maximum temperature (45°) from March to May. Primary industries are food processing and textiles.

Abidjan (5° 19' N, 4° 02'W) is the economical capital of Ivory Coast. It is the largest city of the country and the third-largest French-speaking city in the world, after Paris and Kinshasa. With a population of 8.9 million in 2006, Abidjan will soon become a megacity. The city has a tropical monsoon climate with a long rainy season from May to July, a short rainy season (September–November) and two dry seasons, though rain is observed even during these dry seasons. Abidjan is generally humid throughout the year, at levels generally higher than 80 percent. Total rainfall is

about 2,000 mm per year. Temperature is almost constant (about 27°C). The observatory of air quality envisaged in the Code of the Environment for air quality control is not yet set up.

Dakar, the capital of Senegal (14° 41' 34" N, 17° 26' 48" W), located on the Cap-Vert Peninsula at the Atlantic coast and the westernmost city of Africa mainland is a major regional harbour. Dakar and its metropolitan area have a population of 3.5 million. Dakar has a hot semi-arid climate with a short rainy season (July to October) and a long dry season (November-June). Total rainfall is 540 mm per year. Temperatures are not as hot as other African cities. Diesel fuel is predominantly used in traffic activities.

Bamako is the capital of Mali (12° 39' N, 8° 0' W) with a population of 1.8 million. Bamako is ranked as the fastest growing city in Africa, the sixth one in the world. Bamako, in a basin site surrounded by sandstone plateaus indented with small valleys favouring air channelling, is regularly swept by harmattan winds. Based on the Köppen climate classification, Bamako has alternately a tropical wet and dry climate. Annual temperatures are over 30°C with higher temperatures from March to May (46°C) and lower ones from November to February (16 to 19°C). The rainy season is from July to September (total annual rainfall about 990mm). Local manufacturing includes textiles, meat processing, and metal goods. There is commercial fishing on the Niger River. Bamako has a large fleet of 2-stroke vehicles.

Accra (5° 33' 00" N, 0° 12' 00" W), capital of Ghana, has a population of 5.7 million. The average annual rainfall is about 730 mm during two rainy seasons (April - mid-July) and in October. There are only small temperature variations throughout the year with minimum temperature (24.7°C) in August and maximum in March (28°C). Relative humidity is generally high, varying between 65% in mid-afternoon and 95% at night. Predominant wind directions are from the WSW to NNE sectors with speeds ranging between 8 and 16 km/h. Economical activities in Accra are in the financial and agricultural sectors, Atlantic fishing and manufacturing of processed food, lumber, plywood, textiles, clothing and chemicals.

Cotonou (6° 22' N, 2° 26' E) is Benin's economical capital with a population estimated between 761,137 to 1.2 million, the population in 1960 was only 70,000. The urban area continues to expand, notably towards the west. Based on Köppen's climate classification, Cotonou displays alternately a tropical wet and dry climate with two rainy seasons (April–July and September–October with 800 to 1,200 mm of rain per year) and two dry seasons. In December and January, the city is affected by harmattan winds. Temperatures are relatively constant throughout the year, with higher temperatures at 30°C, and lower temperatures at 25°C. A familiar feature of the city is the motorcycle–taxis (Zémidjans).

Lagos (6° 27'N; 3° 21' E) was the federal capital of Nigeria until 1991 with a population growing very rapidly from 346,137 in 1950 to 1.1 million in the 1963 census. With an estimated annual growth rate of 6% per annum, the estimated population of Lagos is now expected to reach 25 million by 2015. Despite the high poverty index for Nigeria, Lagos State has a relatively High Human Development Index (HDI) with a 25% contribution to National GDP, hosting most of the industrial and commercial Nigerian activities. Based on Köppen's climate classification, Lagos has a tropical savanna climate with two rainy seasons (April-July and October-November) and two dry seasons (December-March and August-September). Total annual rainfall amounts to about 1800mm, with temperatures in the range 24°-30° throughout the year.

2.2.2 Emissions of air pollutants in West African megacities

Recently, an urgent need has emerged to develop regional emission estimates including African specificities within global inventories. Indeed, diesel consumption from the United Nation database has been shown to highly differ from regional database such as Africaclean. Significant differences were also noticed between traffic emission factors measured in Europe and West Africa [*Assamoi and Liousse*, 2010]. For all these reasons, a regional African emission inventory has been developed for carbonaceous aerosols, including African specificities for 2005 and 2030 projections [*Assamoi and Liousse*, 2010; *Liousse et al.*, 2012]. Estimates given by this new inventory are different from ACCMIP and RCP global inventories, especially for organic carbon

particles [*Lamarque et al*., 2010]. In 2005 and 2030, mean OC estimates are higher by at least a factor 2 in the regional inventory versus ACCMIP/RCP inventories. Such investigations still need further development. Indeed, a study on traffic intensity in Burkina Faso for the period 2001 to 2020 has shown that vehicle inventories (cars and motorbikes) are evolving very rapidly from 2001 to 2005 (Figure 4). In Lagos (Nigeria) also, more information is now available for road transportation with details on gasoline and diesel consumption by types of vehicles (motorcycles, light duty trucks, cars and heavy duty trucks). Such examples underline the urgent need to fully collect regional fuel data in every West-African country.

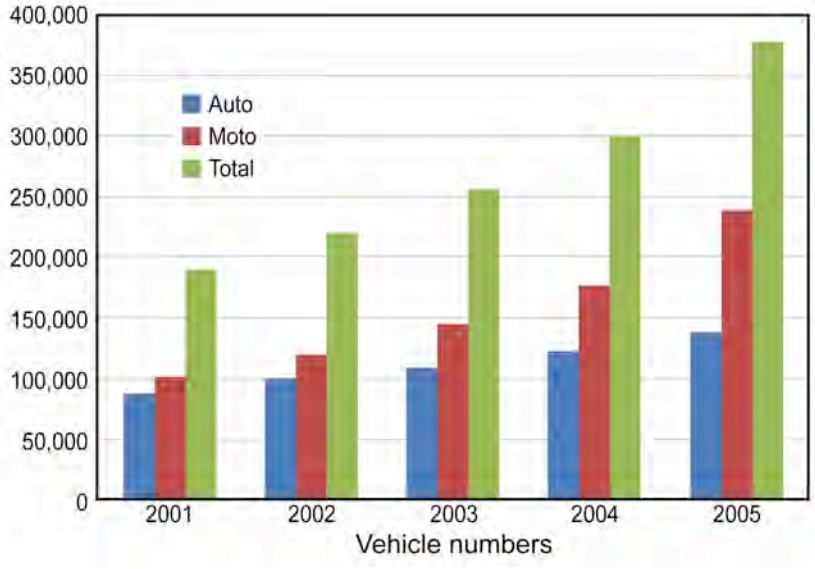

Figure 4 - Evolution of the number and composition of the pool motor vehicles in Burkina Faso

This work has also shown that all sources present in West Africa are not well characterized. That is the case for air transport and the small industries sector in Nigeria and, more importantly and generally, in West Africa, emissions from unpaved roads, waste management and burning (such as electronic wastes) and processing of used lead batteries.

2.2.3 Atmospheric pollution

Unexpected high pollution

Unexpected high pollution levels have been recently measured at traffic sites in West African megacities in the frame of IGAC DEBITS in Africa (IDAF) and Pollution des Capitales Africaines (POLCA) programmes. NO_2 and $PM_{2.5}$ concentrations have been found to be particularly high in many cities (Figure 5 and 6), much higher than WHO norms [*Liousse and Galy-Lacaux*, 2010; *Doumbia et al.*, 2012]. This has been confirmed with more details in Dakar and Bamako. For example, Figure 7 shows the annual NO_2, HNO_3 and SO_2 concentrations in Dakar and Bamako, which are much higher than at the rural site of Katibougou (Mali) [*Yoboué et al.*, 2012]. Another example given in Table 2 shows that black carbon pollution measured in West Africa is of the same order of the rest of the world. Note that the high $PM_{2.5}$ concentrations reported in *Doumbia et al.* [2012] were also obtained in *Guerreiro et al.*, [2005] and *Dieme et al*. [2012] in Dakar. Finally, studies in Ouagadougou in December 2007 and April 2010 and in Abidjan in January 2008 by *Linden et al.* [2012] and *Kouassi et al.* [2010] show urban centers, urban residential, suburban residential and industrial sites have exceptionally high concentrations of gases and particles. PM_{10} mass concentrations found in Ouagadougou, for example, also largely exceed WHO 2006 air quality guideline.

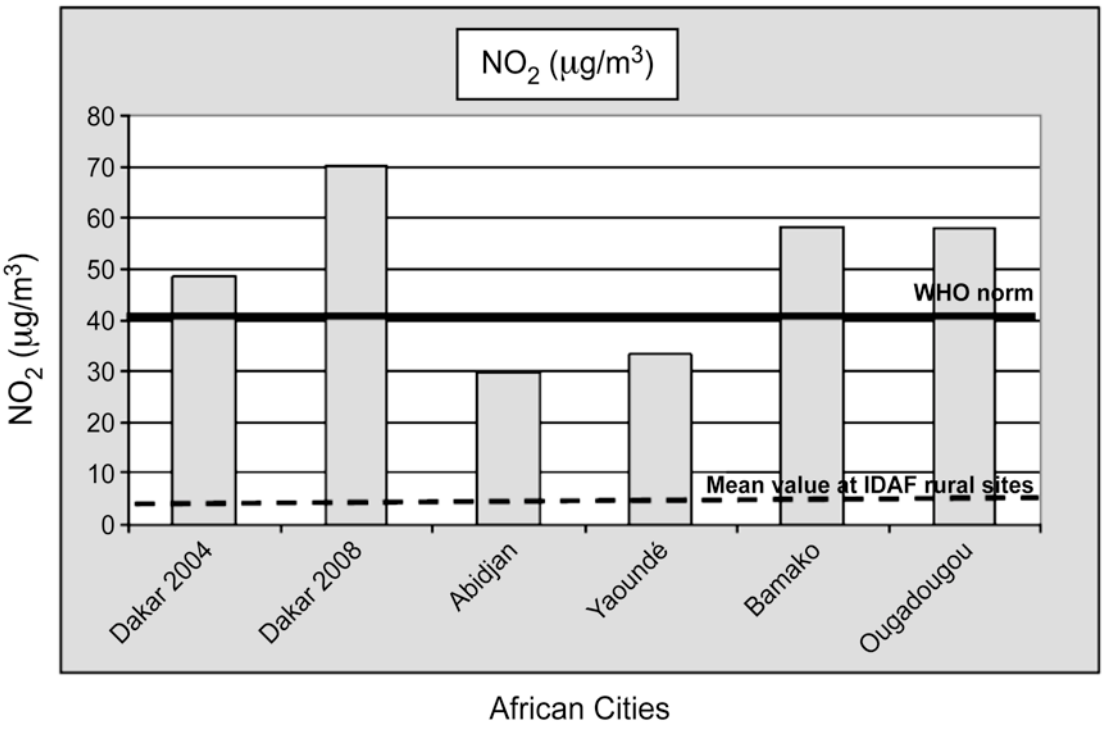

Figure 5 - NO_2 measurements in African capitals [*Liousse and Galy-Lacaux*, 2010].
(Also shown in Chapter 7 as Figure 15)

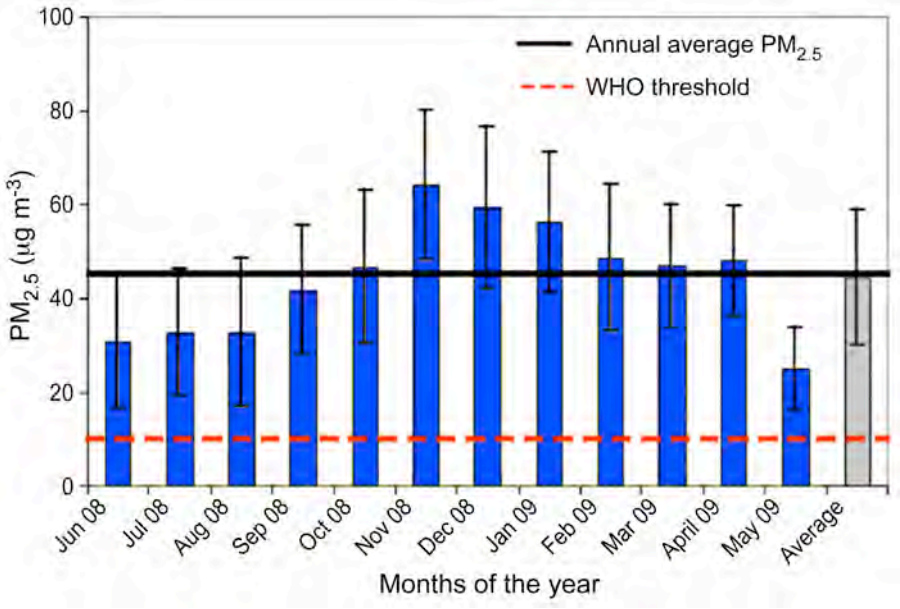

Figure 6 - Monthly $PM_{2.5}$ concentrations in Dakar (from June 2008 to May 2009) [*Doumbia et al.*, 2012]
(Also shown in Chapter 7 as Figure 16)

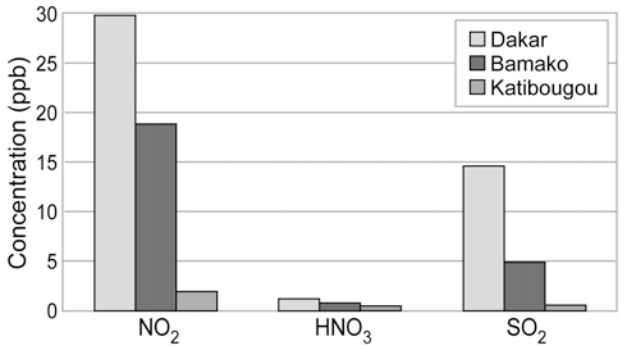

Figure 7 - Mean annual concentrations of NO_2, HNO_3 and SO_2 at Dakar and Bamako traffic sites. Values for Katibougou (IDAF rural site) are added for comparison (*Mali,* 2008-2010), (Yoboué et al., 2013, in prep.)

Table 2 - Black carbon concentrations at urban sites (adapted from Table 1 in *Doumbia et al.*, 2012)

Location	Period	BC (µg m^{-3})	References
Dakar. Senegal	June 08-July 09	**5.7-15.4**	*Doumbia et al*., 2012
Bamako. Mali	April 2008	**19.2 ± 8.9**	*Doumbia et al*., 2012
Cotonou. Benin	May 2005	**4.9 ± 3.9**	*Doumbia et al*., 2012
Yaounde. Cameroon	June-July 2004 August 2010	**2.4 ± 0.8** **5.7 ± 2.3**	Ouafo, Personal Com.
Paris. France	Aug.-Oct. 97	**10-20**	*Ruellan et Cachier*, 2001
Singapore. Asie	Jan.-Dec. 00	**3-14**	*Balasubranmanian et al*., 2003
Kanpur. India	December 04	**12.3**	*Tripathi et al*., 2005
Karachi. Pakistan	Apr. 06-Apr. 07	**2-15**	*Vincent et al*., 2009
Beijing. Chine	Jan. 03- Aug. 04	**1.2-16.3**	*Guinot et al.,* 2007
Xi'an. Chine	Sept 03-April 04	**16.5 ± 9.8**	*Li et al*., 2004
Marylebone. Londres	Oct-Dec 06	**12.2**	*Green*, 2007

Spatial and temporal variability of gases and particulate concentrations in West African cities

In West-African cities, concentrations are spatially highly variable : this was observed for example in Cotonou (Benin) and Bamako (Mali) during AMMA and POLCA programmes respectively, from real time measurements of BC and CO in a taxi [*Doumbia et al.*, 2012]. It was also noticed in Dakar (Sénégal) and in Bamako (Mali) with passive samplers for NO_2 and SO_2 gases in the frame of POLCA [*Yoboué et al*., 2012]. *Doumbia et al.* [2012] show that differences by a factor of 5 may be found between different BC measurements downtown in Bamako, with maximum and minimum levels respectively located near the markets and the Niger river. In Burkina Faso, *Linden et al.* [2012], *Boman et al*. [2009], *Eliasson et al*. [2009] and *Arku et al.* [2008] have observed high variability in PM concentrations at street level, in downtown Ouagadougou, with maximum values near unpaved roads and vegetated areas. Also in this study, it is underlined that PM_1 and combustion pollutant concentrations are much higher at traffic sites than in suburban areas, whereas PM_{10} concentrations are much higher in suburban zones, due to the important relative impact of household fuel use in such areas.

In West-African cities, concentrations display high temporal variability. In most of them, we can underline morning and evening peaks associated with combustion pollution. In Ouagadougou, background concentrations are highly affected by diurnal meteorological stability and dynamics : the highest CO concentrations are found in conditions of extreme stability during evening rush hours with a CO peak much larger than in the morning (at 7am) [*Linden et al.*, 2012]. Evening peaks are also observed in BC concentrations at Bamako traffic sites (Figure 8). However, such peaks are lower in Dakar and Cotonou than in Bamako [*Doumbia et al.*, 2012]. This can be explained by meteorological effects as previously shown in Ouagadougou, but also by important use of biofuel at this time of the day at Bamako. Note that this late and large peak never exists for NO_2 concentrations, since its considered only as a traffic tracer. Weekly variations display higher concentrations from Monday to Friday than for Sundays, independent of the type of pollutant (gaseous or particulates).

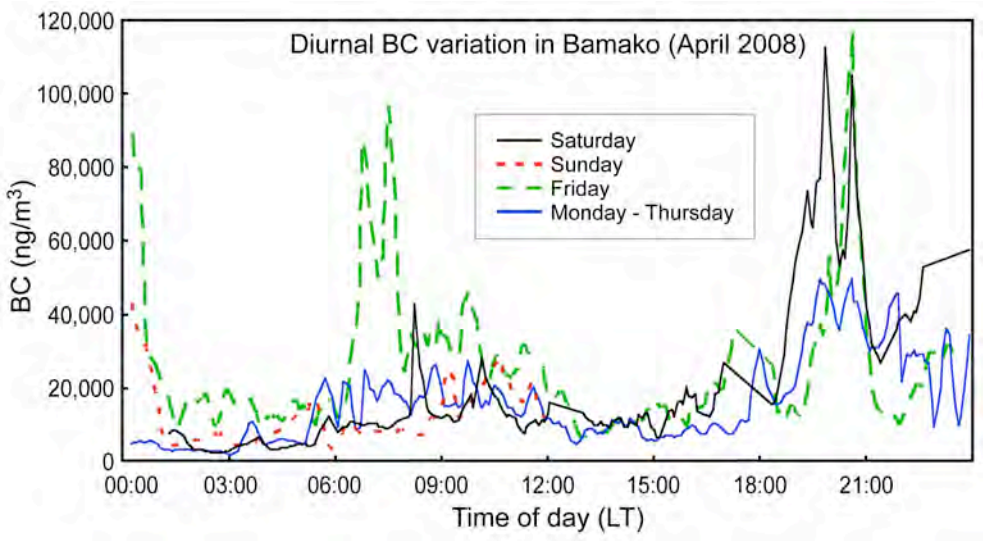

Figure 8 - Weekly variations of BC concentrations in Bamako [*Doumbia et al.*, 2012]

As already mentioned, gaseous and particulate concentrations are affected both by local pollution and/or long range transport. This is reflected in monthly variations (not shown here) of BC and NO_2 concentrations, displaying higher values during the biomass burning season. This is also seen during dust tranport events, particularly in Bamako and Ouagadougou. Finally, note that during the wet season, wet deposition also contributes to lower particulate concentrations than in the dry season in all capitals.

Link between pollution chemical characterization and main combustion sources

As discussed above, main prevailing combustion sources in West-African cities are traffic, domestic fuel combustion, dust resuspension from unpaved roads, waste burning and industries. In Ouagadougou, toluene/benzene, NO_X/benzene and PM1/PM10 ratios are used to discriminate between traffic and domestic fuel combustion sources [*Linden et al.*, 2012].

In Abidjan, where rural/urban/industrial sites have air pollution monitors, the influence of both natural and anthropogenic emissions are observed [*Kouassi et al.*, 2010]. Higher benzene concentrations are found in urban and industrial sites than in the rural one, whereas other organic gaseous compound concentrations are higher at the rural site. Transition metals are retrieved at every sampling site (Table 3). Due to coastal influences linked to the location of the industrial site, chlorides are more important at this site than at other sites. Iron (Fe) is more important in urban than in industrial sites, whereas aluminium (Al) is maximum at the rural site. The same compounds have been studied in Dakar at traffic and rural sites by the same team [*Dieme et al.*, 2012]. They have drawn similar conclusions for Fe, i.e. more abundant at urban than at rural sites due to anthropogenic influence, but lower concentrations of Al are retrieved at the rural site near Dakar

than near Abidjan. In contrast to Abidjan, lower concentrations of lead are measured in Dakar : this is due to unleaded gasoline effectively used in Dakar, which has not yet been implemented in Abidjan. Same conclusions appear for SO_4 concentrations in both studies, higher at urban than at rural sites due to unregulated combustion of fossil fuels in the traffic and industrial sectors. Both studies also suggest possible coating by VOC, inorganics and PAH onto PM particles, with higher specific surface areas at urban than at rural sites.

Table 3 - Inorganic and ionic compositions of collected PM. Inorganic compounds were detected in particle matter (PM) by inductively coupled plasma–atomic emission spectrometry. Ionic compounds were quantified with ionic chromatography [*Kouassi et al.*, 2010]

Ions;(µg;mg^{-1};PM)	Rural	Urban	Industrial
Cl−	6.17	3.35	31.37
NH4+	6.16	2.65	4.30
NO3	12.23	19.62	20.04
SO4	19.62	13.36	17.24
Metals;(µg;mg^{-1};PM)			
Al	78.79	60.59	56.01
Ca	31.57	35.54	36.31
Cr	0.11	1.44	0.86
Cu	0.14	0.24	0.11
Fe	46.32	54.25	34.30
K	16.20	9.69	17.45
Mg	11.55	7.19	10.75
Mn	2.79	0.68	0.57
Na	9.59	10.06	24.64
Ni	0.07	0.05	0.07
Pb	0.06	0.49	0.40
Sb	<LOQ	0.04	0.01
Sn	0.02	0.05	0.01
Ti	5.18	3.93	3.29
V	0.10	0.10	0.10
Zn	0.18	0.39	0.50

Table 4 - BC/OC ratios in Bamako and Dakar for ultra fine (UF), fine (F) and coarse (C) particles. BK1 and BK2 are for Bamako, associated or not to dust events, DK is for Dakar

	BK1	BK2	DK
	BC/OC		
UF	0.17	0.18	**0.49**
F	0.09	**0.35**	**0.54**
C	0.11	**0.34**	**0.36**
Bulk	0.10	0.31	**0.45**

Focus of source emission impacts on aerosol and gas characterization is now shown for Bamako and Dakar in the frame of POLCA programme. Figure 7 shows higher NO_2 and SO_2 concentrations in Dakar than in Bamako, whereas NH_3 (not shown here) has observed concentrations higher in Bamako than in Dakar [*Yoboué et al.,* 2012]. This pattern may be explained by the relative importance of different combustion sources prevailing in both capitals. Indeed, in Dakar, traffic seems to be the predominant combustion source mainly with diesel fuels. In Bamako, there is a mix of sources with more incomplete combustion than in Dakar with an important park of two-wheel vehicles using gasoline-oil mixed fuels, domestic fires and waste burning. This difference may also be highlighted with particulate measurements (Figure 7). Black carbon to organic carbon ratios (Table 4) are much higher in Dakar than in Bamako : such lower

ratios in Bamako suggest higher organic carbon content, in agreement with more incomplete combustions in Bamako. This feature is confirmed with PAH concentrations being much higher in Dakar (diesel sources) than in Bamako. Finally, water-soluble organic carbon (WSOC) concentrations are found much higher in Bamako than in Dakar.

2.2.4 Atmospheric Pollution and Health

Exposure/Epidemiological studies

In West Africa, air pollution exposure is expected to be responsible for most respiratory diseases [*WHO*, 2006], with lung affections and respiratory irritations respectively due to CO, NO_X, PM_1, benzene and toluene. However, potential exposures vary with populations, depending on their daily activities. As shown by *Linden et al*. [2012] for Ouagadougou, careful consideration of both location and time resolution of measurements is required for accurate exposure assessments. There is an urgent need for exposure scenarios, through displaying for example individual data monitoring measurements. Heavy exposures are expected for workers near traffic sources and for females and young children near domestic fuel combustion. In Nigeria, occupational exposure intensive measurements are available, linking air pollution at workplaces to adverse implications for the health of workers [*Baumbach et al*., 1995; *Adejumo et al.,* 1994; *Ogunsola et al*., 1994; *Oyedele et al*., 1995].

During the POLCA programme, groups of people have been identified for health studies linked to pollution. In Bamako for example, *O. Koita* [personal communication] has shown that the selected group is mainly affected by traffic and domestic fuel sources prevailing at the site (only 15% of this group is exposed to waste burning, 18% to cigarette smoke, 7% to industries). In Dakar, the concept of bio-indicators for air quality was examined: several cytokines such as Interleukin 1 β (IL-1β), interleukin 5 (IL-5) interleukin 8 (IL-8) and proteins (Clara Cells, CC16) have been evaluated for subjects exposed to air pollutants prevailing at the POLCA traffic site [*Gueye*, PhD Thesis 2008]. About 67 subjects are enrolled, including 33 merchants (traders) with collection of their blood for pro-inflammatory cytokine study. This study has shown a glutathione status alteration implying a defense system alteration due to atmospheric pollution [*Gueye,* PhD Thesis 2008 ; *Gning,* PhD 2011].

Finally, only few results are presently available for epidemiological studies linked to air pollution in West Africa using long-term observations. An example is given in Table 4 with a follow up realized in Kossodo (Burkina Faso) and carried out by the Laboratoire de Physique et de Chimie de l'Environnmement of the University of Ouagadougou over 3 years (1999, 2001 and 2002), detected an increase of upper and lower respiratory track infections (Table 5). Such results are encouraging and need to be largely extended to other cities in West Africa and linked to atmospheric pollution. Indeed, up to now, all estimates for asthma, morbidity or mortality in West Africa are based on dose-response functions only established for northern developed countries. As shown earlier, and due to African pollution specificities, such functions urgently need to be specifically determined for West Africa, using in parallel both long-term pollution measurements and health registrations: this is quite a important point to be stressed.

Table 5 - Prevalence of respiratory infections in the Health District of Kossodo (Ouagadougou). Source: LPCE/UO. The number of cases of respiratory infections was 3 times greater in 2002 than observed in 1999 (35 405 cases versus 11 466)

Kossodo Health District/Year	1999	2001	2002
Upper respiratory track infections	4 819	9 428	15 072
Lower respiratory track infections	6 647	11 847	20 331
Total	11 466	21 275	35 403

Several studies in Nigeria have dealt with the consequences, such as mental development in children, of high blood lead concentrations [*Ogunsola et al*., 1994]. However, recent lead measurements in Lagos and Abuja show that atmospheric lead concentration are now between 1 and 2 orders of magnitude lower than previous measurements, following the gasoline lead phase out in 2006 [*Ezeh et al*., 2010].

Aerosol biological reactivities of human bronchial epithelial cells from in vitro studies

The fine and ultrafine aerosol fractions are now recognized to induce biological effects due to their capacity to reach the distal lung, together with specific compositions including transition metals and organic compounds [*Happo et al*., 2008; *Seagrave et al*., 2006; *Huang et al*., 2003]. Particle toxicity results from their ability to trigger intracellular production of reactive oxygen species in epithelial cells and macrophages, the first cells encountered by particles in the respiratory tract. This oxidative stress activates signalling pathways, leading to the release of pro-inflammatory biomarkers (IL-8; IL-6, GM-CSF….) [*Mitschik et al*., 2008]. Such comprehensive processes of particle health effects have been extensively studied in developed countries, leading to specific regulations [*Ramgolam et al*., 2009]. Only a few studies have been conducted in developing countries, especially in West Africa [*Val et al.,* 2012; *Kouassi et al*., 2010; *Dieme et al*., 2012].

An example of such results was part of POLCA, with the aim of characterizing atmospheric particulate pollution and to determine induced toxicity potential of particles in epithelium cells (16HBE) according to their sizes (coarse, fine and ultrafine particles). Three main situations were scrutinized: Bamako during (BK1) and after (BK2) a typical dust event and Dakar (DK) (aerosol chemical specificities given in Figure 9 and in Table 4). PM biological reactivities were characterized from measurements of the expression of a panel of biomarkers.

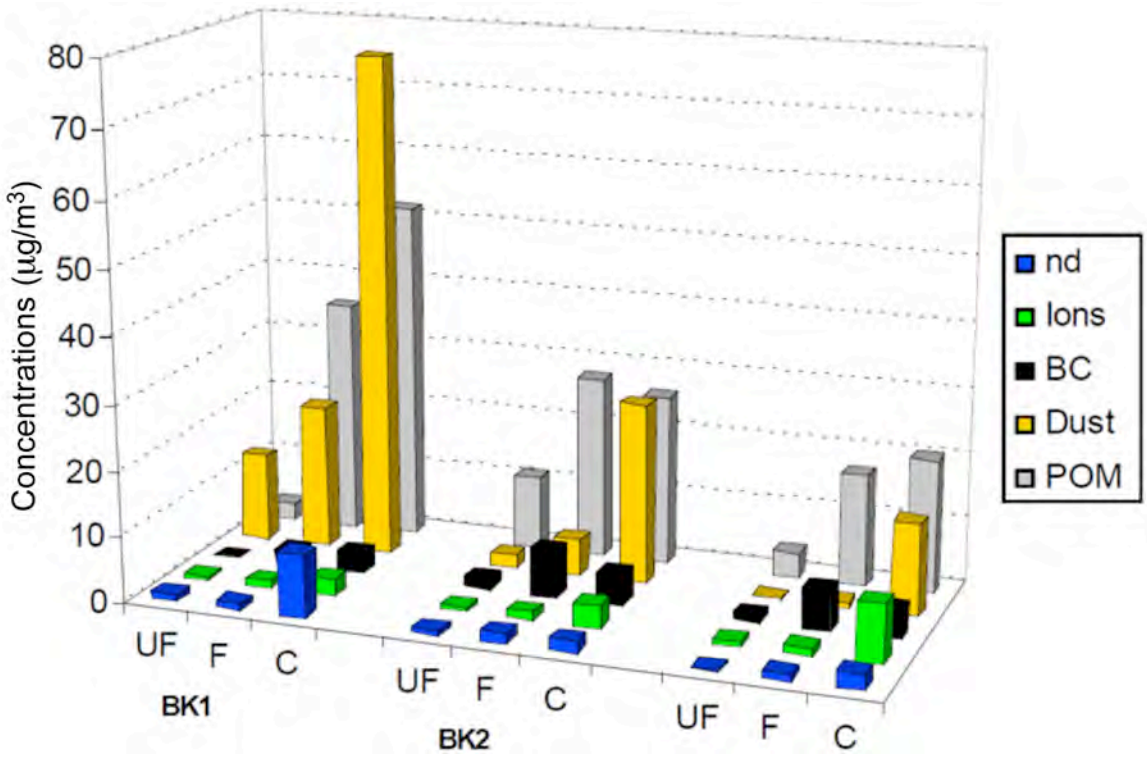

Figure 9 - Size speciated chemical aerosol composition in Dakar and in Bamako during (BK1) and after (BK2) a dust event [*Val et al.*, 2013]
(Also shown in Chapter 7 as Figure 17)

As shown in Figure 10, ultra fine and fine PM induce higher biological effects (GM-CSF cytokines) as compared to coarse particles, whatever the sites. Bamako aerosol can be distinguished by an impressive biological reactivity associated to local sources, since less reactive when diluted by external input such as dusts. This reactivity is stronger in Bamako than in Dakar. Considering aerosol chemical speciation in the two cities, aerosol biological responses seem to be closely related to organic compound contents [*Val et al*., 2013].

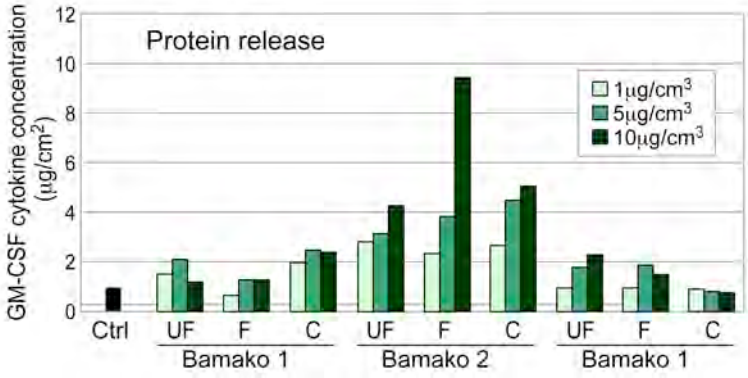

Figure 10 - GM-CSF cytokine concentrations as a function of aerosol size classes for the reference sample (Ctr), and Bamako (BK1 and BK2) and Dakar samples. Different doses (1-5 et 10 g/cm^2) have been tested [*Val et al.*, 2012]

Other in-vitro studies dealing with air pollution effects possibly involved in lung toxicity in epithelium cells have been recently conducted in Abidjan for urban, industrial and rural sites on A549 cells and in Dakar for traffic and rural sites on BEAS-2B cells [*Kouassi et al*., 2010; *Dieme et al.,* 2012]. Both studies have confirmed that both anthropogenic transition metals (Fe, Al, Al, Pb, Mn, Zn) and organic compounds (PAH) are involved in a time and dose dependent secretion of inflammatory mediators and ensuing oxidative damages. Note that high levels of reactive oxygen species have been measured at Abidjan sites, not in Dakar.

Finally, in the frame of POLCA (O. Koita, pers. com.), atmospheric bacterial population has been studied in Bamako for a six-month period (57% Bacillus against 34% Cocci). The presence of these bacteria is an important factor for the prevalence and incidence of respiratory and cardiovascular diseases in Bamako. Note that dust-carrying bacteria are generally associated with meningitis epidemics in the Sahelian belt. Indeed, meningitis outbreaks typically occur a few days after dust events: mechanistically, combination of lung system damage due to atmospheric pollution, dust vectors and bacteria could jointly act in developing epidemics. Many studies are still on going to decipher such occurrences.

2.2.5 Conclusion

As mentioned in the present chapter, air pollution and health is a real joint problem in West African capitals. The main sources of emissions are linked to traffic, domestic fuel combustion and waste burning. The importance of industrial activities has been pointed out in South Africa only. The main emissions sources are controlled with only a few regulations (e.g. fuel desulfurization in Senegal, lead decrease in Nigeria). The situation is expected to rapidly deteriorate in the absence of any further actions. Moreover, political conditions in West Africa will result again and again in population migration and urbanization. It is now urgent for research studies to focus on integrated projects combining emissions, air quality, health studies (epidemiological monitoring, toxicology, hospitalizations), acid deposition and impacts on soil and water resources together with local and regional climate change studies to provide possible emission mitigation options resulting in positive impacts.

2.3 SOUTH AFRICA

2.3.1 Johannesburg conurbation and the Vaal triangle area: characteristics, geography, population, meteorology

South Africa is experiencing large-scale social and economic changes, coping with both developed and developing world problems that affect its regional environment. For example, Greater Johannesburg is faced with great differentiation in terms of service deliveries between different areas.

Significant impacts are due to population growth, population migration, industrial development, water shortages and change in agriculture practices. Increasing demand for domestic food sources and exports in southern Africa lead to growing needs for agricultural land as well as intensification of industrialization. South Africa is under severe anthropogenic pressure with increasing emissions due to rapid growth in energy needs, particularly in the Vaal Triangle close to Johannesburg (Figure 11).

Figure 11 - South Africa map with a focus on Johannesburg conurbation and the Vaal triangle area

The Johannesburg conurbation and the Vaal triangle area is a large industrial area with 18 million inhabitants, 13 million of which live in Johannesburg conurbation. Four “industrialized” cities are included in the conurbation, including huge industries such as Sasol, Natref, Dow, Arcelor Mittal, Samancor, Emsa (electro chemical) and many coal mines such as Sigma, Coalbrooke, Eskom mines and Wonderwater. In this 3600km^2 area, there are six big townships with strong emissions also linked to domestic combustion of firewood, coal and other biofuels.

Johannesburg is, by population, the largest city in South Africa, ranking amongst the 50 largest metropolitan areas in the world. According to the 2007 Community Survey, the population

of the city of Johannesburg was 3.9 million and the population of the Greater Johannesburg Metropolitan Area was 7.2 million. A broader definition of the Johannesburg metropolitan area, including Ekurhuleni, the West Rand, Soweto and Lenasia, has a population of 10.3 million. Johannesburg is the provincial capital of Gauteng, with the largest economy of any metropolitan areas in Sub Saharan Africa. Johannesburg owes its location to the presence of gold, comprising 40% of Gauteng's GDP. Though Johannesburg is not one of South Africa's three capital cities, it is the location of the Constitutional Court.

Johannesburg is located in the eastern plateau area of South Africa (Highveld) at an elevation of 1753m. Johannesburg features a subtropical highland climate. Temperatures in Johannesburg are usually fairly mild, with average maximum daytime temperatures in January of 25.6 °C, dropping to an average maximum of 16°C in June. Regular cold fronts pass over in winter bringing very cold southerly winds but usually clear skies. The annual average rainfall is 713mm, mostly concentrated in the summer months.

In Johannesburg, 80% of households have access to running water, and 80% use electricity as the main source of energy. Thirty percent of Johannesburg residents live in informal dwellings with less than adequate accommodations. Some 15% of households are still burning fossil fuels for their cooking and heating requirements. With an inadequate public transport system, traffic volumes are rapidly growing with a 25% increase between 1996 and 2000 for example.

The Vaal triangle area (also called dirty triangle) is notorious for its poor air quality, with industrial and mining emissions combining with large-scale domestic coal burning emissions in informal settlements and townships to produce a formidable air quality problem in the region. Moreover this outdoor pollution is often related to problems of indoor air quality. This region, the industrial heartland of South Africa, is located roughly 40km south-southwest of Johannesburg. Major towns within the region are Vereeniging, Vanderbijlpark, Sasolburg and Meyerton. However, the majority of the population lives in the townships of Boipatong, Bophelong, Evaton, Orange Farm, Sebokeng, Sharpville and Zamdela. The annual average rainfall in the Vaal triangle area is between 500 and 700mm, with cold winters.

Note that the Vaal triangle area has been recognized by the Air Quality Act from National Environment Management in 2010 as one "priority area", where hot spot of pollution needs to be managed. The other one is the Highveld area, not detailed here.

Finally, another important combustion source of pollution in South Africa to note that is well established in publications is biomass burning during the dry season (from August to November). The different pollutants from a variety of emission sources (e.g. fossil fuel, biomass burning and natural sources) are mixed together and transported, sometimes as far as Amsterdam Island in the Indian Ocean or over the Atlantic Ocean [*Piketh et al*., 1996; *Tyson et al*., 2002].

2.3.2 Emission sources of air pollutants

As depicted in Figure 2, South African emissions are mainly linked to domestic, industrial and power plant activity sectors.

The South African power generation sector is heavily dependent on coal (*Scorgie et al.,* 2004). Based on 2001 Eskom data, coal is responsible for about 90% of Eskom's power generation, other sources including nuclear (5.7%) and hydro (1.1%). These power stations are designed to use low-grade coal associated with higher emissions than high-grade coal available for export. South African coal has a relatively lower sulphur content than coal elsewhere. Due to high ash content in combusted coal, particulate emissions and ash production are higher than would be the case with low-ash coals. As a consequence, Eskom's pollution control policy has been concerned primarily with the control of particulate emissions using electro-static precipitators (ESPs) to remove bulk particulate emissions from flue gases [*Scorgie et al*., 2004].

The industrial sector (non-power generation) includes coal, anthracite, coke, heavy fuel oil, gas, diesel and paraffin. This sector has a large variety of industrial and waste disposal processes

ranging from waste incineration (including medical, toxic, animal and other waste types), petroleum refining, inorganic and organic chemical processing (e.g. carbon black processes), mineral product industries (e.g. brickworks), cement manufacturing, coal conversion (e.g. Sasol oil from coal plants at Sasolburg and Secunda), glass manufacture and metallurgical industries. Note that the Sasol process of coal gasification in the production of synthetic liquid fuels from gas is a relatively unique process, whose emissions are expected to be very high.

Fuel other than electricity, used for household for cooking, lighting and/or space heating purposes primarily include: coal, wood, LPG, paraffin and candles. Waste material, including old shoes and tires, is also burned by households unable to afford other fuel carriers. In rural areas, some households burn animal dung to meet their energy, heating and cooking needs. Continued use of coal and wood by a large section of the population in South Africa is a cause of concern with regard to air pollution and health risk potentials. These fuels continue to be used for primarily two reasons: (i) rapid urbanization and growth of informal settlements (ii) Coal is readily available and inexpensively in Gauteng and Mpumalanga. Given the availability of coal and relatively low temperatures experienced during winter months, coal consumption is high in these regions. Wood is burned in place of coal in coastal regions including Cape Town and Ethekwini. Household fuel burning tends to peak during early morning and evening at which times the atmosphere is characterised by limited mixing depths and stable conditions favourable to stagnation of pollution [*Engelbrecht et al*., 2000]. Due to emissions in a confined space (indoor pollution) and due to winter maximum uses being associated with minimum atmospheric dispersion, domestic fuel burning emissions have a greater potential impact on air quality as compared to equivalent emissions from, for example, industrial sources.

Focus on coal use in the different sectors shows that 2% is for industry, 2% for the domestic sector (producing 25% of the emissions), 43% for electricity, 30% for export and 21% for coal liquefaction (synthetic fuel production).

Diesel and petrol (including leaded and unleaded petrol) represent the main fuels used by vehicles. Note that reductions in ambient lead concentrations (consequently in blood lead levels of children) have been observed. This could be due to the introduction of unleaded fuels, with leaded fuel phased out in January 2006. Other mobile source uses a variety of fuels, e.g. aircraft mainly use jet fuel, ship engines typically use marine diesel oil and non-electrified trains use primarily diesel and coal.

Local inventories

Since 1990, a national effort has taken place to develop local emission inventories. Certain metropolitan areas have emission inventories for common pollutants such as SO_2, NO_X, CO, CO_2, hydrocarbons and particulates (TSP or PM_{10}).

In Johannesburg, combustion-related emission sources have been identified and efforts have been made to quantify some of these sources, but no exhaustive emission inventory currently exists. For example, in the National emission inventory database (1994), emission inventories for coal power plants in the area are available. Domestic coal combustion emission inventories have been developed from different assessments and measurements [*Scorgie et al*., 2003a], giving a mean total of 3878 tons/yr for PM_{10}. Emissions data for the transport sector only included diesel fuel with 1625 tons/yr of PM_{10}. Data is also available for spatial emission based on road density, industry and power plant location maps.

In 1995 an emissions inventory for municipal areas in the Vaal Triangle was established [*Van Nierop*, 1995]. The emissions inventory includes industrial, domestic, vehicular and power plant sectors. In parallel, a few experiments took place on domestic fuel emissions for characterization. During the winter of 1997, *Engelbrecht et al*. [1998; 2002] and *Terblanche* [1995b; 1998] found 62% of PM25 came from domestic burning, 14% from biomass burning, 11% from dust and only a minor contribution from power plants and vehicles. More interestingly, two types of coal were studied: D grade coal (low quality usually used) and low smoke coal (as an alternative source of energy). The results showed a 25% reduction of particulate emissions when

low smoke coal was used instead of D grade coal.

To conclude, it is important to note that the existing inventories were not followed up on. Also, these emission inventories are sometimes incomplete and need to be extended to other pollutants and to include other sources (particularly industrial and institutional fuel burning, industrial processes, household fuel combustion and vehicle emissions).

Note that much work is going on to compensate for the current lack of information. For example, a PhD study was just completed on emission inventories linked to domestic burning in South African townships [*Naidoo*, 2012]. Other measurements on emission characterization are being carried out and results are expected soon (power plant, coal liquefaction).

Regional emissions

As mentioned in the introduction, a regional emission inventory has recently been developed for Africa, including some African specificities [*Assamoi*, PhD 2011; *Liousse et al.*, 2012]. For example, a particular effort has been made on spatial emission characterization of power plant and industrial activities. Figure 12 displays industrial and power plant OC emissions in 2005, either spatially distributed with CIESIN (2005) population density (left) or with emission locations after Flemming data (right). Other important work has been conducted on emission projections for 2030. Indeed, in the socio-economical POLE model used to obtain fuel consumption in 2030 for different scenarios, South Africa was considered as a developing country. A new scenario (2030ccc*) has been constructed by applying projection factors typical for semi-developed countries and by applying improved emission factors for domestic fuels, both actions thus reducing OC emissions by 62%, as compared to the previous “best” scenario.

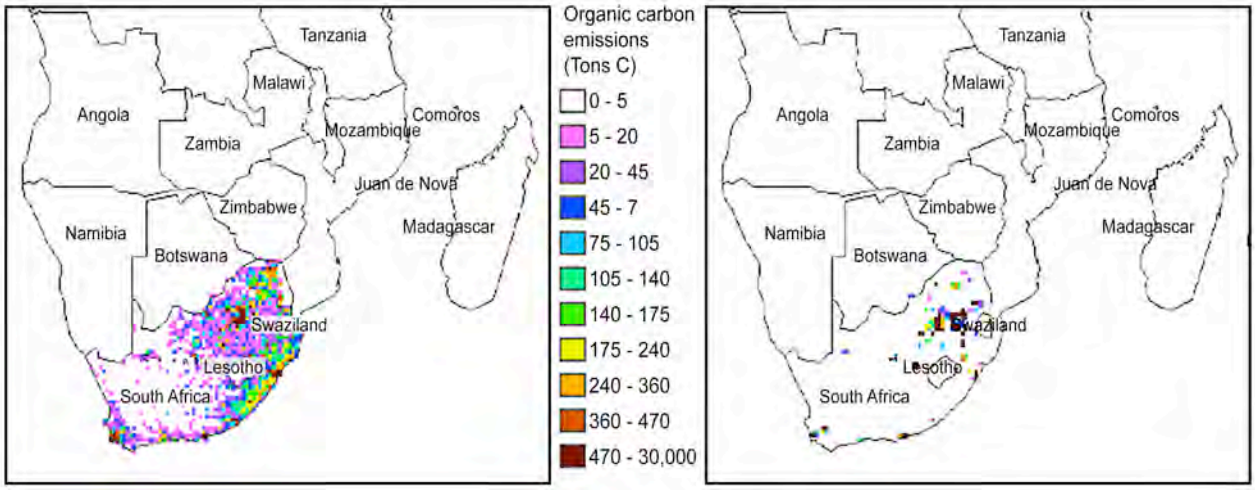

Figure 12 - Organic carbon emissions (in tons C) in South Africa in 2005 with the new regional African inventory [*Liousse et al.*, 2012]: left with CIESIN spatialization (0.25° x 0.25°), right with the data of *Flemming et al.* (Fourié, pers. com)

Finally, Figure 13 presents OC emissions obtained with the regional inventory for 2005 and for the three 2030 scenarios. Data given by global ACCMIP/RCP inventories are added for comparison [*Lamarque et al.*, 2010]. OC is higher in the regional inventory than in ACCMIP inventory for 2005 (BC, not shown here, is quite comparable). OC projections differ, especially for the BAU scenario. Differences are smaller for the “best” scenarios. Such a study underlines the need to construct local and regional inventories for the past, the present and the future.

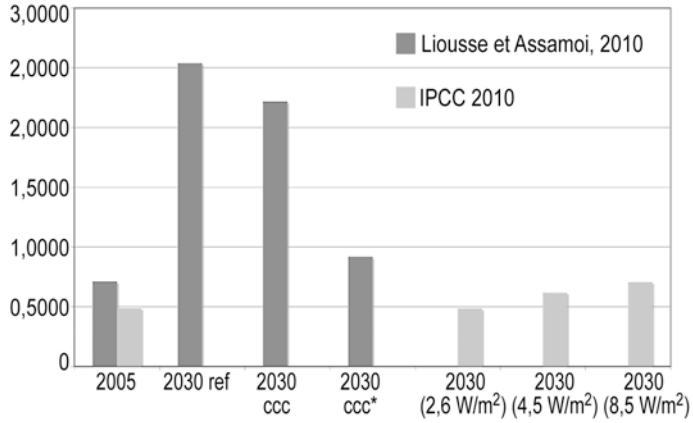

Figure 13 - OC anthropogenic emission annual budget for 2005 and different 2030 scenarios. In black, *Liousse et al.* [2012], in grey; ACCMIP/RCP inventories [*Lamarque et al.*, 2010]

2.3.3 Air Pollutants

As displayed by satellite retrievals (e.g. Sciamachy), there are NO_2 pollution hot spots in South Africa Highveld. Indeed, tropospheric NO_2 column densities of this area are comparable to those observed in central and northern Europe, eastern North-America and south-east Asia. In addition, existing ground measurements in South Africa confirm such a picture, showing the huge impacts of domestic coal burning, industrial and power plant sources and traffic on air quality. A recent example, including a box model with detailed gas-phase chemistry developed along with new measurements of trace gases at several locations in and around Johannesburg conurbation investigated the impact of industrial activities on tropospheric photochemistry and showed that NO_2 concentrations in the megacity have diurnal peaks during early morning and late afternoon, which coincide with peak traffic hours and domestic combustion [*Josipovic et al.*, 2010; *Lourens et al.,* 2012]. During these periods, NO_2 concentrations in the megacity were even higher than in the Highveld hotspot. These diurnal NO_2 peaks in the megacity have generally been overlooked by satellite observations, since satellites have fixed local overpass times that do not coincide with such peak periods. This implies that the importance of NO_2 in the megacity has been underestimated (Figure 14).

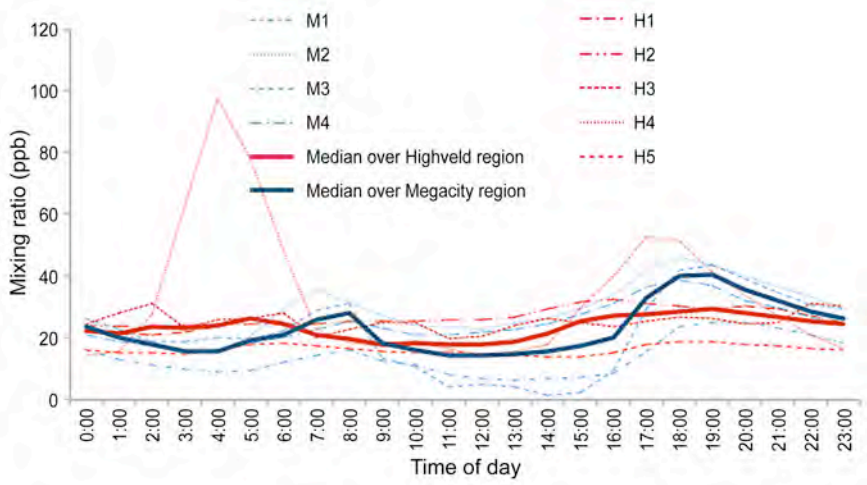

Figure 14 - Median diurnal NO_2 mixing ratios at all ground-based stations for the period March to May 2009 together with average values over the regions. M denotes a Megacity station and H a Highveld hotspot station [*Lourens et al.,* 2012]

Such results highlight numerous previous studies conducted since the 1990s in South African that measured elevated concentrations of particulate matter, sulphur dioxide (SO_2), nitrogen oxides (NO_X), carbon monoxide (CO), carbon dioxide (CO_2), ozone (O_3), volatile organic compounds (VOCs) and semi-volatile organic compounds (SVOCs), methane (CH_4), hydrogen sulphide (H_2S) and various trace elements at urban, industrial and coal burning site. These air quality studies are fragmented and unsystematic: though a number of studies have been

completed, their results have not been yet integrated and made easily accessible (South Africa country report, 2005). Despite this limitation in information, South's Africa air quality is considered as being relatively good on the whole, though with a number of air pollution "hot spots" where severe problems are encountered. As already mentioned, such a concern is tackled by the National Environmental Management agency with the AQA program providing measures for air quality improvement (ambient air quality standard, control of emissions ...). A few examples of studies focused on Johannesburg conurbation and on the Vaal triangle priority area follows.

Johannesburg

Scorgie et al. [2003a], gather ambient air quality monitoring data in Johannesburg from field campaigns at occasional sites and at continuously operating monitoring stations. Note that many monitoring sites exist in the Johannesburg conurbation (Mintek, New town, Lapa, Kemton Park, City Deep, City Hall, Soweto), some of them since the 1980s (*Piketh* 2007). High heterogeneity in environmental air quality in this conurbation is noticed with poorest situations in the northern part of the metropolitan area, Alexandra, the mining and industrial belt and the Klip River area, Roodeport vicinity and the area around Soweto, Orange Farm and Poortjie (http://ceroi.net/reports/johannesburg/csoe/navpoll.htm).

Ambient air quality monitoring data at the Soweto site taken as part of the Soweto Air Monitoring Project (Project SAM, Department of Environmental Affairs and Tourism (DEAT), Soweto Health Department, Soweto Branch of the National Association of Clean Air (NACA)) examined ambient particulate concentration and their health risks from domestic coal burning in Soweto residential areas between 1991 and 1999. Seasonal trends show increasing particulate concentrations during winter months, associated with an increase in coal burning for space heating and unfavourable atmospheric dispersion potentials. Winter time concentrations are in exceedance of the US-EPA standard by a factor as high as 4.8. This is confirmed by PM2.5 source apportionment also conducted in Soweto in 1996-1997, with 57% to ~75% of PM2.5 concentrations due to domestic coal burning emissions [*Annegarn and Grant*, 1999]. A downward trend in airborne particulate concentrations was observed during the studied period (1992-1999), confirming results in *Annegarn and Sithole* [1998]. The downward trend is mainly explained by a decrease in coal use in favour of less polluting fuels, such as electricity, in the immediate vicinity of the monitoring site. Morning and evening peaks in airborne particulate concentrations have been associated with cooking and heating activities and peak hour commuter traffic. Distinct increases in sulphur dioxide, oxides of nitrogen and volatile organic carbon concentrations are also apparent during winter months, with temporal trends also indicative of domestic fuel burning periods. NO_2 levels measured during this period exceeded WHO guidelines, whereas O_3 concentrations exceeded both the South African and WHO guidelines [*WHO,* 2000].

A regional study conducted by IVL Swedish Environmental Research Institute and CSIR and appointed by the City of Johannesburg, allowed SO_2 and NO_2 air pollution level mapping across Johannesburg, from a passive diffusive sampling campaign at almost 300 sites in July 1999. Elevated sulphur dioxide levels are evident across the industrial areas with significant contribution of domestic coal burning in townships and informal settlements during winter months. The zones of NO_2 maxima closely coincide with the areas of high vehicle activity.

Vaal triangle

The Vaal Triangle Air Pollution Health Study (VAPS) was initiated in 1990 to study the potential impacts of air pollution on human health. Several other sampling campaigns have also occurred in this region [*Burger,* 1994; *van Nierop*, 1995; *Engelbrecht et al.*, 1998; *Terblanche*, 1998]. Main findings of VAPS and related studies are as follows:

- Particulate matter appears as the pollutant of greatest concern in the region with annual average levels exceeding international health standards by at least a factor of 2.5.
- High ambient particulate concentrations are found to coincide with low ambient temperatures and low rainfall. As shown earlier for Soweto, the highest concentrations were predicted to occur in or near coal burning residential areas (townships and informal settlements), with maximum values during winter months (2 to 4-fold increase) and a

distinct diurnal trend. Within coal burning townships, levels of TSP were 4-6 times worse than regional averages.

- Sulphur dioxide levels were found to exceed air quality guidelines less than 5% of the time in the Sasolburg industrial area, with SO_2 levels in residential areas being below air quality guidelines. Concentration peaks observed during the early morning are associated with emissions from tall stacks in the region.
- No exceedances of ambient air quality guidelines for NO_X were observed to occur during VAPS. An increase in annual averages occurred during the 1990 to 1993 period, with increasing NO_X concentrations at the Sasolburg Industrial site.

A similar pattern was observed 10 years later.

More recent data are now available in the Vaal Triangle area, in connection with Sasol and Eskom monitoring sites. Also new measurements are being developed in the frame of the GRDI ARSAIO programme, a collaboration between North-West University (South Africa) and Laboratoire d'Aérologie (France). Efforts will focus on aerosol chemical speciation and biological impacts on human health in the Vaal triangle. An example of expected results is shown in Figure 15. Aerosol chemical speciation was obtained during the 2006 winter months for PM2.5 and PM10 particles at the Amersfoot rural site, which has an industrial influence (IDAF/DEBITS network, http://idaf.sedoo.fr). The data show high relative contribution of organic carbon particles followed by sulphate and black carbon. Corresponding results obtained from a modelling study are added for comparison.

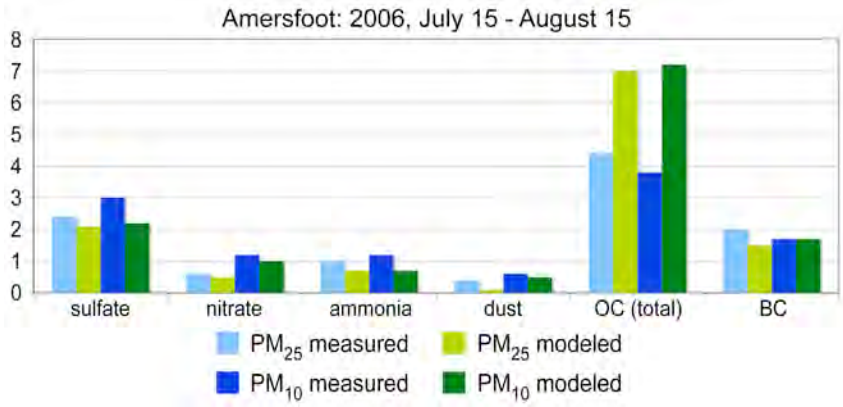

Figure 15 - Modelled/Measured size speciated aerosol chemistry at the Amersfoort site for the 2005-2007 period [*K. Martin*, PhD 2008; *Guillaume et al.*, 2007]

Finally, as previously indicated, this review shows that air quality is a national concern and air quality national legislation is becoming more and more stringent. The National Environmental Management (Air Quality Act, Act No. 39, 2004, 2010) is now replacing the Atmospheric Pollution Prevention Act (APPA), Act 45 of 1965. In 2010, national ambient air quality standards have been defined for SO_2, NO_2, PM10, 0_3, benzene, lead, CO (http://www.naca.org.za) with addition of PM2.5 in September 2012. Note that a recent workshop in 2011 "Changing Chemistry in a Changing Climate: Human and Natural Impacts over southern Africa – C4-SAR" organized by Burrows, Piketh and Thompson also gathered many studies here mentioned.

2.3.4 Health

Many epidemiological studies related to indoor and outdoor air pollution have been conducted in South Africa since 20 years, using different indicators (mortality rate, risk assessment studies, health index…). A few of them (this is not an exhaustive list), is now presented to underline the evidence of air pollution impacts.

General studies in South Africa

The mortality rate of acute respiratory infections (ARI) in South Africa is reported to be 270 times greater than for children in Western Europe [*Terblanche et al.*, 1993]. Recent

epidemiological data have indicated that ARIs are one of the leading causes of death for black South African children. More recently, Barnes et al. (2009) reviewed evidence of the association between household energy, indoor pollution and child ALRI (Acute Lower Respiratory Infections) in South Africa, showing high relative mortality risks from different studies in South Africa.

In *Scorgie et al.* [2003, 2004] the focus is on epidemiological studies in many South African cities (Cape Point, Johannesburg, Vaal triangle, Highveld). Using northern developed country dose-response functions (no values exist for Africa) of 1.20 x 10^{-5} for PM10 daily exposure and respiratory hospital admissions and 2.01 x 10^{-6} for SO_2 (values for PM10 are more harmful than for SO2) the studies showed:

Table 6 - Numbers of health endpoint in Johannesburg and in the Vaal triangle [*Scorgie et al.,* 2004]

HEALTH ENDPOINT	CITY OF JOBURG & EKURHULENI	VAAL TRIANGLE
Respiratory hospital admissions (due to PM10, SO_2 and NO_2 exposures)	34,021.1	9,440.0
Cardiovascular hospital admissions (due to PM10 exposures)	262.2	71.0
Premature mortality (due to PM10 and SO_2 exposures)	71.5	19.9
Chronic bronchitis (due to PM10 exposures)	38,550.4	9,457.5
Restricted activity days (RAD, due to PM10 exposures)	238,326.3	62,546.5
Minor restricted activity days (MRAD, due to SO_2 exposures)	12,396,320.4	6,128,743.4
Leukemia cases (due to 1.3 butadiene and benzene exposures)	67.4	9.1
Nasal carcinoma cases (due to formaldehyde exposures)	1.5	0.2
Number of children exposed to Pb> 2µg/m^3 & hence to potential for IQ point reductions	5,285.8	0

- Combustion sources are responsible for 0.64% of hospital.
- Domestic sources account for 70% of all respiratory related hospital admissions (RPHA) and 75% of all premature mortalities
- Traffic is responsible of 12% of all RPHA and 6% of all premature mortalities.
- Electricity generation accounts for 6% of all RPHA and 5% of all premature mortalities

Due to paramount importance of coal burning sources in townships and informal settlements, health indices (HI) were calculated by *Terblanche* [1996] to demonstrate the magnitude of household exposures and associated risks. The health index (HI) represents the ratio of the sum of pollutant doses over acceptable doses, summed for all pollutants with the same health effects. Generally, HI <1 stands for negligible risks, HIs of 1 to 10 for low risks, and indices above 10 reflect unacceptable risks. Health indices were calculated as follows:

- A HI of 27 was calculated for winter-time exposures to outdoor pollutant concentrations within coal-burning areas, as compared to a HI of 5 in electrified smoke-free zones in the Vaal Triangle.
- Winter indoor particulate exposures in coal-burning households were associated with an HI of 34, with the HI for summer exposures being estimated to be 14.
- The HIs for SO_2, NO_2 and CO exposures were calculated to be below 10 in all cases, indicating a low to moderate risk for current exposures to these pollutants.

Terblanche [1996] concluded that, based on calculated HIs, a reduction in particulate concentrations near 100% would be required to meet with WHO health standards. More recently, *Norman et al.* (2007a and b) have estimated the burden of diseases attributable to urban areas (including Johannesburg and Vaal triangle area) and to indoor air pollution from solid fuel burning respectively by using the Comparative risk assessment methodology developed by WHO. PM2.5 and PM10 were used as exposure metrics. They showed that outdoor air pollution was estimated to cause 3.7% of national mortality in 2000 from cardiopulmonary diseases, 5.1% from respiratory track cancers in adults aged 30 and older and 1.1% of mortality from ARI in children under 5 years old. Regarding indoor pollution, it is estimated 20% of South African households are exposed to solid fuel burning with marked variations between population groups. Such exposure was estimated to have caused 0.5% of all deaths in South Africa in 2000.

Vaal triangle

Exposure studies have been conducted in Sebokeng (Vaal triangle) in 1991 by CSIR and the Medical Research Centre. Forty-five children were monitored between the ages of 8 to 12 years old. The study revealed extremely high levels of exposure to total suspended particulates, in exceedance of the US-EPA air quality standard [*Terblanche et al.*, 1992]. Exposures to indoor CO concentrations were found to be up to 180% higher in coal-burning households as compared to wood-burning ones within the Vaal Triangle [*Terblanche et al.*, 1995a].

More recently, the VAP programme (1990-1993) using six monitoring sites studied the health status of adults who had spent their developing years in a polluted area in the Vaal triangle and for whom their respiratory health status was known [*Oosthuizen*, 2004 Master; *Oosthuizen et al*., 2008]. Approximate 14,000 children (10 years old) were involved in this programme. Both outdoor and indoor measurements with personal monitoring on 30 children (teenagers = 15 male/15 female) were performed. The results indicate that the upper prevalence of respiratory health effects is 65% for 10 year olds children compared to 72% for young adults (i.e. 20 years old). The risk was the same whether or not the young adults has stayed or left from the Vaal triangle region. Consequently, exposure to pollution only cannot explain such an increase. This could be due to external risk factors such as pollution perception, allergies, smoking, family history, weight, etc.

Johannesburg conurbation

An epidemiological study took place in Soweto lead by the Medical Research Council. The programme, Birth to Ten, focused on neonates to ten years old children (3275 children total). This program now called Birth to twenty is focused on growth and pubertal development in relation to the many risk factors facing young people. Questionnaires on a monthly basis were distributed over a period of one year. Information was obtained on housing factors, fuel usage and health status of children involved. It was reported that 54% of the children in the sub study experienced a high frequency of colds and chest illness since birth.

To conclude, epidemiological studies show interesting results on domestic coal burning and urban areas source impacts. Such work still needs to be linked to target aerosol species (such as organic particles, PAH, etc.). Also parallel long-term atmospheric and epidemiological survey should be organized to produce dose response functions typical to South Africa. Finally, integrated studies from processes (biological impacts) to epidemiological studies can provide a better understanding of links between pollution and health.

2.4 NORTHERN AFRICA

2.4.1 The Greater Cairo Area; City characteristics, geography, population, meteorology

Egypt is located in Northern Africa, bordering the Mediterranean Sea, between Libya and the Gaza Strip, and the Red Sea north of Sudan, and includes the Asian Sinai Peninsula. It has a population of 73.5 million with approximately 43% of the population living in urban areas [*WMO*, 2008]. In 2009 it was estimated that Egypt's population increased at a rate of 1.64 % per year and urban population increased at an annual rate of 1.8% (http://world.bymap.org/

PopulationGrowthRates.html). According to *WHO* [2006], the life expectance at birth in 2004 was 68 yr (66 yr for men and 70 yr for women). The percentage of population older than 60 years old increased from 6.4% in 1994 to 7.1% in 2004 [*WHO*, 2006]. Thus, in 2009 the life expectance was 72.12 yr for the whole population (69.56 yr for males and 74.81 yr for females) [*CIA*, 2009].

Cairo (Al-Qāhirah), the capital of Egypt, is the largest city in Egypt, followed by Alexandria. The Greater Cairo Area (GCA) consists of 16 agglomeration with a population of combined population of 15.2 million (including the close-by Cairo cities of Giza (Al-Jizah), Helwan (Hulwan) and Shubra al-Khaymah over a total surface of 8815 km^2 [*Brinkhoff*, 2009]. Cairo is a rapidly expanding city, which has led to many environmental problems. The Greater Cairo Area is situated south of the delta in the Nile basin. The main populated area is ~200 km^2, an area 4 km wide and stretching 50 km along the banks of the Nile River (Figure 16). Climatologically, GCA is in the subtropical region. Dust and sand storms frequently occur in spring and autumn and hot desert cyclones known as the "Khamasin" depressions pass over the desert during spring [*Zakey and Omran*, 1997]. These cyclones are always associated with strong hot and dry winds often carrying dust and sand that increase PM levels. During winter the general climate is cold, humid and rainy; while during the summer season the predominant weather is hot and dry [*Zakey et al.*, 2008].

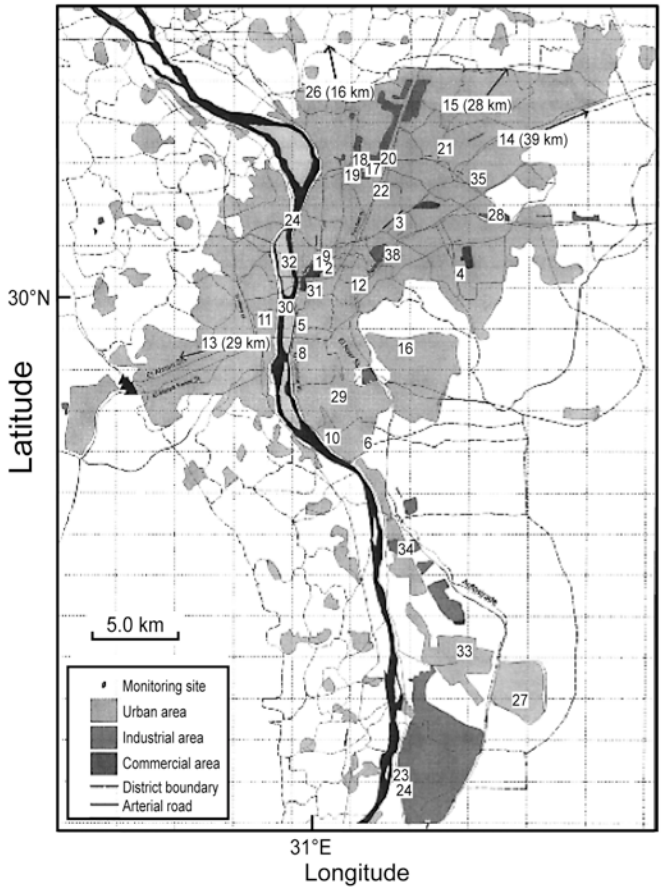

Figure 16 - Map of the Greater Cairo area with significant land features- grey from light to dark: urban, industrial and commercial areas respectively. Cairo Air Improvement Project (CAIP) monitoring site locations are also shown [*Abu-Allaban et al.*, 2002]

2.4.2 Emission sources of air pollutants

In Egypt the main industrial sectors consist of cement industry, metal smelters, brick factories, fertilisers, aluminium, petrochemical, chemical, sugar factories, and textiles (http://www.eeaa.gov.eg/eimp/typicalsourcesof%20pollution. html). About 52% of the industries and about 40% of the electricity production in Egypt are located in the GCA [*Nasralla*, 2001]. Cairo

also has many unregistered lead and copper smelters that heavily pollute the city. There are over 2 million cars on the streets of Cairo, 60% of which are over 10 years old and therefore lack modern emission cutting features like catalytic converters. Cairo has a very poor dispersion factor because of the lack of rain and its layout of tall buildings and narrow streets. This resulted in a permanent haze over the city with particulate matter in the air reaching over three times WHO levels. Open fire burnings is a common practice in Egypt and a major contributor to air pollution in the area. Their signal is seen in the AOD seasonality derived from satellite data (Figure 17).

The information regarding the amounts of pollutants released in the atmosphere of Cairo is very limited [*El Mowafi and Atalla*, 2005]. The only available emissions inventories in the area are those of EMEP in 50km resolution as well as emission inventories developed for global modelling such as, for example, EDGAR, GEIA, and ACCMIP [*Vestreng et al*., 2006; *Olivier et al.*, 2001; *Graedel et al*., 1993; *Lamarque et al*. 2010]. These inventories are found to be inadequate to support contemporary air quality applications in a large urban agglomeration such as Cairo due to their coarse resolution. Thus a more updated and detailed emission inventory such as the regional inventory developed for Africa in *Liousse et al*., [2012] and *Assamoi* [PhD 2011] will greatly improve our understanding of air pollution levels in the area.

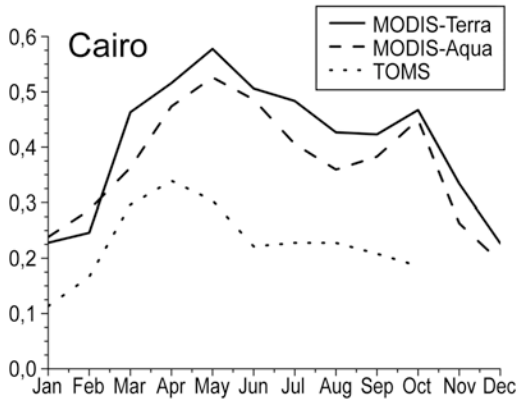

Figure 17 - Intra-annual variation of visible aerosol optical thickness (AOT) over Greater Cairo Area (29°N-31°N, 30°E-31°E), based on data taken from MODIS-Terra (2000-2005, solid lines), MODIS-"qua (2002-2005, dashed lines) and TOMS (1980-2001, dotted lines). The AOT values are provided at λ = 500 nm and λ = 550 nm for TOMS and MODIS, respectively (Figure adopted from *Hatzianastassiou et al*., 2009)

Some information on the sources responsible for air pollution levels is derived from source apportionment analysis based on simultaneous observations of several non-methane hydrocarbons (NMHC), including Benzene, toluene, ethylbenzene and xylene (BTEX), or of aerosol components, including metals [*Abu-Allaban et al.,* 2002; 2007; 2009]. They point to mobile emissions and industrial emissions (lead smelting and liquefied petroleum gas- LPG, considering that industrial processes may be fuelled by LPG) as the major source of NMHC during both summer and winter. Mobile evaporative emission contributions were higher during warmer periods. Mobile sources and open burning are major contributors to particulate matter levels (PM). The large particles, PM10, have also important geological contribution, instead of the secondary species that are contained in the smaller particles (PM2.5).

2.4.3 Air pollutants

Ozone and its precursors

As a consequence of the high pollutant emissions and the meteorological conditions that affect GCA, volatile aromatic hydrocarbon levels are higher than many other similar cities [*Khoder*, 2007]. Air quality measurements in Cairo have also been recording dangerous levels of lead (Pb), sulphur dioxide (SO_2) and suspended particulate matter concentrations due to decades of unregulated vehicle emissions, urban industrial operations, and chaff and trash burning [*Khoder*, 2002]. Ozone in the southwestern Cairo area has been observed to exhibit a seasonal and diurnal

cycle with levels reaching 140 μg.m^{-3} in summer 2001 (Egyptian Environmental Affairs Agency, Environmental Information monitoring programme; http://www.eeaa.gov.eg/ eimp/news8.html). *Khoder* [2009] reported a year (Dec 2004-Nov 2005) of observations of ground level O_3, nitrogen dioxide (NO_2) and nitric oxide (NO) concentrations at Giza in the GCA. The mean values of O_3 were about 44, 65, 91 and 58 ppb in daytime during the winter, spring, summer and autumn seasons, respectively. The diurnal cycles of O_3 concentrations during the four seasons revealed a uni-modal peak in the mid-day time, with highest O_3 levels in summer due to the local photochemical production. The diurnal variations in NO and NO_2 concentrations during the winter and summer showed two daily peaks linked to traffic density. The highest levels of NO_X were found in winter. Year-around, the observed mean daytime O_3 concentrations exceeded by about 35% (winter) to 100% (summer) of the days the Egyptian and European Union air quality standards of 60 ppb for daytime (8-h) O_3 concentrations.

Vrekoussis et al. [2009], based on SCIAMACHY satellite-sensor observations gridded into 0.125x0.125^o girds over the GCA, deduced mean tropospheric NO_2 columns over the period 2003-2007 higher than 6 x 10^{15} molecules cm^{-2}. They also derived an increasing trend that corresponds to 0.2-1.0 ppbv y^{-1} for this 4 years period.

Abu-Allaban et al. [2009] also reported very high NMHC, including BTEX, in the GCA varying from 365 ± 102 to 1849 ±298 ppb C in later winter and 462±315 to 2037±1369 ppb C in late fall 1999. In summer 2004, observations of NMHC by *Khoder* [2007] at three locations, two urban areas in GCA and background one in the rural area in Menofiya province point to road traffic as the major source of aromatic NMHC. The mean concentrations of n-hexane, n-heptane, benzene, toluene, ethylbenzene, (m, p)-xylene, o-xylene, 1,3,5-trimethylbenzene and 1,2,4-trimethylbenzene were about 124, 71, 87, 214, 43, 141, 74, 31 and 65 μg m^{-3}, respectively in city centre of Cairo where the total average concentration of NMHC was about 2 times higher than those found in other urban sites, and 22 times than those found in background sites. These levels were among the highest reported for megacity regions [*Khoder*, 2007].

Particulate matter

Abu-Allaban et al. [2002; 2007] reported pollutant observations at 6 sites in the greater Cairo area from 1999 to 2002, showing average PM_{10} ($PM_{2.5}$) mass ranged from 265 μg m^{-3} (216 μg m^{-3}) at an industrial site to 88 μg m^{-3} (30 μg m^{-3}) at a residential location. High levels of trace metals were also observed, with an average PM2.5 Pb level of 26.8 μg m^{-3}. Based on weekly observations from Jan 2003 to May 2006, *Favez et al.* [2008a] reported bulk aerosol seasonal mean levels of 115, 165, 215 and 190 μg m^{-3} during summer, autumn, winter and spring respectively at two urban sites in Cairo.

Khoder [2002] measured sulphate ($SO_4^=$) and nitrate (NO_3^-) aerosol components and their precursor gases SO_2, NO_2, nitric acid (HNO_3) and O_3 during winter 1999-2000 and summer 2000. The average concentrations were 6.2 and 9.8 μg.m^{-3} for particulate nitrate, 1.1 and 6.7 μg.m^{-3} for gaseous nitric acid and 15.3 μg.m^{-3} and 25.1 μg.m^{-3} for particulate sulphate, during the winter and the summer seasons, respectively. The highest average concentration ratio of gaseous nitric acid to total nitrate was found during the summer season. Particulate sulphate and nitrate and gaseous nitric acid concentrations were relatively higher in the daytime than in the nighttime.

Favez et al. [2008a,b] reported more than 2 years of weekly observations of bulk aerosols and their chemical characterization with respect to selected ionic species and carbonaceous aerosols (sum of elemental carbon (EC) and organic carbon (OC)) at two Cairo urban sites. Dust aerosols have been derived from calcium (Ca^{2+}) measurements and displayed high background concentration levels (50 μg m^{-3}) all year long and maximum concentrations during the dust storm periods [*Favez et al*., 2008a]. About 40% of Ca^{2+} on these dust aerosols was found to be associated with ions of anthropogenic origin like $SO_4^=$, NO_3^- and/or Cl^-, pointing out human driven processes that alter the chemical characteristics of dust and thus its climatic impact on a regional scale. High concentration levels of non-sea-salt chloride (up to 15 μg m^{-3}on a monthly basis), likely of industrial origin, were observed in autumn and winter.

During autumn, biomass burning aerosols originating from rice straw burning in the Nile Delta, known as the ''Black Cloud'' event, have shown to account for 12%, 35% and 50% of Cairo EC, water insoluble organic carbon (WIOC) and water soluble organic carbon (WSOC) mass concentrations, respectively. Overall, non-dust aerosols were equally distributed between carbonaceous aerosols and ions, and their concentrations were of the order of 100 μg m^{-3} in autumn and winter, and of 60 μg m^{-3} in spring and summer. Remarkably, relatively low WSOC/OC ratios (about 1/3) were obtained all the year-long [*Favez et al*., 2008a]. *Favez et al.* [2008b] further investigated the carbonaceous content in the sub micron fraction of aerosols by at an urban site in Cairo in spring 2005. They found well-marked diurnal patterns for the WSOC/EC and WIOC/EC ratios, with minima during the traffic-influenced morning period and maxima during the intense photochemical period, suggesting significant formation of both water-soluble and water insoluble secondary organic aerosols during the afternoon. Applying the EC-tracer method, freshly formed secondary organic carbon was found to possibly account for more than 50% of OC concentrations measured during the early afternoon period, and this fresh SOC was calculated to be mainly (~60%) composed of water-insoluble species. The latter (unexpected) result has been suggested to be due to low ambient relative humidity as well as to the importance of anthropogenic volatile organic compounds in Cairo [*Favez et al.,* 2008b].

Regulations

In 1995, the first environmental acts were introduced and the situation has seen some improvement with 36 air monitoring stations and emissions tests on cars. Twenty thousand buses have also been commissioned to the city to improve congestion levels, which are very high. In 2003, Egypt initiated an enforced vehicle emission-testing programme in Greater Cairo. The limits of CO, hydrocarbons and opacity for the vehicles before and after 1995 have the values of (7, 4.5 percent) (1000, 900 ppm) and (65, 50 percent), respectively. The publicized information indicated an overall failure rate of about 10 percent [*El Mowafi and Atalla*, 2005].

2.4.4 Health

Lead smelters have been found to be major sources of lead in Cairo's ambient atmosphere [*Abu-Allaban et al.,* 2007]. Melting down of old circuit boards and other electronic components for their metal content has been shown to expose communities to extremely high levels of dioxins and metals such as lead, cadmium and mercury [*Carroll and Essik*, 2008]. About 40% of the total population in Egypt is below 25 years old while only 3% exceeds the age of 65. In 2006 the birth rate was significantly higher than the death rate with 25.3 ‰ births compared to 6.2‰ deaths.

As reported by *El Mowafi and Atalla* [2005] health risks studies due to air pollution in Cairo conducted by *Smith* [1999] indicated that approximately 3% of the population is chronically exposed to PM_{10} levels above 100*μ*g/m^3, compared to 48 % exposed to 50-100 μg/m^3 and 49 % exposed to 5-50 μg/m^3 PM_{10}. Thus, it was suggested that Cairo air pollution causes about 3,400 premature deaths, 28 million restricted activity days and other additional cases of air pollution-related diseases, e.g. asthma attacks and chronic bronchitis. Based on ambient atmospheric concentrations of criteria pollutants, notably total suspended particles (TSP; 593 μg.m^{-3}), SO_2 (37 μg.m^{-3}), and nitrogen dioxide (NO_2; 59 μg.m^{-3}), *Gurjar et al.* [2008] have classified Cairo as extremely poor air quality megacity where measures for air pollution reduction need to be taken urgently. It is estimated that 10,000 to 25,000 people a year in Cairo die due to air pollution-related diseases. *The World Bank* [2002] evaluated that environmental degradation in Egypt at 0.7-2.3 of GDP per year, accounting that air pollution causes about 20,000 premature deaths every year, in the two metropolitan areas and 450,000 disability adjusted life year with about 92 % of them being in the Greater Cairo because of the higher air pollution and larger population. These findings indicate the significant benefits that could be achieved by implementing the proper abatement measures to improve air quality in Cairo.

2.5 GENERAL CONCLUSIONS

This chapter strongly stresses the need to seriously considering air pollution and associated health risks in African megacities as a subject of its own. Air pollution levels are

comparable to those encountered in the most polluted cities of the world. Due to anthropogenic pressure and lack of present regulations, pollution and associated risks are expected to increase more and more in the absence of any mitigation plan. These are not only national problems, but international problems due first to pollutant transport but also to technological trades (e.g. China to Africa for two wheel vehicles, Africa to China for coal liquefaction process).

The actions to be taken are complex and multiple because Africa has a wide representation of development levels. Africa has to fight against problems of both developed, semi-developed and developing countries, e.g. food and energy development, political instability, high polluting technologies related to industries and domestic fuel burning conversion, growing traffic reduction, health diseases, etc. Indeed, in most places, importance of health diseases due to air pollution is now competitive to ones due to infectious diseases. This is very critical due to expected interactions between both diseases and public health deterioration.

Moreover, air pollution and health problems have a significant cost on national economics linked to hospital costs but also to absenteeism, job losses, and deaths. All of the proposed plans will need to be tested from an economical point of view as well as from the consequence of morbidity/mortality effects.

These actions can also have an important impact on climate change due to reduction/increase of greenhouse gases and aerosol emissions, not only due to pollutant concentrations but also due to quality with relative composition of atmospheric pollution mixtures.

In conclusion, it is time to act, and the actions for research studies have to focus on integrated projects combining emissions, air quality, health studies (epidemiological monitoring, toxicology, hospitalizations), acid deposition and impacts on soil, crop and water resources, local and regional climate change and cost-benefit studies to provide possible emission mitigations option that include air quality and climate change feedback impacts. A required condition for successful action could be found if a strong link between these research programmes is constructed regionally with educational systems and policy makers.

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CHAPTER 3 - ASIA

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3.1 ASIAN MEGACITIES: GENERAL CHARACTERISTICS

Population and geography

With more than 50% of the world population, Asia is the most intensely populated continent on the earth. Based on the World Urbanization Prospects of the United Nations (2009 Revision), in 2010, 10 of the 21 world megacities and 15 of the worlds 30 largest cities are in Asia (Table 1).

Table 1 - The 15 of the 30 world largest cities in Asia in 2010.
Source: the World Urbanization Prospects of the United Nations (2009 Revision)

Rank order	Country	Megacity	Population (Million)
1	Japan	Tokyo	36.67
2	India	Delhi	22.16
4	India	Mumbai (Bombay)	20.04
7	China	Shanghai	16.58
8	India	Kolkata (Calcutta)	15.55
9	Bangladesh	Dhaka	14.65
10	Pakistan	Karachi	13.12
13	China	Beijing	12.39
15	Philippines	Manila	11.63
16	Japan	Osaka-Kobe	11.34
22	Republic of Korea	Seoul	9.77
23	China	Chongqing	9.40
24	Indonesia	Jakarta	9.21
26	China	Shenzhen	9.01
28	China	Guangzhou, Guangdong	8.88

The bottom-up and top-down methods for estimating population in Asia may have large differences. For example, Beijing Municipal Bureau of Statistics reports that Beijing has 15.38 million long-term residents and 3.57 million temporary migrants in 2005 (http://www.bjstats.gov.cn/sjfb/pcsj/rkpc/200607/t20060704_45124.htm), while the World Urbanization Prospects of the United Nations (the 2009 Revision) reports that Beijing has a population of 11.45 million in 2005 an 12.39 million in 2010 (Table 1).

One of the major reasons for this large difference is that urbanization in Asia is occurring at a fast speed due to the dynamic social and economical development. Projections based on the census data years ago might not be able to reflect the real situation in many cities in Asia.

The continent of Asia is characterized by high diversity and inhomogeneity in geography, this leads to large spatial variation in the population density in Asia. Figure 1 shows that, besides the hot spots of large cities with high population density, Asia also has large regions with high population density, such as Bangladesh, Indo-Gange plain, and North China plain. Bangladesh is one of the most densely populated regions in the world.

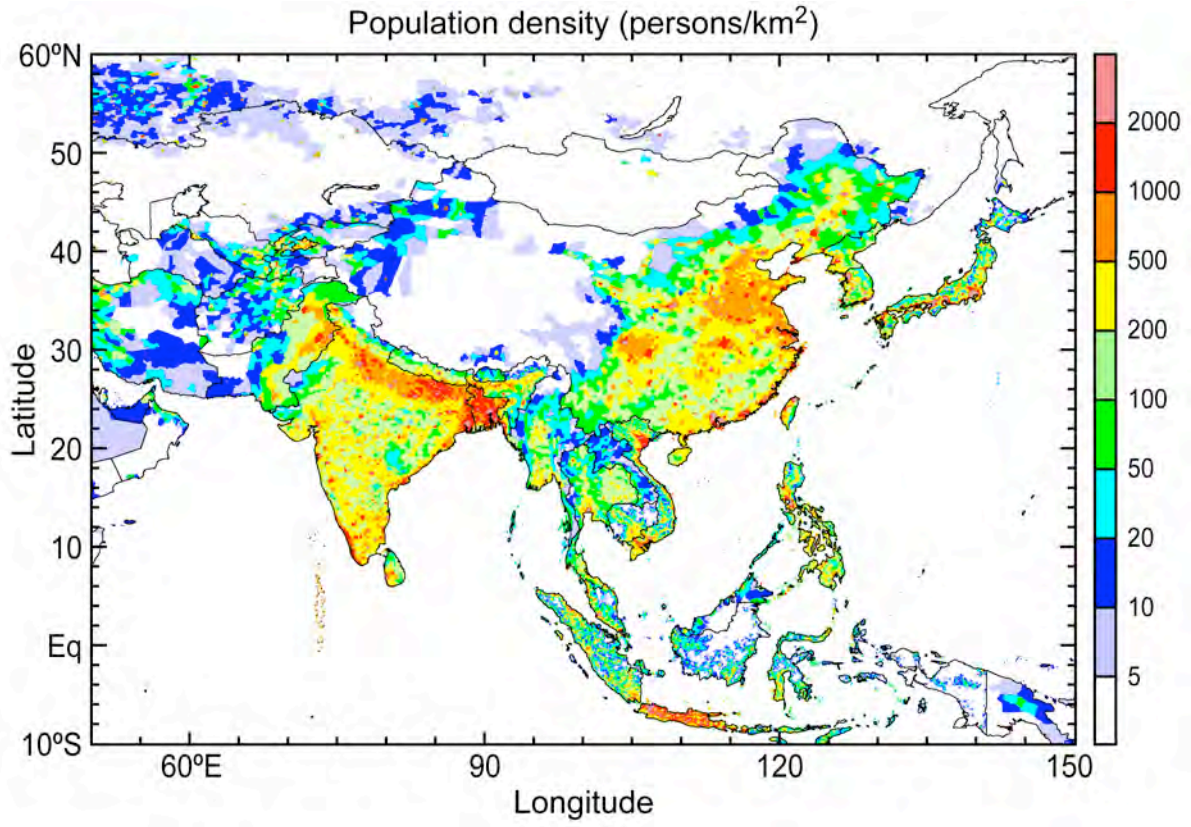

Figure 1 - Map of the population density in Asia (persons per km2), based on 0.25º gridded data for 2000 from the Center for International Earth Science Information Network (CIESIN) at Columbia University [*http://sedac.ciesin.columbia.edu/gpw/*]

Asian megacities

This chapter covers 12 Asian megacities and major population centers (MPC) (Table 2) that reflect the high diversity of geography and socio-economical development of megacities in Asia. Tables 1 and 2 clearly show that there is a large difference in the population of each of these cities. Due to the inconsistence of the population data from different sources, the selection of the cities covered in this report is not only based on population, but also based on country and geographical representation, and equally important, the availability of authors and data to report on these megacities.

Table 2 also lists the area, latitude, and longitude of each city. One has to keep in mind that the definition of the area of each city may vary greatly as some estimation are based on administrative borders and others are based on the urban area.

Table 2 - Twelve Asian megacities reported in this chapter. The source of information can be found in the section of each megacity in this chapter

Megacity	Country	Population [Million] (year)	Area (km^2) (urban area)	Latitude	Longitude
Bangkok	Thailand	[10.1] (2007)	7762	13°45'	100°35'
Beijing	China	[16.3] (2008)	16808(735)	39°54'	116°24'
Delhi	India	[16] (2007)	900	28°36'	77°13'
Dhaka	Bangladesh	[13]		23º76´	90º38´
Hong Kong	China	[6.98]	1104	22º15'	114º10'
Jakarta	Indonesia	[8.7] (2005)	661	6º12' S	106º48'
Manila	Philippines	[11.56] (2007)	636	14 º 34'	120 º58'
Osaka-Kansai	Japan	[22.7] (2008)		34.67°	135.53°
Pearl River Delta	China	[47.9] (2009)	4170	21°17´– 23°56´	111°59´– 115°25´
Seoul	Korea	[23.9] (2008)		37° 33'	126° 58'
Shanghai	China	[19.2] (2009)	6341	30º40'	31º53'
Tokyo	Japan	[12.9]		35.69°	159.69°

Emission

Figure 2 shows the emissions of CO, NO_X, and SO_2 in Asia for the year 2000 based on the EDGARv3.2 FT2000 database, reflecting fossil fuel based energy consumption. The bottom-up and top-down methods may also have large differences in emission data, especially in Asia where the emission inventories are constantly changing due to dynamic economic development. However, the large spatial variations of CO, NO_X, and SO_2 emissions are clearly associated with the high diversity and inhomogeneity of the population density.

Asia is also highly diversified and inhomogeneous in its economical development stages. Tokyo and Osaka-Kobe are well-developed megacities, while the rest of megacities in Asian are still in the developing stage. This is an important factor when determining how pollution is generated and controlled in the megacities.

In the well-developed megacities, such as Tokyo, the sources of air pollutants and greenhouse gases (GHGs) are dominated by the vehicle emissions, whereas in the developing megacities, biomasses burning from agriculture waste and forest fires are also important sources to local air pollution.

One unique source of air pollution to the East Asian megacities is dust and sand storm (DSS) that originate in the Mongolia desert and Gobi area every spring. The air quality in megacities like Beijing, Seoul, and Tokyo is frequently deteriorated due to DSS.

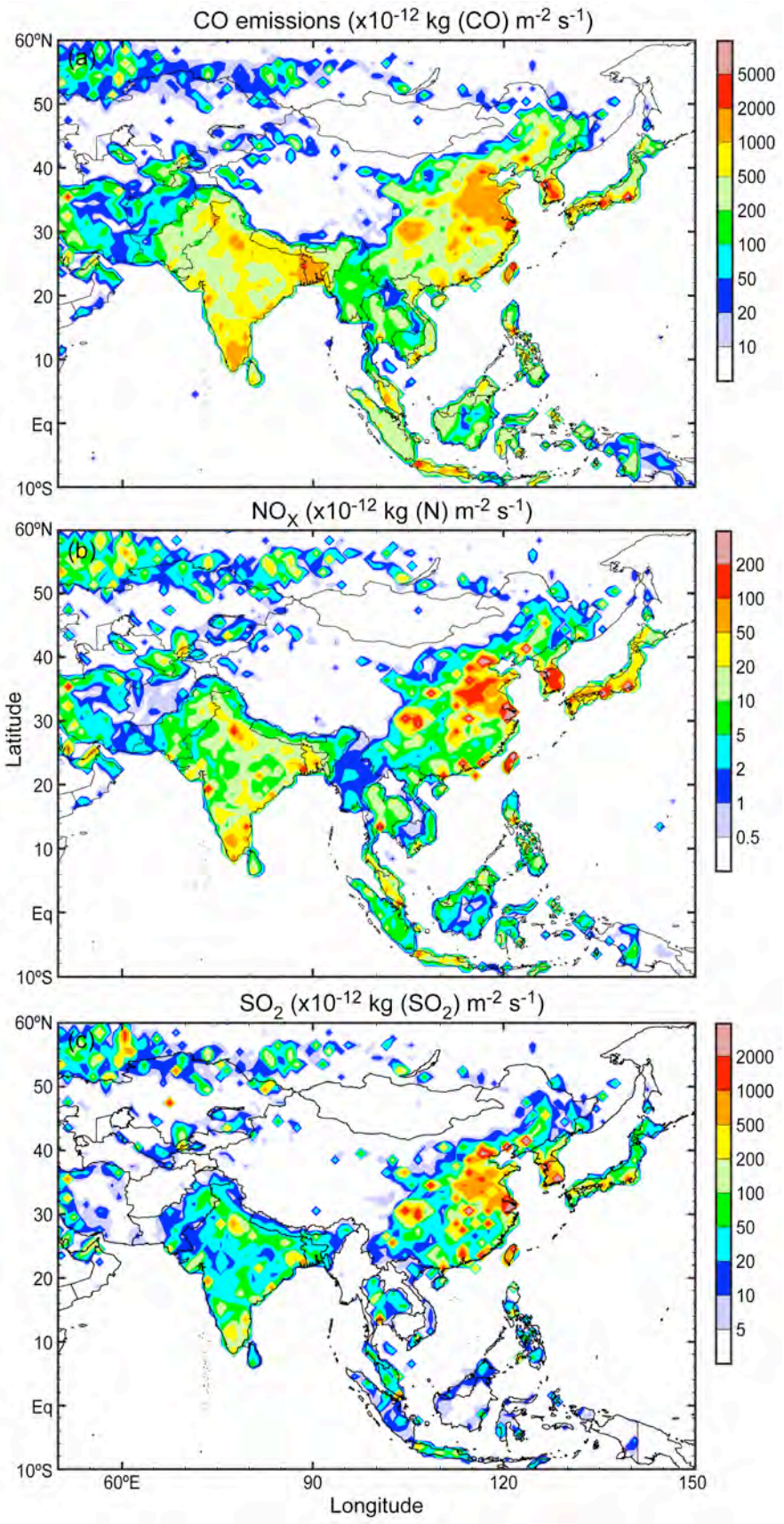

Figure 2 - (a) CO emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (b) NO_X emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (c) SO_2 emissions for the year 2000 based on the EDGARv3.2 FT2000 database

Air pollution levels

Air pollution is a serious environmental problem in Asian megacities. Concentrations of air pollutants, especially particulate matter, are frequently higher than those of the WHO guidelines and the ambient air quality standards of their own countries.

Megacities in Asia have taken various measures to control air pollution; some measures implemented in developing megacities are equally strict as those in the developed ones. For example, in 2008, Beijing implemented vehicular emission standards on new cars equivalent to EURO-IV. Introducing lead-free gasoline in many developing megacities proved to be effective in reducing lead concentrations in particulate matter. Clean energy and advanced public transportation systems have also played important roles in improving the air quality in Asian megacities.

Monitoring network data show that air pollution in Asian megacities remains a serious risk to human health and ecosystems. This indicates that all the air pollution mitigation efforts put forward by megacities to date still have not gone far enough. With the fast social and economical development and urbanization in Asia, new emission sources are being added to the old emission sources that are currently not well controlled. How to realize the co-benefits of controlling air pollution and reducing climate change in Asian megacities requires systematic research efforts.

3.2 BANGKOK, THAILAND

Introduction and specific features of the city

Bangkok, the capital city of Thailand, is recognized as a megacity in South East Asia and is facing serious air pollution problems. Bangkok and five bordering provinces including Samut Prakarn, Nonthaburi, Pathumthani, Nakorn Pathom, and Samut Sakorn are known collectively as the Bangkok Metropolitan Region (BMR), which is an economic centre of Thailand. BMR has an area of 7,762 km^2 and is situated in the central part of Thailand in the Chao Phraya River Basin. Bangkok city is located around 13°45' Northern latitude and 100° 35' longitudes in an immediate proximity to the Gulf of Thailand. BMR has a total population of about 10.1 million (as of 2007), which comprises around 16% of the total population in Thailand (63.0 million, as of 2007). In 2007, the population density in Bangkok was 3,644 persons per km^2 while the average population density in BMR is 1,297 persons per km^2 [NSO, 2009]. Bangkok land use comprises 23.4% for residential area, 23.6% for agricultural area, 8.2% for road transportation, 3.9% for commercial area, 24.2% un-use land, and the rest are for other purposes [Department of City Planning, 2008].

The Gross Domestic Product (GDP) of Thailand is continuously increasing. In 2008, Thailand GDP was 9,075 Billion Baht, a growth of around 6.4% from 2007, and the national income was 6,687 Billion Baht [BOT, 2010] (the exchange rate is around 35 Bath for 1 USD in 2007). Accordingly, the demand for electricity has increased. In 2008, a total of 148,200.93 Million kWh was required across the country, 0.87% increase from 2007 [EGAT, 2008]. Electricity consumption in BMR is also increasing, similar to the trends of electrical consumption in Thailand, especially in the inner Bangkok area.

BMR experiences the tropical monsoon climate with two distinct seasons. The wet season extends from mid-May to mid-October, when the southwest monsoon dominates and brings in warm, moist air from the Gulf of Thailand and the Andaman Sea. The southwest monsoon and the Inter Tropical Convergence Zone (ITCZ) cause heavy rainfall during the wet season. The dry season can be further classified into two periods. The first period (mid-October to mid-February), known as the local winter, is characterized by mild weather under the influence of the northeast monsoon that brings in cold, dry air associated with a high-pressure ridge extending from the anticyclone in China. The second period of the dry season (mid-February to mid-May) is known as the local summer when weather is the hottest of the year, with the highest temperature occurring in April [TMD, 2009]. Bangkok is also under the influence of a sea land breeze. The southerly sea breeze counteracts the northeasterly monsoon during the winter, which results in low wind conditions over the city. This, in turn, reduces the mixing and enhances pollution build-up [*Zhang*

and Kim Oanh, 2002]. Meteorological conditions, averaged over 1981-2007 period, show the annual average temperature of Bangkok ranges between 26.7-28.0^{o}C and average annual rainfall between 1,320-1,950 mm. Low wind, average of 1.3 m/s and mostly below 3 m/s, is observed throughout the year [TMD, 2009]. Monthly windroses of a 10-year period, 1991 – 2000, show high percentages of calm conditions, ranging from 30% in March to 61% in October [*Zhuang*, 2001]. Monthly meteorological variables in Bangkok, averaged for 1971 to 2000, are presented in Figure 3.

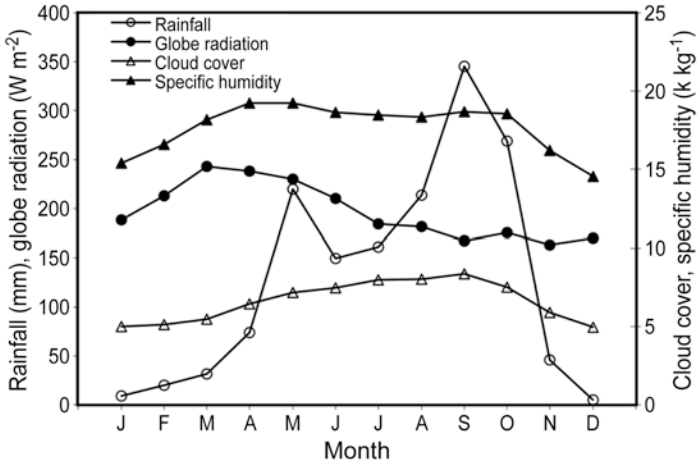

Figure 3 - Monthly observed average meteorological conditions in Bangkok, Thailand (1971-2000) [*Zhang,* 2001]

Emission sources of air pollutants

Major sources of air pollution in BMR include traffic, power plants, industries and incinerators. In addition, agroresidue field open burning also contributes significantly to urban air pollution but has not yet been properly quantified. Point sources (industry, power plants) are the major source of SO_2, whereas mobile source contributes significantly to NO_X, CO, VOC/HC, and PM emission. Area sources also contribute a remarkable amount to CO emission.

The average growth rate of vehicles in Bangkok, over an 18-year period starting from 1989, is around 7% per year. There were approximately six million vehicles registered in Bangkok as of May 2009 [Department of Land Transport, 2007]. A high growth rate of vehicles and inadequate road infrastructures are major causes of traffic congestion in Bangkok that lead to high air pollution emissions. In recent years, in order to improve air quality, the European standards have been imposed for new vehicles progressively. For example, in 1998 new heavy-duty diesel vehicles registered in Thailand had to meet EURO1 standards. The EURO2 standard was implemented in 2003 and since 2007, new heavy-duty diesel vehicles have to meet the EURO-3 standard [PCD, 2009]. For over a decade, development of fuel specification has been continuously implemented in Thailand. Leaded gasoline was phased out as of January 1996. Sulphur content in diesel fuel is being reduced, e.g. 500 ppm sulphur was enacted on July 1998; 350 ppm sulphur was enacted on January 2004; and 50 ppm sulphur is currently proposed for 2010 [ESMAP, 2008]. Thailand has a policy of encouraging use of compressed natural gas (CNG) and ethanol in transport [ESMAP, 2008]. Nowadays, the CNG use is widely spread, especially in taxis and trucks.

For point sources, there are two thermal power plants under EGAT operated in BMR. Fortunately one is oil-based and the other is natural gas (NG) based, which are cleaner than coal-based power plants. Around 38,593 industries of different types and sizes are registered in BMR [DIW, 2007]. Half of them (19,082 industries) are located in Bangkok city. Most of these industries are of small scales and potentially more polluting. A few waste incinerators are also in operation in BMR to treat the hygiene and hospital wastes [*IPEP*, 2006] and also emit air pollutants.

Data available on air pollutants

Monitoring of ambient air quality in Thailand was established in 1983 under the Pollution Control Department (PCD) of Thailand. The ambient air quality monitoring network consists of over 50 permanent automated ambient air quality monitoring stations located throughout the country [PCD, 2006]. Ambient air pollutants measured include CO, NO_X, SO_2, O_3, TSP, PM_{10}, Pb, and HC.

Most of them also have 10 or 30 meter meteorological masts. Data are generated on hourly basis (except for TSP) and are transmitted daily to the central data processing system at the PCD through a dial-up telemetric communication system [*Supat*, 1999].

Bangkok has 17 permanent automatic monitoring stations (as of 2006), whereas each surrounding province has only one station located at the most densely populated area. In addition, a number of temporary monitoring stations have also been set up in the Bangkok area [PCD, 2006]. The monitoring stations in Bangkok are categorized into two types, namely, general and roadside stations. General ambient air quality monitoring stations are located within 50-100 m from main roads, whereas roadside street-level stations are situated within 2-5 m from main road [*Supat,* 1999a].

The status and trend of the pollution

Results of ambient air quality monitoring for more than 10 years indicate that the air pollutants of greatest concern in Bangkok are suspended particulate matter, especially PM_{10}, and ground level ozone. Both of them usually exceed the air quality standard of Thailand: PM_{10} at roadside (annual average standard: 50 $\mu g/m^3$) and ozone at ambient sites (one hour maximum standard: 100 ppb). Other pollutants, such as CO, SO_2, NO_X, and lead (Pb), are generally lower than the standards. Table 3 summarizes the quality of the ambient air in Bangkok as of 2008. Trends of the ambient air pollution levels over the period 1995-2008 for PM_{10} and other pollutants are shown in Figure 4 and Figure 5, respectively.

$PM_{2.5}$ data, available from monitoring of the AIRPET research network, show high levels of $PM_{2.5}$ especially during the dry season with the average of 50 $\mu g/m^3$, which is significantly higher than the wet season level of 18 $\mu g/m^3$ [*Kim Oanh et al.*, 2006].

Table 3 - Ambient air quality in Bangkok as of 2008. (Source: PCD, 2008)

Pollutant	Unit	Range	95-percentiles	Standard	Times exceeding standard / measurement times (%)	Annual average	Annual standard
General areas							
24-hr avg TSP	mg/m^3	0.01-0.33	0.17	0.33	0/541 (0)	0.08	0.1
24-hr avg PM_{10}	$\mu g/m^3$	12.1-180.9	86.9	120	30/2,540 (1.2)	47.9	50
1-hr avg CO	ppm	.0.0-6.8	1.7	30	0/80,728 (0)	0.7	-
8-hr avg CO	ppm	0.0-4.4	1.5	9	0/83,758 (0)	0.7	-
1-hr avg O_3	ppb	0-153	56.0	100	194/77,541 (0.3)	17	-
1-hr avg SO_2	ppb	0-53	10.0	300	0/80,981 (0)	4	40
24-hr avg SO_2	ppb	0-16	8.7	120	0/3,337 (0)	4	40
1-hr avg NO_2	ppb	0-152	52.0	170	0/81,534 (0)	23	-
Roadside areas							
24-hr avg TSP	mg/m^3	0.03-0.86	0.28	0.33	25/695 (3.6)	0.14	0.1
24-hr avg PM_{10}	$\mu g/m^3$	8.1-205.4	113.6	120	82/2,000 (4.1)	61.8	50
1-hr avg CO	ppm	0.0-16.4	3.5	30	0/64,716 (0)	1.4	-
8-hr avg CO	ppm	0.0-10.0	3.4	9	7/65,491 (0.01)	1.4	-
1-hr avg O_3	ppb	0-116	37.0	100	10/25,988 (0.04)	11	-
1-hr avg SO_2	ppb	0-45	12.0	300	0/25,566 (0)	5	40
24-hr avg SO_2	ppb	0-18	9.3	120	0/1,089 (0)	5	40
1-hr avg NO_2	ppb	0-177	70.0	170	1/26,169 (0.004)	34	-

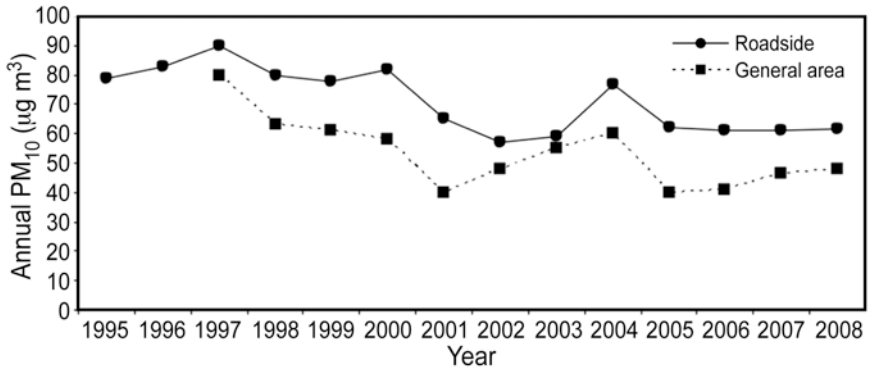

Figure 4 - Annual average PM_{10} concentration in Bangkok from 1995-2008 [*PCD*, 2008]

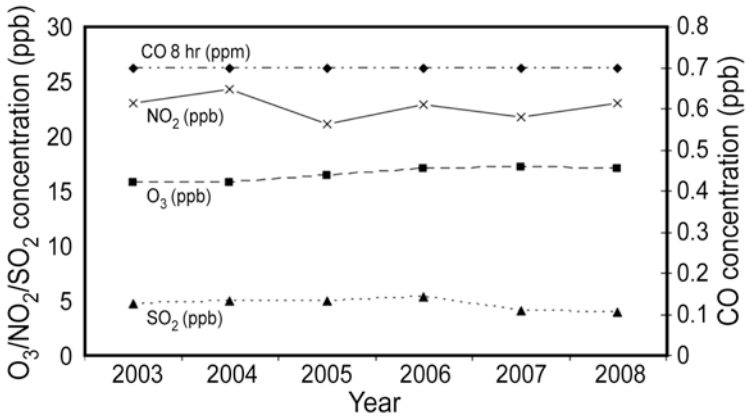

Figure 5 - Annual average air pollutant concentrations in the general area of Bangkok from 2003-2008

Relationships of the trends to regulations

The air quality management programme in BMR follows the policy and prospective plan of Thailand. This programme covers enhancement and conservation of natural environmental quality, which is a cross-sectoral operation involving the environment, transport, and urban dimensions. The air quality management policy of Thailand for 1997-2016 focuses on the following key issues: (1) air quality in pollution control zones and urban areas, particularly on dust, (2) other pollutants in ambient air, particularly on carbon monoxide, and (3) air pollutants in industrial zones and general communities, particularly on sulphur dioxide and nitrogen oxides, to be within designated Ambient Air Quality Standards [*Supat*, 1999].

Successful stories include the phasing out of leaded gasoline, establishment of oxygenated gasoline, reducing sulphur content in diesel, and development of the standard for low-smoke 2T lubricating oil. Phasing out of lead in gasoline, starting in January 1996, has substantially reduced Pb in ambient air (Figure 6). At present, lead concentration in ambient air is much lower than the standard. Recently, due to the concern of toxic effects of volatile organic compounds (VOCs), VOC ambient air quality standards have been established [PCD, 2006].

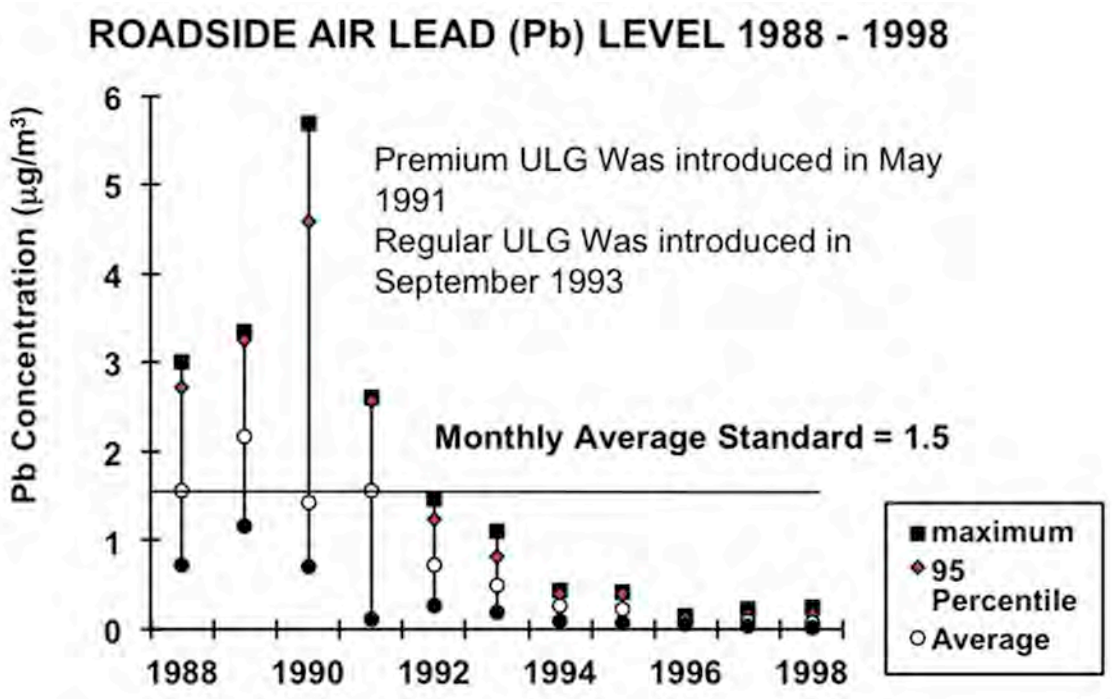

Figure 6 - Annual average curbside lead concentration in Bangkok from 1985-1998 [*Supat,* 1999]

Climatic change issues

The Intergovernmental Panel on Climate Change (IPCC) estimated that, as of 2004, Thailand contributes around 265 million tons of CO_2 per year, ranked 25th in the world. The data shows that 56.1% of greenhouse gases (GHG) in Thailand are from energy uses (as of 2003), whereas the remaining GHG emissions are from agriculture (24.1%), wastes (7.8%), land use change (6.65%), and industrial sector (5.4%) [TGO, 2009].

Thailand signed the United Nations Framework Convention on Climate Change (UNFCCC) in June 1992 and ratified the Convention in March 1995, as a Non-Annex I country. In addition, Thailand signed the Kyoto Protocol in February 1999, and ratified it on 28 August 2002. According to the agreement, a promotion of Clean Development Mechanisms (CDM) under the guideline from the Kyoto Protocol has been implemented in order to encourage clean and environmental friendly technologies for GHG reduction in the country, as well as to promote the country's capability by developing sustainable business practices [TGO, 2009].

In 2007, the Thailand Greenhouse Gas Management Organization (Public Organization) (TGO) under the Ministry of Natural Resources and Environment (MNRE) was established. The organization is responsible for implementation of GHG emission reductions, promoting low carbon activities, investment and marketing on GHG emission reductions, establishing GHG information centre, reviewing CDM (Clean Development Mechanism) projects for approval, and providing capacity development and outreach for CDM stakeholders. The organization in particular performs the role as the Designated National Authority for CDM (DNA-CDM) office in Thailand. As of 2007, there were five CDM projects approved by the Cabinet. They are expected to reduce CO_2 emission in Thailand by around 578,700 tons per year [TGO, 2009].

Research projects on air quality in BMR

Several governmental air quality projects have been implemented. A project on measurement of agro-residual open burning in Thailand has been established in order to assess the status of agro-residual open burning and to provide a database for implementation of abatement programmes. PCD also conducted a project on the comparison of PM emissions from internal combustion diesel engine using commercial diesel and bio-diesel fuel. An inspection and

maintenance programme is being promoted in BMR among private bus companies in order to reduce vehicle emission. Development of strategies for reduction of diesel emissions and demonstration project on emission control devices for diesel vehicles has been conducted since the middle of 2006 [*PCD*, 2006].

There are also a number of international projects funded by donors and carried out in Thailand through research networks. These include the DIESEL project [*DIESEL*, 2008], which aims at development of integrated emissions strategies for existing land-transport. Japan International Cooperation Agency (JICA) has also provided funding and technical supports through many projects in Thailand e.g. development of environmental and emission standard of VOCs, emission inventory and modelling for acid deposition assessment. BMR is also a target city of the research network for "Improving Air quality in Asian Developing Countries" or AIRPET (http://www.serd.ait.ac.th/airpet), funded by Sida. ARRPET conducts monitoring of toxic pollutants, modelling of air quality, and develops integrated management strategies for agro-residues open burning.

Problems remaining

Presently, high PM_{10} and ozone remain the most serious air quality issue in Bangkok. Traffic is the major sources of air pollution in the city. Air toxics, in particular the volatile organic compounds (VOCs/HC), are also of concern in BMR and need continuous monitoring and standard enforcement. Open burning of agro-residues contributes significantly to air pollution in BMR and appropriate regulations have to be established and enforced in conjunction with the educative measures.

3.3 BEIJING, CHINA

Introduction and specific features of the city

Beijing, China (Figure 7) is a megacity that is experiencing fast air quality improvements. The city is located between $39^0$28' N to $41^0$05'N latitude, and between $115^0$25' and $117^0$30' longitude. Beijing is surrounded by mountains on three sides: the Taihang Mountain lies to the west, while the Yanshan Mountain lies to the north and the northeast; the Great North China Plain links the south of Beijing all the way to Bohai Bay. The average height of the surrounding mountains is about 1000 m.a.s.l with a descending trend from northwest to southeast. The administration area of Beijing is 16,808 km^2, of which 61% is mountainous area. The special topography results in a mountain-plain breeze: north wind at night and south wind during the day. This effect leads to an isolated local circulation that is unfavourable for the dispersion of air pollutants within the city.

Figure 7 – Map of Beijing city in China

Beijing has a population of 16.3 million as of 2008. As the capital of China, the city has had fast economic growth in the last two decades: the GDP per capita has been increasing at a rate of more than 13% within the last 5 years, and reached 9075 USD in 2008. In the same period, the energy consumption increased at a rate of 6% ~7% per year. The number of vehicles in Beijing, which increased at an annual growth rate of 13.3 % from 1999 to 2008 and reached to 3.5 million in 2008, is also an important driving force influencing air quality trends.

Emission sources of air pollutants

The trend of annual emissions of SO_2 and dust as well as the increase in the number of vehicles are shown in Figure 8. According to a study by *Hao et al.* [2005], the major SO_2 emission sources are power plants, domestic heating, and industrial sources; accounting for 49%, 26% and 24% of the total emissions, respectively, in 1999. Due to stringent air pollution controls, SO_2 emissions show a decreasing trend from 2.24×10^5 tons in 2000 to 1.52×10^5 tons in 2007. In 2008, the SO_2 emissions dropped by 12.9 % of the 2007 level. The emission of NO_X in Beijing appears to be stabilized and a constant, NO_2 concentration in Beijing from 1998 to 2005 suggests that there is no significant increase in NO_X emissions in Beijing [*Chan & Yao*, 2008]. National Ambient Air Quality Standards for NO_X were relaxed in 2000; this is one of reason that the NO_X emission control in China is not as stringent as SO_2. However, the NO_X emission reduction policy is one of the top priorities for the coming the 12th five-year-plan of China.

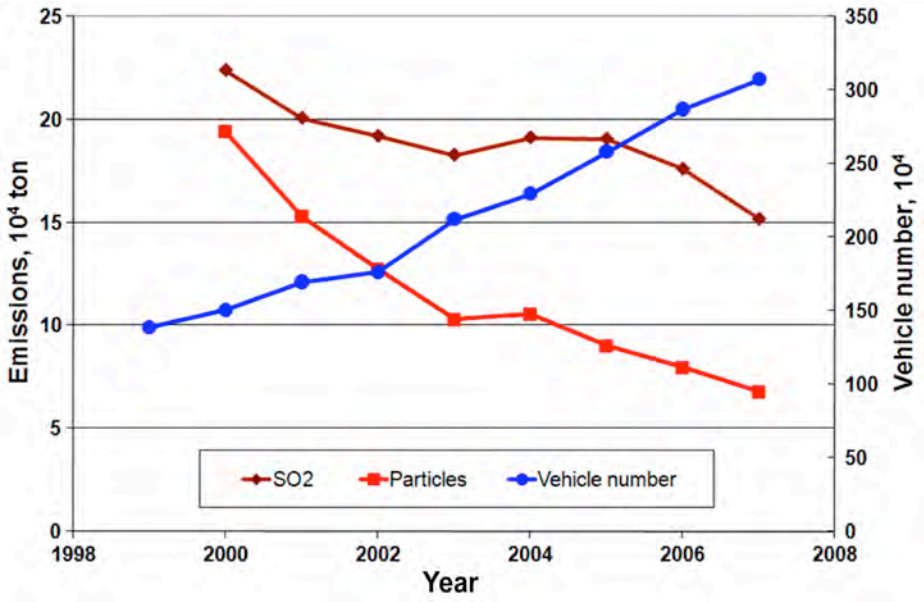

Figure 8 - The variation of emissions of SO_2 and particulate matter and the increase of vehicle numbers in Beijing [*Beijing Municipal Environmental Protection Bureau*, The communiqué on the Environmental Status of Beijing City, 1999 ~ 2007]

The information for volatile organic compounds (VOC) emission inventories, e.g. emission factors of sources and correspondent activity data is very limited in China. Vehicular emissions have been extensively studied for measuring the emission factors [*Fu et al.,* 2005]. A more recent work was to map the anthropogenic VOCs emissions in China from 1980 to 2005 based on updated activity data and available emission factors [*Bo et al*., 2008]. This study revealed a very fast increase of VOCs emissions in China, from 3.91 Tg in 1980 to 16.49 Tg in 2005 with the annual growth rate at 10.6%. Beijing, Yangtze River Delta, and Pearl River Delta region were the most intensive VOCs emissions areas. Beijing city alone contributed 0.32 Tg VOC. On the other hand, the VOCs source apportionments using receptor models were performed for Beijing city, the results show that mobile sources contributed more than 50 % of VOCs (in mass concentrations) to ambient air, other important sources are gasoline evaporation, painting, and solvent use [*Lu et al*., 2007; *Song et al*, 2008]. More importantly, alkenes and aldehydes were found to be the most reactive species for ozone formation and vehicle exhaust was the largest contributor to reactive alkenes,

nearly half of the C1–C3 aldehydes were attributed to secondary sources, while regional background accounted for 21-23% of the mixing ratios of aldehydes. Primary anthropogenic emissions contribution to aldehydes was comparable to biogenic emissions contributions (10–16%) [*Liu et al*, 2009].

Data available on air pollutants

The air quality in Beijing is routinely measured by Beijing Municipal Environmental Protection Monitoring center. The center runs 27 automatic monitoring stations with 13 stations in the urban center and 14 stations in sub-urban regions. The air pollutants measured at these stations are ambient concentrations of SO_2, NO_2, and PM_{10}. The monitoring data are available via Beijing Municipal EPB.

From 2006, the air chemistry in Beijing city and surrounding areas were studied in the CareBeijing field campaigns. More species including O_3, $PM_{2.5}$, CO, volatile organic compounds (including carbonyls), HONO, and size distributions and chemical compositions of particles were measured. A database for CareBeijing-2006, 2007, and 2008 was established and assessable based on the data protocol of the project.

The status and trend of the pollution

The trends of the concentrations of SO_2, NO_2, CO, and PM_{10} were evaluated based on the annual communiqué released by Beijing Municipal Environmental Protection Bureau from 1999 to 2007 (Figure 9). The SO_2 levels have been decreasing and went below the national ambient air quality standard (NAAQS) for SO_2 at grade II in 2005. The annual average NO_2 concentration has stayed at a level of about 70 μg/m^3 and almost remained constant. PM_{10} is reported to be the major air pollutant on about 90% of the days in the last nine years in Beijing and its annual average concentrations were all above the NAAQS for PM_{10} from 1999 to 2007. On 30% of the days each year, the daily average PM_{10} concentration exceeded the Grade-II standard of 150 μg/m^3.

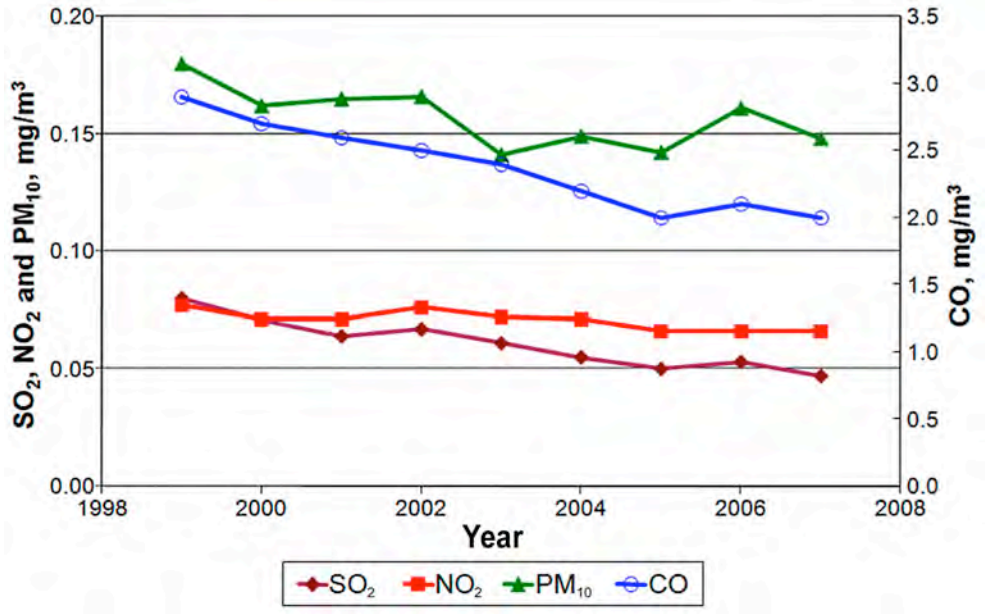

Figure 9 - The change of annual average concentrations of SO_2, NO_2, CO, and PM_{10} in Beijing between 1999 and 2007 [*Beijing Municipal Environmental Protection Bureau*, The communiqué on the Environmental Status of Beijing City, 1999 ~ 2007]

The air quality is assessed by an Air Pollution Index (API) that reflects the concentrations of three pollutants; SO_2, NO_2, and PM_{10}. However, the secondary air pollutants cause more significant adverse effects on public health and eco-systems. From a recent evaluation of air quality in Beijing [*Tang et al*., 2009], NO_X concentrations decreased linearly at a rate of 3.9±0.5 ppbv/yr after 2002,

while ozone concentrations increased at a rate of 1.1±0.5 ppbv/yr in a two-year cycle during 2001-2006, and O_X (O_X = NO_2+O_3) concentrations remained nearly constant (Figure 10). The reduction of NO_X emissions and elevated non-methane hydrocarbon (NMHCs) emissions may have contributed to the increased O_3 concentrations in Beijing.

The concurrence of high levels of ozone and fine particles suggest that air pollution in Beijing is very severe and complex [*Shao et al*., 2006]. Beijing is a typical city suffering from complex air pollution with recent ambient $PM_{2.5}$ concentrations ranging between 96.5 μg/m^{-3} and 154.3 μg/m^{-3}, six to ten times as high as the limit recommended by the US EPA (15 μg/m^{-3} annual average) [*Chan and Yao*, 2008]. In Beijing, the non-attainment days for ground-level ozone (number of days with hourly ozone concentration >100 ppbv) accounted for more than 10% of days from 1999–2007 [*Beijing Municipal Environmental Protection Bureau*, 1999-2007]. At one site in Beijing, about 20 km northwest of Beijing downtown area, an evident increase in ozone concentrations was found from long-term diurnal variations in ozone between 1987 and 2003 as shown in Figure 10 [*Shao et al*., 2006]. *Wang et al*., [2006] measured ambient ozone concentrations in a northern mountainous site in Beijing in summer 2005 where ozone levels exceeded 120 ppbv on 13 out of 39 days and maximum hourly average concentration reached 286 ppbv, the highest reported ozone concentration in China. The concurrent high concentrations of fine particles and ground-level ozone, especially the interaction between atmospheric oxidation processes and the formation of fine particles is of great concern for air quality improvement in Beijing.

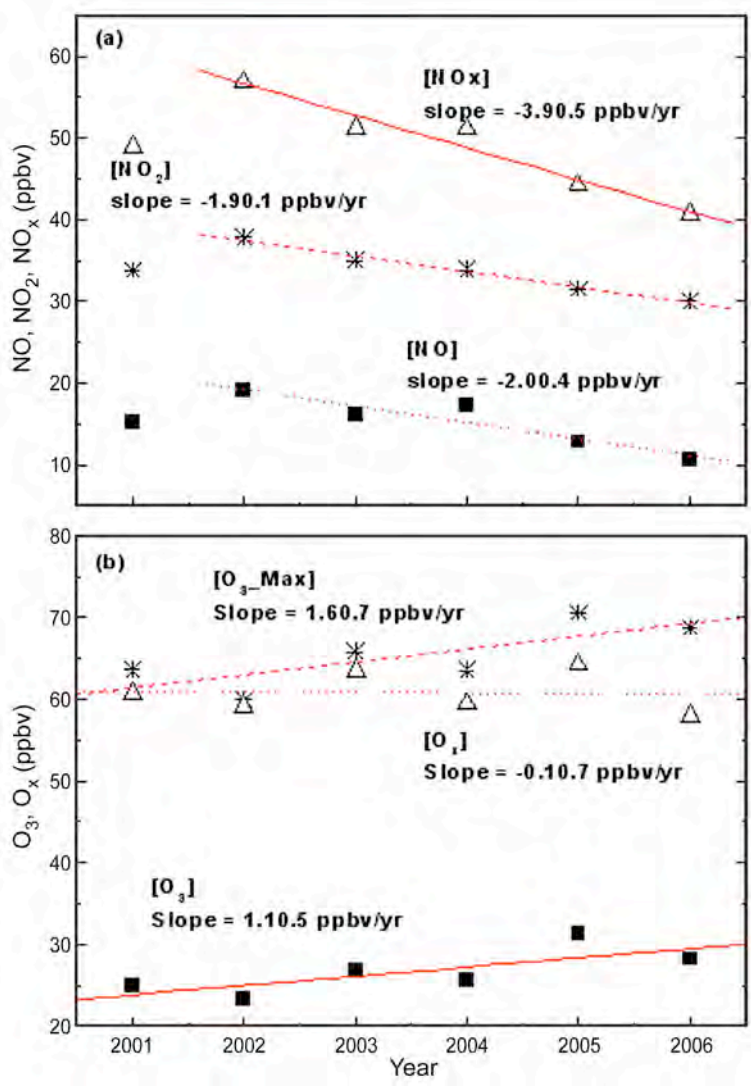

Figure 10 - Concentration trends for (a) NO, NO_2, NO_X, (b) O_3, O_X, 2001–2006. The concentration of each species represents an average of measurements from all six representative stations in Beijing [*Tang et al.,* 2009]

Relationships of the trends to regulations

From 1998 to 2009, Beijing municipal government implemented 15 stages of air pollution control countermeasures (*http://govfile.beijing.gov.cn*). These measures covered a wide range of air pollution controls including the use of low sulphur coal and promotion of cleaner fuels, a ban on high emitter vehicles in the urban fleet and implementation of a vehicle inspection/maintenance system, the prevention of fugitive dust from road and construction sites, and inspection of the operation of emission control devices, amongst others. More stringent control measures including the control in Tianjin city and surrounding provinces were enacted in 2008 for the Olympic and Paralympics Games. The 15th stage measures began in 2009 and includes the use of renewable energy, control of several VOCs emission sources, and emergent actions under extreme weather conditions were required.

The decrease of SO_2 emissions shown in Figure 9 was mainly due to mandatory phase-out of high emission, low efficiency enterprises and facilities, especially in sectors such as electricity, metallurgy, building materials, and chemical industries, as well as due to the use of coal with sulphur content lower than 0.5% and the increase use of liquefied petroleum gas and compressed natural gases in the city.

Beijing is the first city in China to adopt the Euro-IV vehicular emission standards as of March 2008, making Beijing the leader in China in upgrading vehicular emission standards. Beijing has also very stringent controls for already in-use vehicles. Vehicles in the urban fleet with lower than the Euro I emission standard have been banned from circulation since October 2009 and the city buses will be upgraded to electricity/natural gas powered and hybrid buses. Beijing is also the first city in China to implement the gasoline vapour recovery for gasoline stations and gasoline tanks across the entire city.

Climatic change issues

From a study by the National Climate Center and Beijing Municipal Meteorological Bureau, the air temperature in Beijing showed an evident increasing in the last 40 years: the growth rate in urban areas was observed to be 0.43^0C/10a, while in rural areas the rate was 0.21^0C/10a, indicating a clear heat island effect [*Song et al*., 2003]. However, the study on ambient levels and the emissions of greenhouse gases are very limited for Beijing. A 10 years study (1993-2002) done in Beijing by *Liu et al*., [2005] indicates that the ambient CO_2 increased at an annual rate of 0.57%, higher than the annual growth rate obtained at a global background site in China, 0.46% (Waliguan). The ambient N_2O was more or less constant for the first 5 years and then increased very quickly at annual rate of 2.4%. The ambient CH_4 concentration increased from 1993-1998, but changed to decreasing from 1998 to 2002, dropping at 1.33% per year. Black carbon is an important radiative active species and recent work estimated that the black carbon emission in Beijing was 7.77 Gg in 2000 and would be 2.97 Gg in 2008 due to the energy restructuring [*Liu &Shao*, 2007]. No literature was found regarding CO_2 emission inventory in Beijing.

Research projects on air quality of Beijing city

Starting in late 1990s, Beijing city organized a series of air quality research programmes. The first one was launched in 1998, known as the Blue Sky Project, aimed at exploring the relationship between atmospheric visibility and fine particles. The Blue Sky Project was followed by a project from 2000-2003 addressing mainly emission sources, including both primary pollutants such as SO_2, NO_2, CO, and PM_{10} and also the sources for high levels of ozone in Beijing. From 2004-2006 a third project was funded to study secondary air pollution, i.e. the formation of ground-level ozone and fine particles. The research goal was to formulate a strategy for Beijing to be in compliance with the NAAQS of China. From 2006-2008, the project CareBeijing was implemented to investigate the air quality problems with a regional perspective. The most important objective of the CareBeijing project was to propose control measures for 2008 Beijing Olympic Games for both Beijing, Tianjin, and surrounding provinces (Hebei, Shanxi, Inner Mongolia). In May 2009, the Beijing Municipal Government initiated a 3 year project to support long-term control strategies of air quality improvement.

Problems remaining

The success of air quality improvements for Beijing Olympic Games proved that the reduction of precursors of ozone and fine particles were correct in the short-term, but will be tough in long term. *Xu et al.,* [2008] used a 3D air quality model to show that the Beijing urban area was in a VOC-limited regime, while the downwind area changed gradually to a NO_X-controlled process. Furthermore, 35–60% of the ozone during pollution episodes at the Beijing Olympic Stadium was found to come from sources of ozone precursors outside Beijing, e.g. from Hebei Province and Tianjin [*Streets et al*., 2007]. Fine particles have significant origins of both primary emissions and secondary production and therefore are not very well understood yet for Beijing. Furthermore, the grey haze and fine particles issue is known to be a regional problem. Therefore, a regional perspective is required to provide efficient control measures to overcome ozone and fine particle problems. A better understanding of the interaction between the formation of ozone and fine particles, shown in Figure 11, is essential in order to implement effective measures to abate both fine particle and the ground-level ozone pollution in the next 20 years.

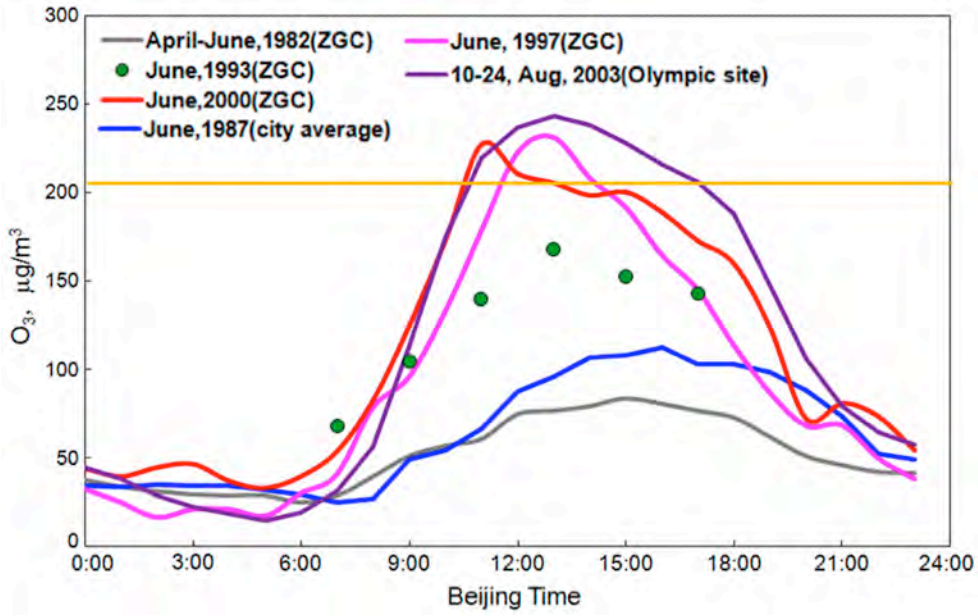

Figure 11 - The change of peak concentrations of ambient ozone levels at Zhongguancun site in Beijing during ozone episodes [*Shao et al*., 2006]

3.4 DELHI, INDIA

Delhi, the capital city of India, hosted the Commonwealth Games in October 2010 (CWG 2010). With the Games, the debate on air quality in Delhi and athletes health during the Games took center stage, similar to the debates on air quality in Beijing before and during the Olympics Games in 2008 [*UNEP*, 2009].

The National Capital Region (NCR) of Delhi has grown rapidly in the past two decades. It now covers an estimated area of 900 km^2, which includes new townships and satellite centers such as Noida, Gurgaon, Ghaziabad, and Faridabad, all of which are a combination of information technology firms and industrial clusters (a graphical representation of the NCR is presented in Figure 12). In 2007, the population of NCR was estimated at 16 million. It is expected to reach 22.5 million in 2025 [*UNHABITAT,* 2008].

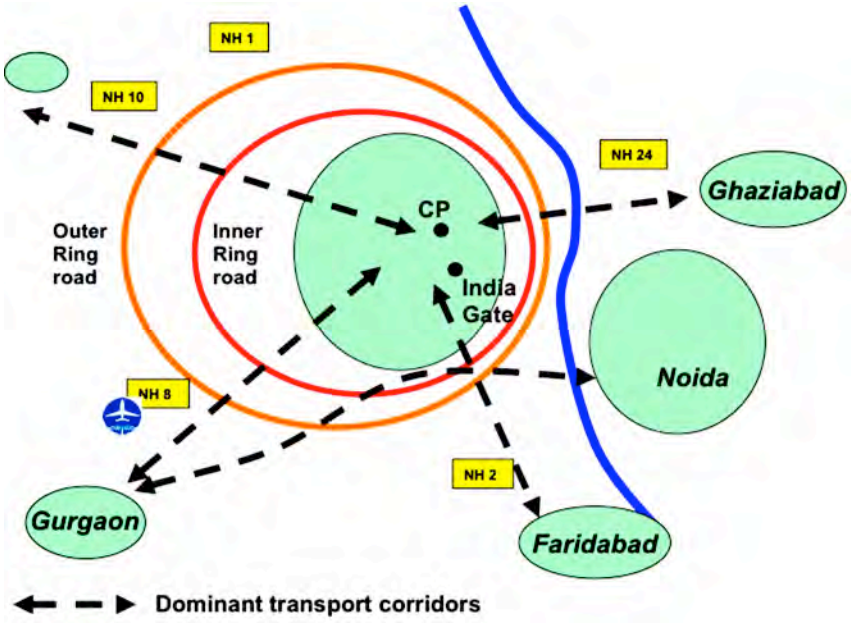

Figure 12 - Graphical representation o the National Capital Region and the travel demand from the satellite cities

Air pollution in Delhi

Delhi is a rapidly expanding city; transportation, energy generation, construction, domestic burning, and industrial activity are contributing to increasing air pollution and its resulting health and respiratory impacts. Figure 13 presents the summary of measured daily averages for the period of 2004-07 for PM_{10} and NO_X concentrations from the four monitoring stations in Delhi. The collection efficiency at each of the stations is ~25 percent [*CPCB, 2010*].

A summary of the PM and ozone pollution observed at one of the continuous air monitoring stations in Delhi (located at the Income Tax Office - ITO) is presented in Figure 14. Data is collected from the period of 2006-09 for the ITO station covering a range of pollutants and meteorology [*CPCB, 2010*]. On an average, the PM pollution exceeded 2-3 times the daily ambient standard of 100 $\mu g/m^3$ for PM_{10} and 60 $\mu g/m^3$ for $PM_{2.5}$ and ozone remained lower than the daily standard of 80 $\mu g/m^3$ but exceeds the 8-hr standard (plotted as thick blue line).

Over the past decade, the government has introduced some green initiatives to address the air pollution problem in the city. In 1998, the Supreme Court ruled that the city of Delhi should take concrete steps to address air pollution in the transport and industrial sectors. The timeline of implementation (in the transport and industrial sector) and the experience for instituting change has become a model for other Indian cities and is described in detail in *Narain et al.* [2005].

For the transport sector, this ruling led to the largest recorded compressed natural gas (CNG) switch in the world for public vehicles. This resulted in a dramatic decrease in the reduction of the air pollution. Since 2000, Delhi has enforced Euro II emission standards (five years ahead of schedule), Euro III standards in 2005 for all passenger vehicles, and Euro IV fuel standards in April, 2010 (in Delhi and 11 other cities).

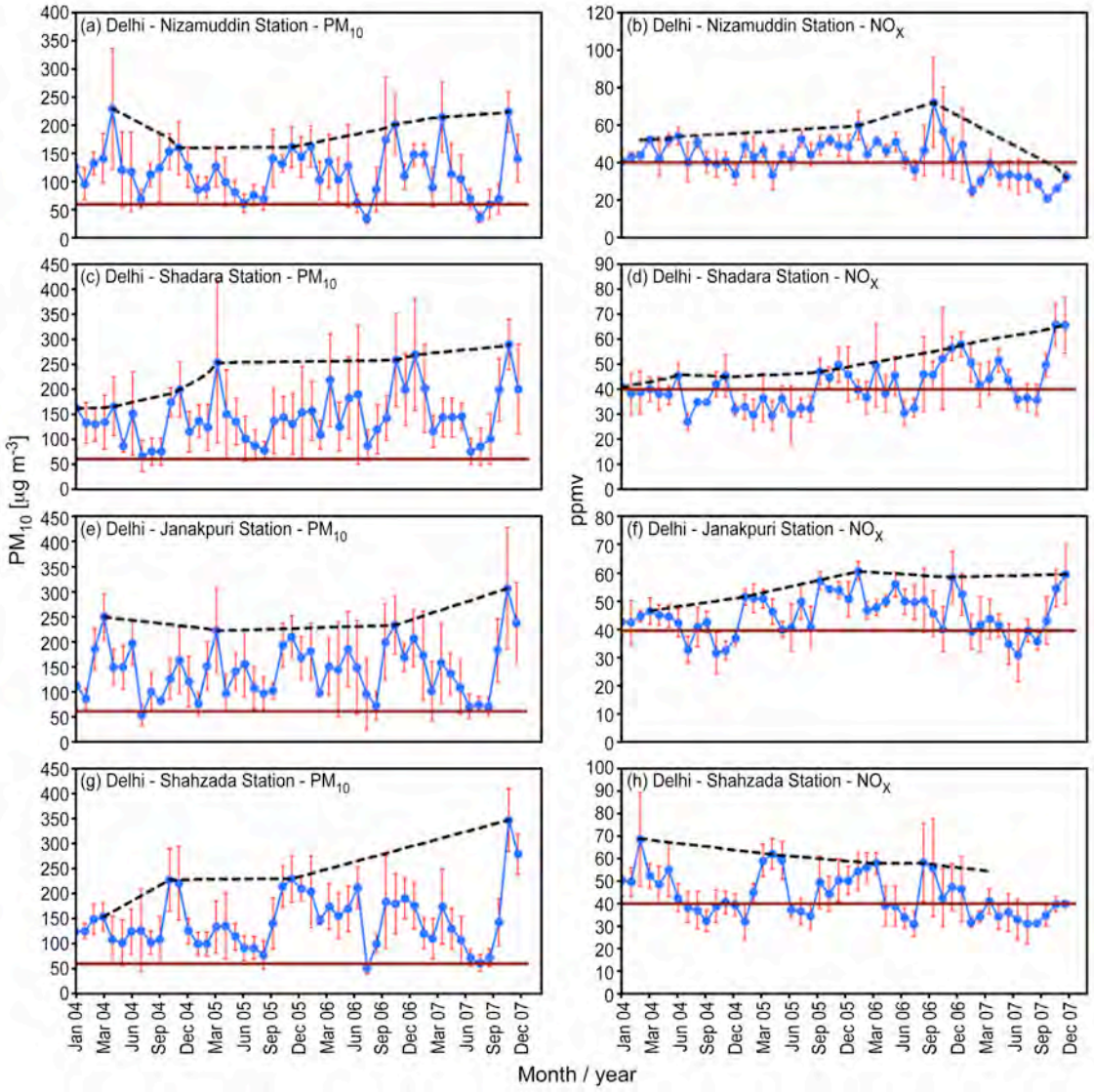

Figure 13 - PM_{10} and NO_X measured at four manual stations in Delhi, India. The error bars indicate one standard deviation for the daily averages over each month; thick line indicates the annual ambient standard

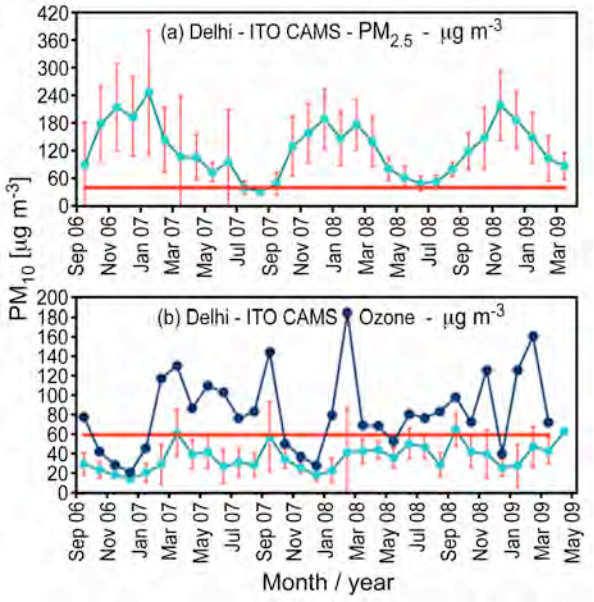

Figure 14 - Daily average measured at a continuous air monitoring station located near ITO in Delhi, India

Another significant fallout of the Supreme Court ruling was in the industrial sector. Approximately 500 heavy industries were shut down and relocated to areas outside the Delhi administrative boundaries. This not only led to a significant drop in air pollution, but also energy efficiency as several relocated industries took the opportunity of the relocation to upgrade their systems.

Yet, there still remains a tremendous amount of potential to reduce air pollution and its impacts as the demand rises for infrastructure and services.

Pollution sources

No single sector is responsible for all of Delhi's air pollution. Rather, it is a combination of factors including industries, power plants, domestic combustion of coal and biomass, and transport (direct vehicle exhaust and indirect road dust) that contribute to air pollution [*Garg et al.,* 2006*; Gurjar et al.,* 2004*; Reddy et al.,* 2002*; Shah et al.,* 2000]. Seasonal changes in demand for fuel and natural pollution result in differing sources during the summer and the winter months. These need to be taken into account to maximize the effectiveness of anti-pollution initiatives. Figure 15 presents the results of particulate matter source apportionment of the urban air pollution in Delhi, conducted by the Georgia Tech University (USA) in 2005.

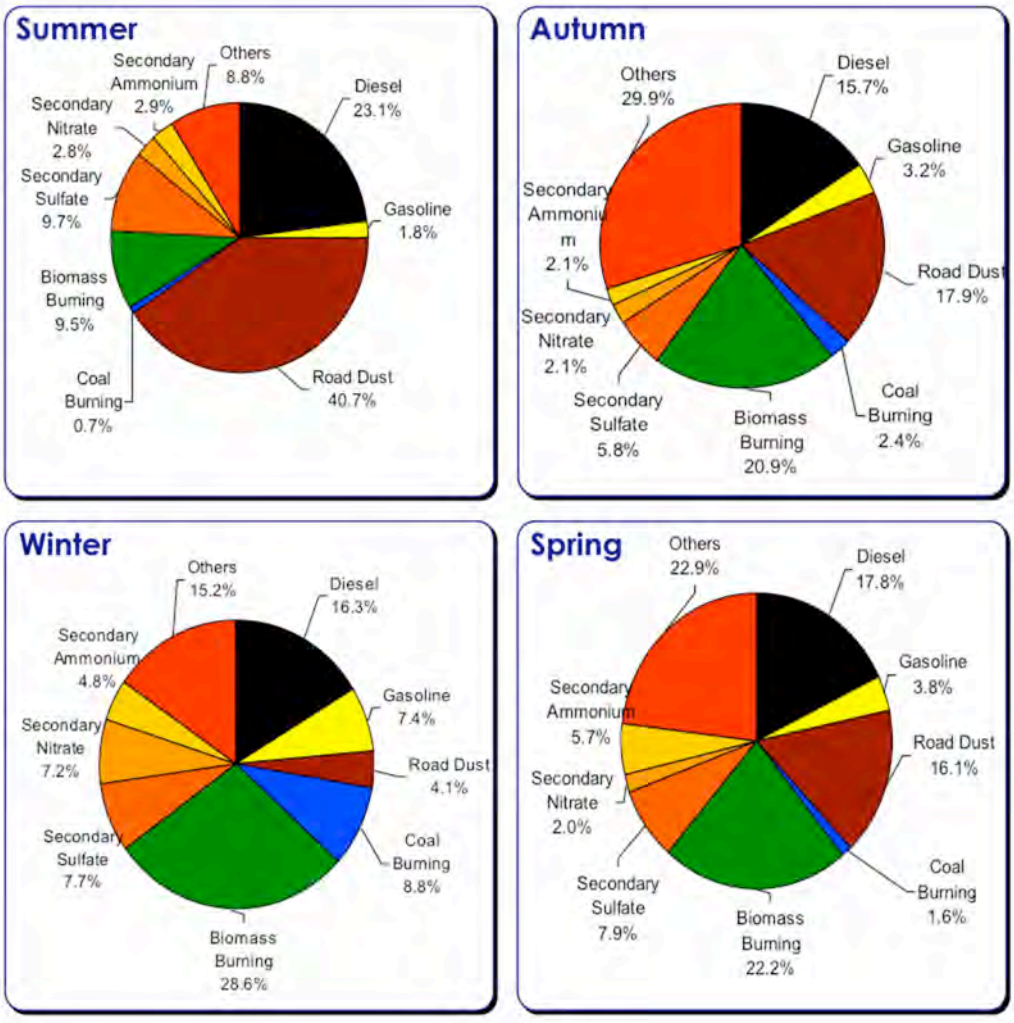

Figure 15 - $PM_{2.5}$ Source apportionment results for Delhi

In summer, in addition to the road dust already present on the Delhi roads, dust storms from the desert to the southwest [*Earth Observatory,* 2008] contribute to increased fugitive dust, which is enhanced by growing vehicular movement. This is exacerbated by the low moisture content in the air, leading to higher resuspension of road dust (40 percent of particulate pollution in summer, compared to 4 percent in winter). In the winter months, the mix of pollution sources changes dramatically. The use of biomass, primarily for heating contributes to as much as 30 percent of particulate pollution in winter. Most of this burning takes place at night, when the "mixing layer height" is low due to inversion. In summer, biomass accounts for only 9 percent of particulate pollution.

Another external factor to air pollution in Delhi is agricultural clearing [*Earth Observatory,* 2008]. After harvesting crops, the land is cleared, a common practice in surrounding (largely agricultural) states. The smoke from clearing crops reaches Delhi and contributes to the smog formation and ozone pollution.

Apart from biomass burning and ambient dust, transportation and industries are major contributors. With a growing city, transportation needs are increasing, creating a rise in private vehicles (2 and 4 wheelers), taxis and auto-rickshaws. Idling time and pollution are corresponding factors that have increased due to increased use of vehicles. The largest gain in the air quality was observed at the peak of the CNG conversions for the public transport buses, taxis, and 3-wheelers in 2001-2002. Air quality levels since have declined gradually over the years in the residential areas and along the major corridors, which is directly linked to the growing passenger fleet, especially diesel based, increasing commuter times leading to idling, and more vehicle kilometres travelled.

The efforts to address this by building flyovers that connect and bypass major junctions in the city have not yielded results as expected. For one, this solution addresses only the supply side of the equation and does not influence demand management. In fact, as it becomes easier to take a private vehicle, the number of vehicles has increased (about 1000 new registrations per day in 2006) thus negating many of the planned improvements. In addition, the increase in the on-street parking and encroachments by hawkers has exacerbated the situation.

Industry, the other major source – accounts for about a fifth of the air pollution in Dehli and includes five power plants at Indraprastha, Faridabad, Badarpur, Pragati, and Rajghat (using a mix of coal and natural gas for electricity generation) and ~3,000 industries ranging from pharmaceutical to metal processing that use coal, fuel oil, and biomass [*CPCB, 1997*].

The growing industrial conglomerations and information technology (IT) parks, under the Special Economic Zone (SEZ) schemes have also led the way in increasing the travel demand.. This leads to a significant change in the geographical settings, the travel behaviour and the mode of transport (transformed to motorized transport), and not only increased vehicle kilometres travelled per day, but also exerting pressure on the limited infrastructure that results in congestion, idling, and pollution (Figure 11). On a daily basis, in and out travel between Delhi and the satellite cities accounts for nearly 30-40 percent of the passenger trips [*CRRI,* 2008].

These satellite cities are also prone to regular power cuts, leading to increasing use of generators for in-situ needs. This includes cinemas, hotels, hospitals, farmhouses, apartment buildings, and institutions. The rapid growth in generator use has consequently led to increased fuel combustion, poor traffic management, and lack of sufficient public transport. This results in deteriorating air quality, increased trip costs, extended commuter times, thus means longer exposure to increasing air pollution and health impacts [*Guttikunda,* 2009a].

Seasonality in pollution

The seasonal variation of the mixing layer height is very prominent in Delhi. During the winter months, the mixing layer height is low which leads to increased air pollutant concentrations. The recurring impacts include heavy persistent smog during the months of November to February, higher pollution levels for all criteria pollutants, frequent delay or cancellation of flights (domestic

and international), and reduced visibility causing minor and major accidents along the roads. This wintertime phenomenon is of utmost importance because it starts forming in the month of October, the start of the CWG 2010 [*Guttikunda,* 2010a].

Correlations for air pollutants like CO, NO, SO_2, and $PM_{2.5}$ measured at the ITO station are presented in Figure 16. Note that the figure indicates measured concentrations and not emissions and most likely indicate the sources in the vicinity of the monitor rather than as a city average. The graphs also provide a distinction between the summer (dark dots) and non-summer months that are linked to the seasonal differences in the mixing layer heights [*Guttikunda, 2009c*].

The correlation between $PM_{2.5}$ and CO concentrations is an indication of direct emissions, most likely transport and fresh plumes from the industrial areas to the East. The CO concentrations are also sourced to the chemical conversion of VOCs via photochemistry and the fraction of the PM also originates from the chemical conversion of SO_2 and NO_X emissions. The fractional analysis of the secondary contributions is not presented.

For the NO_X emissions in the transport sector, the nitric oxide (NO) is close to 90 percent of the emissions and readily oxides to nitrogen dioxide (NO_2) in the presence of sunlight. Figure 16 (top right) indicates a direct emission source with a strong correlation between NO and SO_2, which in this case is linked to diesel combustion from the transportation sector and possibly generators in the vicinity. Lower concentrations of NO in the summer months coincide with the faster oxidation to NO_2 in sunlight. The ozone pollution is higher in the summer months and linked to the presence of VOCs (CO as a proxy in Figure 16, bottom left) and the oxidizing capacity of NO_X, details of which are described in the following section.

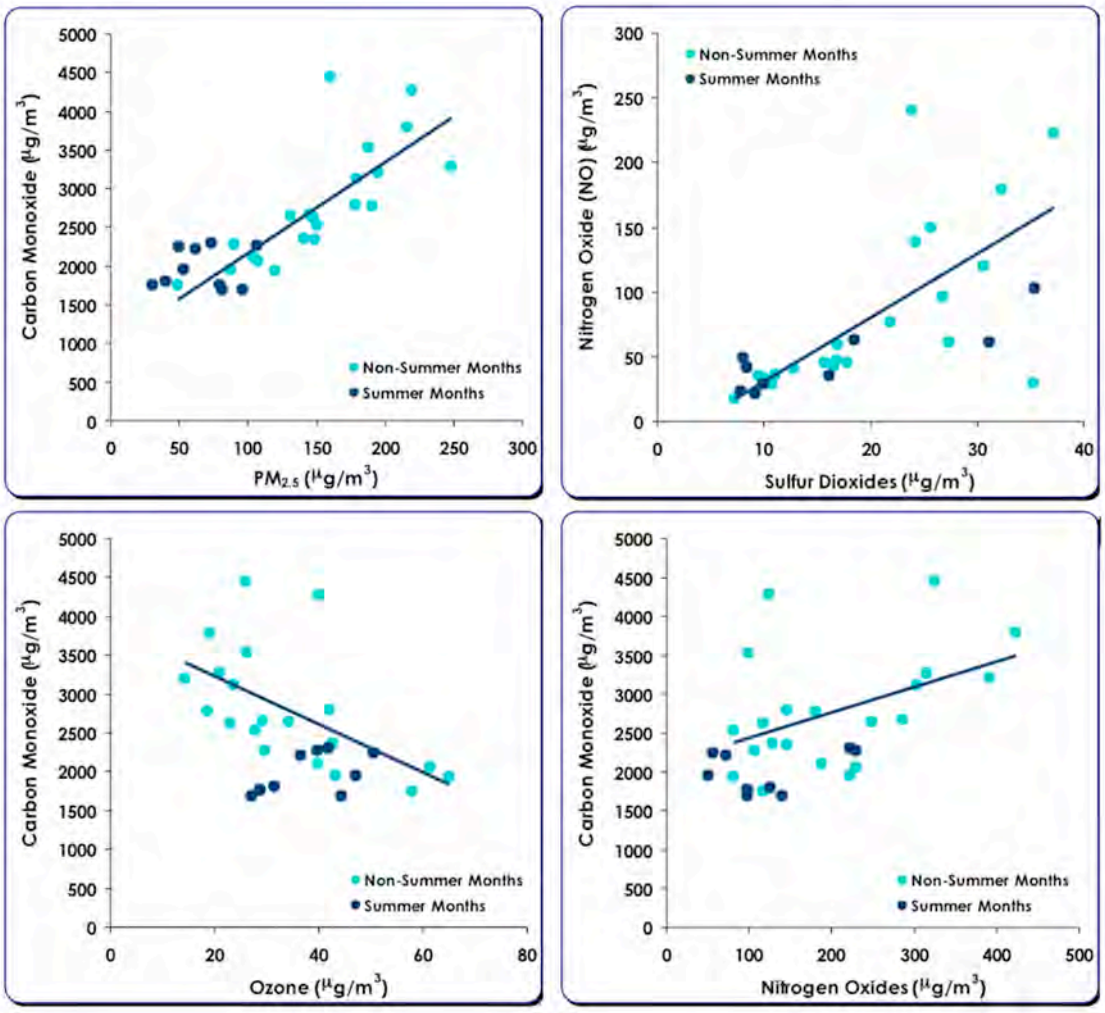

Figure 16 - Correlations between criteria pollutants of measured daily averages (2006-09) at the ITO station in Delhi, India

Tracer model simulations over Delhi, India

A snapshot of the variation in monthly tracer concentrations compared to the annual average over Delhi area for 2008 is presented in Figure 17(a). This is the result of a dispersion model simulation conducted over the Delhi area with constant emissions from all the grid cells and varying only the meteorological conditions based on NCEP Reanalysis data [*Guttikunda,* 2010a].

A clear conclusion is that irrespective of the constant emissions over each month, the observed concentrations are invariably 40% to 80% higher in the winter months (November, December, and January) and 10% to 60% lower in the summer months (May, June, and July) when compared to the annual average tracer concentrations for the emissions domain. The pattern is consistent over the years and the shift is primarily due to the variability in the mixing layer heights and wind speeds between the seasons (and years). During the day, similar patterns are also evident, when the mixing layer height is routinely lower during the nighttime compared to the day, irrespective of the seasons.

Mathematically, this is better illustrated in Figure 17(b) as a box model. By definition, the ambient concentration is defined as mass over volume. Assuming that the emissions are equally mixed in an urban environment under the mixing layer, for the same emissions, a lower mixing height means higher ambient concentrations.

Similar to the mixing layer height, the wind speed is also very relevant. Figure 17(c) presents a summary of the surface layer wind speeds in 2008 from ECMWF for Delhi. The higher wind speeds observed in the summer months are responsible for driving part of the pollution out of the city limits, as evident in the months of June, July, and August (Figure 13) when the predominantly southerly winds move the pollution more north, thus reducing the average contribution of the local emissions. The mixing layer height, presented in Figure 17(d), shows the highs in the spring and summer months for the period of 2007-2008.

It is important to note that while the modelling is conducted using the meteorology pertinent to the city area, the emissions are not. The simulations provide a better understanding of the dispersion of the air pollution within the city. However, the pollution patterns would be best studied using a local emissions inventory, including the contributions of emissions originating outside the city (transboundary pollution). For example, in case of Delhi, a constant traffic between Delhi and its satellite cities in the south (Gurgaon and NOIDA) is a growing emission source along with all the industrial estates in the northeast and northwest sectors.

Back-trajectory analysis of meteorological fields reaching Delhi

The analysis presented in the previous section focused primarily on understanding what happens to the emissions originating in Delhi city limits, but the question of how much is the contribution of emissions outside Delhi to the pollution experienced in Delhi is explored via a back-trajectory analysis.

The back-trajectory analysis was conducted for the year 2008 using the HYSPLIT (Hybrid Single Particle Lagrangian Integrated Trajectory) model, a web-based portal for trajectory analysis, utilizing the NCEP Reanalysis data [*Draxler et al.,* 2003*; Guttikunda,* 2010a]. The trajectories are generated once per day and advected backwards for 24 hours to indicate the meteorological origin of the air parcel. The source point, Delhi, India is assigned at 28.6^{o}N latitude and 77.2^{o}E longitude at 100m above ground level. Each line in Figure 18 indicates a trajectory for one day.

The winter and summer months see a strong influence of northern and northeasterly winds, passing cold temperatures over the city and the resultant low mixing layer heights. The summer and late fall months experience a mix of southern winds, increasing the possibility of pollution entrainment from the southern satellite cities.

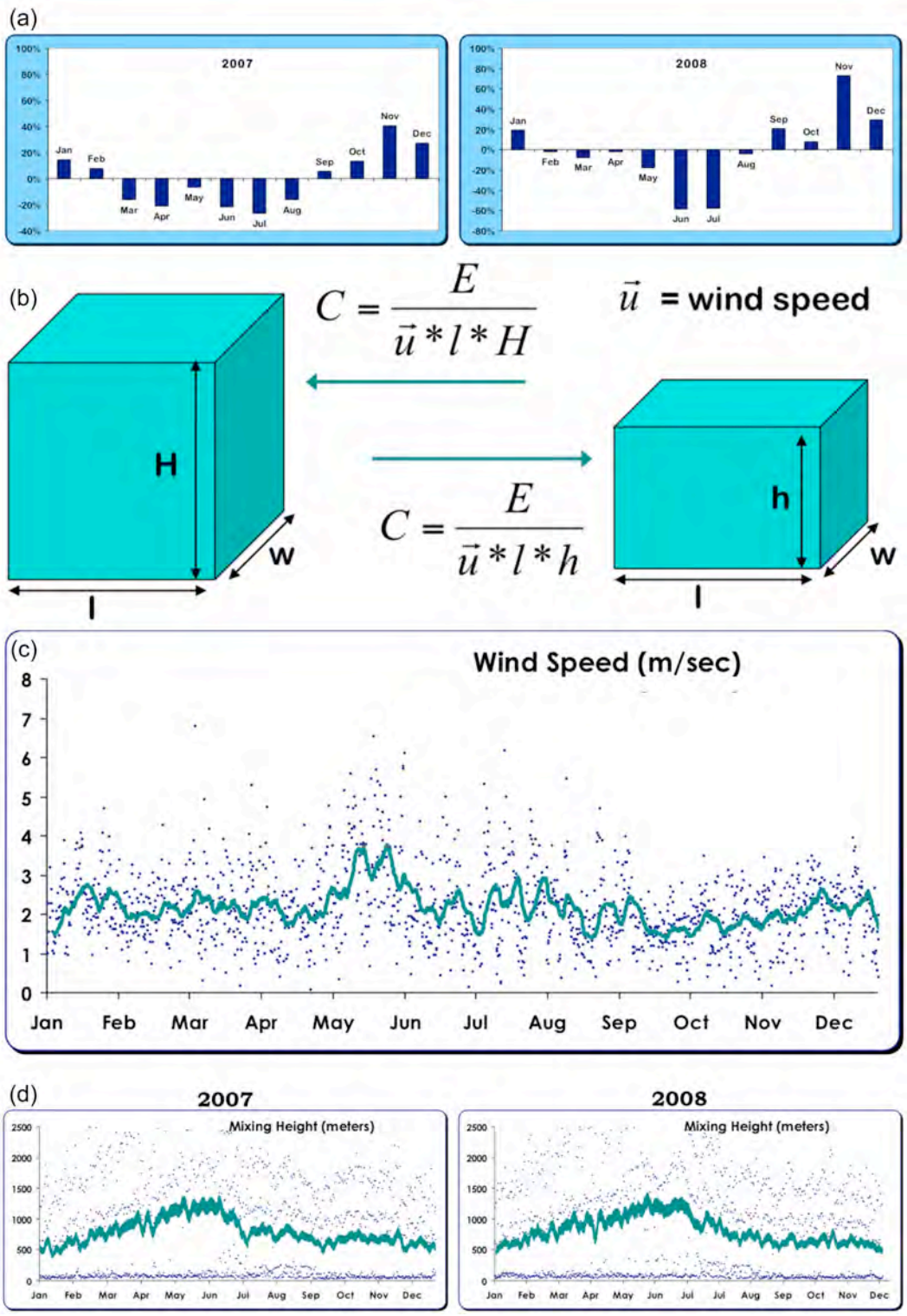

Figure 17 - (a) Variation of monthly averaged tracer concentrations compared to the annual average concentration for the Delhi emission domain; (b) Box model illustration of the impact of the mixing layer height; (c) Figure 6: Wind Speed for Delhi domain, estimated from ECMWF; (d) Mixing layer height in Delhi, estimated from NCEP Reanalysis data

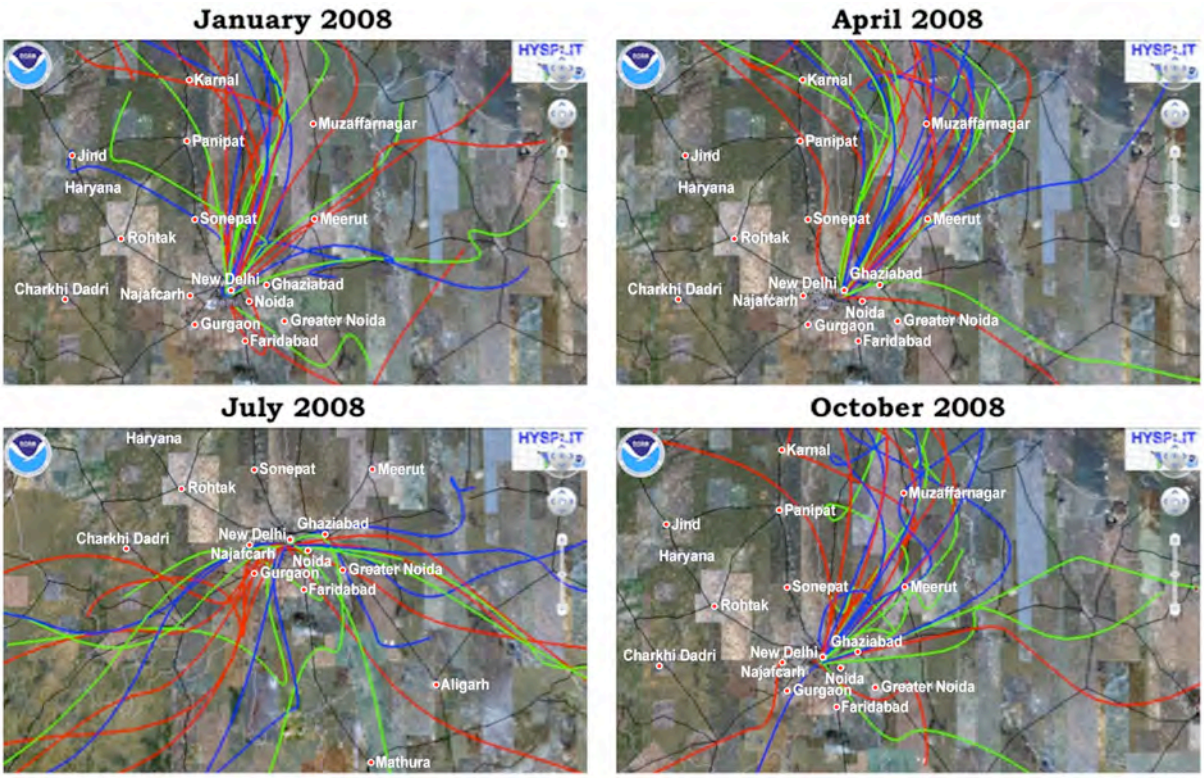

Figure 18 - Back trajectories for Delhi, India for four months in 2008

Diurnal cycles in pollution

In the morning and evening traffic hours, the transport sector is the predominant source of pollution at the ITO monitoring site. Figure 19 shows typical variation of $PM_{2.5}$, Ozone, CO, SO_2, NO_X, and NO_2, over a 24-hour period in 2008. The graphs indicate an average of the measured concentrations for each hour over all days in 2008.

Truck pollution at night

An important observation in Figure 19 is the diurnal variation of the $PM_{2.5}$ pollution. Besides the rush hours bumps (8-10 in the morning and 6-9 in the evening) the steady increase in the pollution levels is attributed to two reasons – a direct source from trucks, which are allowed to pass through the city after 9 PM and a change in the mixing layer height. The influence of the truck emissions is more evident in the direct correlation of the $PM_{2.5}$ and SO_2 cycles, possibly originating from the diesel combustion in trucks.

While the passenger travel in the city has grown over the last decade [*Guttikunda,* 2009b], the importance of the freight transport (via trucks) during the night should not be neglected, since the high concentrations observed during the night tend to linger during the rush hours (mixed with the passenger travel) and beyond (through ~11 AM) and hence increasing the exposure times and related health concerns along the major corridor.

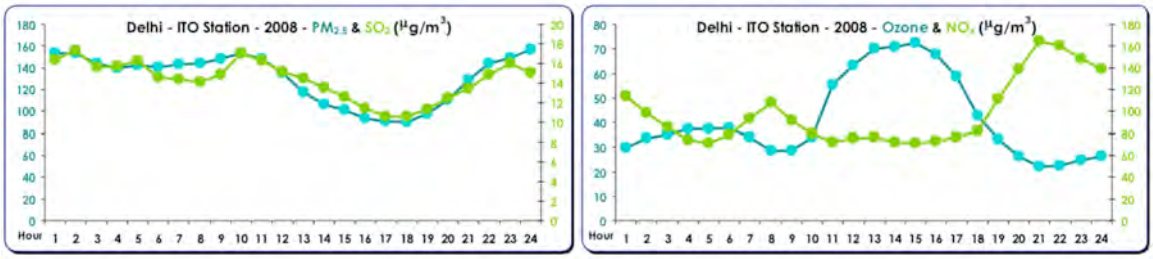

Figure 19 - Diurnal variation of pollution at the ITO station in Delhi, India averaged over all days in 2008

Emissions inventory for Delhi, India

For the area covering the NCR of Delhi an emissions inventory that reflects the trends and sources observed in 2010 is under development, including all key species – PM, SO_2, NO_X, BC, CO, and VOCs, and covering the primary sources – transport (especially passenger travel), industrial clusters, power plants, residential fuel use for cooking and heating (including biomass), generator sets (in households, industries, cinemas, institutions, hospitals, hotels, apartment buildings, and farm houses) and garbage burning (especially for the areas where the waste collection efficiencies are small) [*Guttikunda,* 2010b].

The urban inventory is further segregated spatially (Figure 20) to allow for diurnal and geographical variations among all the sectors. For this inventory, data was collected from many sources – including surveys conducted by local agencies, such as Center for Road Research Institute on traffic density on main corridors, CPCB on the industrial clusters in NCR, and fuel usage for cooking and heating in the residential sector by the project team. The inventory presented in Figure 20 is updated to reflect the consumption patterns in 2010.

A summary of the emissions inventory is presented in Table 4, segregated into major sectors. The largest sectors are transport and power plants. However, due to the release of air pollutants at higher altitudes, the pollution from the power plants is felt less than the low lying emissions from vehicle exhaust, waste burning in the residential areas, low stack emissions from industries, and fuel (fossil and biomass) for residential cooking.

The emissions inventory is maintained in a geo-referenced system to further analyze the vulnerable areas, residential vs. industrial, hot spots for the monitoring air pollution, transport corridors, venue locations and the Games Village (specific product for the 2010 CWG). The model-ready emission inventory also includes diurnal cycles for the transport sector emissions to distinguish between rush and non-rush hours for all modes, operational hours for the industrial sector, and cooking and heating hours for the domestic sector.

This emission inventories are now in use for dispersion modelling, as part of air quality forecasting for Delhi, including scenario analysis for future projections. The “Clean Air for Delhi 2010 and beyond” project is funded by the Government of France under their FASEP bilateral funds; implemented by Aria Technologies SA and Leosphere SA (Paris, France), in technical collaboration with the Central Pollution Control Board (CPCB), Delhi, India. The analysis results will be presented in 2011.

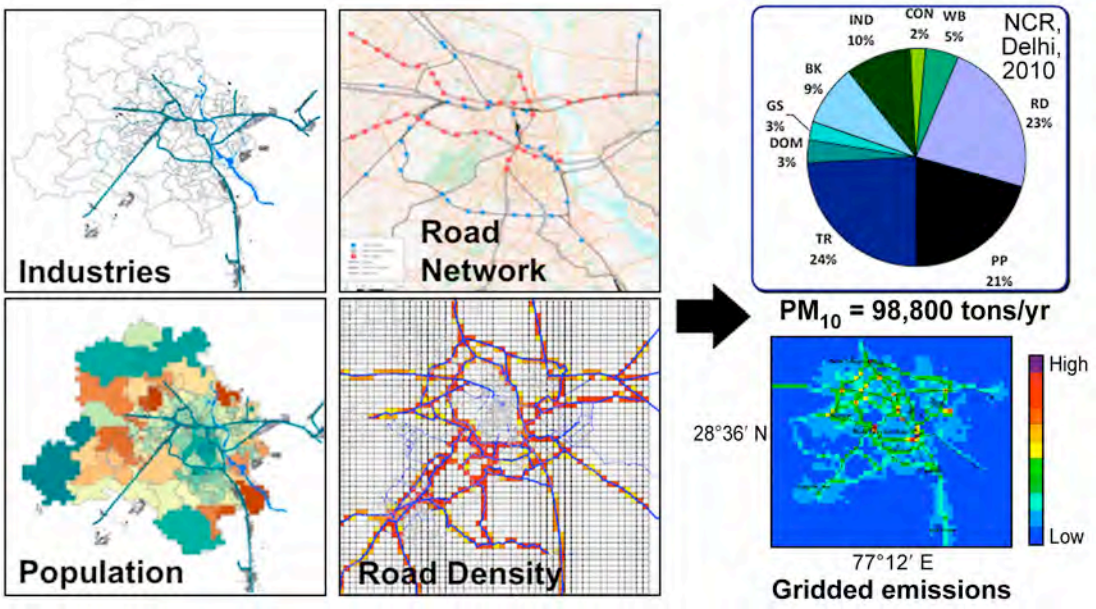

Figure 20 - Distribution mechanisms utilized for urban scale emissions. (IND = industries; PP = power plants; DOM = domestic; TR = transport; RD = road dust; WB = waste burning; CON = construction activities; BK = brick kilns; GS = generator sets)

Table 4 - A summary of the emissions inventory (Tons/yr) for 2010 for NCR of Delhi, India

	PM2.5	PM10	SO2	NOx	CO
Transport	17,900	24,000	900	170,800	323,500
Domestic	2,800	3,150	1,600	2,250	128,550
Diesel Gen Sets	2,100	2,800	700	53,650	56,200
Brick Kilns	4,450	9,050	5,700	10,100	203,550
Industries	4,100	9,600	8,500	41,500	219,600
Construction	750	2,400	50	850	1,100
Waste Burning	3,300	4,750	200	1,250	17,500
Road Dust	3,400	22,650			
Power Plant	12,250	20,400	21,600	9,850	135,600
Total	**51,050**	**98,800**	**39,250**	**290,250**	**1,085,600**

Air Quality Index for Delhi

Applied methodology

The methodology to calculate the air quality index (AQI) is presented below, along with the supporting data for various ranges (Table 5).

$$AQI = \frac{AQI_{hi} - AQI_{lo}}{BP_{hi} - BP_{lo}} * (CONC - BP_{lo}) + AQI_{lo}$$

Where
CONC = concentration of the pollutant
AQI = air quality index for the pollutant
BP_{hi} = the breakpoint that is greater than or equal to CONC
BP_{lo} = the breakpoint that is less than or equal to CONC
AQI_{hi} = the AQI value corresponding to BP_{hi}
AQI_{lo} = the AQI value corresponding to BP_{lo}

The break point concentrations (high and low) are adjusted to the national ambient standards of India for each of the pollutants [*NAAQS,* 2010]. The AQI ranges are adjusted with 150 as the threshold, corresponding to the ambient standard for that pollutant. This is not an official AQI methodology for India, but an attempt to consolidate the available information and put together a reasonable methodology based on applications from across the world.

Applications of the methodology in various in cities across the world and how the information is utilized for public awareness is presented in *Guttikunda* [2010c].

Table 5 - The applicable ranges for AQI methodology for Delhi, India

Range		Healthy		Moderate		Unhealthy1		Unhealthy2		Very Unhealthy		Hazardous	
Values	AQI-low	0		51		101		151		201		301	
	AQI-hi	50		100		150		200		300		500	
Concentrations		BP_{low}	BP_{hi}	BP_{low}	BP_{hi}	BP_{low}	BP_{hi}	BP_{low}	BP_{hi}	BP_{low}	BP_{hi}	BP_{low}	BP_{hi}
PM_{10} ($\mu g/m^3$)		0	40	40	80	80	120	120	200	200	300	300	
$PM_{2.5}$ ($\mu g/m^3$)		0	25	25	50	50	70	70	100	100	150	150	
SO_2 (ppm)		0	0.01	0.01	0.02	0.02	0.04	0.04	0.10	0.01	0.15	0.15	
NO_2 (ppm)		0	0.02	0.02	0.04	0.04	0.06	0.06	0.10	0.10	0.20	0.20	
O_3 (ppm)		0	0.03	0.03	0.06	0.06	0.10	0.10	0.15	0.15	0.25	0.25	
CO (ppm)		0	2	2	7	7	12	12	15	15	30	30	

Air Quality Index results

Using the methodology above, AQI was calculated for the period of August 2006 to June 2010 utilizing the data for two stations across Delhi; (1) Income Tax Office (ITO) and (2) Delhi College of Engineering (DCE), where the PM monitoring data is available. For this analysis, AQI's are calculated in 6 bins, explained in Table 6. The first three bins, at par with the national ambient air quality standards are considered "safe mode" or "clean air" days and the others "polluted" days [*Guttikunda,* 2010d].

Table 6 - The AQI colour codes and their definitions

0-50	"**HEALTHY**" – this range poses little or no risk to the general public. No cautionary actions are prescribed.	51-100	"**MODERATE**" - is acceptable for general public. However, unusually sensitive people should be cautious.
101-150	"**UNHEALTHY**" - is borderline unhealthy, particularly for members of sensitive groups.	151-200	"**UNHEALTHY**" - is considered unhealthy for most of the public where everyone may begin to experience some discomfort.
201-300	"**VERY UNHEALTHY**" - can trigger a health alert, meaning everyone may experience more serious health effects.	>300	"**HAZARDOUS**" – this range triggers health warnings under emergency conditions, affecting all age groups.

Of the six criteria pollutants – PM (coarse and fine), SO_2, NO_X, CO, and Ozone - the PM pollution is routinely above the daily average standards (Daily average national ambient air quality standards for PM_{10} and $PM_{2.5}$ are 100 µg/m^3 and 60 µg/m^3, respectively) and the conditional pollutant for calculating the AQI for health impacts assessments – presenting the worst AQI. Figure 21 presents estimated AQI due to $PM_{2.5}$ pollution at ITO and estimated AQI due to PM_{10} pollution at DCE.

Observations on AQI

- At the monitoring sites, AQI is often more than the healthy levels of 150. The ITO is located at a traffic junction. On 19% of the days the AQI is less than 100 and on 33% of the days less than 150. The DCE is a background station in the north and tends to measure lower than the city averages and yet struggled to stay in the green with only 24% of the days with an AQI less than 150.
- In Figure 21(a) and 21(b), the winter months, which are highlighted with a blue box for each year, show the worst pollution each year starting in October and leading up to February the following year.
- A large portion of the AQI's greater than 300 (summary presented in Figure 21(c)) are associated with the winter season.

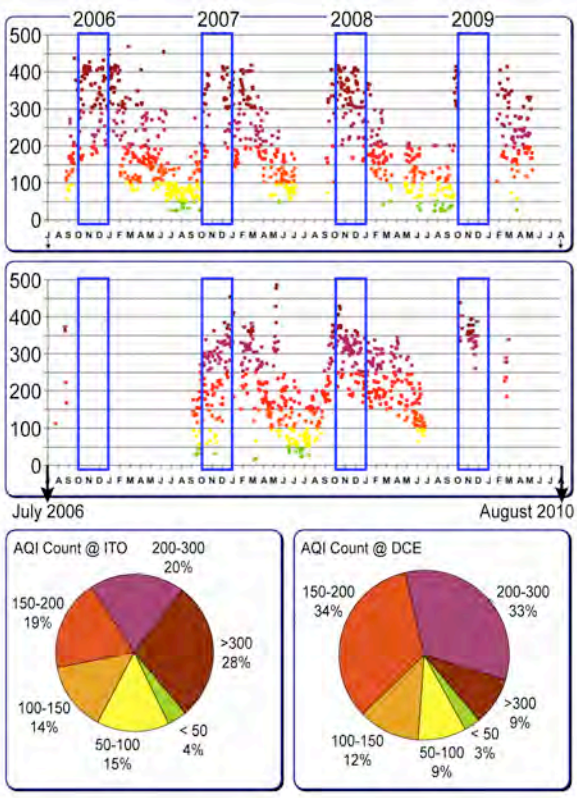

Figure 21 - (a) AQI based on $PM_{2.5}$ measurements at ITO monitoring station; (b) AQI based on PM_{10} measurements at DCE monitoring station; (c) Frequency of AQI between July 2006 and August 2010

Air quality management in Delhi

The air quality in Delhi improved in the early 2000's due to a number of interventions, including the large-scale conversion of the bus fleet and the three wheel fleet from the conventional gasoline and diesel to CNG (The benefits of CNG conversion in Delhi on global climate and local pollution are summarized in *Reynolds et al.* [2008]). However, the large increase in the demand for personal transport and construction activities reversed these trends in the last 3 years.

A major intervention that Delhi is banking on is the extension of the metro rail system in order to shift the motorized transport trends to the metro rail system. The expected level of the shift is uncertain and depends on a number of factors. A "what-if" analysis revealed a potential reduction of 20-45 percent in the criteria pollutant emissions due to completion of the metro rail system in 2010. This is also very consistent with other cities like Mumbai, Shanghai, Beijing, Bangkok, and Hong Kong, which experienced significant changes in the air quality after the expansion of the public transport systems with a metro system. Certainly, the challenge will be public awareness, promotions, and incentive schemes for the public to use metro rail system more frequently than their personal mode of transport.

In the transport sector, the emphasis is on the public transport. The JNNURM funds for buses and urban transport strategy of India are promoting the need for infrastructure for new buses (via two of the largest manufacturing firms - Tata and Ashok Leyland). A good public transport system, including substantial support for non-motorized transport (NMT), is expected to help reduce the congestion levels, energy demand, and thus emissions from the transport sector. However, the initial phase via the introduction of the Rapid Bus Transport for ~5km and promoting NMT along the path has had a difficult time gaining traction in Delhi.

In the short term

In order to improve air quality during the 2010 Commonwealth Games, Delhi implemented lessons learned from when Beijing hosted the Olympics in 2008. The lesson focused on the local air quality in order to bring the pollution to a manageable level, fast and efficiently. Some of the lessons learned from Beijing that were suggested to Delhi include:

- Improve the number of air quality monitors operating in the city. There are more monitoring stations than those that can actually deliver monitoring data. Even if we consider the most important pollutants, such as PM, and not worry about the other pollutants (in order to keep the costs low), the stations that measure this are limited.
- More than operations, the data should be made public as frequently and consistently as possible so the general public is aware of the consequences of their actions.
- Better understanding of emission source contributions of both within and outside the city. Currently, a lot of emphasis is put on the transport sector. However, the contributions from the industrial, power, and residential sources are very significant and in October (and in the winter months), with low inversions, these low lying residential sources (via biomass burning) will hinder the "clean air" goal.
- The hot spots of industrial and residential areas (many of them outside the Delhi area, but included in the National Capital Region) should be monitored to manage the emissions in real time.
- In the case of Beijing, stringent regulations and policy measures were implemented months in advance to ensure clean air days before and during the 2008 Olympic Games. However, this remains a challenge for the Delhi authorities.

During the Commonwealth Games, the Delhi Government was unable to shutdown industries nor stop half of the vehicle fleet, making it more difficult to improve the air quality. Therefore, some innovative interventions were needed in order to implement fast and effective air quality management. Some of the innovative interventions were:

- Shut down part of the industries, depending on the meteorological and air quality forecasts (either daily or weekly).

- Strict restrictions on garbage burning during the winter months, especially the open burning for heating purposes in the residential areas.
- One-way transport along the major corridors for better flow of the traffic. Some corridors be dedicated for the movement of the athletes, but similar provisions should be made for passenger travel.
- Aggressive procurement of buses and incentives to promote the use of bus and metro rail systems.
- Promote telecommuting where possible, especially along the satellite cities like NOIDA and Gurgaon, which experiences the largest rush hour traffic during the week days
- Promote wet sweeping of the all the major roads, at least once every two days to reduce the amount of dust loading and thus reduce the resuspension due to increasing vehicular movement.

In the long run Delhi should implement some of the following air pollution management strategies:

- The air quality monitoring network needs serious improvement in all respects – the number of operational monitors, placement of the monitors across the city, dissemination of the monitoring data to the public and the media, access to the archives of monitoring data, and provisions for further analysis.
- In general, there is greater understanding of the pollution sources in the city now than ever before. There is also a greater awareness among the public on the harmful impacts of growing air pollution. Systematic programmes should be introduced in all sectors, starting with tackling low lying sources, such as road dust and residential open waste burning to reduce the daily pollution levels.
- For power plants, coal was slowly being replaced with natural gas, at least in the smaller power plants, in order to reduce pollution during the 2010 Commonwealth Games. This intervention should be part of the long-term strategy.
- In the transport sector, the public transport (via buses) and the NMT should be promoted equally, along with better traffic management for the passenger cars.

3.5 DHAKA, BANGLADESH

Introduction and specific features of Dhaka, Bangladesh

Bangladesh is situated in the eastern part of South Asia. The country is surrounded by India to the west, the north, and the northeast, Myanmar to the southeast, and the Bay of Bengal on the south (Figure 22). Bangladesh is one of the most densely populated regions of the world. Meteorologically, the year can be divided into four seasons [*Salam et al.*, 2003; *Azad and Kitada*, 1996] pre-monsoon (March - May), monsoon (June - September), post monsoon (October - November), and winter (December - February). The average temperatures vary between 7^{o}C and 25^{o}C in winter and high values between 25^{o}C and 38^{o}C in summer. Dhaka (23º76´N, 90º38´ E, and 8 m above sea level) is the capital of Bangladesh (Figure 22). Dhaka is a rapidly growing megacity with a population of approximately 13 million. Dhaka is the center for commercial, political, and cultural activities in Bangladesh. With the modernization of its transport, communication, public works sectors as well as industries, the capital city Dhaka is facing severe air pollution challenges.

Emission sources of air pollutants

There are many emission sources for air pollutants in Dhaka, e.g. large number of motor vehicles, construction activities (road and building), industry, brick fields, etc. Natural gas burning for cooking and long-range transport also contributed to the air pollution in Dhaka. *Begum et al*. [2004] sampled fine and coarse fractions of ambient particulate matter (PM) in Dhaka between June 2001 and June 2002. Six and seven different factors of elemental compositions for coarse and fine PM fractions were identified with positive matrix factorization (PMF) technique. The sources are soil dust, road dust, cement, sea salt, motor vehicles, and biomass burning. A large fraction, more than 50%, of the $PM_{2.5–10}$ mass came from soil dust and road dust. Motor vehicles, including two strokes, contributed about 48% of the $PM_{2.5}$ mass in Dhaka.

Particulate matter (PM) is the main pollutant of concern in Dhaka, Bangladesh, especially during the winter months. There have been several studies to characterize atmospheric pollution in Dhaka. Data are available for particulate matter, heavy metals, and trace gases [*Salam et al*., 2008 and 2003; *Mahmud*, 2008; *Bilkis et al*., 2008 and 2004; *Nasiruddin*, 2006 - DoE, Government of Republic of Bangladesh].

Salam et al. [2008] studied aerosol particulate matter (SPM, PM_{10} and $PM_{2.5}$) and trace gases (SO_2, NO_2, CO and O_3) in Dhaka between January and April 2006. The total average concentrations of SPM, PM_{10} and $PM_{2.5}$ were 263, 75.5, and 66.2 μgm^{-3}, respectively. The mass of $PM_{2.5}$ is approximately 88% of the PM_{10} mass, indicating fossil fuels as the main source of particulate matter in Dhaka. The total average concentrations of SO_2, NO_2, CO, and O_3 were 48.3, 21.0, 166.0 and 28 μgm^{-3}, respectively. The total average concentrations of As, Cd, Cu, Fe, Pb, and Zn in $PM_{2.5}$ were 6.3, 13, 94, 433, 204, and 381 ngm^{-3}, respectively. The measured concentrations for trace gases and metals were much lower than the ambient standard values for Bangladesh.

Begum et al. [2008] measured particulate matter (PM) in Dhaka (Farmgate), which is a hot spot (HSD) with very high pollutant concentrations due to its proximity to major roadways. Fine particulate matter concentrations at HSD have decreased over this period (from January 2000 to March 2006) to less than half of the initial value, even with an increasing number of vehicles. This decrease is likely the result of governmental policy interventions such as the requirement of vehicle maintenance, training of repair workers, and phase-wise removal of two-stroke three wheelers from the roads in Dhaka. Other policy interventions include the banning of old buses and trucks from operating in Dhaka, promotion of the compressed natural gas, introduction of pollution control devices on vehicles, control of emissions from industries, etc.

Salam et al. [2003] studied aerosol chemical composition of atmospheric aerosol particles in Dhaka under pre-monsoon conditions (March–April 2001). The elemental carbon (EC), organic carbon (OC), organic acids, major inorganic ions, and trace elements were measured in TSP. The reconstructed average particulate mass (TSP) was 516 μgm^{-3}. On average about 76% of the aerosol is from soil-type materials, around 18% from carbonaceous material, and around 6% are soluble ions, and trace elements (without iron) are about 0.3%. Coal fly ash is likely a main source for Cd, Pb, and Zn in Dhaka aerosol, while as appears to be of geogenic origin. High concentration of elemental carbon and organic carbon concentrations were observed in Dhaka, e.g. about 22 μgm^{-3} of EC and 45 μgm^{-3} of OC. The correlation between EC and OC was quite high (0.81) indicating a potential joint source of emission for carbonaceous aerosols. The EC/total carbon (TC) and K/EC ratios indicated that biomass combustion was not a main contributor to EC in Dhaka, which implicates that the fossil fuel combustion is the major contributor to EC levels in Dhaka aerosol.

Nasiruddin [2006] reported the variation of the particulate matter (TSP, PM_{10}, $PM_{2.5}$), and trace gases (SO_2 and NO_2) from a continuous monitoring station on the premises of the national Parliament Building, Dhaka. Annual average PM concentrations (PM_{10} and $PM_{2.5}$) in the city of Dhaka showed a slight increasing tendency from April 2002 to July 2006 (2002 data is average of concentrations from April to December and 2006 data is average of concentrations from January to July 2006). Both PM_{10} and $PM_{2.5}$ concentrations exhibit more than twice the national standards of annual PM_{10} (50 $\mu g\ m^{-3}$) and $PM_{2.5}$ (15 $\mu g\ m^{-3}$). Annual average concentration of SO_2 in 2003 (6.67 ppb) is within the national ambient air quality standard of 30 ppbv. The annual average concentration of NO_2 in 2003 was 27.6 ppb, which is also bellow the annual ambient standards value of Bangladesh of 53ppb. *Khaliquzzaman* [1998] reported the SO_2 concentrations were from 64 to 143 $\mu g\ m^{-3}$ at Dhaka (Tezgaon), where as NO_2 concentrations were between 25 and 32 $\mu g\ m^{-3}$ at Dhaka (Farmgate) in 1996.

The status and trend of the pollution

The overall air quality situation in Dhaka, Bangladesh is improving day by day due to the awareness among the city dwellers and also due to control measures implemented by the government of Bangladesh. The major achievements in improving air quality in Dhaka, Bangladesh

is due the ban of leaded gasoline, introduction of lubricant standards, switching to CNG fuel, and banning of baby taxis, old trucks, and buses. For example, the lead concentration in the Dhaka city air (493ngm^{-3}) was the highest in the world during the dry season in 1996, which falls down during the periods of medium and heavy rainfall. However, the lead concentration in Dhaka air is decreasing [*Salam et al.* 2008] due to the ban of leaded fuel. *Core* [1998] also reported the measurement of the Department of Environment, Bangladesh for suspended particulate matter (SPM) concentrations at several locations for eight hours along busy roads of Dhaka city between 1996 and 1997. Results showed that the SPM concentrations of 665-2456 μg m^{-3} at a hot spot in Dhaka (Farmagte). The SPM trends are highest during the dry season (December-March) due to an increase in roadway dust, dust from dust-carrying vehicles, and also increased open burning. *Salam et al*. [2003] measurements show that the SPM concentrations were also decreasing remarkably.

Relationships of the trends to regulations

The Ministry of Environment and Forestry is the key institution for air quality monitoring and management in Dhaka, Bangladesh. The primary legislation instituted to mitigate air pollution problems is from the Department of Environment (DoE), Government of Bangladesh. They have adapted the 1995 Bangladesh Environmental Conservation Act and the 1997 Environmental Conservation Rules (ECR) to control air pollution (DOE 1997, DOE 2002).

Climatic change issues

There have been several socioeconomic studies done for climate change issues. The Government of Bangladesh as well as the Department of Environment, educational institutes, and research organizations are aware of the climate change issue. No literature values could be found regarding CO_2 and other trace gas emissions in Dhaka.

Research projects on air quality of Dhaka city

Academic institutions like the Department of Chemistry, Dhaka University, Bangladesh University of Engineering and Technology, and Jahangirnagar University are researching various issues in order to understand air pollution in Dhaka. The AQMP Project (BoE, the Government Republic of Bangladesh) has long-term measurements for PM and trace gases in Dhaka. Organizations such as the Forum of Environmental Journalists, Bangladesh Paribesh Andolon, Bangladesh Environmental Lawyers' Association, and the Society for Urban Environment Protection help raise awareness among the public on environmental issues including air pollution through conferences, reports, and ad campaigns. There is certainly a need for long-term problem research of air pollution in Dhaka, Bangladesh.

Problems remaining

Dhaka still needs to assess the status of the air quality more completely, as well as monitor the impact of reduction measures in emissions. The lack of information on the positive impacts of measures implemented by the government may undermine the efforts to improve air quality in the country further. However, there is a need to conduct further studies to understand the impacts of air pollution on climate and health as well as to guide policy.

3.6 HONG KONG, CHINA

Introduction and specific features of the city

The Hong Kong Special Administrative Region (HKSAR; 22º15'N 114º10'E) is located at the eastern bank of the Pearl River Delta facing the South China Sea [*GovHK*, 2009]. Covering a land area of 1104 km^2, the territory is composed of New Territories that adjoin Mainland China, Kowloon Peninsula, Hong Kong Island, Lantau Island, and 262 outlying islands (Figure 23). Due to the mountainous terrains, less than 25% of the land area is developed, whist ~40% of the remaining landmass is designated as country parks and nature reserves. HKSAR experiences a subtropical, monsoonal climate, with a hot (mean air temperature: ~26ºC), wet (80% of the 3066 mm annual rainfall) summer influenced by the southwest monsoon, when typhoons are most common, and a

cool (mean air temperature: ~15ºC), dry winter when the northeast monsoon prevails (http://www.hko.gov.hk/wxinfo/climat/climahk.htm).

HKSAR has a population of ~6.98 million. Being an international trade and financial centre, HKSAR experiences a fast growing economy. The GDP has increased by >170% since 1990, reaching HK$1620 billion in 2007. During 1990 - 2007, the energy consumption rate has increased ~20% (i.e. 290000 TJ in 2007) and the vehicle-kilometres travelled (VKT) increased by ~50% (Figure 24).

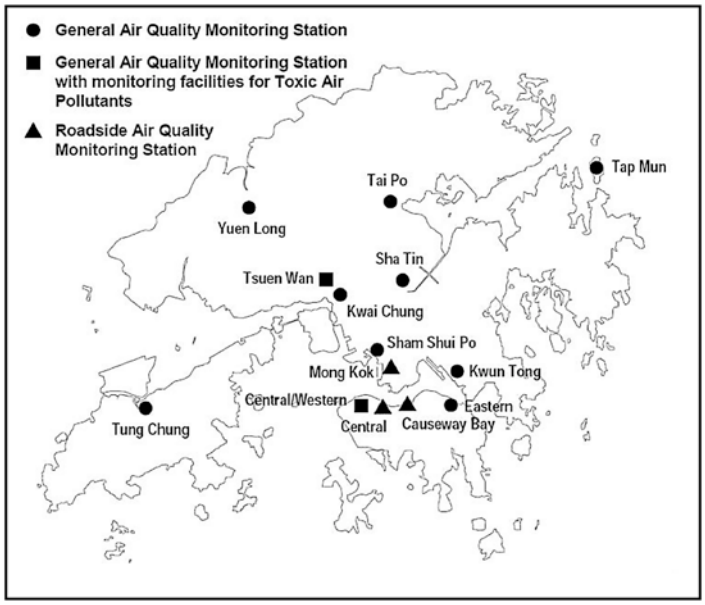

Figure 23 - Locations of EPD's Air Quality Monitoring Stations

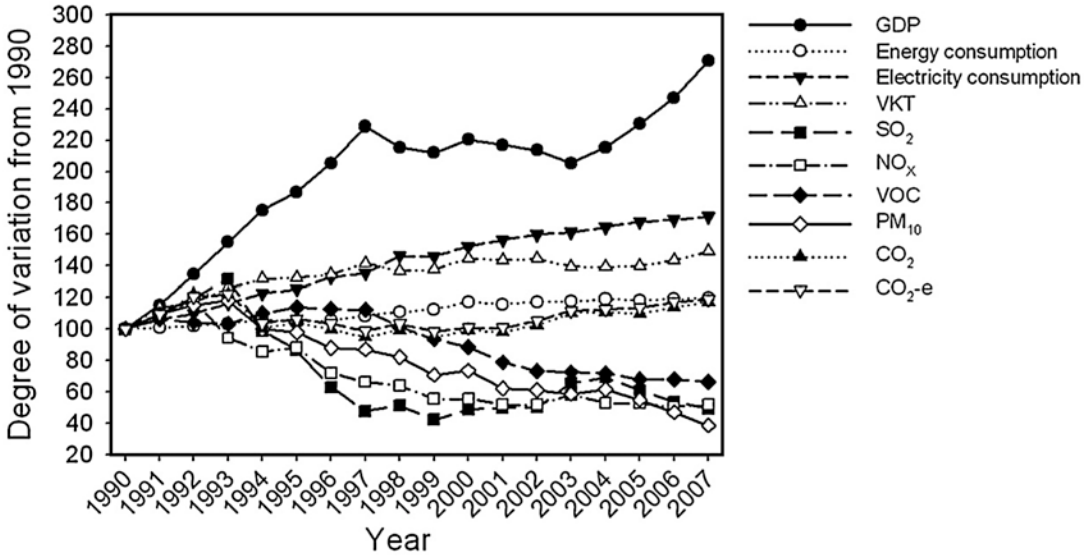

Figure 24 - Trends of gross domestic product (GDP), energy and electricity consumption, vehicle kilometres travelled (VKT), and atmospheric emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_X), volatile organic compound (VOC), respirable suspended particulates (PM_{10}), carbon dioxide (CO_2) and carbon dioxide equivalent (CO_2-e) in HKSAR in 1990 – 2007 (1990 = 100). Note: Energy consumption stands for energy consumed via town gas, liquefied petroleum gas, oil and coal products, and electricity. CO_2-e is a measure used to compare the emissions from various greenhouse gases (e.g. carbon dioxide, methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride (SF_6)) based upon their global warming potential

Availability of air pollutant data

Owing to the potential negative human health impacts, air quality is one of the most pressing environmental topics in HKSAR [*Leverett et al*., 2007; *Civic Exchange*, 2008]. To better understand the patterns, sources, and origins of air pollutants in HKSAR, as well as to report Air Pollutant Index (API) to the public, the Environmental Protection Department (EPD) has developed a fixed network of ambient air quality monitoring stations (AQMS) to measure the concentrations of nitrogen oxides (NO_X), respirable suspended particulates (PM_{10}), carbon monoxide (CO), sulphur dioxide (SO_2), and ozone (O_3) (Figure 23). This 14-station network involves 11 general and three roadside stations. Two toxic air pollutant (TAP) monitoring stations are co-located with Tsuen Wan and Central/Western AQMS to measure total polychlorinated biphenyls, polychlorinated dibenzo-p-dioxins/dibenzofurans, polycyclic aromatic hydrocarbons, volatile organic compounds (VOC), carbonyls and hexavalent chromium [*Lau et al*., 2007]. In addition, three stations (Tsuen Wan, Tung Chung, and Tap Mun) are incorporated with another 13 regional stations to form the 16-station Pearl River Delta (PRD) regional air quality monitoring network, which reports the PRD regional air quality index (RAQI) to the public [*Zhong et al.,* 2007]. Hourly, daily, and annual data are posted to relevant websites for data sharing with the public and academic communities (e.g. http://www.epd.gov.hk/epd/english/environmentinhk/air/data/air_data.html and http://www.epd.gov.hk/epd/english/ environmentinhk/air/air_quality/air_quality.html).

Emission sources of air pollutants and emission inventory

HKSAR is currently facing serious particulate and photochemical smog problems (http://www.epd.gov.hk/epd/english/environmentinhk/air/air_maincontent.html). One of the major emerging particulate problems is fine suspended particulates ($PM_{2.5}$), which comprises ~70% of PM_{10} concentration, and, are emitted from a number of sources, including vehicle exhaust, electricity generation, navigation, other fuel combustion, road dust, etc. See *Louie et al*. [2005a and b]; *Huang et al*. [2009]; *Yuan et al*. [2006b and c] for PM_{10} source apportionment information. Photochemical smog is produced from a photochemical reaction between nitrogen oxides (NO_X) and volatile organic compounds (VOCs). In HKSAR, NO_X are mainly produced by electricity generation, vehicle exhaust and shipping (http://www.epd.gov.hk/epd/english/environmentinhk/air/data/emission_inve.html), while VOCs are mainly emitted from consumer products, paint application and printing process [*Guo et al*., 2004a and b, 2007].

The current status and air pollution trends

The particulate and photochemical smog problems in HKSAR have severely impaired local visibility since the early 1990s [*Wang*, 2003] Intensification of the particulate problem is likely from ambient concentration of SO_2 and NO_X, but the regional influence should not be neglected. (Figure 25) Whilst SO_2 is predominantly originates from regional industrial sources [*Wang*, 2003; *Louie et al*., 2005b] and shows similar patterns amongst the fixed network monitoring stations, NO_X and carbonaceous matters which explain ~50% of $PM_{2.5}$ mass are emitted mainly from local vehicle exhaust and is more concentrate in urban than in rural sites [*Yu et al*., 2004; *Louie et al*., 2005a; *Lee et al*., 2006]. *So et al*. [2007] compared $PM_{2.5}$ levels in HKSAR between 2000/2001 to 2004/2005 in two differente $PM_{2.5}$ speciation campaigns that suggest over 36% growth of ambient sulphate concentrations most likely resulted from an increase in regional sulphate pollution.

Intensification of the regional photochemical smog problem, on the other hand, has resulted in the increase in O_3 concentration in HKSAR (Figure 26). Similar results were obtained by *Wang et al*. [2009b], in which the ozone concentration increased by an averaged rate of 0.55 ppbv/yr in 1994 – 2007. *Wang et al*. [2009b] suggested that this rising O_3 trend is predominantly due to the increase in nitrogen dioxide (NO_2) and probably VOC emitted from regional sources. Under the rapid urbanization and industrialization of the PRD region in recent decades, this trend will probably persist for the foreseeable future [*Louie et al*., 2008]. The box-plots of ozone trends as observed in different sites in Hong Kong as depicted in Figure 27, show the regional nature of the photochemical ozone problem. Amongst the four fixed network monitoring stations, O_3 concentrations were higher in rural than in urban sites (Figures 26 & 27). Nitrogen monoxide (NO) from vehicle exhaust would remove some ambient O_3 (http://www.info.gov.hk/gia/general/200411/24/1124240.htm), while

biogenic emission (~10% in Tap Mun, Yuan et al. 2006a) would enhance O_3 formation. These two processes explain the higher O_3 concentration in rural sites.

Visibility impairment is usually more severe during the winter than in the summer [*Wang*, 2003]. During winter, the northeast monsoon facilitates long-range transport of regional pollutants to HKSAR [*Louie et al*., 2005b; *Guo et al*., 2009]. Land-sea breeze circulation would further trap and accumulate air pollutants in the PRD region [*Lo et al*., 2006; 2007], resulting in higher air pollutant concentrations. Winter accounts for ~80% of high PM days [Huang et al., 2009] but O_3 concentration is highest in autumn when more sunlight is available for photochemical formation of this pollutant. The southwest monsoon in summer, in contrast, brings in fresh air to HKSAR and the high rainfall help washing away air pollutants in the atmosphere [*Wang*, 2003; *Louie et al*., 2005b]. Particulate and photochemical smog problems in summer usually occur with typhoon episodes when the outskirt subsidence of airflow restricts the dilution and dispersion of air pollutants in the PRD region [*Huang et al*., 2009].

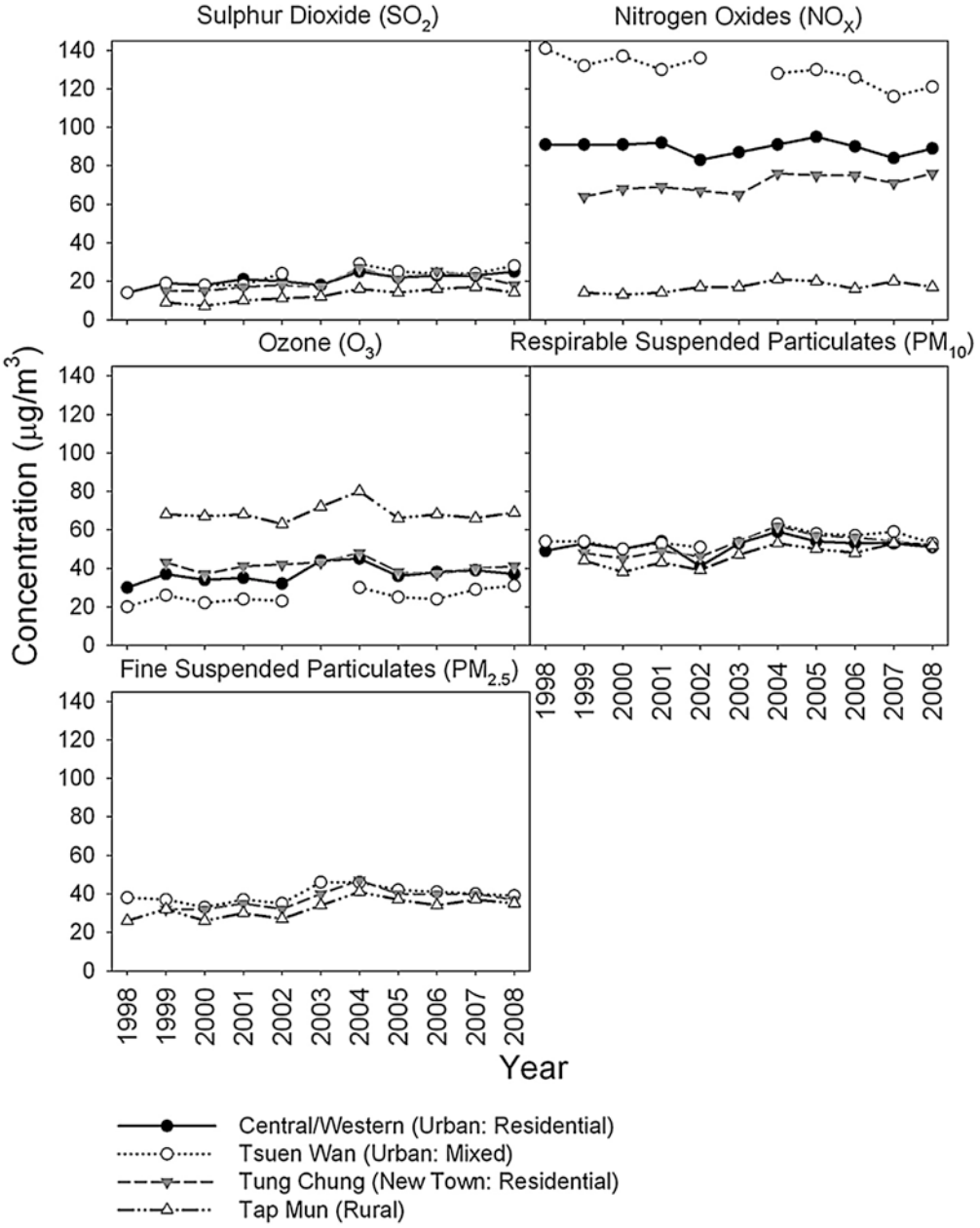

Figure 25 - Concentration of sulphur dioxide, nitrogen oxides, ozone, respirable suspended particulates, and fine suspended particulates at Central/Western, Tsuen Wan, Tung Chung and Tap Mun Air Quality Monitoring Stations from 1998 to 2008

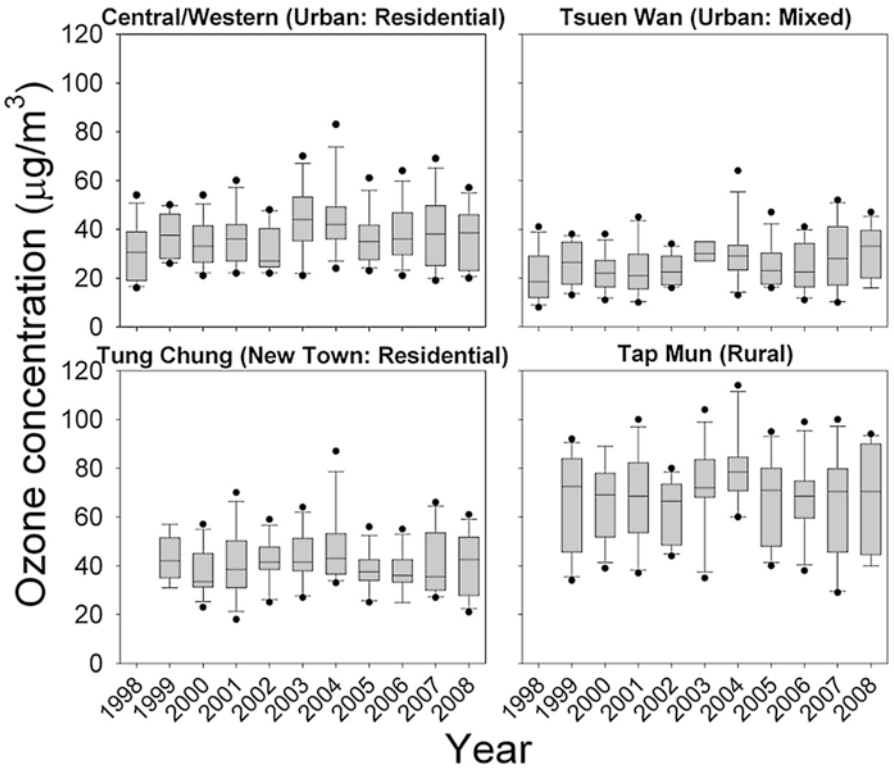

Figure 26 - Box plots of ozone concentrations at Central/Western, Tsuen Wan, Tung Chung, and Tap Mun Air Quality Monitoring Stations from 1998 to 2008. The boundaries of the box represent the 25th and 75th percentiles, the line within the box represents the median, the error bars illustrate the 10th and 90th percentiles, and the dots indicate outlying points. Note: An outlier is a point which falls >1.5 times the interquartile range above the third quartile or below the first quartile

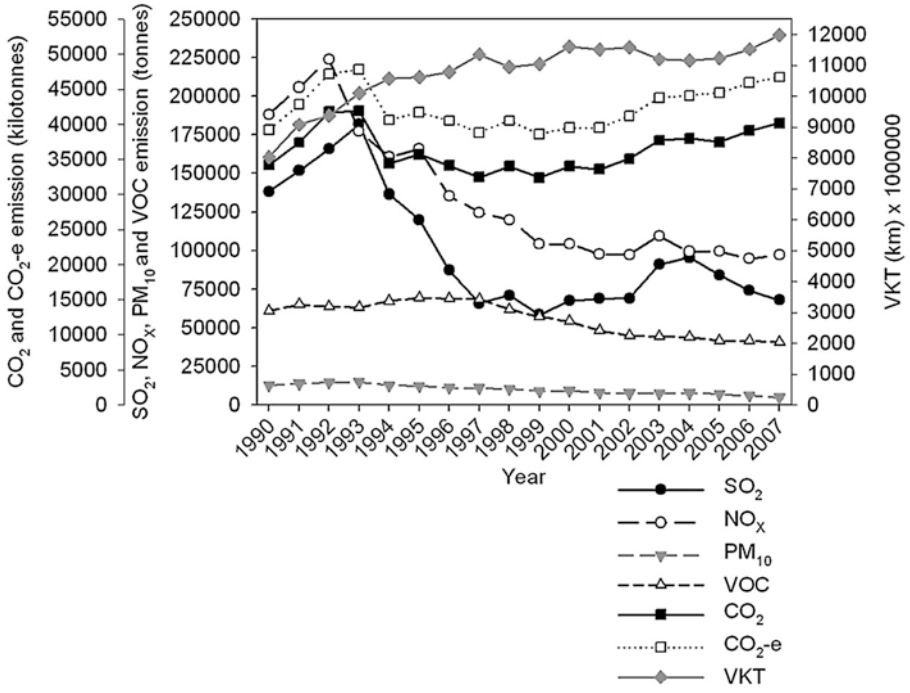

Figure 27 - Vehicle kilometres travelled (VKT) and sulphur dioxide (SO_2), nitrogen oxides (NO_X) respirable suspended particulates (PM_{10}), volatile organic compound (VOC), carbon dioxide (CO_2), and carbon dioxide equivalent (CO_2-e) emitted in HKSAR from 1990 to 2007

Relationships of the trends to regulations

The Air Pollution Control Ordinance empowers the HKSAR Government to establish Air Quality Objectives (AQO) and to control air pollution from industry, commercial operations, and construction work [*Trumbull*, 2007]. To improve local air quality, the HKSAR Government has introduced a wide range of control measures (e.g. increase fuel efficiency, reduce consumption of pollutant-emitting products, restricted emissions, etc.) since 1990 in order to reduce emissions from motor vehicles (PM_{10} and NO_X), power plants, industrial, commercial (SO_2) and VOC sources (see *Guo et al.,* 2004a and b;http://www.epd.gov.hk/epd/english/environmentinhk/air/prob_solutions/vocs_smog.html for details). These control measures have effectively reduced local emission of these pollutants (~50% reduction in SO_2, NO_X, and PM_{10} and ~34% reduction in VOCs) despite the increase in VKT (~50%) and energy (~20%) and electricity consumption (~70%) over the years (Figures 24 & 27). In addition to local emission measures, the HKSAR Government and Guangdong Provincial Government reached a consensus in April 2002 to prevent air quality from further deterioration and, in the long-term, to achieve good air quality for the Pearl River Delta region. Specifically, this consensus aims to reduce the regional emission of SO_2, NO_X, PM_{10}, and VOCs by 40%, 20%, 55%, and 55%, respectively by 2010, using 1997 as the base year [*Trumbull,* 2007].

Climatic change issues

Like other parts of the globe, HKSAR is undergoing climatic change in recent decades (rural and urban temperature increased by 0.2ºC/decade and 0.6ºC/decade, respectively) [*Lam et al*., 2006]. This climatic change is likely a result of the ~20% increase in greenhouse gases (carbon dioxide (CO_2 = ~80% contribution; ERM 2000), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride (SF_6)) content in the atmosphere due to increased energy use in recent decades (increase in VKT, energy and electricity consumption = ~50%, ~20% and ~70%, respectively, for HKSAR; Figure 27). *Lam et al.* [2006] discovered that the ozone concentration in Tung Chung was positively correlated with temperature, wind speed, and solar radiation. The increase in greenhouse gases is believed to trap solar radiation in the atmosphere which, in turn, raise air temperature and intensify photochemical formation of smog, resulting in further deterioration of visibility at a rate of 1.9%/decade [*Lam et al*., 2006]. *Wang et al*. [2009a] further suggested that aerosols reflect and absorb solar radiations at Earth‘s surface, as well as modify cloud cover and other cloud properties, resulting in considerable uncertainty in the estimation of aerosol impacts on radiation.

The United Nations Framework Convention on Climate Change (UNFCCC) and Kyoto Protocol were extended by the Central Peoples' Government (CPG) of China to Hong Kong as of 5 May 2003. As with other developing countries that have ratified the Kyoto Protocol, China (including the HKSAR) is not required to achieve any greenhouse gas (GHG) emission limits. Instead, it is required to submit national communications in accordance with the specific requirements of the Protocol.
(http://www.legco.gov.hk/yr08-09/english/panels/ea/ea_iaq/papers/ ea_iaq0113cb1-531-4-e.pdf).

A host of measures have begun to reduce the level of GHG emissions in Hong Kong. Measures include Buildings Energy Efficiency Funding Scheme and the Mandatory Energy Efficiency Labelling Scheme, banning the construction of coal-fired power generating units, and providing economic incentives to encourage development of renewable energy sources. At the same time, policy measures to reduce energy intensity are being pursued to address GHG emissions from other major emission sources including the transportation sector and landfills. These include continuously extending the coverage of the public transport system (in particular the railway network), promoting the use of electric vehicles, and enhancing the utilization of landfill gas as an alternative fuel (http://www.info.gov.hk/gia/general/200911/04/P200911040130_print.htm). Notwithstanding the concerted efforts mentioned above, if more specific GHG mitigation measures are put in place, more tangible rewards could be resulted.

Research projects on air quality in HKSAR

To date, over 600 scientific articles and technical reports related to air quality in HKSAR/the PRD region, focusing on the techniques, source characterization and transport, ambient concentrations, and effects on health and visibility, have been published (see *Lee et al*., 2009 for review). In addition to routine air quality monitoring, the EPD has commissioned various research projects on air quality in HKSAR and the PRD regions (http://www.epd.gov.hk/epd/english/environmentinhk/air/studyrpts/air_studyrpts.html). Periodic campaigns in $PM_{2.5}$ speciation (2000/2001 and 2004/2005) study, for example, were carried out to better understand the source apportionment of $PM_{2.5}$ in HKSAR [*So et al*., 2007; *Guo et al*. 2009]. The EPD has been monitoring VOC speciation at Tung Chung AQMS, in which the data can be applied to projects related to VOC source apportionment, temporal variation, etc.

With an aim to enhance the air quality in the PRD region, the EPD commissioned two air quality feasibility studies in the 3rd quarter of 2007: (i) major industrial pollution sources study (http://www.epd.gov.hk/epd/english/news_events/press/press_071018c.html) and (ii) VOC and photochemical ozone pollution study (http://www.epd.gov.hk/epd/english/news_events/press/press_070927a.html). In order to improve the overall air quality in the PRD region, the EPD has worked closely with local and international scientists and academics to estimate the feasibility to develop a cross-boundary air quality monitoring network [*Lee et al*., 2009].

Apart from research projects, the EPD has developed various computer models, including the PATH (Pollutants in the Atmosphere and their Transport over Hong Kong) model to investigate the transport patterns of aerosol pollutants and photochemical ozone problems [*Louie et al*., 2008] and, with the aid of *Wang* [2003], HK-WinHaze to calculate and visualize the extent of visibility impairment. These research projects and models highlighted the importance of collaboration among HKSAR, mainland China, local, and international scientists and academics [Lee et al., 2009].

Guangdong and Hong Kong Regional Cooperation Efforts

The HKSAR Government makes controlling both street-level air pollution and smog as high priority. In particular, it has been working closely with Guangdong Provincial Authorities to implement a joint plan to tackle the regional smog problem. (http://www.epd.gov.hk/epd/english/environmentinhk/air/air_maincontent.html). In December 2003, the two governments jointly created the PRD Regional Air Quality Management Plan (the "Management Plan") with the goal of meeting the emission reduction targets set in 2010. The PRD Regional Air Quality Monitoring Network, established under the Management Plan between 2003 and 2005, is now in full operation and provides comprehensive and accurate air quality data, which are made available to the public. (http://www.epd.gov.hk/epd/english/resources_pub/publications/m_report.html). The Network also evaluates the effectiveness of the air pollution control measures through long-term monitoring.

Problems remaining

Notwithstanding the considerable studies on air pollutions and the effectiveness of local emission control measures, HKSAR still suffers from elevated levels of air pollution [*Leverett et al*., 2007]. This implies the air quality of HKSAR is not only influenced by local emission sources, but also strongly affected by regional contribution, especially during winter and typhoon episodes [*Louie et al*., 2005b; *Huang et al*., 2009]. Under the influences of global climate change, as mentioned above, the extent of air quality deterioration and related health and environmental impacts in HKSAR and the PRD region will need to be studied further. In order to minimize air pollution problems in HKSAR and to improve the overall air quality in the PRD region, there is, as a consequence, an emerging need to (i) conduct more scientific studies between academia and government to fill knowledge gaps from previous studies (ii) integrate multi-pollutant emission reduction policy framework based on sound science, including: establishment of mitigation measures in greenhouse gases, as well as reviewing (see ERM 2000 for details) and updating AQO periodically (http://www.epd.gov.hk/epd/english/environmentinhk/air/air_quality_objectives/air_quality_objectives.html), and (iii) intensify cross-boundary collaboration with mainland China and foreign countries

and fully implement all possible control measures [*National Academy of Engineering and National Research Council*, 2008, *Lee et al*., 2009].

3.7 JAKARTA, INDONESIA

Introduction and geographical information of the city

Jakarta, the capital city of Indonesia is located in a low land area with an average height of around 7 m.s.a.l and centred at 6°12' South latitude and 106°48' East longitude with 661 km^2 of land area (see Figure 28). Jakarta is bounded by a mountain range extending along the south coast of the western Java Island with the average height of 1500 m and the highest mountain named Salak Mountain is 3019 m. The climate of the city is characterized by monsoons that bring about 2 major seasons, the dry (centred in September-November) and wet (centred in January – March). In the dry season the southeasterly (SE) monsoon dominates while in the wet season the northwesterly (NW) wind dominates [*Sofyan et al*., 2007]. As commonly found in coastal areas, sea-land breeze is also prevalent in Jakarta. The Salak mountain range, with an almost V- shape directed along Northeast on one side and Northwest on the other, tends to modify wind direction surrounding the city [*Permadi and Kim Oanh*, 2008].

Jakarta is inhabitated by approximately 8.7 million people as of 2005 with the average growth rate of 0.16 % from 1990 to 2005 [*JBS*, 2006]. Jakarta's annual Gross Domestic Product (GDP) increased at a stable annual growth rate of approximately 9% from 1990 to 2005. Over the same period, fuel consumption increased at an annual average of around 2.6%. Major fuels used by households are kerosene and LPG while for transportation gasoline and diesel oil are used. In the industrial sector, diesel oil, coal, and natural gas are used. Number of motor vehicles population, which is a main source of air pollution in the city, has been increasing rapidly with an average annual rate of around 15 %. Almost 50 % of motorized vehicles registered in Jakarta are motorcycles that still consist of numerous polluting two-stroke motorcycles. Road capacity is insufficient compared to the massive increase of vehicles, which leads to high traffic congestion and hence further urban air quality deterioration.

Emission sources of air pollutants

Investigation of pollution sources is one of the most important tasks in air quality management. *Soedomo et al*. [1992a] reported the first emission inventory of Jakarta, which was reported for the base year of 1990 (Table 7). The emissions inventory includes five pollutants, carbon monoxide (CO), total hydrocarbon (THC), nitrogen oxides (NO_X), particulate matter (PM), and sulphur oxides (SO_X) from four sectors (industry, residential, solid waste burning, and transportation). The transportation sector was the major contributor for all pollutants except for SO_2, for which industry was the major source. The residential sector was found to also contribute significantly to PM emission.

Japan International Cooperation Agency (JICA) and National Environmental Protection Agency (BAPEDAL) jointly conducted emission inventory for Jakarta and its surrounding cities for the base year of 1995 [*JICA*, 1997]. The similar 5 target pollutants were addressed but not for all sectors. Note that, the mobile sources in JICA study include road transport, ships, and aircraft. A higher total emission was recorded in JICA's study, which may reflect the increase in the emissions during the five-year period. Transportation remained the main source of air pollution. However, the relative contribution by industry was higher for PM and SO_2 as compared to the previous estimate by *Soedomo et al*. [1992a].

Air Quality Monitoring Network (data availability)

The air quality monitoring network of Jakarta is operated by Jakarta Environmental Protection Agency (JEPA) and includes both manual and automatic monitoring stations. The intermittent manual monitoring was initiated in 1986 at twelve monitoring sites that are operated on a rotational basis. Each station measures selected air pollutants for 24-hrs every eight days [*ADB*, 2002]. In early 2002, continuous air quality monitors for ambient air quality at 5 (five) JEPA automatic monitoring stations (JAF1, JAF2, JAF3, JAF4, and JAF5) began. All of these automated

monitoring stations are equipped to monitor CO, NO, NO_2, SO_2, PM_{10}, and O_3 as well as meteorology (wind speed and direction, temperature, relative humidity, and global radiation) producing 30-minute average data online. Methane (CH_4) and non-methane hydrocarbons (NMHC) are not monitored at any station. The operation and maintenance of the stations are handled by the Regional Center (RC) in JEPA, which calibrates the equipment once per year. Data is also sent to JEPA center in order to be converted to daily air pollution index (API). Daily API data is then sent to 4 data displays automatically that are located at several public spaces.

All of the automated stations are classified as general ambient stations that are not located at the curbside of main roads. Accordingly, JAF1 is located in a dense residential area while JAF2 is located near a commercial area and between office buildings. JAF3 is located in an open space of mixed land use including commercial facilities and residential areas. Both JAF4 and JAF5 are located near the city center with dense population and road traffic. JAF5 is in the city park inside the Senayan Sport Complex Stadium and JAF4 is in a municipality building complex [*Permadi and Kim Oanh*, 2008]. Detail information of each monitoring station is given in Table 7.

Table 7 - Information on air quality monitoring stations Source: Permadi and Kim Oanh, [2008]

No	Stations[a]	Area Type	Latitude Coordinate	Longitude Coordinate	Height m ASL[b]
1	JAF 1 (East Jakarta)	Municipality building	6° 12'53.28''	106°56'41.72''	6
2	JAF 2 (Kemayoran)[c]	Commercial and open space	6° 09' 12.54''	106° 51' 22.56''	2
3	JAF 3 (Pondok Indah)[c]	Open space (City Park)	6° 15' 40.94''	106° 47' 00.73''	23
4	JAF 4 (West Jakarta)	Municipality building	6° 11' 07.17''	106° 44' 12.36''	7
5	JAF 5 (Senayan Sport)	Sport complex	6° 12' 53.28''	106° 48' 18.43''	12

[a] *All five stations are of the general ambient type*
Monitored parameters include CO, NO, NO_2, SO_2, O_3, WS – Wind speed, WD – Wind direction, Temperature, Relative Humidity and Global radiation.
[b] *ASL : Above Sea Level*
Height of air intake: 3m
[c] *much missing data*

Air pollution trend and status of Jakarta

Jakarta experienced serious air pollution problems due to intensive industrialization, urbanization, and economic development. *WB.* [1997] reported the initial assessment of air quality in Jakarta, Indonesia addressing concentrations of NO_2, CO, O_3 and TSP during the period of 1981 – 1993. Annual average NO_X was reported around 20-160 μg/m^3 while its maximum 24-hr concentrations were observed around 200-500 μg/m^3. The annual values are well above the corresponding National Air Quality Standard (NAAQS) of 100 μg m^{-3} and the 24-hr NAAQS of 150 μg m^{-3}, respectively. Average 8-hour CO levels were reported around 3.5 mg m^{-3} in residential area and 27 mg m^{-3} in a central commercial area, well above the WHO Air Quality Guideline of 10 mg m^{-3}. 24-hr average values of TSP were observed of 300-450 μg m^{-3}, which were well above the corresponding NAAQS of 230 μg m^{-3} while the annual TSP averages in the most polluted areas (city center) were reported to be 5-6 times the national air quality standard of 90 μg m^{-3}. Ozone concentrations ranged from 2-15 μg m^{-3} for the annual mean and 86-200 μg m^{-3} for the highest 1-hr concentration, which were within the NAAQS (annual: 50 μg m^{-3} and hourly: 235 μg m^{-3}). Thus, photochemical smog pollution was not serious a problem at that time.

More recent trends, for the period of 2001 to 2007, of annual average concentrations are reported in the Jakarta State of Environment Document [*JEPA*, 2008]. As seen in Figure 29, CO, NO_2, PM_{10}, and O_3 seem to follow similar variation patterns, i.e. decreases in recent years. This

may be indicating the improvement in transportation emissions that are related to the operation of Trans-Jakarta busway, a rapid transit system started in January 2004 that links most of the main roads in the city. Nevertheless, PM_{10} levels are observed to be consistently above the WHO annual average guideline of 20 μgm^{-3}. The highest annual average of SO_2 was observed in 2004 (110 $\mu g\ m^{-3}$) exceeding the standard of 60 $\mu g\ m^{-3}$. Annual average O_3 in this period was consistently above the standard of 50 $\mu g\ m^{-3}$, which indicates photochemical air pollution in the city.

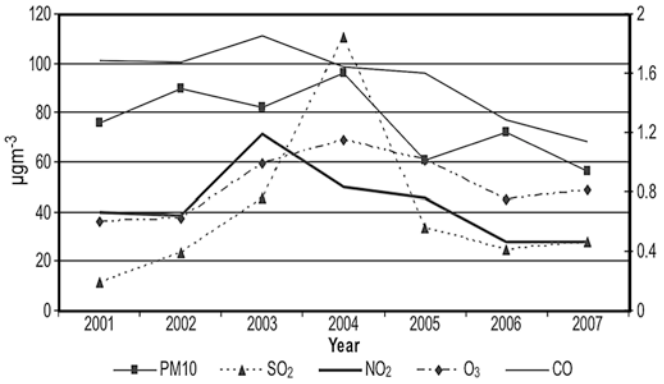

Figure 29 - Trends of air pollution in Jakarta. [modified from *JEPA*,2008]. Annual NAAQS: CO: na; PM_{10}: 20 $\mu g\ m^{-3}$ [*WHO*, 2006]; SO_2: 60 $\mu g\ m^{-3}$; NO_2: 100 $\mu g\ m^{-3}$; O_3: 50 $\mu g\ m^{-3}$

Suhadi et al. [2005] and *Permadi & Kim Oanh* (2008) provided a detailed study of photochemical smog pollution in Jakarta in relation to its precursors and meteorology. During the period of 1996 to 1998, the annual ozone averages in ambient stations located off the city center have exceeded the annual NAAQS and showed an increasing trend [*Suhadi et al*., 2005]. Frequent exceedances of the hourly NAAQS were also observed from 2002 to 2003 with over 450 hourly measurements or 0.7 % out of 66,000 hourly ozone measurements at the 5 stations [*Permadi and Kim Oanh*, 2008]. Both studies suggested that the highest monthly average ozone occur during the dry season (September-November) and the lowest during the wet season (December-March). The diurnal cycle of ozone in Jakarta and its precursors is clearly shown and is typical for polluted urban areas [*Permadi and Oanh*, 2008].

Air quality management

This section highlights the attempts of the local government in implementing regulations and actions under the scheme of urban air quality management. During past decades, initiated by JEPA, many attempts have been initiated in order to improve air quality in Jakarta. Urban air quality monitoring stations have significantly improved by updating the monitoring method from rotational manual basis to the automated continuous monitoring systems with QA/QC. Air pollution Index (API) is used to display the results in public spaces to enhance public participation and awareness. Local government is also working to reduce traffic congestion by improving the urban transportation system. Busway corridor VII – X has been built covering the entire city. This has been recognized as the largest busway system in Asia and won a BAQ award in 2006. A rail-based Mass Rapid Transit (Monorail) is still under construction. Intensive development of urban green space has occurred in recent years. Car free day programme has been campaigned intensively calling for public participation in more sustainable ways. Unleaded gasoline has been promoted in the city since June 2000 with support from the government and industry [*JEPA*, 2008]. As of January 2008, the government is said to be enforcing the Euro 2 standard [*MOE*, 2003].

Climate change issue

According to *IDRC – EEPSEA* [2009], Jakarta is one of the most vulnerable cities to climate change in Southeast Asia. Therefore, climate change issue has attracted more attention from the local government and scientists. However, the studies on ambient levels and the emissions of greenhouse gases or other climate forcing agents such as black carbon in the city are still in the

preliminary stage. *Kosasih* [2004] presented the total CO_2 emitted from the transportation sector in Jakarta, which is around 12 million tons/year while the residential sector contributes around 10 million tons/year. So far, there has been no other reported climate forcing agent inventories for Jakarta. Several potential Clean Development Mechanism (CDM) projects have been proposed such as CDM for Trans-Jakarta bus way (Bus Rapid Transport) project, clean buses, reducing methane emission from municipal solid waste disposal site, and fuel shifting in residential sector [*Pelangi Foundation*, 2007].

Recorded research on air pollution in the city

There have been several research projects on air pollution in Jakarta since early 1990's. MOE initiated the "blue sky" project focusing on implementation of emission controls of stationary and mobile sources. In 1993, the Urban Air Quality Management in Jakarta (URBAIR) project was implemented to collect benchmark information on air quality in Jakarta [*WB*, 1997]. The study included monitoring, modelling, and cost benefit and health assessment. It provided the first recorded emission inventory of air pollutants for Jakarta and applied a simple dispersion model. The study on the Integrated Air Quality Management for Jakarta Metropolitan Area was carried out in 1997 by Japan International Cooperation Agency [*JICA*, 1997] providing more detailed information on emission inventories and dispersion modelling of Jakarta and the surrounding area. Asian Development Bank sponsored the RETA Project on the "Study on Air Quality in Jakarta: Future Trends, Health Impacts, Economic Value, and Policy Options" with a focus on projected health assessment of air pollution in Jakarta region [*ADB*, 2002]. A more recent study was conducted under the project of Asian Regional Research Programme in Energy, Environment, and Climate, Phase III (ARRPEEC III) coordinated by the Asian Institute of Technology and implemented by the Institut Teknologi Bandung that aimed to develop scenarios of emission reductions of air pollution and GHG from the transportation sector. A number of different research projects conducted by universities and research institutions are also available (see publications cited above).

Recent air quality problems

Photochemical smog has become more serious in Jakarta in recent years [*Permadi and Oanh*, 2008] with frequent exceedances of NAAQS. Higher emission strength of ozone precursors (NO_X, VOC) and favourable meteorological conditions are the driving forces to the increased O_3 concentrations. Ozone concentrations are expected to remain the same or even get worse since there is no reported systematical attempt to formulate mitigation strategies. This may endanger the human health and agricultural crops of Jakarta since ozone is toxic and phytotoxic.

Particulate matter pollution remains a major air quality issue in Jakarta as well. Efforts should be put forward to monitor fine particles as well as their composition.

The transportation sector is still considered the main contributor of emissions in Jakarta. The number of two-stroke motorcycles in Jakarta is reduced every year but high levels of pollutants are still observed. The difficulties may be due to high growth rate of vehicle population in Jakarta, which was almost 100% from 1995 to 2000. The government has also promoted the Trans-Jakarta busway, unleaded gasoline, and bio-fuel that may improve air quality. New sources such as open biomass burning in areas surrounding the city have not been addressed adequately. Therefore, more comprehensive air pollution research needs to be carried out in order to provide relevant information for decision makers.

3.8 MANILA, PHILIPPINES

Manila demography, topography and climate

The Manila Metropolitan Area, or Metro Manila**,** is the administrative region encompassing the city of Manila, the national capital of the Republic of the Philippines. Metro Manila is a megacity made up of 17 cities and municipalities of the National Capital Region (NCR). A 2007 government census estimated its population at approximately 12 million over an area of only 636 km^2. The 2009 UN population estimate for the same region was over 20 million and projected to increase to 29.5 million by 2025. Manila proper (i.e., excluding surrounding cities and municipalities) has one of the

highest population densities of any major city in the world, currently estimated to be 46,000 people/km^2 [*Columbus*, 2003].

Metro Manila is the political, economic, social, and cultural center of the Philippines and is one of the more modern metropolises in Southeast Asia. Metro Manila's topography ranges from flat fluvial and deltaic lands in the west to the rugged Marikina Valley and Sierra Madre mountains in the east. It is bordered by the larger and more fertile plains of Central Luzon to the north. Metro Manila is bisected by the Pasig River and sandwiched by two bodies of water: Manila Bay to the west and the Laguna de Bay to the southeast (see Figure 30). Lying as it does within the Philippine archipelago very close to the equator at 14˚N, 121˚E, Metro Manila's climate is hot and humid, with a temperature range of about 20-38 ˚C. It has a distinct, short dry season from January through April, and a longer wet season from May through December.

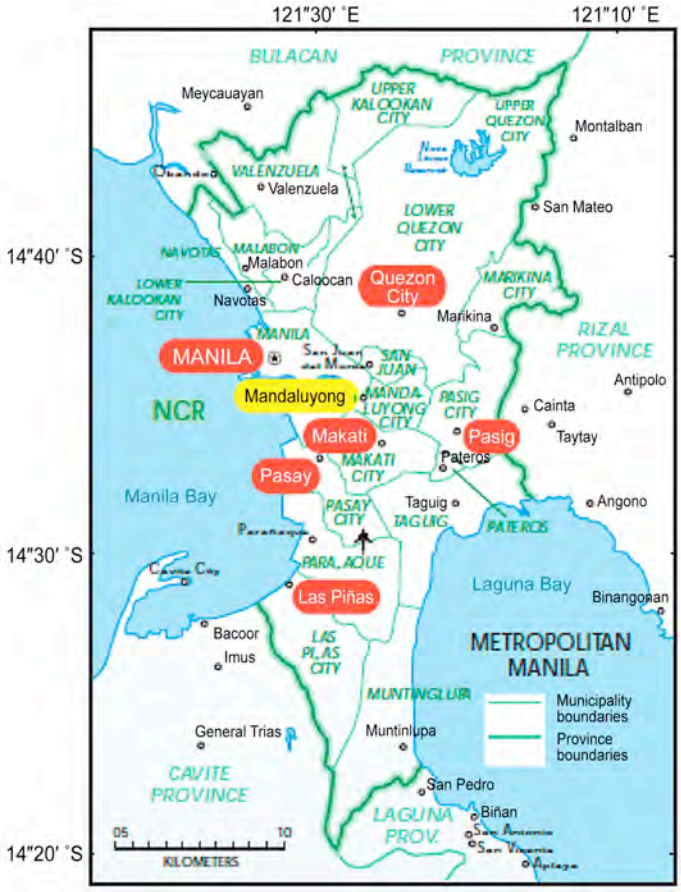

Figure 30 - Metro Manila Region

Air quality in Metro Manila

In 1999, the Philippine government established the Clean Air Act (CAA), which set legal limits for ambient levels of major air pollutants to protect public health and the environment. Table 8 presents the air quality standards set by this legislation.

Table 8 - National Ambient Air Quality Guideline Values (NAAQGV) for Criteria Pollutants

Air Pollutant	Averaging Time	Standard in µg/m³
Total Suspended Particulates (TSP)	Daily Annual	230 90
Particulate Matter Less than 10 µ (PM10)	Daily Annual	150 60
Sulphur Dioxide (SO_2)	Daily Annual	180 80
Nitrogen Dioxide (NO_2)	Daily Annual	150 150
Ozone (O_3)	1 Hour 8 Hours	140 60
Carbon Monoxide (CO)	1 Hour 8 Hours	35,000 10,000
Lead (Pb)	3 Months Annual	1.5 1.0

The Department of Environment and Natural Resources (DENR) operates a network of 44 air quality monitoring stations in 15 regions located throughout the country. A network of ten automated continuous monitoring stations was implemented by DENR within Metro Manila to measure criteria pollutants (PM_{10}, SO_2, NO_2, CO and O_3) and meteorological data [*EMB*, 2006], of which two stations could also measure $PM_{2.5}$, methane, non-methane hydrocarbons, benzene, toluene, and xylene [*ADB and CAI-Asia*, 2006]. The stations were operational from 2004-2005; however, operations were halted due to a contractual dispute. In addition to the monitoring activities of the DENR, the Philippine Nuclear Research Institute (PNRI) of the Department of Science and Technology (DOST) monitors PM_{10} and $PM_{2.5}$ at three sites in Metro Manila.

Of the different types of air pollution prevalent in Metro Manila, PM has received the greatest attention. Figure 31 presents annual average TSP concentrations in Metro Manila for the period 1987-2008 as measured by the Environmental Management Bureau (EMB) of the Philippine Department of Environment and Natural Resources (DENR). The data show that substantial progress has been made in reducing TSP concentrations since 1995, despite the fact that annual mean concentrations still exceeded the national standard of 90 µg m^{-3}.

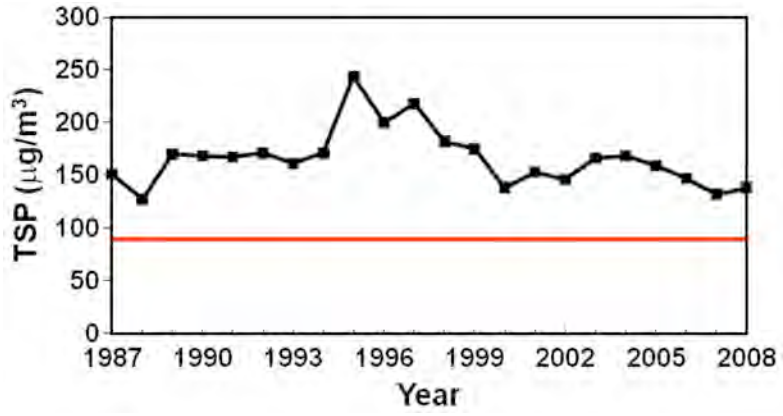

Figure 31 - Average annual TSP concentrations in Metro Manila for the period 1987-2008. Red line indicates NAAQGV value of 90 µg m^{-3}. Sources: ADB, 2002 (data from 1987-2001); CAI-Asia, 2009 (data from 2002-2008)

Figures 32 and 33 show the average annual PM_{10} and $PM_{2.5}$ concentrations at the PNRI sampling sites in Metro Manila from 2001 to 2008. Although PM_{10} is consistently below the national standard of 60 µg m^{-3}, it exceeds the WMO Air Quality Guidelines (AQG) of 20 µg m^{-3}. The Philippines currently does not have a $PM_{2.5}$ standard. However, $PM_{2.5}$ annual mean concentrations exceed the WHO AQG of 10 µg m^{-3}.

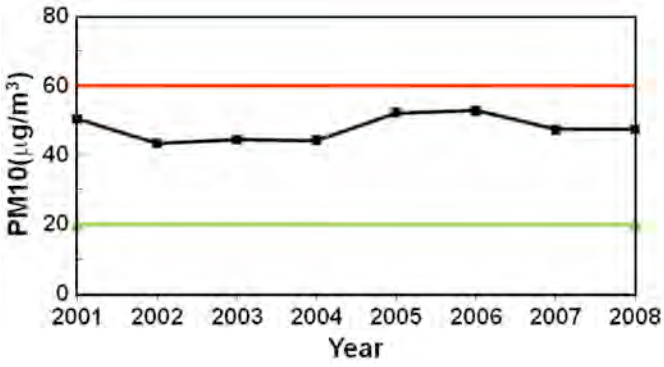

Figure 32 - Annual mean concentrations of PM_{10} in Metro Manila for the period 2001-2008.
Red line: NAAQGV value of 60 μg m^{-3};
Green line: WHO AQG of 20μg m^{-3} [*CAI-Asia*, 2009]

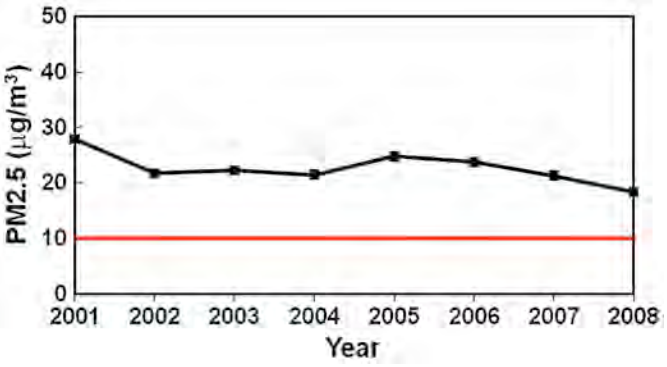

Figure 33 - Annual mean concentrations of $PM_{2.5}$ for the period 2001-2008.
Red line indicates WHO AQG of 10 μg m^{-3}
[*CAI-Asia*, 2009]

The Manila Observatory (MO) participated in the Asian Regional Air Pollution Research Network (ARPET), coordinated by the Asian Institute of Technology [*Oanh et al*., 2006]. During Phase 1 of ARPET (2001-2004), MO collected over a thousand $PM_{2.5}$ and PM_{10} samples at five sites in Metro Manila using both MiniVol and dichotomous samplers. Table 9 presents the results of ARPET Phase 1 for Metro Manila for both the dry and wet seasons.

Table 9 - ARPET Phase 1 Results for Metro Manila averaged over all samples. [*Oanh et al*., 2006]

PM Size Range	Season	Number of Samples	Mass in μg m^{-3}
$PM_{2.5}$	Dry	407	44
PM_{10}	Dry	122	54
$PM_{2.5}$	Wet	376	43
PM_{10}	Wet	136	55

From the data in Table 9, it is clear that seasonal average $PM_{2.5}$ concentrations in Metro Manila already exceed the current WHO 24-hour AQG of 25 μg m^{-3}in both dry and wet seasons, while seasonal average PM_{10} concentrations are almost within the WHO 24-hour AQG of 50 μg m^{-3}in both dry and wet seasons.

Limited monitoring of other criteria pollutants (SO_2, NO_2, and O_3) has been conducted in Metro Manila. The concentrations of SO_2 and NO_2 are below the NAAQGV at the monitoring stations, while O_3 concentrations exceed the one-hour guideline on most days at one station between 2001-2002 [*World Bank*, 2002], but are relatively low elsewhere.

Sources of air pollutants and emission inventories

The DENR conducted a national emissions inventory of air pollution sources in 1990 and subsequently updated the inventory in 2001, 2004, and 2006 as mandated by the Philippine Clean Air Act. The emissions inventory includes criteria pollutants such as PM, SO_2, NO_X, CO, and VOCs. According to the latest (2006) inventory, the transportation sector is the major source of air pollution contributing about 65% of the pollutants, while 21% is from stationary [point] sources and 14% from area sources, with total national emissions of seven million tons per year from all sources [*DENR*, 2009]. The same emissions inventory shows that CO has the biggest pollution load contribution of 50%, followed by NO_X at 15%, VOCs at 15%, PM at 11%, and SO_X at 9%. CO emissions are caused by the increasing population of gasoline-powered vehicles, which include cars (13.6%) and motorcycles/tricycles (47.9%) [*DENR*, 2009].

Control strategies

Among the environmental success stories in the Philippines is the phase-out of lead in gasoline that was implemented after 1993, as summarized in Table 10. Control strategies for other criteria pollutants have also been implemented in the Philippines with very limited success, as summarized in Tables 11-12. (Source: World Bank: Philippines Environment Monitor, 2002).

Table 10 - Lead phase-out in the Philippines. Source: World Bank: Philippines Environment Monitor, 2002

Date	Action
April 1993	Lead content in gasoline was reduced from 0.6 g/l to 0.15 g/l
February 1994	Introduction of unleaded gasoline (ULG)
January 1995	Oil Deregulation Law lowered tax on ULG, and priced it cheaper than leaded gasoline; ULG sales rose
April 2000	Leaded gasoline phased out in Metro Manila
December 31, 2000	Leaded gasoline completely phased out in the Philippines.

Table 11 - PM control options and their use in the Philippines (Source: World Bank: Philippines Environment Monitor, 2002)

Sources	Control options	Status
Mobile		
Motorcycles	- Regular maintenance - Switch to 4-stroke engines	- 75 percent are 2-stroke motorcycles
Trucks / buses	- Regular maintenance - Reduced sulfur in fuel - Trap oxidizer	- Existing inspection/ maintenance system not functioning effectively
PUVs		
Stationary		
Industry and power	End-of-pipe control measures (e.g., venturi scrubbers, electrostatic precipitators, fabric filters, and cyclones)	- Less than 10 percent of the identified air polluting industries have installed control measures, which are poorly operated. - Power plants switching to cleaner fuels
Area		
Refuse burning	- Disposal in sanitary landfills	- Widespread burning of garbage - No proper disposal facilities
Reentrainment	- Regulating movement of trucks carrying debris - Better construction practices - Paved roads	- Truck movement unregulated - Ineffective controls at most construction sites
Agricultural waste burning	- Composting of waste	- Unknown

Table 12 - NO_X Control options and their use in the Philippines (Source: World Bank: Philippines Environment Monitor, 2002)

Sources	Control options	Status
Vehicles		
Cars	- Regular maintenance - Catalytic converters	- Use of catalytic converters for gasoline vehicles not mandated - High sulfur content in diesel deters the use of catalytic converters
Motorcycles		
Trucks/ buses	- Regular maintenance - Catalytic converters	
PUV's		
Stationary		
Power & Industry	- Low NO_x burners	- Not prevalent

Table 13 - SO_6 Control options and their use in the Philippines (Source: World Bank: Philippines Environment Monitor, 2002)

Sources	Control options	Status
a. Stationary		
Power	- Low sulfur fuel - Flue gas de-sulfurization	- Two diesel power plants closed, remaining will be decommissioned by 2010 - Few plants converted to natural gas, while the use of low sulfur coal is increasing - The recently approved Energy Plan proposes to expand natural gas use during the next 10 years
Industry	Same as above	- Few enterprises have installed control measures and are poorly regulated - Availability of natural gas in Southern Luzon will allow replacement of fuel oil and coal with natural gas
b. Mobile		
Trucks/ buses	- Efficient public transport system - Low sulfur diesel - Use of Compressed Natural Gas (CNG)	- Sulfur content in diesel reduced from 0.5 to 0.2 percent in 2001 - Functioning Light Rail Transit (LRT) system in Metro Manila, but is only 2 percent of the daily ridership - Introduction of CNG being studied
PUVs		

The EMB through its regional offices is in charge of monitoring industrial firms and issuing notices of violations (NOVs) and permits-to-operate (POs). According to *DENR* [2009], from 2005-2007 a total of 18,697 firms were monitored. Within that three-year period, a total of 1,676 NOVs and 24,391 POs were issued *DENR* [2009].

The Land Transportation Office (LTO) enforces compliance with emission standards for motor vehicles. EURO 2 standards for new motor vehicles took effect in 2008. A Motor Vehicle Inspection System (MVIS) project is currently being implemented so that all private motor vehicles will only be allowed renewal of annual registration upon inspection by the LTO or other authorized private motor vehicle inspection center. In addition, vehicles observed to be emitting excessive smoke while operating in any public highway may be subjected to an emission test by properly-equipped law enforcers and other deputized agents. Total annual apprehensions of smoke-belching vehicles exceeded 15,000 in both 2005 and 2006, but declined to under 12,000 in 2007 [*DENR*, 2009].

Another major control strategy is the Alternative Fuels Programme, which was intended to facilitate the country's attainment of 60% energy self-sufficiency by 2010, as well as to curb air pollution. The Programme has four major subprogrammes: Biodiesel, Bioethanol, Natural Gas Vehicle Programme for Public Transport (NGVPPT), and the Autogas Programme. Other technologies advocated under the Programme are hybrid, fuel cell, hydrogen, and electric vehicles. In addition, the Bio-fuels Act of 2006 has enabled a bio-diesel blend of 1% by volume to be available at all gas/pump stations nationwide since 2007. Likewise, a gasoline fuel consisting of 10% bio-ethanol blend (E10) by volume is distributed and sold by all oil companies and dealers in the country [*DENR*, 2009].

Metro Manila Air Quality Improvement Sector Development Plan

Beginning in 1999, the Asian Development Bank agreed to loan the Philippine government approximately $296 million to address air pollution in Metro Manila. This programme, known as the Metro Manila Air Quality Improvement Sector Development Plan, was intended to help the government implement its 1998 Air Quality Action Plan (AQAP). While the programme helped to eliminate lead from gasoline in Metro Manila, it did not fully achieve other intended outcomes, which included: (i) mitigating air pollution from mobile sources; (ii) mitigating air pollution from stationary sources; (iii) improving fuel quality; (iv) reducing emissions from vehicular use; (v) reducing traffic congestion and improving traffic flow; (vi) strengthening ambient air quality monitoring, evaluation, and reporting; (vii) intensifying public awareness; (viii) monitoring coordination and implementation of the AQAP; and (ix) strengthening the capacity of relevant institutions.

The programme's complexity demonstrated design inadequacies during its implementation, a problem compounded by the lack of permanent staff positions at the DENR to sustain the programme efficiently and to maintain the air quality monitoring network.

Climate change policies

The Philippines is a signatory to the UN Framework Convention on Climate Change and ratified the Kyoto Protocol in 1998. The National Action Plan was formulated in 1997, which provides guidance on prioritizing mitigation measures. In 2006, the DENR launched the Philippine Greenhouse Gas Accounting and Reporting Programme (PhilGARP) to encourage the private sector to undertake voluntary actions on the inventory of greenhouse gas (GHGs) and identification of mitigating measures. The President of the Philippines signed the Climate Change Act of 20 in 2009, which will establish a commission to develop a Framework Strategy on Climate Change and to formulate and implement a National Climate Change Action Plan (see DENR website).

Future challenges

The contribution of the transport sector to the worsening air pollution requires immediate action and poses an increasing threat to the health of the city dwellers. Unsustainable urbanization had led to growing traffic congestions necessitating transport planning and management [*DENR*, 2009].

Among the challenges facing the Philippines in improving air quality in Metro Manila are: 1) the lack of trained technical personnel, particularly in regulatory agencies; and 2) the difficulty of fostering a culture that can address the complex social, organizational, scientific, and engineering issues involved in improving air quality amid a scarcity of financial resources. The implementation of existing regulations is thus rendered extremely difficult without adequate monitoring and government oversight and enforcement. Complex and overambitious programmes are not likely to work under these circumstances. Rather, a step-wise approach with realistic goals is more likely to succeed, especially if it is aimed at increasing capacity within the research, educational, industrial, policy, and regulatory sectors in the Philippines.

3.9 OSAKA, JAPAN

Introduction

Osaka is located in south-central Japan (Osaka Prefectural Government: 34.67°N, 135.53°E). The population of Osaka Prefecture was about 8.8 million, and the population of the Kansai area (Osaka Prefecture and the five surrounding prefectures) was about 22.7 million in 2008 (Figure 34).

Because the Osaka Plain is narrower than the Kanto Plain and is surrounded by mountains, the geographically complex distribution of urban areas, mountains, and sea plays an important role in local atmospheric circulation [*Ohashi and Kida*, 2001]. In summer, sea breezes blow from Osaka Bay to the Kyoto basin (southern part of Kyoto Prefecture) and the Nara basin (northern part of Nara Prefecture) until late afternoon.

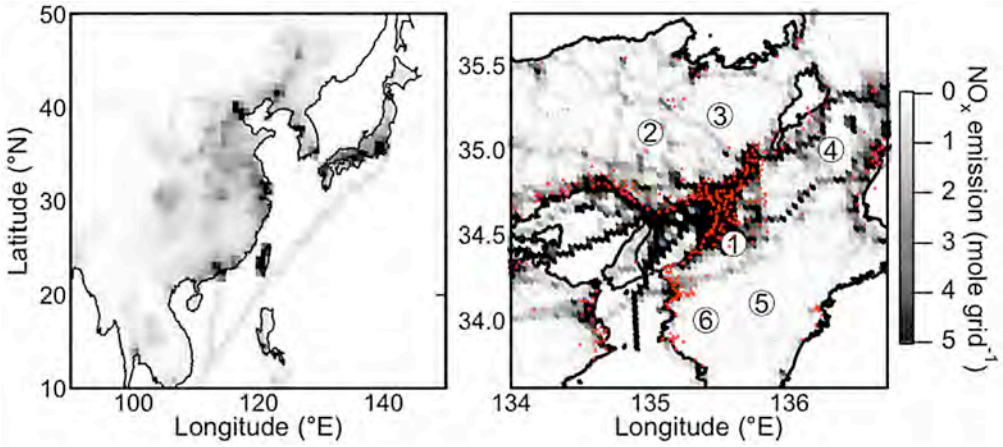

Figure 34 - NO_X emission rates over the East Asia (left) and Kansai area (right). Numbers indicate six prefectures in the Kansai area (1, Osaka; 2, Hyogo; 3, Kyoto; 4, Shiga; 5, Nara; 6, Wakayama). Red points in the right panel show the locations of monitoring stations in the AEROS network

Emissions sources and regulations

In the entire Kansai area, motor vehicles accounted for 75%, 50%, 42%, 15%, and 5% of emissions of CO, $PM_{2.5}$, NO_X, VOCs, and SO_2, respectively, in 2000 [Table 14, *Kannari et al.*, 2007]. Large point sources were the largest contributors to SO_2 emissions (~46%) and other transport sector sources (mostly ships) were the second largest contributors (37%). Stationary evaporative sources were the largest contributors to VOCs emissions (~48%) and biogenic sources were the second largest (35%).

Table 14 - Source contributions to NO_X, SO_2, $PM_{2.5}$, VOC, and CO emissions (Gg/year) over the Kansai area in 2000 estimated in the EAGrid inventory (*Kannari et al.*, 2007)

	NO_X		SO_2		$PM_{2.5}$		VOC		CO	
Large point sources	85	24%	42	46%	4.0	21%	2.9	1%	148	19%
Other point sources	16	5%	11	11%	1.2	6%	1.6	0%	19	2%
Motor vehicles	149	42%	4.2	5%	9.7	50%	79	15%	598	75%
Off-road vehicles	24	7%	0.5	0%	0.8	4%	3.0	1%	26	3%
Other transport	78	22%	34	37%	3.8	20%	3.7	1%	8.9	1%
Stationary evaporative sources							248	48%		
Biogenic							179	35%		
Total	352		92		20		517		800	

Trends of ozone and its precursors

Wakamatsu et al. [1996] reported that maximum hourly oxidant concentrations in the Osaka Bay area showed a decreasing trend during 1978–1990 (Figure 35). By contrast, increasing trends were observed in the inland Kyoto and Nara areas. The locations where daily maximum oxidant concentrations were observed tended to move further away from the emission area of the Osaka Bay area to the Kansai area. Concentrations of non-methane hydrocarbons (NMHCs) greatly decreased during the 1970s and 1980s, whereas NO_X concentrations did not show any significant trend. As a result, ratios of the NMHCs concentration to the NO_X concentration decreased. The decrease of NMHCs indicates a decrease in photochemical reactivity, which leads to a decrease in the photochemical reaction rate. By contrast, the increase in the NO_X/NMHCs ratio caused an increase in the O_3 formation potential. Higher concentrations of O_3 are usually observed near the shore in the morning, and as the sea-breeze penetrates inland, the high O_3 concentration area also

moves inland and O_3 concentration increases with time. Under these meteorological conditions, the decrease in the NMHCs/NO_X ratio caused the location where maximum O_3 is typically observed to move inland in the Kansai area.

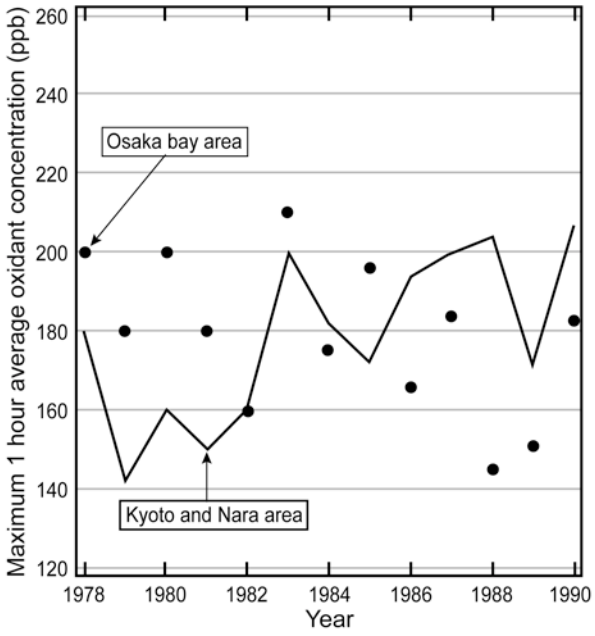

Figure 35 - Trends in maximum hourly oxidant concentrations during 1978–1990 in the Osaka Bay area (filled circles) and the Kyoto and Nara areas (line). Reproduced by permission of Elsevier from Wakamatsu, S., T. Ohara, and I. Uno (1996), Recent trends in precursor concentrations and oxidant distributions in the Tokyo and Osaka areas [*Wakamatsu et al.,* 1996]

Sadanaga et al. [2008] analyzed weekday–weekend differences in O_3, NO_X, and NMHCs concentrations in Tokyo and Osaka from 2001 to 2004. In Osaka, weekend O_3 concentrations were found to be greater than weekday concentrations during most periods, although the precursor concentrations were higher on weekdays. Weekday O_3 concentrations in Osaka decrease owing to the reaction of O_3 with NO, which results in higher O_3 but lower O_X (the sum of O_3 and NO_2) concentrations on weekdays than on weekends.

Research project

In spring, the Osaka area is generally located closer to the center of an anticyclone than Tokyo. Thus, the temperature is higher and photochemical air pollution occurs more often in Osaka than in Tokyo. Also, because the Osaka area is closer to the Asian continent than Tokyo, it is more strongly affected by the outflow of Asian pollutants. To investigate the structure and mechanism of springtime photochemical air pollution in the Osaka area, intensive measurements of O_3 from aircraft were carried out by the National Institute of Environmental Studies and the Osaka City Government in March of 2001 and 2003 [*Itano et al.*, 2006]. The concentrations of particles in the ultrafine to supermicron size range were also measured in March 2003 [*Hasegawa et al.*, 2007]. These studies showed that transboundary air pollution contributes significantly to the springtime O_3 [*Itano et al.*, 2006] and aerosol [*Hasegawa et al.*, 2007] concentrations in the Osaka Metropolitan Area in spring. By combining filter sampling of particulate matters in the boundary layer at 300–1300 m over Osaka Prefecture with a numerical simulation, *Hasegawa et al*. [2007] concluded that soil dust particles as well as black carbon (BC) particles are transported from the Asian continent to Japan (flight ME0307 in Figure 36).

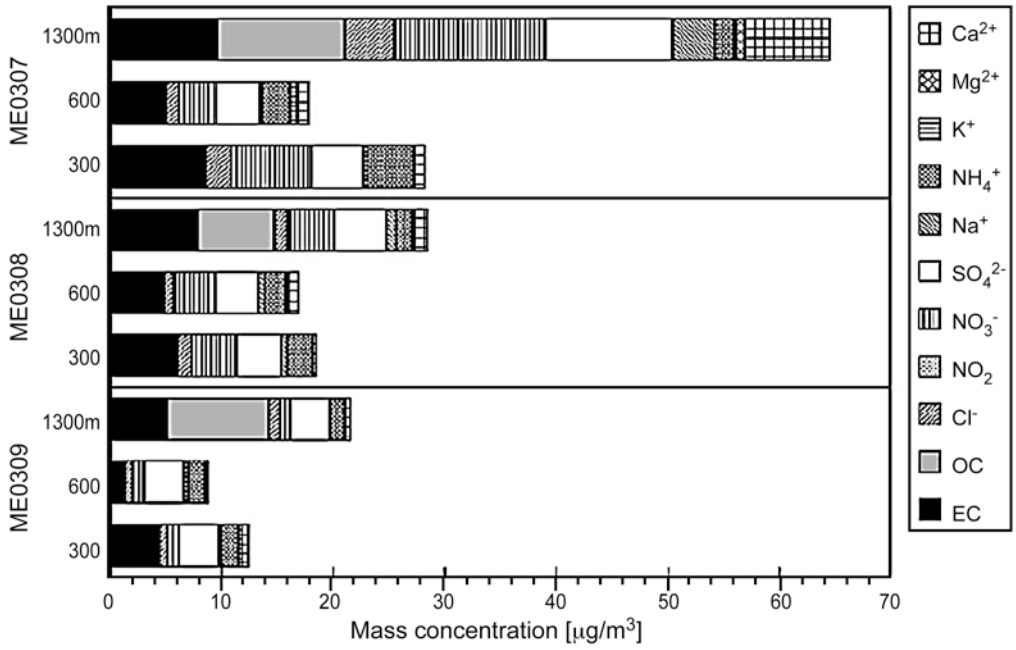

Figure 36 - Mass concentrations of carbonaceous and ionic species of total suspended particles during level flights made at 300, 600, and 1300 m on 19 March 2003. ME0307, ME0308, and ME0309 correspond to flights (~2 h) that took off at 08:30, 12:30, and 16:30 local time, respectively. Reproduced by permission of Elsevier from Hasegawa, S., S. Wakamatsu, T. Ohara, Y. Itano, K. Saitoh, M. Hayasaki, and S. Kobayashi (2007), Vertical profiles of ultrafine to supermicron particles measured by aircraft over Osaka metropolitan area in Japan [*Hasegawa et al.,* 2007]

Traffic-related air pollutants have been monitored near major roads at 10 sites in the Kanto, Chubu, and Kansai areas by the Ministry of the Environment of Japan (MOE) since 2005 [e.g., *Naser et al.*, 2009]. Nitrogen oxides (NO_X), suspended particulate matter (SPM, 100% cut-off aerodynamic diameter at 10 mm), $PM_{2.5}$, and BC were instantaneously and continuously monitored at four stations at various distances (about 5, 35, 70, and 150 m) from each of the target roads. *Naser et al*. [2009] used these measurement data and a Gaussian plume model to derive the emission factor of BC to SPM at roadside sites. They obtained good agreement between the observed and estimated ratios with a proportionality constant (BC/SPM) of 0.4, indicating that the same BC/SPM emission ratio.

MOE also measured VOCs (63 compounds) at four urban sites in the Kansai area throughout 2004 [*Sasaki and Sakamoto*, 2007]. Five samples were collected at each site in every season. The analysis, made by using a Chemical Mass Balance model, suggested that gasoline vapour and gasoline vehicle exhaust were significant sources of atmospheric VOCs.

Remaining problems

The effects of regulations of NO_X, VOCs, and PM emissions on O_3 and PM concentrations should be assessed by using the monitoring data from MOE and various research groups. Also, source contributions of O_3 and PM should be evaluated by analyzing these measurement data in combination with model simulation. Because the Osaka area is closer to the Asian continent than Tokyo, it is more strongly affected by the outflow of Asian pollutants. More systematic assessment of the effect of transboundary pollution on pollutants in the Kansai area is needed.

3.10 PEARL RIVER DELTA

Introduction and specific features of the city

The Pearl River Delta (PRD) situated at 21˚17´–23˚56´N and 111˚59´–115˚25´E is connected to the South China Sea. The PRD region covers 4.17×10^4 km^2 with a population of over 38 million. Most of the PRD region includes Guangdong Province and two Special Administrative Regions of

Hong Kong and Macau. The PRD region mainly consists of floodplains between the Nan Ling Mountains to the North and the South China Sea to the south, except in Hong Kong, where more than 50% of the land is mountainous area. The region has a subtropical marine monsoon climate with an annual average temperature of 22.8°C and average relative humidity of 68%. A large number of rainfalls occur annually due to its subtropical location and weather conditions and the annual rainfall at the urban area is over 1,600 mm. The annual variation of rainfall can greatly affect air quality in the whole PRD region because rainfall is a major pathway to remove air pollutants [*Wai and Tanner*, 2005; *Chan and Yao*, 2008].

The PRD region has been a leader in economic reform, economic development and urbanization in Guangdong and China under the "opening up" policy since 1978. The GDP in the PRD excluding Hong Kong and Macau increased at a rate of about 19.6% annually and reached more than 3.11×10^6 million yuan in 2007 (Figure 37) (Guangdong Statistical Yearbook). The vehicular population in Guangzhou and Shenzhen increases by 10% per year over the last decade. The expansion of the economy in this region causes higher demands for energy, mobility, and communications. As a consequence, both coal combustion and traffic exhaust cause serious photochemical smog and particulate pollution from the urban to regional scale.

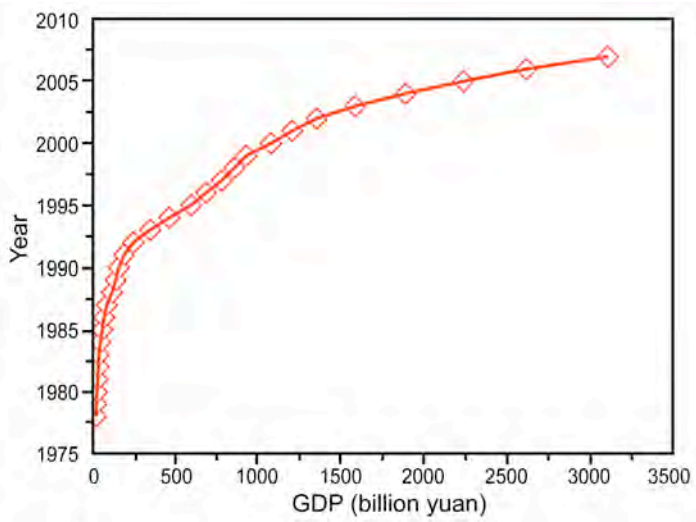

Figure 37 - GDP of Guangdong Province. (Guangdong Statistical Yearbook)

Emission sources of air pollutants

According to the report by *Chan and Yao* [2008], the annual SO_2 emission in Guangzhou exhibits a decreasing trend from 2000 to 2005, and was 14.9×10^4 tons in 2005. In Hong Kong, SO_2 emissions slightly increased from 5.65×10^4 in 1999 to 6.76×10^4 tons in 2002 and increased by about 40% in 2004 due to the increased emissions from public utilities. *Wang et al*. [2005] estimated that 32.9% of SO_2 concentrations were due to the power plant emissions and the transportation source was the main contributor to NO_X, CO, and O_3 concentrations accounting for 34.2%, 33.1%, and 17.8% of their total concentrations, respectively. NO_X concentrations were rather uniformly distributed in the different sites of Guangdong Province [*Zhang et al*., 2008a]. The annual NO_X emissions varied from 8.65 to 10.2×10^4 tons from 1999 to 2005 in Hong Kong. The public electricity generation accounted for about 50% of the total NO_X emissions in 2003 [*Chan and Yao*, 2008].

Limited data of both ambient VOCs concentrations and VOC emission inventories in the PRD region is available. Receptor models were used to apportion the source of ambient VOCs in the PRD region. The results showed that VOC emission sources include vehicle exhaust, liquid petroleum gas leakage, paint vapours, biomass and coal combustion. Vehicle exhaust was the largest source of VOCs, contributing to more than 50% of ambient VOCs at urban sites, and liquid petroleum gas leakage also played an important role [*Liu et al*., 2008].

Data available on air pollutants

Comprehensive and accurate information on air quality in the PRD is available from the PRD Regional Air Quality Monitoring Network, which has 16 automated air quality monitoring stations jointly established by the Guangdong Provincial Environmental Monitoring Centre and the Environmental Protection Department of the Hong Kong Special Administrative Region. Thirteen stations are operated in Guangdong while three stations are located in Hong Kong. The concentrations of air pollutants like SO_2, NO_X, NO_2, O_3, RSP (Respirable Suspended Particulate, PM_{10}), and CO are regularly and continuously monitored. On November 30, 2005, the PRD Regional Air Quality Monitoring Network allowed for daily publication of the Regional Air Quality Index on the Internet (http://www-app.gdepb.gov.cn/raqi3/RAQI_en.htm)

The Programme of Regional Integrated Experiments of Air Quality over Pearl River Delta (PRIDE-PRD) provides a comprehensive understanding of the air pollutants. More species including NO, NO_2, NO_Y, VOCs, SO_2, CO, CO_2, O_3, HONO, and $PM_{2.5}$ as well as the size distribution, chemical composition, and optical property of particles were measured. These data can help to appraise the air quality situation and pollution problems in the PRD region. The results are published in a special issue of *Atmospheric Environment* (42(25), 2008) for PRIDE-PRD 2004 as well as on *Atmos. Chem. Phys.* for PRIDE-PRD 2006.

The status and trend of the pollution

Previous studies showed that PRD is suffering from a serious air quality degradation problem, which is characterized by the coexistence of the high levels of O_3, $PM_{2.5}$, NO_X, and SO_2 (*Zhang et al.,* 2008a). High aerosol concentration events are observed in the MODIS aerosol optical depth data (Figure 38) [*Wu et al*., 2005]. The high concentration of aerosol particles is a major reason for the low visibility and haze day occurrence remain very high, e.g. around 150 days year^{-1} between 1980 and 2006 (Figure 39) [*Wu et al*., 2005; *Deng et al*., 2008].

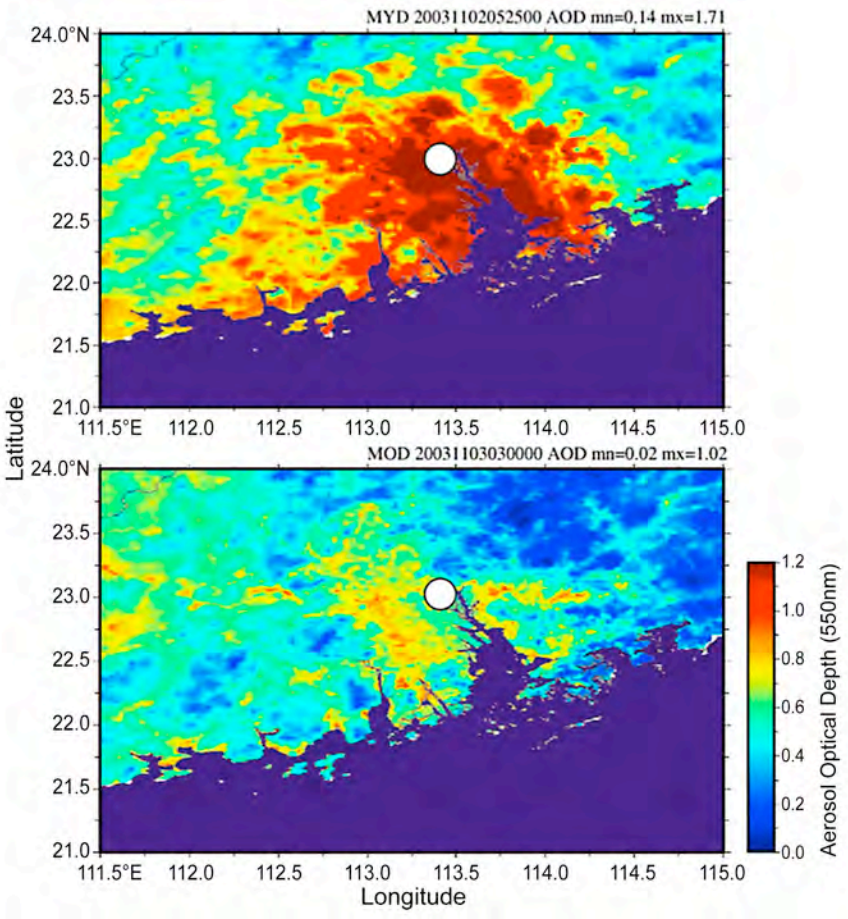

Figure 38 - MODIS measurements of aerosol optical depth form on 2 November (upper panel) and 3rd November (lower panel) 2003 at 8am local time. The white circle indicates the location of Guangzhou [*Wu et al*., 2005]

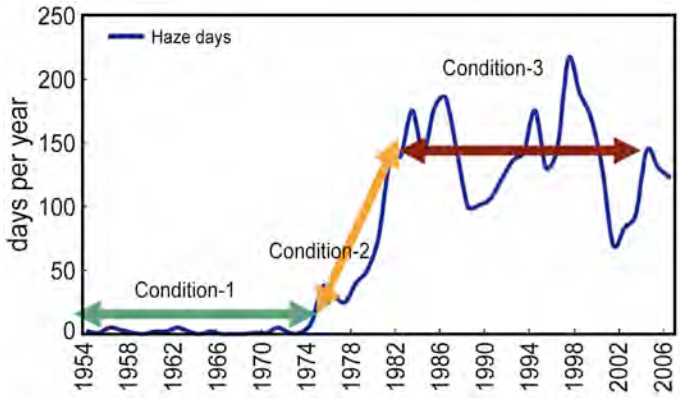

Figure 39 - A 52-year observation of haze day occurrences in Guangzhou [*Deng et al.*, 2008]

As seen from Figure 40, the annual average SO_2 concentration in Guangzhou and in Hong Kong increased from 1999 to 2005. At the same time, decreases in CO concentrations were observed at Guangzhou and Tsuen Wan, but not at other sites in Hong Kong. There was no increase in the annual averages of O_3 and (NO_2+O_3) concentrations at the three sites in Hong Kong (Figure 41) [*Chan and Yao*, 2008]. The average O_3 exhibited maximum concentrations during the fall in Hong Kong and O_3 levels increased during the days with intense solar radiation, light winds, and the presence of unique wind convergence circulation [*Wang et al*, 2001]. The experiments revealed O_3 precursor emissions from vehicle exhaust are likely one of the reasons for high O_3 concentrations in the PRD and a regional air quality model indicated the ozone formation was controlled by ambient VOCs [*Zhang et al*., 2008a and b].

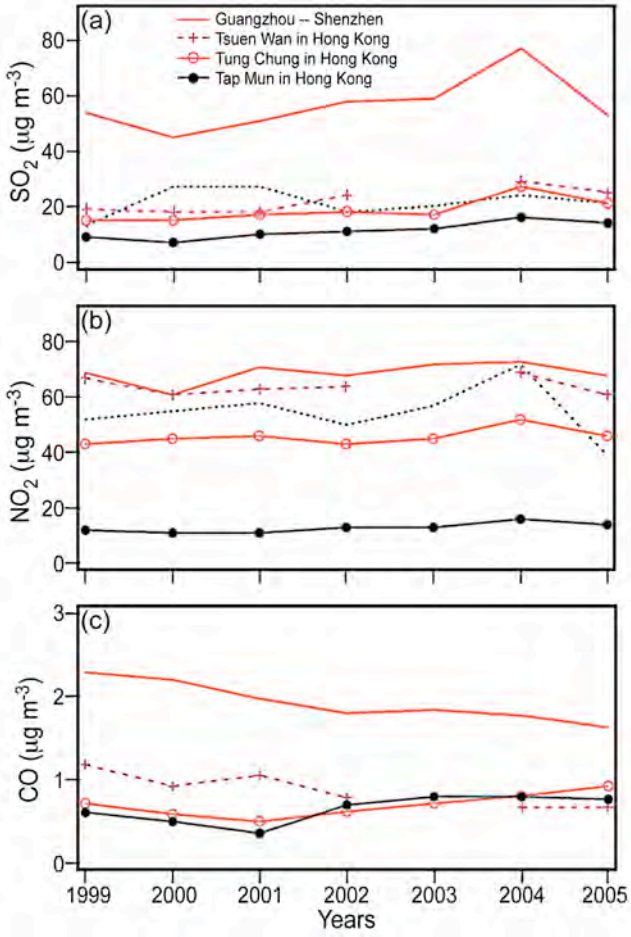

Figure 40 - (a) SO_2, (b) NO_2, and (c) CO concentrations in the PRD region from 1995 to 2005 [*Chen and Yao*, 2008]

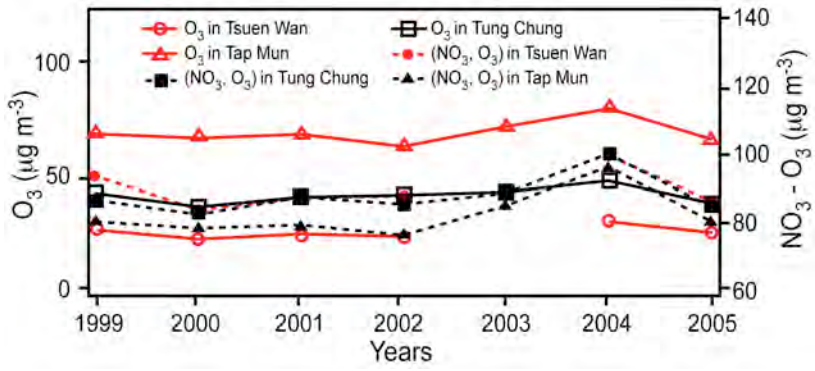

Figure 41 - O_3 and (NO_2 +O_3) concentrations in the PRD region from 1999 to 2005 [*Chen and Yao*, 2008]

Relationships or the trends to regulations

The Hong Kong and Pearl River Delta Pilot Air Monitoring Project was initiated in May 2002. The Project aimed to study the cause of ground-level O_3 pollution in Hong Kong and to study the characteristics of fine particles ($PM_{2.5}$) in the Pearl River Delta. The project aims to reduce regional emissions of SO_2, NO_X, RSP, and VOCs by 40%, 20%, 55% and 55%, respectively, by 2010 in PRD region, using 1997 as a baseline year. Enhanced control measures of the HKSAR is implemented, including encouragement to replace light diesel buses with ones using clean fuel, reduce VOC emissions from the printing process and consumer products, reduce emissions from power stations, and enhance energy efficiency of buildings. In 2007, the Environmental Protection Bureau (EPD) of Hong Kong and Guangdong announced the implementation framework for the Emissions trading Plot Scheme for Thermal Power Plants in the PRD region.

The results for 2008 from the regional Air Quality Monitoring Network indicated that the air quality in PRD region showed improvement in 2008. The overall average annual concentration of SO_2 and RSP decreased by 19% and 11%, respectively, as compared to the 2007 levels. The annual averages of NO_2 and O_3 remained the same. (http://sc.info.gov.hk/gb/www.epd.gov.hk/epd/textonly/english/news_events/press/press_090422a.html). The SO_2 levels in Guangzhou decreased from 0.077 mg m^{-3} in 2004 to 0.046 mg m^{-3} in 2008 and were lower than the national ambient air quality standard for SO_2 at grade II (annual average 60 μg m^{-3}) except in 2004. The annual average NO_2 concentration had a slightly decrease during 2000–2008 (Figure 42).

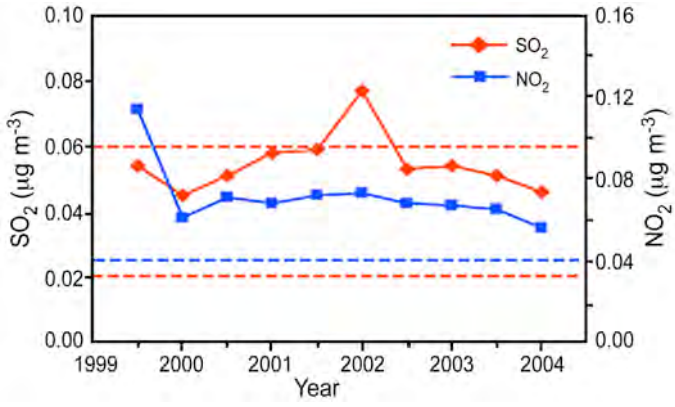

Figure 42 - The change of the annual average concentration of SO_2 and NO_2 in Guangzhou from 1999 to 2007 [*Guangzhou Environmental Buttetion*, 1999–2008 Guangzhou Environmental Protection Bureau, Guangzhou, www.gzepb.gov.cn]. The red dashed up and bottom lines indicate the grad II (60 mg/m^3) and grad I (20 mg/m^3) Chinese National Ambient Air Standards for SO_2, respectively. The blue dashed line is the grad I/II (40 mg/m^3) Chinese National Ambient Air Standard for NO_2

Climatic change issues

The regional climate in the PRD is sensitive to changes from urban expansion (*Lin et al.,* 2007). Climate data of south China indicates the rapid urbanization contribute a warming trend in winter temperature by approximately 0.32°C/10y from 1960 to 1996 (*Liang and Wu*, 1999). Figure 43 shows that that the annual mean temperature had an increasing trend from 1990 to 2000 (*Chen et al.,* 2006). The urban heat island (UHI) effect is very important to global change. The UHI effect has become more prominent in areas of rapid urbanization (e.g., Shenzhen) in the PRD region. It's reported that the total annual contribution of land use/cover changes in Shenzhen (0.042 °C) to increases in temperature is greater than that in the PRD (0.027 °C) (*Chen et al*., 2006). CO_2 of coal-fired power units remained quite constant at about 84–88% of the total GHG emissions from 1990 to 2005 (*Leung et al*., 2009).

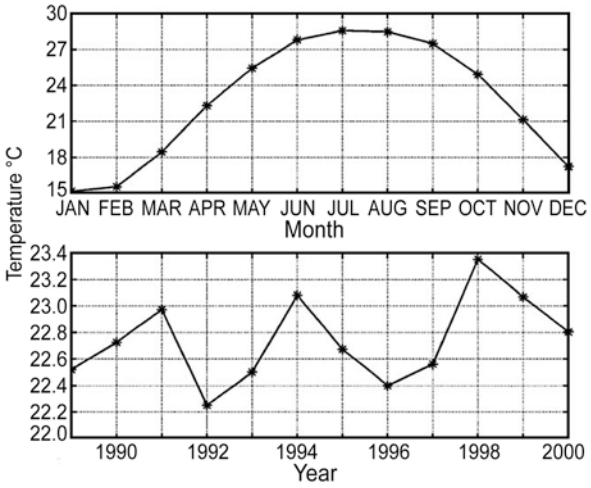

Figure 43 - Fluctuations of yearly and monthly mean temperature form 1989 to 2000 in the PRD region [*Chen et al.*, 2006]

Research projects on air quality of PRD

From 2005 to 2007, a project on the effect of atmospheric haze on the deterioration of visibility over the PRD was implemented to study the formation mechanism and forecast method of haze. The Programme of Regional Integrated Experiment of Air Quality over Pearl River Delta (PRIDE-PRD) was developed by Peking University. The objective of PRIDE-PRD is to investigate major air pollutants and provide a comprehensive understanding of the air pollution in the PRD. Gaseous pollutants (SO_2, O_3, and NO_X) and particulate matter (PM_{10} and $PM_{2.5}$) were measured in 2004. The study measured high levels of O_3 that exceed China's National Standard at grade II (hourly maximum 160 μg/m^3), throughout the PRD region. The research of PRIDE-PRD 2006 indicated that hydroperoxides play an important role in the formation of secondary sulphate in the aerosol phase. A major project "Synthesized prevention techniques for air pollution complex and integrated demonstration in key city-cluster region" (3C-STAR, 2006–2010) was funded to build up the capacity of regional air pollution control and to establish regional coordination mechanism.

Problems remaining

Reduced pollutant concentrations show that both the Guangzhou governments and Hong Kong are committed emission reduction efforts for further improving the regional air quality. But fine particle is still a serious pollution problem as well as O_3. Aircraft measurements showed a high level of O_3 pollution in the PRD region (*Wang et al*., 2008). The previous study from PRD campaigns (PRD2004 and PRD2006) indicated concurrent high concentrations of O_3 and $PM_{2.5}$ throughout the PRD region. Such high concentrations of both primary and secondary pollutants cause an "air pollution complex" problem. Furthermore the high concentration of aerosol particles is a major cause for the low visibility and the grey haze that has been a regional problem of concern. Both the Guangzhou governments and Hong Kong are committed to continuing to enhance emission reduction efforts to further improve regional air quality.

3.11 SEOUL, KOREA

Introduction and Specific Features of the City

Seoul is a megacity located at the heart of the Korean Peninsula. It stretches from 37° 41' N to 37° 25' latitude and from 126° 47' E to 127° 11' longitude at the level of the temperate zone. The city is also located in the middle of several major Northeast Asian Metropolises, such as Tokyo, Beijing, and Shanghai. It is at the eastern end of the Asian land mass and therefore experiences coastal climate as well. Several mountain peaks of historical significance surround the city of up to 500 meters or more above sea level. The major river that bisects Seoul into a North and South region is the Hangang River, the location of post-Korean War economic development.

Surrounding Seoul are Incheon Metropolitan City and Gyeonggi province, which are emerging megacities as well (Figure 44). Combined, this region comprises 12 % of the area of Korea, yet holds 47% of Korea's population, ranking as the second most populated megacity in the world (next to Tokyo, Japan), with a combined population of approximately 24 million as of 2008. The population explosion in Seoul was due to the city's rapid urbanization and can be traced back to before Seoul hosted the 1988 Summer Olympics. South Korean Industry was already in the grip of a technological revolution before it hosted the Olympics. GDP expanded by around 10% per year from the mid 1980's to the early 1990s due to the creation of one of the largest steel plants in the world (POSCO), drastic demand in foreign and local car sales (Hyundai, Kia, DaeWoo), revolutionizing the electronics industry (Samsung, LG), and establishment of numerous local manufacturing companies. Opportunities for modern livelihood, employment, and a better life lead to migration of people into the region.

As skyscrapers, condominium, and industrial plants established, Seoul developed into a heat island like other urban centres around the world. Also, like other urban cities, Seoul has experienced its fair share of pollution as it has developed. With this, the City has enacted several policies to counter pollution and is one of the earliest respondents to clean air initiatives among the major cities in Northeast Asia.

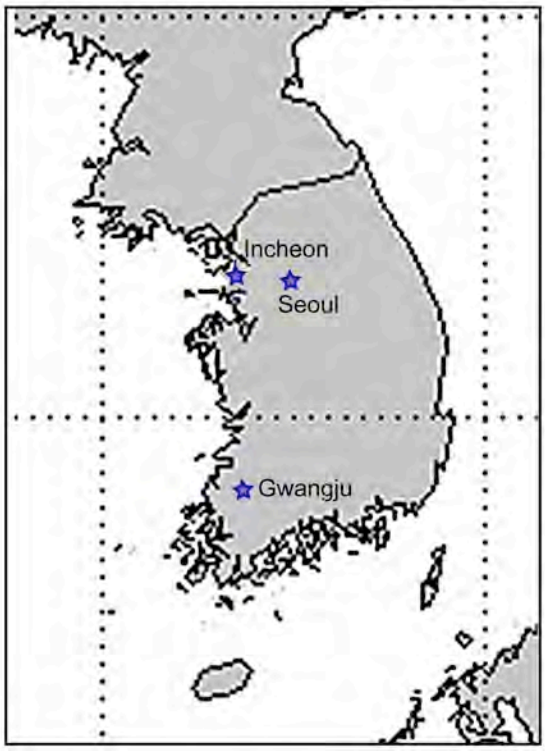

Figure 44 - Map of South Korea and its Megacity Study sites (Seoul, Incheon, and Gwangju)

Emission sources of air pollutants

During the post-Korean War economic development period, South Korea's urban air quality structure was characterized by high-energy intensity associated with primarily fossil fuel energy consumption. As a consequence, urban air pollution caused by the rapid proliferation of cars, growth in heavy and chemical industries, the increasing severity of traffic congestion, and other urban issues stressed the cities during South Korea's high-growth eras. The Clean Air Conservation Act of 1990 was in its initial stage during the post-Korean War economic development. Until 1994, The United Nations Environment Programme (UNEP) reported Seoul to have serious problems with high sulphur dioxide and lesser, though increasing, problems with nitrogen oxides (1). Poor air quality is evident from high PM_{10}, SO_2, and CO concentration since 1991. Greenhouse gas emissions have also been a concern, both regionally and globally. Korea has made various efforts to improve the air quality through, among others, supplying clean energy (natural gas, low-sulphur fuel, etc.) and strengthening emission standards for industries and motor vehicles. These efforts markedly reduced air pollutant emission for SO_2 and CO in major cities. However, the air pollution in Seoul is higher than any other metropolis in the nation, requiring special management. To date, the concentration of vehicles already exceeds the environmental capacity. In 2002, road transport accounted for 77% of all PM_{10} (and $PM_{2.5}$) emissions in Seoul (2, 3). Moreover, periodical Asian dusts storms and haze events add to the existing air quality problem by aggravating PM_{10} levels, making the atmosphere more deleterious to health. Increasing growth of air pollution related sickness and death has become a major issue. As of 2003, the number of premature deaths due to PM_{10} in the Seoul Metropolitan area reached 10,000 per year (4) and research data (5) has estimated the social costs to be 8 billion USD.

Data available on air pollutants

Since the implementation of the Clean Air Conservation Act, the Ministry of Environment (MOE) has established city air quality monitoring sites all throughout Korea. Annual Mean Criteria Pollutant data starting from 1991 are available for public use. In 1991, there were 20 city air quality monitoring stations in Korea that measure PM_{10}, SO_2, O_3, NO_2, and CO. As of 2007, the city monitoring stations have increased to 226. The Ministry on Environment has established and operated Air Pollution Observation Networks that can be classified into six types: suburban atmosphere monitoring, density monitoring of various substances, hazardous air pollutant monitoring, photochemical air pollutants monitoring, acidic deposition monitoring, and earth atmosphere observation, while municipal and provincial governments operate air pollution observations that are classified into four types: urban atmosphere monitoring, roadside atmosphere monitoring, monitoring of heavy metal in the atmosphere, and visibility observation. City atmosphere observation networks represent 55% of the total observation networks. The data collected by the nationwide air pollution observation network is processed by the National Atmosphere Pollution Information Management System (NAMIS, collaboratively operated by Ministry of Environment and the Environment Management Corporation) and is utilized as a resource for regulation of the atmosphere in order to understand air quality, analyze the effectiveness of environmental regulations, and preserve the health of citizens. Moreover, among the data collected by National Atmosphere Pollution Information Management System, the change in concentration of five standard air pollutant (SO_2, CO, NO_2, PM_{10} and O_3) is disclosed to the public in real time on the air pollution homepage (www.airkorea.or.kr).

Data status and the trend of pollution

The concentration trends for SO_2, NO_2, CO, PM_{10} and Pb were evaluated based on the published data by the Ministry of Environment, Republic of Korea (6), the latest being in 2008, with 2007 as the most recent data. The annual average SO_2 levels has dramatically decreased by 20 ppbv from 1991 to 1993, and has been decreasing by 2.5 ppbv per year from 1993 to 2000 (Figure 45). It has maintained an annual average value of 5 ppbv from 2001 to 2006, though it has increased by 1 ppbv in 2007. The annual CO levels has also decreased by 0.11 ppmv per year from 1991 to 2002 (Figure 46).

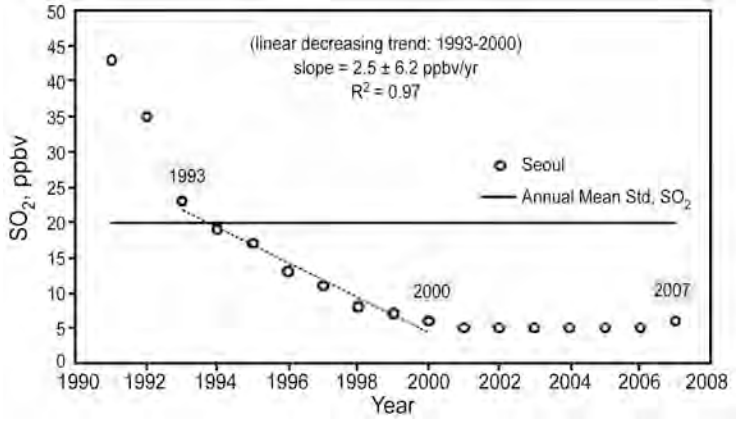

Figure 45 - Annual Mean concentrations of SO_2 in Seoul (1991-2007)

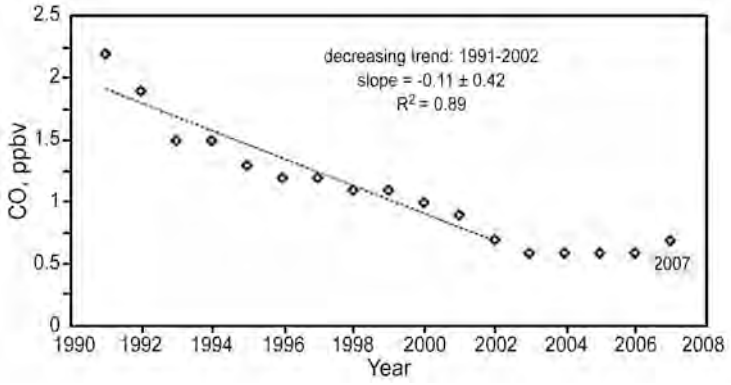

Figure 46 - Annual mean concentrations of CO in Seoul (1991-2007)

It was maintained at 0.6 ppmv from 2003 to 2006 and has increased by 0.1 ppmv in 2007. From 1991 to 2007, the decrease in annual mean NO_2 levels is strongly followed by an increase in annual mean O_3 levels, except in 2006-2007 (Figure 47). In particular, the annual mean increase of 1.21 ppbv/yr by NO_2 was followed by the linear decreasing trend of 0.57 ppbv/yr by O_3 since 1998 to 2004. But on the following years, both NO_2 and O_3 trends has been increasing, which may be attributed to the increasing complexity of the urban atmosphere.

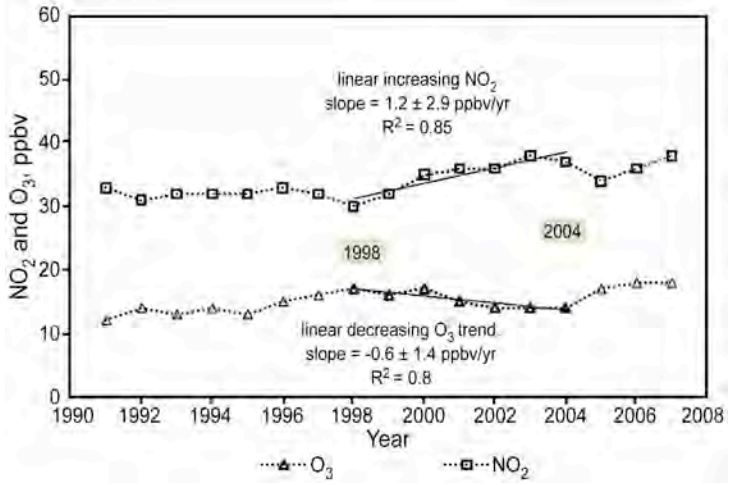

Figure 47 - Annual mean concentrations of NO_2 and O_3 in Seoul (1991-2007)

In the case of PM_{10}, a series of stringency has been implemented since the annual mean standard was implemented in 1993. The annual mean dropped from 80 μg m^{-3} in 1993, to 70 μg m^{-3} in 2001 and lastly, to 50 μg m^{-3} in 2007 (Figure 49). The annual mean standard for PM_{10} was exceeded in 2001-2002 and in 2007.

Particularly in 2002, the annual mean PM_{10} concentration has almost levelled off with the 1995 PM_{10}, mainly due to the extraordinarily severe Asian dust event that happened on that year. For Pb, the annual mean levels have always been met, and the trend linearly decreased from 1993 to 2004 at 0.01 μg m^{-3} yr^{-1} (Figure 50). This is expected trend due to the worldwide shift to unleaded gasoline.

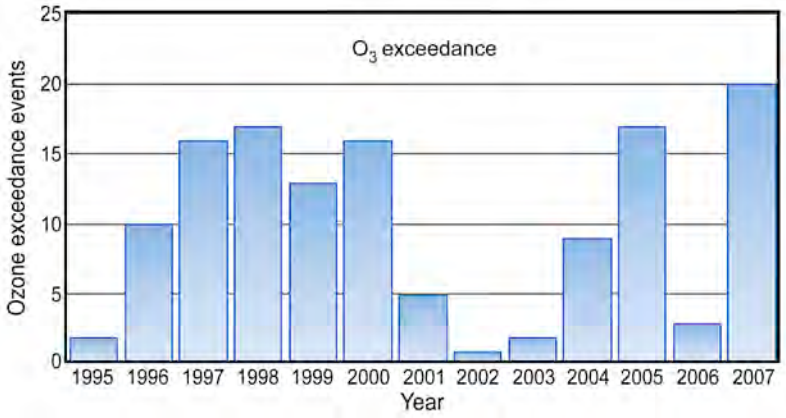

Figure 48 - Frequency of Ozone exceedances per year in Seoul

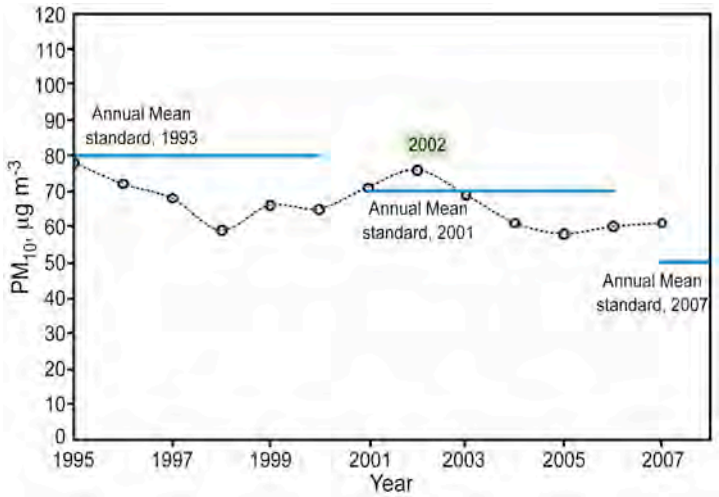

Figure 49 - Annual mean concentrations of PM_{10} in Seoul (1995-2007)

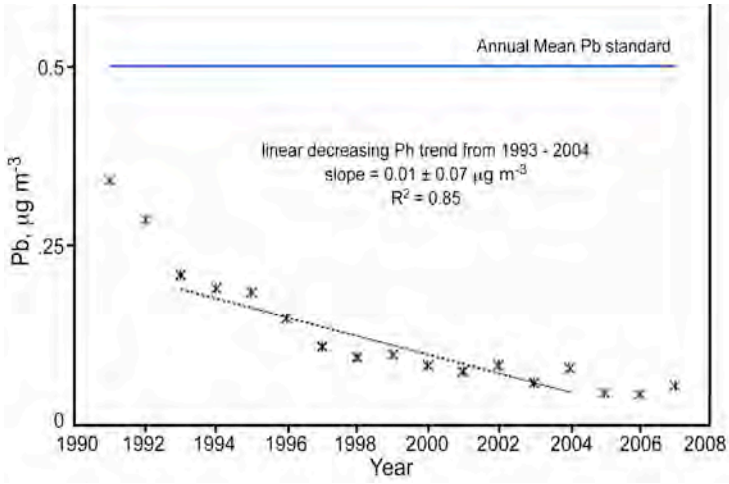

Figure 50 - Annual mean concentrations of Pb in Seoul (1991-2007)

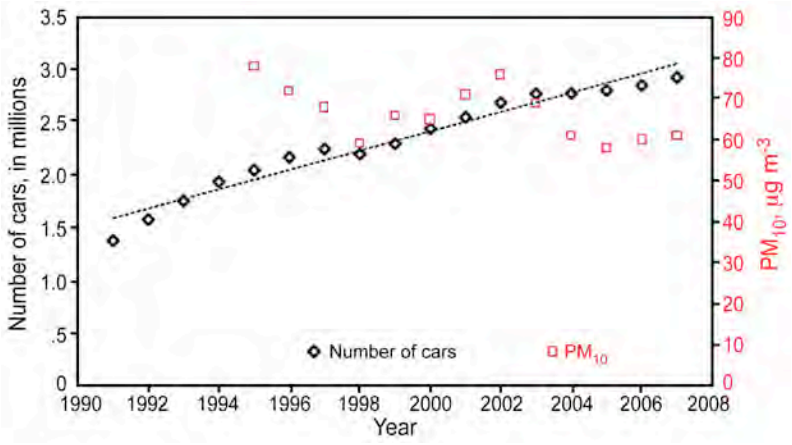

Figure 51 - Increasing trend in Number of Vehicles (1991-2007), in relation to the trend in PM_{10} concentration

Relationship of the trends to regulations

Since the enactment of the Clean Air Conservation Act in 1990, most of the criteria pollutants have decreased dramatically in order to comply with local standards. The SO_2 annual average in Seoul has experienced improvement from 43 ppbv in 1991 to nearly 6 ppbv in 2007 (Figure 45). As seen in Figure 45, the annual WHO air quality standard for SO_2 of 19 ppbv was not met in Seoul until 1995. Since then, SO_2 levels have gradually decreased until it levelled off at 5 ppbv. The same is true with CO levels, but although the two pollutants have experienced a remarkable decrease through the years, PM concentrations experienced a steady increase from 1998 to 2002. Although PM concentrations were declining following the implementation of the Clean Air Conservation Act until 1998, when rising vehicle emissions and dust and sandstorms (DSS) originating from Asian continent increase PM concentrations. In line with this, the Seoul metropolitan area Air Quality Management Plan (SAMQP), which is now in its second phases, was enacted in 1999 to reduce emission levels of major pollutants such as PM_{10} and NO_X to half their current levels by 2014, to improve the metropolitan air quality to the level of advanced countries, and mandates a total local discharge amount management system, a transferable discharge permit system, and the mandatory purchase of low emission vehicles as its major components. Although the general trend on the number of vehicles has been increasing, the annual PM_{10} level has declined as of 2003 (Figure 51), which is believed to be a result of SAMQP's comprehensive policy for air quality improvement. In general, PM_{10} concentrations increase temporarily during the spring

due to yellow sand and low relative humidity. In the case of O_3 and NO_2 (Figure 47), the annual mean has started to increase due to the increase in the number of automobiles in Seoul during the last five years. However, the frequency of ozone exceedances (Figure 48) events is more important rather than changes in annual average of ozone concentrations because exposure to high ozone concentrations during the short term poses a threat to humans. The status of ozone exceedance events during the short term (0.1 ppm/h), which is not shown here, shows that exceedance events have been increasing, from 343 exceedances at 49 stations nationwide in 1996 to 1,090 at 220 stations in 2006 (9).

Climatic change issues

The Korean peninsula has experienced its fare share of climate change trends along with the rest of the world. It has experienced a 0.23°C rise in annual mean temperature per decade, an increasing diurnal range, and more frequent heavy rain events in the recent years (10, 11). The increase in average temperature and precipitation over the Korean peninsula has widened the regional and seasonal weather differences, making the weather more like a subtropical climate.

In terms of emission inventories, the National Institute of Environmental Research conducts regular inventories of several air pollutants by emission sources (point, area, and mobile sources) for Seoul. Emission deductions of major air pollutants (NO_X, SO_X, PM_{10} and VOCs) are better achieved when mobile sources are more augmented (9). Moreover, to evaluate the air quality impacts of these emission reductions, the EPA's Models-3/Community Multi-scale Air Quality (CMAQ) modelling system is used. The impact of changes in air quality on human health is then estimated using EPA's Environmental Benefits Mapping and Analysis Programme (BenMAP). On the average, health benefits could be more substantial if pollutants other than PM_{10} are given attention. Furthermore, aside from CO_2, other greenhouse gases such as NO_X and CH_4 should also be analyzed for emission closure because they also have a high potential for global warming.

Research Projects on Air quality of Seoul Metropolitan City and emerging megacities

The MOE aimed to lower ozone concentrations by reducing two major sources, volatile organic compounds (VOCs) and nitrogen oxides (NO_X) using a stepwise improvement of urban air quality. The need for further air quality improvement led the MOE to conducted an initial performance assessment of measures for improving air quality of Seoul, including the attachment of exhaust emission reduction devices, upgrading the quality of paint supplied in the city, reducing non point pollution sources of VOCs, and attaching additional devises for VOC retrieval. The metropolitan atmospheric environment management plan aimed to reduce the air pollution emissions in the metropolitan area to half of 2001 levels by 2004, while reducing PM_{10} density to 40 $\mu g\ m^{-3}$, and NO_2 to 22 ppb. The provision of eco-friendly fuels was expanded through improving industrial heavy oil, increasing provisions of low-sulphur gasoline, and promoting the use of biodiesel for automobiles running on diesel. Lastly, the MOE developed "city climate maps" which show at a glance green areas, hot spots of air pollution, as well as wind flow through urban street canyons to alleviate the urban heat island phenomenon.

The MOE has also addressed the emerging air quality problems associated with long-range transboundary air pollution, revising the Clean Air Conservation Act, which became a legal basis for formulating a "Dust and Sandstorms (DSS) countermeasure commission" and a "comprehensive measure for preventing damage by DSS". Stations were built to conduct research on the risk assessment of particulate matter and other harmful substances in dust and sandstorms. Furthermore, Korea, China, and Japan have been cooperating since 1996 through the joint Research on Long Range Transboundary Air Pollutants in the Northeast Asia to carry out a joint study to measure the amount of long-range transboundary air pollutants (especially sulphur and nitrogen) into Northeast Asia.

The budget allotment for air quality improvement by the MOE increased from 10.1% in 2003 to 10.8% in 2007 (6). The budget in 2007 for air quality was increased by 5.8% compared to the 2006 budget allotment. This budget is mainly allocated for the project on the improvement of metropolitan air quality, countermeasures for climate change, and measures for the prevention and control of DSS.

Aside from Seoul, intensive research studies on chemical components of urban air were conducted in adjacent urban cities (i.e., Incheon) during August to October 2004 to supplement the air quality information in the Seoul Metropolitan City. Using a high time resolution measurement method for organic carbon and elemental carbon, it was identified that elevated concentration events were brought about by transported aerosols in the direction originating from high traffic highways (12). *Lim,* 2005 retrieved the NO_2 using Imaging Differential Absorption Sprectroscopy (I-DOAS) Technique to understand the rate of plume emission coming from two stacks of a thermal power plant near the Seoul Metropolitan City area (13). In 2005, *Kim et al* studied the fine particulate matter characteristics and its impact on visibility in Seoul and Incheon (14).

In emerging megacities such as Gwangju, South Korea, Asian DSS and haze events are also compromising air quality. Several studies have been conducted to study the aerosol characteristics and surface radiative forcing components during a dust outbreak in Gwangju (15) as well as some aerosol optical properties and satellite parameters (16, 17, 18). For instance, during a severe haze and smoke episodes in Gwangju, aerosol optical properties and microphysical parameters were determined from the ground using a multi wavelength Raman Lidar. Two different aerosol types were identified based on the variability of optical characteristics for different air mass conditions and indicated that there is a distinct light absorbing characteristics (based on Single scattering albedo , SSA) for different haze aerosols from China (haze) and from Siberia (forest-fire smoke). The features of two air mass characteristics (haze and smoke) are now understood in terms of their optical properties and microphysical parameters (16).

Problems remaining

Much effort has to be made to improve air quality in metropolitan areas (Seoul City, Gyeonggi province, and other metropolitan cities) in order to meet the more stringent air quality standards that were applied in 2007 to Particulate Matters (PM_{10} : from 70 μg m^{-3} to 50 μg m^{-3}) and Nitrogen dioxide (NO_2 : from 50 ppb to 30 ppb). Although the MOE has launched an emission cap system, the effectiveness will not be seen for a few more years. Still, there is much to focus on, especially the inventory of VOCs in the Seoul urban area, which is notorious for serious air pollution issues. Although the Clean Air Conservation Act was enacted in 1990, certain revisions were done in order to tailor-fit it to the provincial and municipal jurisdictions for better implementation and to Korean atmospheric conditions. Lastly, the severity of the effect DSS and other long-range transported pollutants needs to be further studied in order to create more efficient countermeasures during the events.

5.12 SHANGHAI, CHINA

Introduction

Shanghai is a coastal megacity in the southeast of China and is located between 30 °40' N to 31° 53' N latitude and between 120°51' and 122°12' longitude. The administration area of Shanghai is 6340.5 km^2, with a width of 100 km in the east-west direction and a length of 120 km in the south-north direction. Shanghai is in the front of the alluvial plain of the Yangtze River Delta. The city is embraced by a river basin: the East Sea lies to the east, the Yangtze River lies to the north, the Hangzhou Bay to the south and Jiangsu and Zhejiang provinces border Shanghai to the west. Shanghai experiences a subtropical monsoon climate with plenty of rainfall. Shanghai's geographic location and climate condition favour the dispersion of air pollutants.

As one of the first regions in China to initiate a market economy, Shanghai has been experiencing very rapid economic growth since the 1980s. The GDP of Shanghai has been increasing at a rate of more than 10%/yr within the last 15 years and the GDP per capita exceeded 10,000 USD for the first time in 2008 [*Shanghai Statistical Yearbook*, 2009]. The energy consumption increased at an average rate of 6% per year correspondingly, reaching 10% per year the past three years. The increase in energy consumption resulted from rapid economic growth, which is followed by the dramatic increase of air pollutant emissions, and deteriorated air quality in Shanghai. Additionally, the increase of vehicles has also played a significant role in decreasing the air quality in Shanghai.

Emission inventories of major air pollutants

The emission of SO_2 and particulate matter, the vehicle number, and GDP of Shanghai over the past years are shown as Figure 52. The emission of SO_2 seemed to be stable in the past with small fluctuations, even though GDP was growing. However, emission of SO_2 has increased since the beginning of the 21st century. From 2003 to 2007 emission of SO_2 increased by 14.3% to 497.8 kt. The emission of particles decreased gradually in past years. From 2003 to 2007, emission of particles decreased by 50.8% to 106 kt. In addition, the vehicle number increased dramatically with the large growth of GDP in the last decades, especially since this century, which resulted in the increase in emissions of NO_X and volatile organic compounds (VOCs). According to the study by *Chen et al*. [2009], the NO_X emission had increased to 599.7 kt in 2007, and the vehicular emission accounted for 14%. More importantly, the contribution of vehicular emission to NO_X in urban areas was much larger than that of the whole city, accounting for 40.8%. Obviously, vehicular emission is the largest source of NO_X in urban area of Shanghai. The VOCs emission was 597.4 kt in 2007 and vehicle emissions accounted for 14.5%, whereas in the case of urban area, the vehicular emission accounted for 18.8%.

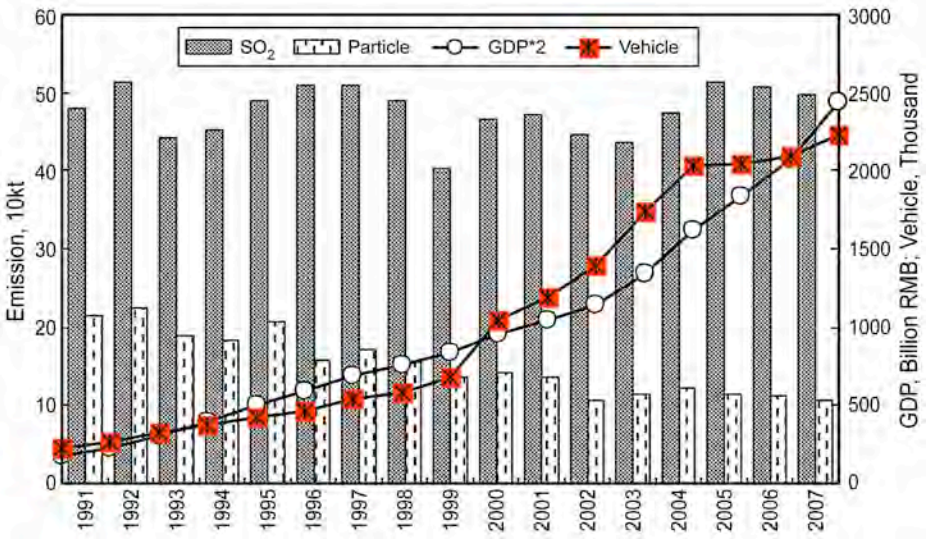

Figure 52 - Variation of emissions of SO_2 and particles, the vehicle number, and GDP in Shanghai from 1991 to 2007 [*Municipal Environmental Protection Bureau*, The communiqué on the Environmental Status of Shanghai City, 1991~2008]

Based on the study by *Chen et al*. [2009], the major SO_2 emission sources are power plants, dispersed combustion processes of industry, and industry furnaces, accounting for 46%, 27%, and 10%, respectively. Similar to SO_2, power plants are also the most important source of NO_X (34%). Dispersed combustion processes of industry and vehicle emissions contribute 18% and 17% to the total NO_X emissions, respectively. In the case of PM_{10}, the largest source is dust resuspension from motorways, accounting for 44%, followed by uncovered yard, dispersed combustion processes of industry, and power plants. Additionally, solvents and other product contributes the largest portion of the VOC emissions, accounting for 28%, followed by industrial processes, 23%, and vehicle emissions, 15%.

In summary, power plants, industry, and vehicular emissions are the major sources of air pollutants in Shanghai and the contribution of vehicular emissions to NO_X and VOCs emissions is much larger in urban area than in rural area. Beside these pollution sources, dust is one of the most important sources of PM_{10}.

The ambient air quality

The profile of annual average concentrations of SO_2, NO_2, CO, and PM_{10} in Shanghai from 2002 to 2007 is shown in Figure 53. The SO_2 annual average concentration was lower than the national ambient air quality standard (NAAQS) for SO_2 at grade II (60 μg m^{-3}) during these six years.

However, the SO_2 annual average concentration kept on increasing gradually from 2002 and the concentration in 2007 increased by 83.3% of the 2002. The trend of SO_2 pollution should be improved by 2010, with the completion of the desulphurization policy of the 11th-5-years-plan of China. The NO_2 annual average concentration seems to be stable between 2002 and 2007, and is lower than the NAAQS for NO_2 at grade II (80 μg m^{-3}). The CO annual average concentration decreased during this period except in 2005 when the concentration reached 1.81 mg m^{-3}, much higher than that in other years. The CO annual average concentration in 2007 had decreased 15.9% compared to 2002 levels. The PM_{10} annual average concentration seems almost to be decreasing, except that there was a slight increase from 2006 to 2007, which could have been due to the sharp increase of construction for the 2010 EXPO. The PM_{10} annual average concentration is lower than the NAAQS for PM_{10} at grade II (100 μg m^{-3}). Obviously, particle control policy has been effective.

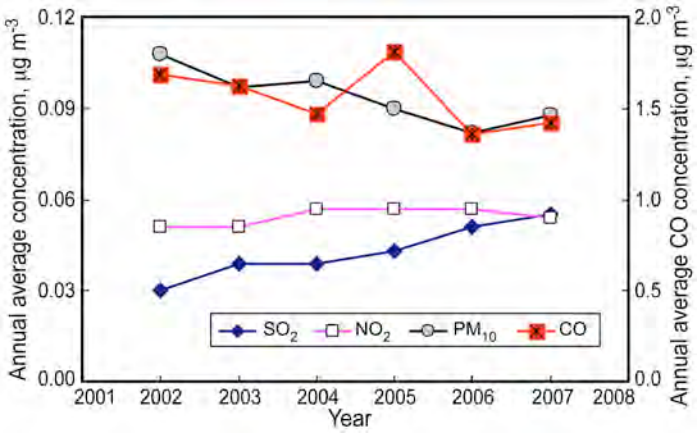

Figure 53 - Profile of the annual average concentrations of primary pollutants in Shanghai from 2002 to 2007 [*Municipal Environmental Protection Bureau*, The communiqué on the Environmental Status of Shanghai City, 2002~2008]

The Shanghai air pollution index (API) at grade II (100) demonstrates the improvement in air quality. The days with good and excellent rate of API was more than 85% of the year, continuously for the past six years and the API did not exceed 100 for 324 days in 2008. However, the API is determined by the concentration of SO_2, NO_2, and PM_{10}, which does not reflect the secondary pollutant, which are even more important than the primary pollutants in megacities with rapid economic development and sharp increases in number of vehicles. The high level of ozone during the summer, poor visibility, and the high frequency of acid rain, almost all result from the secondary pollutants [*Chen et al.,* 2009].

The 2007 monthly average ozone concentration in Shanghai at 14:00 are shown in Figure 54. Ozone pollution is very high during the summer, with the highest hourly average concentration of ozone reaching 380 μg m^{-3}, which is much higher than the NAAQS at grade II of 200 μg m^{-3}. Based on the hourly data in July 2007, the concentration of ozone exceeded 200 μg m^{-3} for 22 days [*Chen et al.*, 2009]. Furthermore, the high levels of ozone are always accompanied with high concentration of fine particles. The high level of ozone therefore cannot be the only cause of adverse effects on public health and ecosystems. The strong oxidative

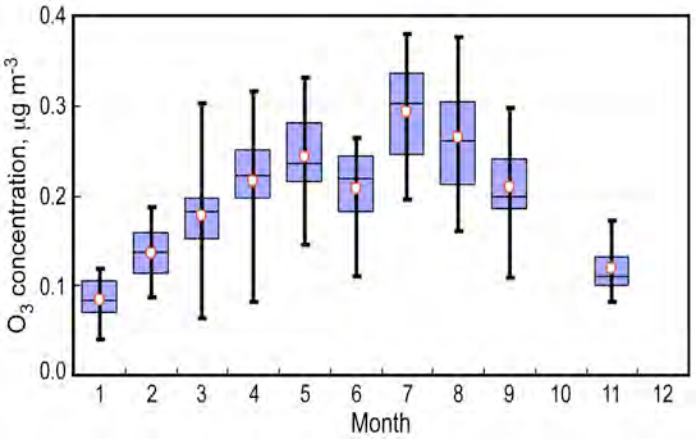

Figure 54 - The 2007 monthly average ozone concentration at 14:00 in Shanghai

atmosphere, which is favourable for the formation of fine particles, results in the decrease of visibility and the transformation of S(IV) to S(VI) causing the occurrence of acid rain [*Tang et al*., 2006]. Consequently, high ozone levels during the summer have been one of the complex problems for the further improvement of the air quality in Shanghai.

Air pollution control countermeasures

Shanghai municipal government (SMG) has carried out many air pollution control measures through three 3-year action plans to improve the air pollution of the city since 2000. The implementation of the first 3-year action plan included several measures that were carried out successively, such as promoting cleaner fuels, shutting down the Yangshupu coal/gas factory, extending the scope of applying natural gas, the use of low sulphur coal, conducting an integrated environmental pollution control policies for industrial parks, closing or suspending the factories with large influence to the air quality.

After the implementation of the second 3-year action plan, according to the national environmental protection requirements and the necessity of the city environmental protection work, Shanghai municipal government implemented the desulfurization project for all coal fired power plants and closed or stopped small coal fired units with a total capability of 695 MW from 2006 to 2008. All these measures largely decreased the emissions of SO_2 and PM_{10}. In addition, during this period, the prevention and control of industrial pollution and the comprehensive improvement of the major industrial parks was announced and carried out and the pollutants emission from Jinshan petroleum industry and Gaoqiao petroleum industry were checked.

Measures were also implemented to deal with vehicle emissions. Shanghai adopted the vehicular emission standards equivalent to Euro-II and Euro-III in 2003 and 2006, and the one equivalent to Euro-IV on November 1, 2009 [*Huang et al*., 2008]. Additionally, in order to reduce the emission from in-use vehicles, “Limits for exhaust pollutants from in-use vehicle equipped ignition engine in short transient loaded mode (DB31/357-2006)” and “Limits for exhaust smoke from in-use vehicle equipped with compression ignition engine under lug-down test (DG31/379-2007)” were implemented. Those vehicles that could not meet the emission standards equivalent to Euro I were forbidden to drive in the city center during the daytime, and this area was expanded in August of 2009. These control measures for vehicles are favourable for the improvement of the air quality in the Shanghai urban area.

Besides the control measures mentioned above, Shanghai municipal government launched a project for environment comprehensive improvement and landscaping of the city for the EXPO 2010, which led to a large drop in the level of particles. Shanghai environmental protection agency organized a project “The control measures for the air pollution during the period of EXPO 2010”. This project focused on local air pollution controls. Meanwhile the large pollution sources of the Yangtze River Delta were also asked to reduce their emissions selectively [*Chen et al*., 2007a], aimed at improving the air quality during the period of EXPO 2010.

Climatic change issues

The increase in temperature in China was similar to the global average of 0.74˚C over the past 100 years (China's National Assessment Report on Climate Change, 2007; IPCC 2007). According to the study by *Li et al.* [2008], the CO_2 emission from energy transformation accounted for 43%, and was followed by industry at 29% and transportation at 18% [*Chen et al*., 2004, 2007b; *Dolf and Chen*, 2001; *Wang et al*., 2001]. Suggestions on the bio-energy research, such as increasing biomass resources, improving biomass processing techniques, and energy transfer efficiency, are recommended, in order to reduce the CO_2 emissions in Shanghai.

Summary

The air quality in Shanghai is gradually improving. The annual average concentrations of primary pollutants are near the NAAQS at grade II over the past years. The good and excellent rate of the API was larger than 85% continuously for the past six years. However, the secondary pollution caused by the photochemical transformation of the primary pollutants is increasing and not considered in the API. The high level of ozone in the summer time, the decrease of the visibility, and

the high frequency of the occurrence of acid rain, suggests there is still much research and emission control strategies needed to further improve the air quality in Shanghai.

3.13 TOKYO, JAPAN

Introduction

Tokyo is located in the southeastern part of central Japan (Tokyo Metropolitan Government: 35.69°N, 159.69°E). The population of the Tokyo Metropolis was about 12.7 million, and the population of the Kanto area (Tokyo Metropolis and the six surrounding prefectures) was about 41.9 million in 2008. The Greater Kanto Area is one of the world's largest megacities in terms of population [*Gurjar et al.*, 2008].

The southeastern part of the Kanto area, called the Kanto Plain (150 × 150 km^2), faces the Pacific Ocean and to the west and north is surrounded by mountains exceeding 1000 m in height.

Local meteorology

From late spring to mid-summer, a sea-land breeze circulation pattern driven by the heating and cooling of the Kanto Plain during daytime and nighttime, respectively, often prevails [*Kondo et al.*, 2006]. On clear, calm days, southerly winds typically dominate during the daytime, and air is brought from over the ocean to the Tokyo Metropolis and further northward. From midnight to early morning, weak northerly winds dominate and air is transported from the northern Kanto Plain to the Tokyo Metropolis.

In winter, northwesterly winds associated with a strong Siberian high-pressure system generally dominate over the Japanese islands. As a result, air is transported primarily from the northwest, over the northern part of the Kanto Plain and then to the Tokyo Metropolis. When the northwesterly winds dominate, sea-land breeze circulation pattern does not develop.

Emissions sources and regulations

Kannari et al. [2007] developed an emission inventory, called EAGrid, which estimated hourly emissions over all of Japan with a horizontal resolution of approximately 1 km for each month in 2000. Over the Kanto area, motor vehicles were estimated to account for 80%, 52%, 43%, 17%, and 5% of emissions of carbon monoxide (CO), particulate matter with a diameter of less than 2.5 μm ($PM_{2.5}$), nitrogen oxides (NO_X), volatile organic compounds (VOCs), and sulphur dioxide (SO_2), respectively, in 2000 (Table 15). Large point sources made the largest contributions to SO_2 emissions (~62%) and stationary evaporative sources were the largest contributors to VOC emissions (~63%).

Table 15 - Source contributions to NO_X, SO_2, $PM_{2.5}$, VOC, and CO emissions (Gg/year) over the Kanto area in 2000 estimated in the EAGrid inventory [*Kannari et al.,* 2007]

	NO_X		SO_2		$PM_{2.5}$		VOC		CO	
Large point sources	164	28%	95	62%	6.7	22%	5.5	1%	230	17%
Other point sources	30	5%	22	14%	2.3	7%	5.8	1%	29	2%
Motor vehicles	256	43%	7.2	5%	15.9	52%	144	17%	1059	80%
Off-road vehicles	80	14%	1.8	1%	2.3	8%	10	1%	0	0%
Other transport	60	10%	27.	18%	3.5	11%	4.2	0%	11	1%
Stationary evaporative sources							535	63%		
Biogenic							148	17%		
Total	589		153		30		854		1330	

Emissions of NO_X from motor vehicles have been regulated by the Automobile NO_X Law since 1992 and PM emissions have been regulated by the Automobile NO_X–PM Law since 2001. Regulations implemented in 2005 ("new long-term regulation") are stricter than the European Union Euro V emission standards for NO_X and PM and regulations implemented in 2010 ("post new long-term regulation") are stricter than the Euro VI standards. The Ministry of the Environment of Japan (MOE) (2005) estimated that from 2000 to 2005 (2010) NO_X emission rates decreased by 16% (41%), and $PM_{2.5}$ emission rates by 42% (77%) over Japan. In addition, MOE estimated that NO_X and PM emission rates will be reduced by 68% and 93%, respectively, from 2010 to 2020 [*MOE*, 2008a]. Emissions of primary pollutants should be greatly reduced within the next 10 years, a prediction that should be validated by the monitoring data.

Emissions of VOCs from stationary evaporative sources have been regulated by the Air Pollution Control Law since 2004. *MOE* [2008b] estimated that VOC emission rates from stationary evaporative sources decreased by 20% from 2000 to 2005 over Japan.

Monitoring networks

MOE has operated an atmospheric pollutants measurement system (Atmospheric Environmental Regional Observation System, AEROS) over Japan since the 1970s. Hourly concentrations of NO, NO_2, SO_2, O_3, CO, non-methane hydrocarbons (NMHCs), and suspended particulate matter (SPM) are observed by this system at about 300 sites in the Greater Kanto Area (Figure 55).

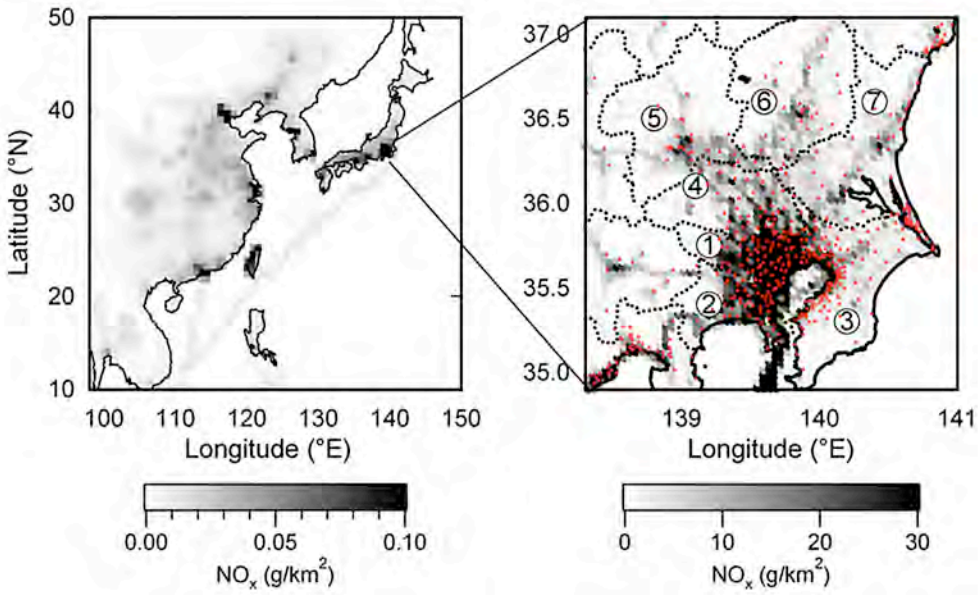

Figure 55 - NO_X emission rates over East Asia (left) and the Tokyo Metropolitan Area (right). Numbers indicate the Tokyo Metropolis and the six prefectures in the Kanto area (1, Tokyo Metropolis; 2, Kanagawa; 3, Chiba; 4, Saitama; 5, Gunma; 6, Tochigi; 7, Ibaraki). Red points in the right panel show the locations of monitoring stations in the AEROS network

Trends and air quality standards

From 1991 to 2004, both NO_X and NMHCs showed continuous decreases [*Tokyo Metropolitan Government*, 2005] (Figure 56) as a result of the regulations to reduce emissions of these species. Their concentrations were considerably lower in the Kanto area than were concentrations in megacities in the developing countries of Asia in the late 1990s to early 2000s [*Molina and Molina*, 2004]. Despite decreases in these O_3 precursors, the average O_3 concentration in the Kanto area during the summer (June–August) increased during 1991–2004. According to Japanese air quality standards (AQSs), hourly O_3 mixing ratios should be lower than 60 ppbv. The frequency with which the concentration exceeds the Japanese AQS for O_3 has increased,

especially since 2000. For example, the fraction of days when the hourly O_3 mixing ratio exceeded 120 ppb (i.e., 2 × AQS) has increased (Figure 57) [*Kondo et al*., 2010].

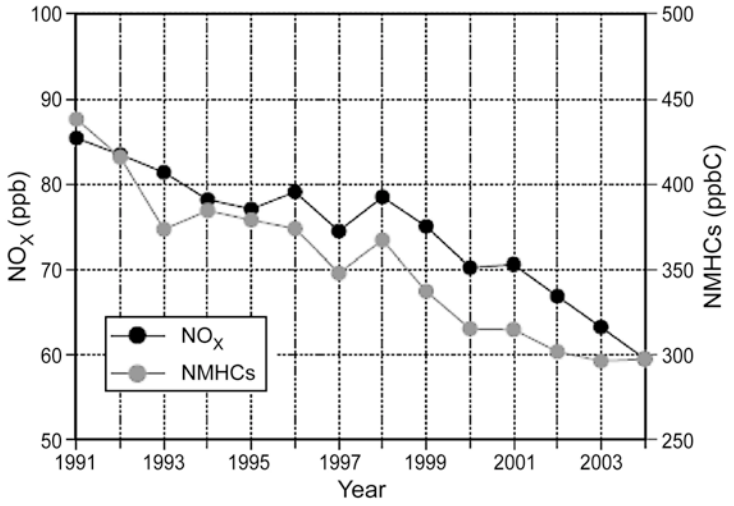

Figure 56 - Annually averaged hourly mixing ratios of NO_X (black, left axis) and total NMHCs (gray, right axis) measured at about 30 sites in Tokyo from 1991 to 2004. The original data used in this figure were provided by the National Institute for Environmental Studies

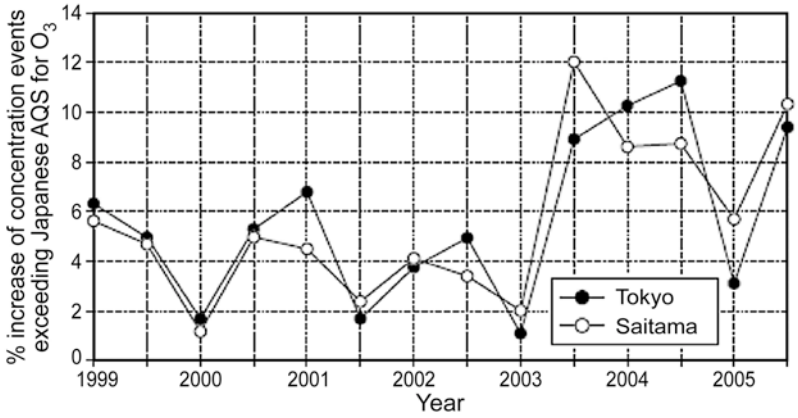

Figure 57 - Three-month summertime (June–August) percentages of days with hourly O_3 mixing ratios exceeding 120 ppb at 41 sites in Tokyo (filled circles) and 57 sites in Saitama (open circles). The original data used in this figure were provided by the National Institute for Environmental Studies

MOE has measured $PM_{2.5}$ concentrations at 12 (8), 14 (8), and 5 (0) monitoring stations at roadsides and in urban and suburban areas, respectively, in Japan (Kanto) since 2001 [*MOE*, 2008c; 2009]. From 2001 to 2008, $PM_{2.5}$ concentrations decreased by ~35% and ~25% at roadside and urban stations (Figure 58). *Minoura et al.* [2006] also measured the concentrations of $PM_{2.5}$ species at an urban station in central Tokyo from 1994 to 2004 and found that mass concentrations of fine-mode particles in Tokyo have decreased since 1996 (2.4 μg m^{-3} yr^{-1} for $PM_{2.1}$) (Figure 59). Most of this decrease was due to decreases in elemental and organic carbon [*Minoura et al.*, 2006], suggesting that the decrease in the $PM_{2.5}$ concentration was achieved by decreasing vehicle emissions. MOE set the AQS for $PM_{2.5}$ in Japan in September 2009 (daily average, 35 μg m^{-3}; annual average, 15 μg m^{-3}). From 2001 to 2008, annually averaged $PM_{2.5}$ concentrations in urban areas were higher than this AQS (Figure 58).

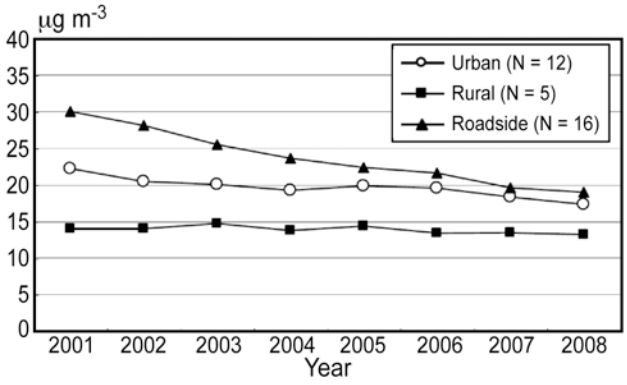

Figure 58 - Annually averaged $PM_{2.5}$ concentrations observed by the Taper Elemental Oscillation Method at roadside, urban, and suburban stations [*Ministry of the Environment of Japan*, 2009]. The numbers of each type of measurement site in Japan and the Kanto area are given in the text

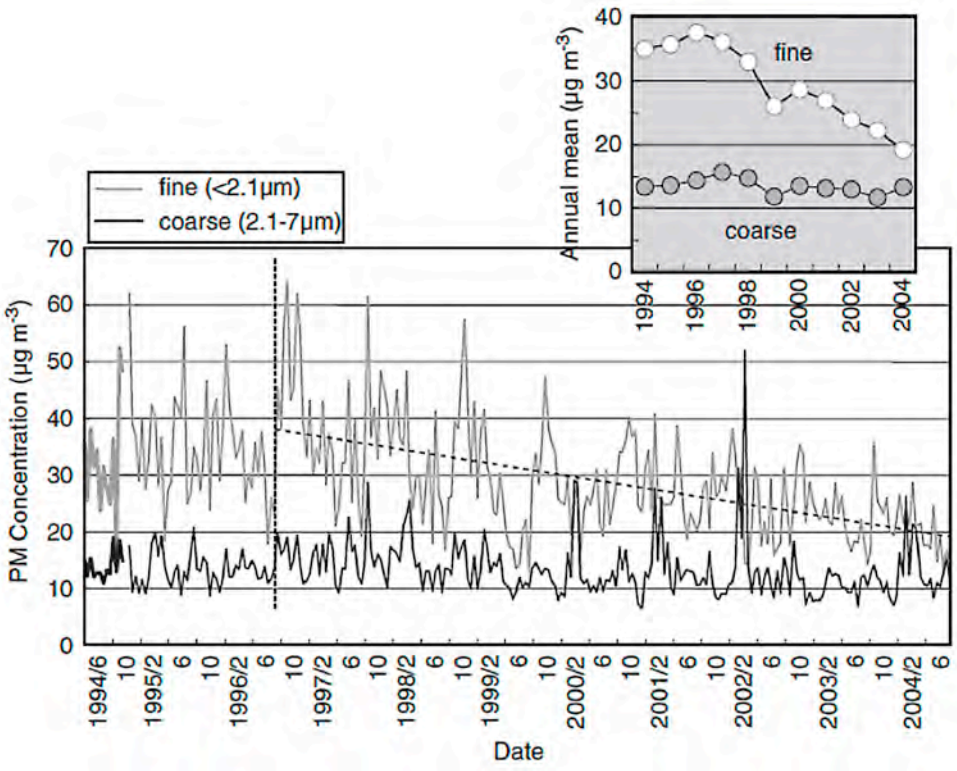

Figure 59 - Fine and coarse aerosol concentrations observed at an urban background site in Tokyo. Reproduced by permission of Elsevier from Minoura, H., K. Takahashi, J. C. Chow, and J. G. Watson (2006), Multi-year trend in fine and coarse particle mass, carbon, and ions in downtown Tokyo, Japan [*Minoura et al.,* 2006]

Research project

Integrated Measurement Programme for Aerosol and oxidant Chemistry in Tokyo (IMPACT) campaigns were conducted within the framework of the International Global Atmospheric Chemistry Project (IGAC), Mega-Cities: Asia. The background, objectives, methodology, and important findings of the IMPACT campaigns are presented in Chapter 7.

Remaining problems

According to the Tokyo Metropolitan Government (2005), the observed increase in the frequency of high-O_3 days over Tokyo cannot be explained by year-to-year variations in meteorological parameters, such as solar radiation and temperature. Thus, improved understanding of the factors controlling O_3 concentrations is needed to assess effective ways to lower the levels of photochemical pollution over Tokyo, which requires more detailed studies of the relationships of O_3 concentrations with its precursors' emissions in various areas.

To establish an effective control strategy for achieving the AQS for $PM_{2.5}$, comprehensive source apportionment of $PM_{2.5}$ is necessary. Organic aerosols account for a large fraction of the total fine-mode aerosol mass concentration in the Kanto area. However, the observed levels of organic aerosols are not well reproduced by 3-D chemical transport models [e.g., *Matsui et al.*, 2009]. Further studies on the formation of organic aerosols are clearly needed.

3.14 TEHRAN, IRAN

Introduction and geographical setting

Tehran, the capital of the Islamic Republic of Iran, has a population of approximately 10 million people. Located at (35° 42'N, 51° 25'E) with an area of 2300 km^2, the city is situated in a semi-enclosed basin just south of the Alborz Mountain chain (with average height of 2000 mASL; Figure 60). Tehran has suffered from poor air quality since the oil boom decade of the 1970s and over the last fifteen years rapid population growth has made matters even worse. On some days, the pollution loading of the atmosphere is so high that the impressive Alborz Ranges become invisible from most vantage points. Tehran's Clean Air Committee stated recently that 10,000 people die every year due to air pollution related cardio-pulmonary disease.

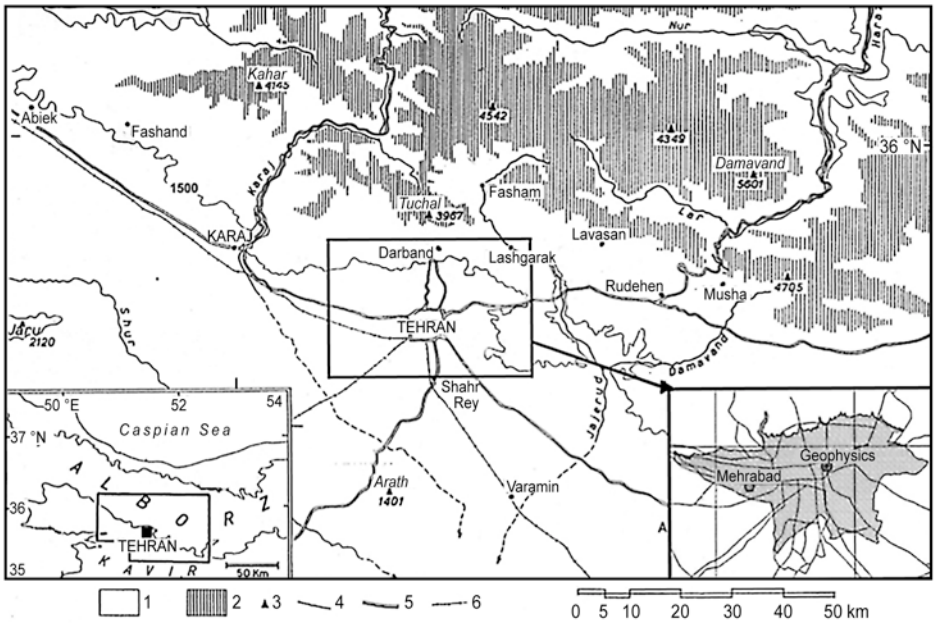

Figure 60 - Map of Tehran

Tehran's location is unusual; unlike most major cities it is not near a river or even close to the sea. Due to high elevation (approximately 1140 m), aridity, and latitude, Tehran experiences four seasons. Climate can be extremely hot in the summer (with midday temperatures ranging between 30 to 40°C) and cold in winter, when night temperatures can fall well below the freezing point (Figure 61). Local precipitation is absent for 6 months of the year in low lying areas. Originating in the Mediterranean, synoptic scale low pressure systems propagate over the region in spring and autumn, while in winter the southward extension of the Siberian high pressure system can advect cold air over the Iranian Plateau. The average annual rainfall is approximately 230 mm, with most precipitation falling in autumn and winter months. The large scale easterly flow that dominates the area in the summer is thought to be associated with a circulation pattern named 'the winds of 120 days' caused by a thermal low over Pakistan [*Zawar-Reza,* 2008]. Outside of the basin where Tehran is situated westerly winds prevail except in summer when the flow tends to be easterly. The airflow over most of Tehran is influenced by the sloping topography as discussed below.

The predominance of a diurnally reversing local wind system is of special importance to air quality, with a major influence on vertical stability and surface-layer meteorology. During the day, a south south-westerly direction prevails, while at night and early morning, the direction of flow is mostly from a north north-east quadrant (Figure 61). Daytime up-slope winds are most frequent in the summer and autumn seasons. The nocturnal northerly drainage winds are also most prevalent during summer and autumn. The combined effect of less isolation to drive thermally generated flows and the regular passage of the eastward propagating depressions that pass over Tehran make the bi-modal behaviour of wind more diffuse in spring and winter [*Zawar-Reza et al*., 2010]. During the transition in the diurnal wind direction, the wind speed tends to drop, hence reducing the ventilation capacity of the atmosphere at a time when peak emission of pollutants from rush hour traffic is occurring (Figure 61). In general, the median wind speed does not go above 3 m/s at any hour, so the ventilation capacity of the urban atmosphere is poor.

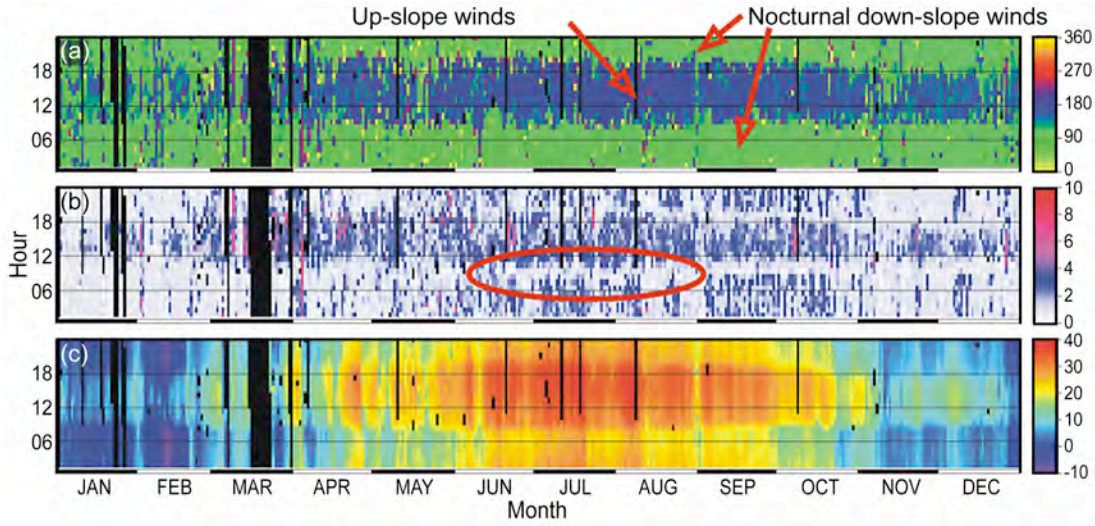

Figure 61 - Meteorological variables at the Foothills Station for 2005 for (a) Wind Direction in degrees; (b) Wind Speed in m/s. Each hour is represented by a coloured pixel, where each day is shown as a vertical strip. The morning transition phase is highlighted by the ellipse; (c) temperature (°C)

To illustrate just how dire the air quality is in Tehran, Figure 62 provides information on daily and hourly averages of particulate matter with aerodynamic radius of 10 μm or less (PM_{10}). The red line indicates the World Health Organization's (WHO) guideline of 50 μg m^{-3}; it is apparent that most days the air quality is at dangerous levels.

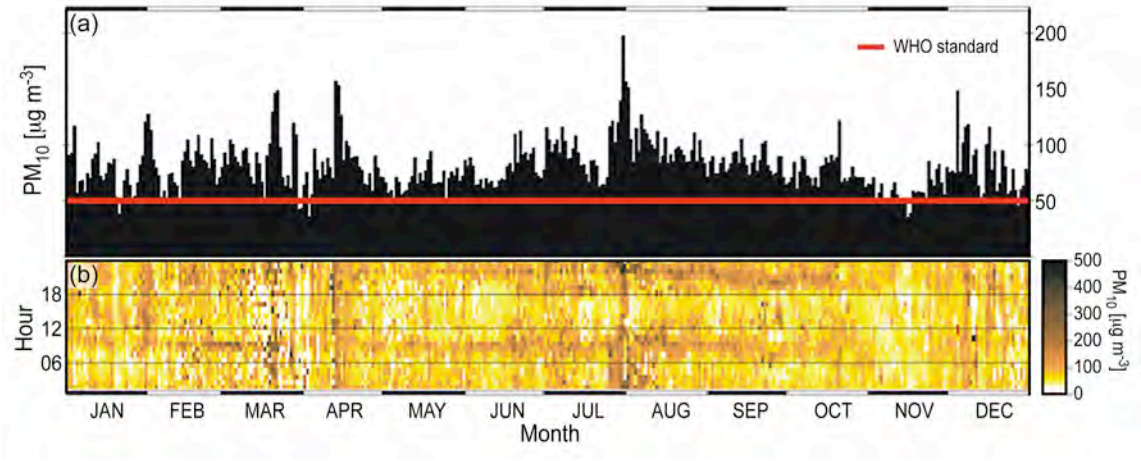

Figure 62 - a) Daily averages of PM_{10}, and b) hourly values for PM_{10} in μg/m^3 averaged over 2001 – 2005 at a monitoring station on the foothills of the Alborz Mountain (northern Tehran)

The scale of this air quality problem is huge, with the economic impact of air quality on Iranian economy and population health estimated at US $7 billion; equivalent to 8.4% of Gross Domestic Productivity [GDP; *Shafie-Pour and Ardestani*, 2007]. Though there are numerous publications on this issue in Farsi, only a few internationally published scholarly papers focus on air pollution and its environmental impacts in Tehran.

Emission sources of air pollutants

Emissions in Tehran are from mobile sources (transport), stationary sources such as industries (mostly in the outskirts), and residences. By far the worst offender, mobile sources are estimated to contribute a massive 89% of the total emission by some studies; other studies suggest that this contribution might be smaller, at approximately 71%. As this would indicate, Carbon monoxide (CO) and PM_{10} are the main concern for air quality in Tehran.

Table 16 - (Data from Air Quality Control Company, AQCC)

Emission (ton/annum)	Mobile Sources	Percentage contribution	Stationary Sources	Percentage contribution
CO	1,354,652	99	18,222	1
SO_2	6,142	10	57,173	90
NO_X	109,917	70	46,253	30
THC	155,609	71	64,761	29
PM	18,777	69	8,444	31

Table 16 summarizes the relative and absolute contribution for each emission source in 2005. Mobile emission inventory surveys for Tehran cover emissions from light duty vehicles, private cars, motorcycles, public transport buses, and trucks. According to an Air Quality Control Company (AQCC) study performed in 2005, the contribution of light duty vehicles to air pollution caused from mobile sources has been estimated to be close to 50%. Factors that determine such a high contribution by the transport sector are complex, including government subsidized inexpensive fuel, which can be of poor quality (unleaded gasoline became available after 2001), and large numbers of older, domestically produced cars.

Data available on air pollutants

There are approximately 13 permanent ambient air monitoring stations dispersed throughout Tehran, which are operated by AQCC and the Tehran Municipality. Daily and hourly averaged data are available for CO, NO_X, SO_2, HC, TSP, O_3 and lead.

The status and trend of the pollution

As mentioned above, international scholarly work on analysis of *in situ* data is rare for Tehran. Yet some good examples can be found; *Halek et al*. [2004] examined the monthly averages of PM_{10} for 2003 and discovered that autumn tends to have the highest concentrations and spring the lowest. Maximum values were observed in September at just over 370 μg m^{-3}; the minimum was reached in April at 65 μg m^{-3}. *Hassanzadeh et al*. [2009] show that SO_2 has a seasonal pattern similar to that of PM_{10}. Monitoring SO_2 levels at 5 monitoring stations for the period of 2000-2005, they concluded that at most sites SO_2 concentration fluctuations were similar.

A link between hospital admissions due to angina pectoris and several pollutants was established by *Hosseinpoor et al*. [2004]. This study established that exposure to CO provided the clearest link to negative health outcomes for the population in Tehran, although the confounding role of other pollutants was acknowledged.

Shirazi and Harding (2001) provide information on trends for some common pollutants such as CO and particulate matter (PM) for the period between 1988 and 1993. Indicating a rapid upward trend for most pollutants except NO_2, they point out that all pollutants except TSM routinely and substantially exceeded WHO guidelines. These findings suggest that as the population continues to grow, and drives increasing numbers of motor vehicles, there will be a corresponding increase in

the negative health effects resulting from exposure to the rise in pollutants. The role of the transport sector on ambient levels of particulates is considered by *Halek et al.* [2004] where the age distribution and polluting potential of the car fleet is examined in conjunction with seasonal variation in PM levels.

SO_2 trends between 1995 and 2002 were the focus of a study by *Aspari et al*. [2002]. Measurements were taken from seven main monitoring stations at different locations in the city. The trend in annual concentrations was downward until 1998, but since then there has been an increasing trend. The seasonality of concentrations is similar to the previous studies.

Information on spatial and temporal concentrations of volatile organic compounds (VOC) is hard to find. *Jafari and Ebrahimi* [2007] provide information on measured concentrations for Benzene. Finding that that benzene concentrations in Tehran average around 0.1 mg/m^3, they note that this is significantly higher than the U.S. Environmental Protection Agency (EPA) recommendation.

Relationships of the trends to regulations

In 1995, Tehran Municipality initiated a two-year project entitled 'Tehran Transport Emissions Reduction Project' to identify strategies for reducing motor vehicle emissions, and also to consider greenhouse gas emissions. The Global Environmental Facility (GEF) and the local municipality supported this research. Key recommendations from this programme included phasing out older more polluting vehicles, cleaner fuel, mandatory inspection and maintenance, public education and better traffic management. However, lack of coordination between local authorities has meant that the key recommendations have not been successfully implemented (Asadollah-Fardi) despite the Clean Air Act passed by the Iranian Parliament in 1995 to address this important issue. This act prohibits the use of polluting (smoking) vehicles in urban areas, amongst other decrees.

A cursory examination of annual averages from a monitoring station in the commercial heartland of Tehran (Bazaar) shows a decreasing trend for all pollutants (Asadollah-Fardi, Unpublished Report), but a more robust assessment is needed. Future studies should consider statistically removing the influence of inter-annual variation in meteorology.

Climatic change issues

A clear link between global climate change and air quality has not been established for Tehran, however a summary of findings up to now follows next. Significant, increasing trends have been found in the annual statistics for temperature (Figure 63). Several climactic metrics, including maximum of daily maximum and minimum temperature, the annual minimum of daily maximum and minimum temperature, the number of summer nights, and the number of days where daily temperature has exceeded its 90th percentile were examined by *Zhang el al*. [2005]. Analysis of daily airport data from 1956 to 2003 has shown significant negative trend in frost days and a significant positive trend for summer days when maximum temperature exceeds 25 °C. Significant negative trends have been found in the number of days when daily temperature is below its 10th percentile. Positive trend in extreme precipitation events for Tehran was also found by *Asgari et al.* [2008].

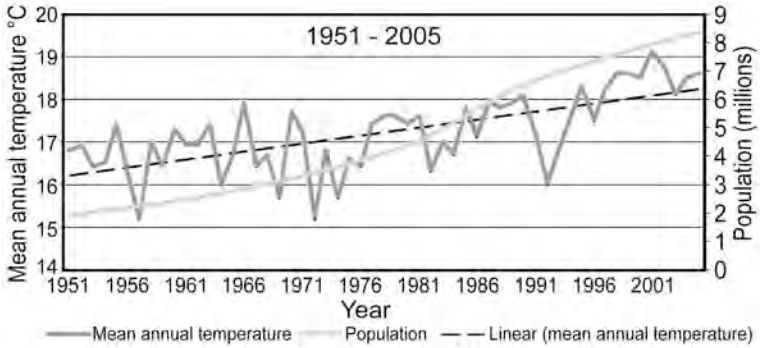

Figure 63 - Trends in mean annual temperature and population for Tehran

Research has shown that temperature trend is not only significantly affected by global climate change, but also by local factors such as the Urban Heat Island (UHI). Recent work by *Soltanzadeh et al*. [2009] on the interaction between nocturnal down-slope winds and the urban boundary layer is still at an early stage, but shows promise in shedding light on the role of this interaction in air pollution dispersion, especially at night time.

Research projects on air quality of Tehran

Extensive systematic air pollution study campaigns of significance have not yet been conducted in Tehran, although an international framework for an 'Integrated Plan for air pollution reduction' is in place. This plan covers topics such as the role that public transport, fuel quality, traffic management, and public awareness play in air quality. Small scale research projects that have looked at constructing emission inventory databases, characterizing air pollutants and limited dispersion modelling with mesoscale models have been performed by various researchers from universities and governmental agencies such as the Islamic Republic of Iran Meteorological Organization (IRIMO). But a co-operative multi-agency research programme that simultaneously measures variables such as emission factors, low level meteorology, and boundary layer height is desperately needed to answer the many key scientific questions that remain.

Various Departments at the University of Tehran are active in air quality research. Work is currently in progress to understand boundary-layer processes over the urban fabric of Tehran, for example, in the Department of Geophysics. A long-term SODAR dataset is the main observational tool here, and extensive postgraduate work is underway to validate mesoscale outputs with this data. Previously, theoretical work in this department concentrated on the climatology of mixed-layer evolution. The Department of Geography is mainly concerned with long-term spatial variation of pollutants, and linkages with synoptic scale flow.

Problems remaining

It is obvious that there is plenty of scope for air pollution research in Tehran. There are many unanswered questions regarding mesoscale meteorology and its role in local and regional advection of pollutants. Basins exhibit peculiar vertical stability characteristics that directly control vertical mixing of pollution, but this aspect has not been examined over Tehran. The twice daily radiosonde from Mehrabad International Airport ascends too rapidly through the first few hundred meters above the ground to provide comprehensive information about stability. So there is very little information on the formation and destruction of the nocturnal inversion layer or its seasonal variation, or on the mixed-layer height evolution throughout the day.

Peer-reviewed published work is badly needed examining long-term pollutant trends after statistically removing the meteorological signal. Given the influence of the surrounding topography on Tehran's meteorology, however, this might not be as vital as it is for cities on less complex terrain. Certainly the AQCC dataset makes a comprehensive overview of trends possible.

In view of the significant amount of air and pollution that can be injected into the upper troposphere via the daytime up-slope flows and transported away by the free atmosphere, it is also vital to research and understand the role this process plays in Tehran's air quality.

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CHAPTER 4 - SOUTH AMERICA

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4.1 OVERVIEW

Air pollution is a reoccurring phenomenon in many South American cities. Like in many cities around the world, population growth and urbanization result in a convergence of a variety of air pollutant emission drivers such at industry, transportation, and energy production and consumption amongst others. This convergence, however, is rather extreme in South America where about 84% of the population today lives in mid- to large size cities (larger than 750 k inhabitants) [*UNPOP*, 2009]. The emptiness of the South American continent is also striking, as cities, and population densities, are mainly located close to coastal areas or a few hundreds of kilometres in land (Figure 1). Although the rate of growth in larger cities of more than 5 million inhabitants has decreased over the last decades (from an average of 3%/yr between 1975 and 1990 to 2% between 1995 and 2010), the growth rate is still enough to ensure the proclivity of these agglomerations to concentrate pollution drivers and hence face environmental problems. Moreover, mid-size cities (<5 million inhabitants) are still growing fast (>2% growth per year between 2005 and 2010).

Oftentimes, South America is referred to as a homogeneous entity, possibly due to our common historic background in connection to the European "conquest" of the territories. However, if one wants to assert global change drivers in the region, one must deal with the actual heterogeneity of South America's physical and human geography. Our economies and political systems vary giving rise to diversity in our policies. Table 1 shows relevant population, economic,

and human development indexes for South American cities. From Table 1 it is apparent that the economic growth has also increased the life expectancy, educational attainment and income measured by the Human Development Index in the region. However, South America is still subject to severe inequity as expressed by the Gini index, which is a measure of income distribution. These inequities act as amplifiers of environmental problems and, in particular, they usually increase the overall air pollution problem as well as domestic air pollution and other concomitant vulnerabilities [*Smith et al.*, 2000; *Adonis and Gil*, 2001; *Emmelin and Wall*, 2007].

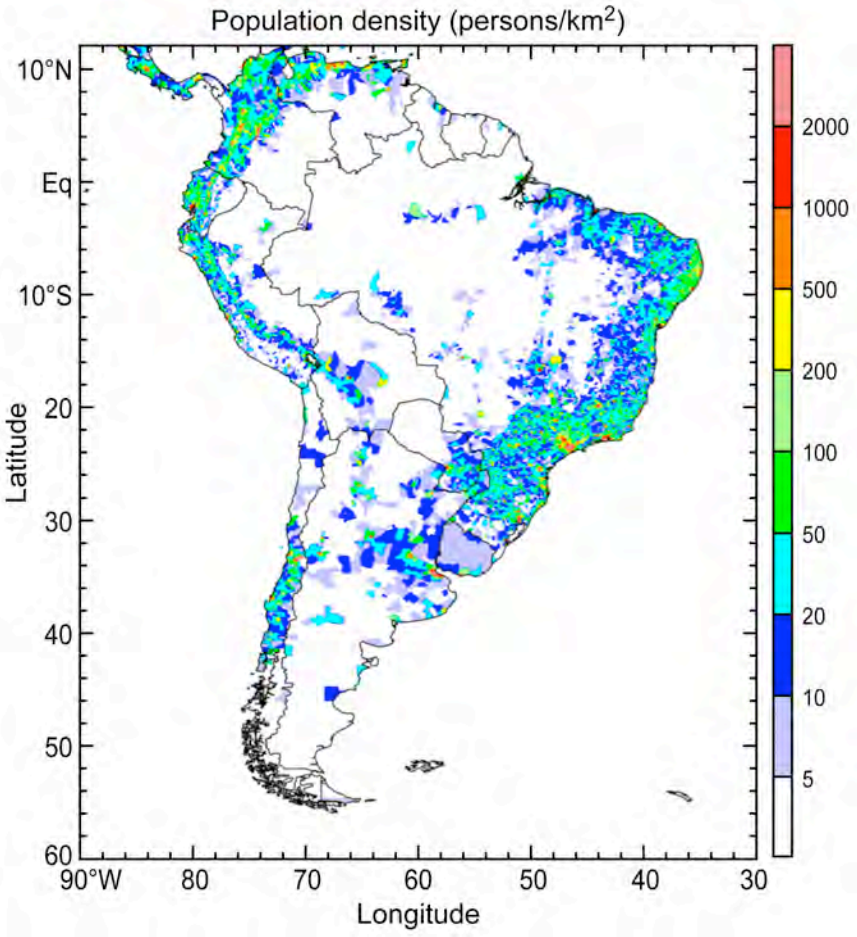

Figure 1 - Map of the population density in South America (persons per km²), based on 0.25º gridded data for 2000 from the Center for International Earth Science Information Network (CIESIN) at Columbia University *[http://sedac.ciesin.columbia.edu/gpw/]*

Table 1 - Location, population *(http://esa.un.org/unpd/wup/index.htm)*, gross domestic product (GDP) per city (Haworths et al. 2009), human development index (HDI) per country *(http://hdr.undp.org/en/statistics/)* and Gini Index in urban areas *(http://www.eclac.org/estadisticas/)* for selected South American cities

Urban Agglomeration	Geographical Coordinates			Population (in k inhabitants)			GDP in US$ at PPP		HDI			Gini Index		
	Lat (S)	Lon (W)	Alt (m.a.s.l.)	1975	2010	2025	2008	2025	1975	2000	2007	Ca. 1990	Ca. 2000	Ca. 2007
São Paulo	23,6	46,8	720	9.614	20.262	21.651	388	782	0,644	0,771	0,813	0,606	0,628	0,586
Buenos Aires	34,5	58,5	20	8.745	13.074	13.708	362	651	0,784	0,854	0,866	0,501	0,590	0,510
Rio de Janeiro	22,9	43,2	30	7.557	11.950	12.650	201	407	0,644	0,771	0,813	0,606	0,628	0,586
Lima	12,0	77,0	110	3.696	8.941	10.530	109	213	0,642	-	0,806	-	0,477	0,421
Bogotá	-4,6	74,1	2600	3.040	8.500	10.537	100	192	0,661	0,771	0,807	-	0,484	0,587
Santiago	33,5	70,5	500	3.138	5.952	6.503	120	207	0,703	0,835	0,878	0,543	0,558	0,517
Medellín	-6,2	73,6	1500	1.536	3.594	4.494	50	97	0,661	0,771	0,807	-	0,484	0,587

Typically, air pollution research and management initiatives have been triggered by severe problems. Therefore, emphasis has been put on short-term and local-scale analyses designed to deal with acute problems, i.e., extreme pollution events, whereas little attention has been paid to long-term, large-scale effects. This is slowly starting to change in places where environmental goals have become more ambitious (e.g., Santiago, São Paulo). It is encouraging to see that the majority of the environmental authorities in the region (Brazil, Chile, Colombia, Peru, etc.) have adopted in principle the Internet as a tool for publishing data, assessments, and procedures. In practice, however, one often finds inconsistencies and information gaps.

A summary of the situation in different urban agglomerations in South America is given here. The following section addresses crosscutting issues related to modelling, emissions, monitoring, and regulation. In addition, we look at the continent as a whole looking at the impact of non-urban sources on megacities and the footprints of megacities. We then describe the situation in Bogotá and Medellín in Colombia, Buenos Aires in Argentina, Lima in Peru, Santiago in Chile, and Rio de Janeiro and São Paulo in Brazil. In all of these cities, except in Buenos Aires, there is concern about air quality, systematic air quality monitoring, and attainment plans have already been implemented. This review is to be considered illustrative, not comprehensive, of the present situation in South American megapolis by showing specific characteristics and features as well as the differences across cities and countries. The review considers relatively smaller cities that are expected to grow and that already show significant pollution levels (e.g., Medellín), medium-size capitals with heavy air pollution (e.g., Bogotá, Santiago, Lima), and cities that already surpassed 10 million inhabitants (Buenos Aires, São Paulo and Rio de Janeiro).

4.1.1 Modelling

Dispersion modelling, and its operational implementation "chemical weather forecasting", is a key-tool for dealing with air quality problems. There is an increasing number of groups in South America developing and applying air quality models [*Andrade et al*., 2004; *Freitas et al.,* 2005a; *Freitas et al*., 2005b; *Martins and Andrade*, 2008a; e.g., *Gallardo et al.*, 2002; *Olivares et al.*, 2002; *Schmitz*, 2005; *Saide et al.*, 2009a; *Jorquera and Castro*, 2009; *Freitas et al*., 2009; *Longo et al.*, 2010; *Saide et al*., 2011a, etc.]. Several institutions, including research centres, environmental authorities, and weather services (See Table 2) have implemented operational chemical weather forecasting systems. Inverse modelling is also being used [*Hoelzemann et al.,* 2009; *Saide et al*., 2009a; *Jorquera and Castro*, 2010; *Saide et al*., 2011b]. Further, fast Internet connections are already available (http://www.redclara.net/) making it possible to develop grid-computing applications, particularly within the framework of atmospheric physical and chemical weather forecasting [*D'Almeida et al*, 2008].

Table 2 - Examples of operational Chemical Weather Forecasting Systems at the urban and continental scale in South America

Institution responsible	Coverage	Species forecasted	Internet
Brazilian Center for Weather Forecast and Climate (INPE/CPTEC)	Continental	Criteria pollutants and AOD from biomass burning	http://www.cptec.inpe.br/meio_ambiente/
Atmospheric Sciences Department, University of São Paulo	São Paulo and surrounding areas	Criteria pollutants	http://www.master.iag.usp.br
Dirección Meteorológica de Chile (DMC, Chilean Weather Office)	Santiago	Ozone	http://www.meteochile.cl/modeloPOLYPHEMUSozono.html
Dirección del Área Metropolitana del Valle de Aburrá	Medellín	Criteria pollutants	http://www.metropol.gov.co/aire/

4.1.2 Emissions

It is well known that developing emission inventories is an extremely complex process, which requires detailed statistics, process-level understanding, and continuous evaluation and updating. South America lacks a sustained and coordinated activity and record for building reliable emission inventories for urban centres. There are a few inventories available for cities for which attainment plans have been implemented (Table 3). Unfortunately, transparency, completeness, consistency, comparability, and accuracy requirements are not always met. For instance, many of these inventories are based on emission factors determined elsewhere that are not representative for the very specific and diverse conditions of South American cities. This is changing due to the implementation of comparable methods and the determination of local emission factors that are now starting to be introduced in new inventories [*Behrentz et al.*, 2009*; Martins et al*., 2006; *Vivanco and Andrade*, 2006; *Sánchez-Ccoyllo et al.*, 2009; *Martins and Andrade*, 2008a; 2008b; *D'Angiola et al.*, 2010]. These inventories are restricted to a few activity sectors such as urban and industrial energy use and transportation, leaving aside rural and natural emissions. Efforts to integrate and reconcile local and global emission inventories for South America have also been initiated [*Alonso et al*., 2010]. Figure 2 shows emissions of CO, NO_X, and SO_2 for South America based on the EDGAR v3.2 FT2000 database.

Table 3 - Emission inventories developed for South American cities. Fluxes are given in kton/yr of species, except for VOCs that are in ktonC/yr and NO_X given in kton of 90% NO and 10% NO_2

City	Base Year	Source Type	Species						Reference
			PM_{10}	$PM_{2.5}$	CO	NO_X	VOCs	SO_2	
Bogotá	2001	Mobile	1.6	-	306	13.7	47.6	2.3	
	2001	Stationary	2.9	-	8	1.4	4.4	5.1	
		Total	**4.5**	**-**	**314**	**15.1**	**52.0**	**7.4**	Zárate et al., 2007
Buenos Aires	2006	Mobile	5.2	-	569	81.0	70	6.6	D'Angiola et al., 2010
	2006	Stationary	2.5	-	9.9	29.8	1.6	15.9	Oreggioni, 2009
		Total	**7.7**	**-**	**579**	**130.8**	**71.6**	**22.5**	
Lima	2000	Mobile	4.6	-	198.5	63.4	32.6	28.7	MTC, 2010
	2004	Stationary	8.8	-	11	7.8	8.4	40.3	DIGESA, 2005
		Total	**-**	**-**	**341**	**78**	**81**	**51.5**	
Medellín	2008	Mobile	2.2	-	204	18.2	29	0.2	
	2008	Stationary	2.8	-	12.8	5.1	5	31.4	
		Total	**5.0**	**-**	**217**	**23.3**	**34**	**31.6**	AMVA, 2008
Rio de Janeiro	2003	Stationary	10.6	-	6.3	30.3	26	55.8	
	2003	Mobile	7.8	-	315	60.2	53	7.5	
		Total	18.4	**-**	**321**	**90.5**	**79.0**	**63.3**	FEEMA, 2004
Santiago	2005	Mobile	1.8	1.4	211	37.9	20	0.2	
	2005	Stationary	3.0	2.4	24	13.8	83	13.2	
		Total	4.8	**3.8**	**235**	**51.7**	**103**	**13.4**	DICTUC, 2007
São Paulo	2009	Mobile	31*	-	1533	362	370	8	
	2009	Stationary	32*	-	39	14	12	17	CETESB, 2010

**For São Paulo it is calculated as total particulate matter including PM_{10}.*

4.1.3 Observations

Considerable resources have been invested by local authorities in various urban areas in South America in air quality monitoring, focusing on exposure to so-called criteria pollutants (Cf. Table 4). However, aerosol measurements in the region, except for some areas of Brazil and Venezuela, are still sparse and are usually limited to concentration levels of PM_{10} and $PM_{2.5}$. Overall, there is an appalling lack of accurate observations of air quality in South America in both urban, rural, and background environments. In addition, oftentimes, the quality and accessibility of the data is not reliable.

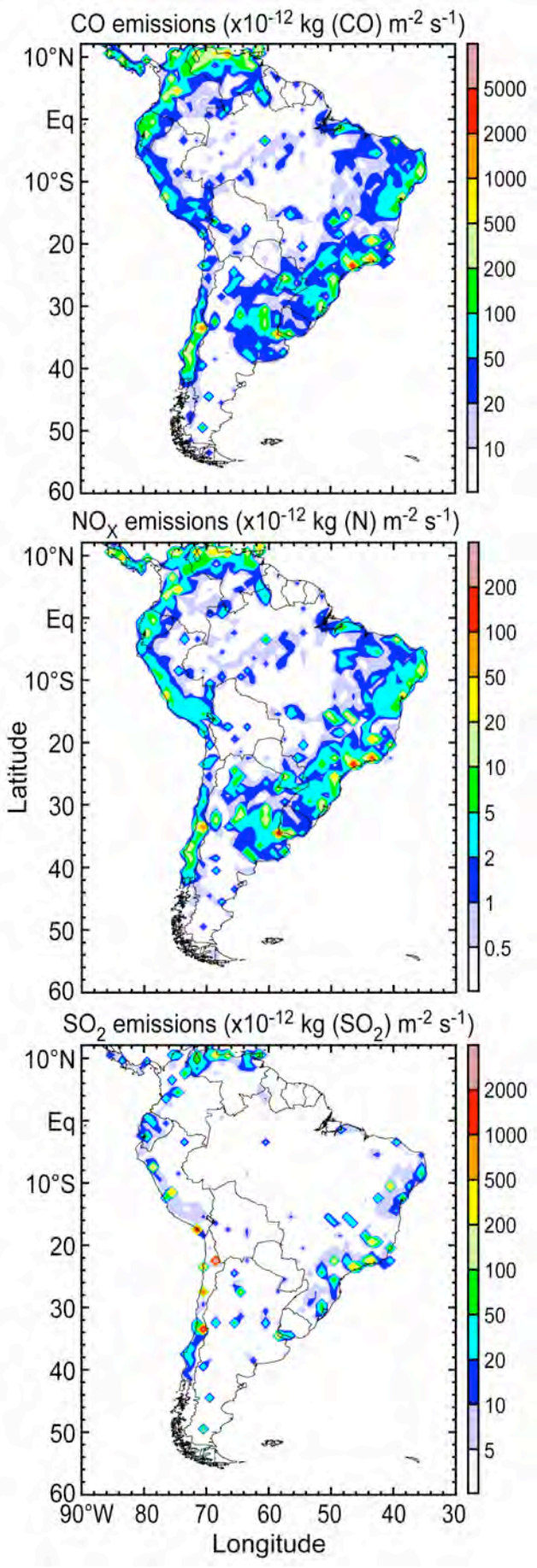

Figure 2 - CO, NO_X and SO_2 emissions for the year 2000 based on the EDGARv3.2 FT2000 database

Table 4 - Available air quality monitoring available in South American cities

City	Current number of stations	Pollutants measured	Beginning of measurements	Note
Bogotá	15	NO, NO_2, SO_2, TSP, PM_{10}, $PM_{2.5}$, O_3, CO, CH_4, HCHO, NMHC, benzene, toluene	1997	Not all stations measure all pollutants http://www.secretariadeambiente.gov.co
Bogotá	15	NO, NO_2, SO_2, TSP, PM_{10}, $PM_{2.5}$, O_3, CO, CH_4, HCHO, NMHC, benzene, toluene	1997	Not all stations measure all pollutants http://www.dama.gov.co
Buenos Aires	4	CO,NO,NO_2,PM_{10} Dust Fall	1985 2009	http://www.agenciaambiental.gob.ar/ At one station the record goes back to 1985 Two out of the four stations have been added in 2009.
Lima	5	TSP Dust fall PM_{10}, $PM_{2.5}$ CO, SO_2, NO_2, O_3	1987 1990 2000 2001	http://www.digesa.minsa.gob.pe/ Only data summaries are available on the internet
Medellín	17	NO, NO_2, SO_2, TSP, PM_{10}, $PM_{2.5}$, O_3, CO, HC	1993	http://www.metropol.gov.co/
Santiago	11	NO, NO_X, SO_2, PM_{10}, $PM_{2.5}$, O_3, CO, CH_4, NMHC	1997	http://www.asrm.cl/ At 4 stations the record goes back to 1988
São Paulo	30	NO, NO_2, NO_X, SO_2, PM_{10}, $PM_{2.5}$, O_3, CO	1981 automated stations 1973 manual stations for SO_2 and smoke	http://www.cetesb.sp.gov.br
Rio de Janeiro	36	TSP NO, NO_X, SO_2, PM_{10}, O_3, CO, CH_4, NMHC	1967 1998	http://www.inea.rj.gov.br 32 are manual stations and 4 automated

4.1.4 Regulations

The World Health Organization (WHO) air quality guidelines represent the current best scientific understanding of the impacts of air pollution on human health. While designing national regulations, local factors such as race related health risks, available technologies, social factors, degree of development, capability to implement regulations, and control compliance should ideally be considered [*WHO*, 2005]. A comparative assessment of environmental regulations in South America shows that most countries first reproduce the standards developed by United States Environmental Protection Agency (USEPA), regardless of their specific national and regional circumstances. However, South American governments do not update these standards at the frequency of the USEPA [*Rincón et al.*, 2007]. Aerosols are regulated in the majority of countries by means of establishing concentrations for PTS, PM_{10} and/or $PM_{2.5}$. All the current levels in South American countries with published air quality information, are several times over WHO standards for PM_{10}, $PM_{2.5}$ and SO_2 [*Romero-Lankao et al*, 2010]. In the case of NO_2, Peru and Colombia established the same standard proposed by WHO. Colombia has established an annual ozone standard more stringent than that of the WHO standard. Chile is the only South American country that follows WHO air quality guidelines for lead. Only Chile and Peru have recently introduced standards for $PM_{2.5}$.

4.1.5 Non-urban sources and megacities

South American cities are not only subject to air pollution that originates in the cities themselves but they are often affected by large industrial sources [*Gallardo et al.*, 2002], volcanic eruptions, and most notably large-scale biomass burning. Despite of the international claims and the Brazilian government commitment to limit deforestation in the Amazonia, the effectiveness of current actions is still open to discussion and their efficacy within the next decades is questionable.

In this context, the perspective of rapid progression of urbanization in South America and any assessment of its impact on atmospheric chemistry must take into account the regional scale emissions associated with biomass burning. The typical atmospheric circulation in South America impose a long range transport of biomass burning smoke from its source areas in the Amazonia to the southern and southeastern part of the South American continent, thus impacting large areas in the subtropics [*Freitas et al*., 2005 a; *Longo et al.,* 2009]. Most of the smoke produced by fires in the Amazonia and central region of South America is transported towards the South Atlantic Ocean throughout the southeastern part of the continent driven by the typical pole-ward warm and moist air flow in the lower troposphere on the east side of the Andes called the South American Low Level Jet [*Vera et al*., 2006]. This lower troposphere smoke transport very often affects cities in the Southern part of the South American continent. *Landulfo et al*. [2009] investigated the impact of the regional smoke from vegetation fires on larger cities. Figure 3 shows a model simulation of the biomass burning aerosol optical thickness (AOT) at 550 nm on 30 August 2005. The smoke river over South America emerging from the biomass burning areas and flowing towards the South Atlantic Ocean is evident and consistent with *in-situ* observations.

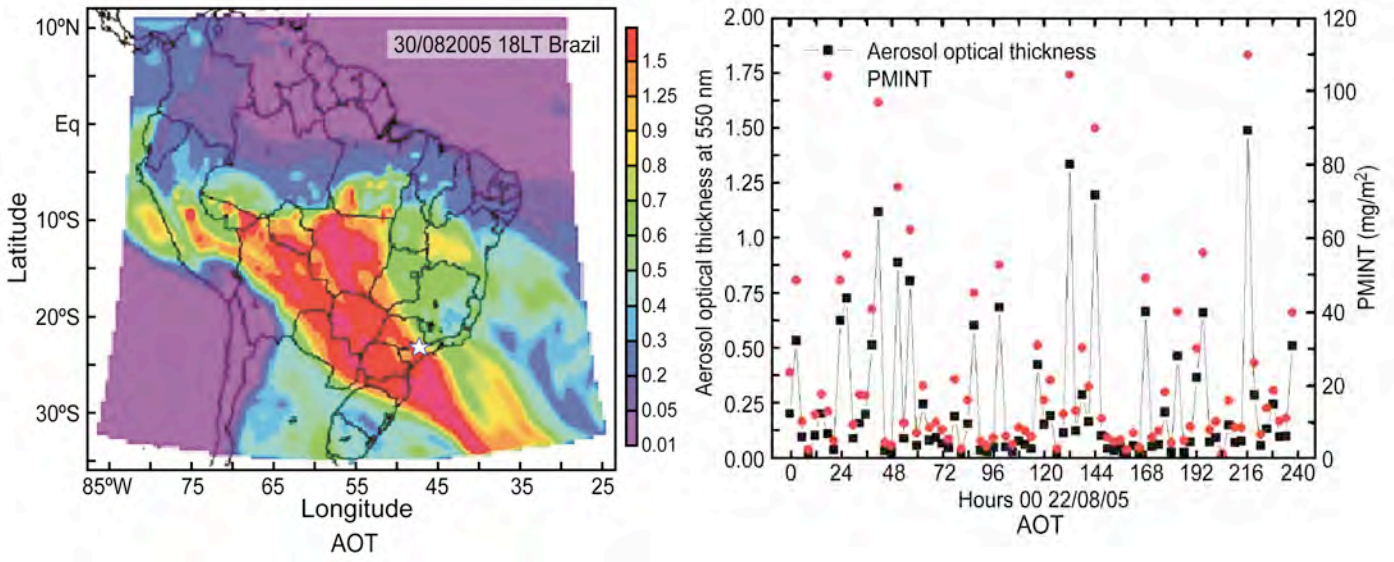

Figure 3 - Left, CATT-BRAMS model simulation of AOT at 550 nm for 30 August 2005 over South America (the star symbol defines the location of São Paulo megacity). Right, AOT observation from CIMEL sunphotometer at 550 nm and $PM_{2.5}$ column integrated model simulated concentration (PMINT) during a period of ten days starting on 22 August 2005 [*Landulfo et al*., 2009]

An assessment of the relative contribution of urban and biomass burning emissions on the near surface ozone on the regional scale was conducted using the regional chemical transport model CCATT-BRAMS [*Longo et al*., 2009]. The transport of O_3 and its precursors by the typical synoptic systems in the planetary boundary layer affects mainly areas on the central part of South America; however long range transport at upper levels affects the atmospheric chemistry far away from the source areas and has the potential to connect the regional smoke plume with local urban emissions.

4.1.6 Air Pollution Footprint of South American Megacities

An estimate of the footprints of South American megacities is shown in Figure 4. The contribution from southeastern megacities in South America to surface ozone levels during October 2007 was in excess of 30%, affecting large areas downwind of these cities. Santiago's footprint is relatively concentrated due to the enclosed basin in which it is located. However, its urban emissions affect larger areas on the West coast of South America, well beyond the city borders. Figure 5 shows the monthly mean O_3 mixing ratio over São Paulo, Santiago, Buenos Aires, and Bogotá with and without urban emissions included in the simulation. São Paulo urban

emissions increase O_3 about two times near the surface and affect the lower troposphere from the surface up to 3 km. A small perturbation is also noted in the middle and upper troposphere (3 to 9 km), which might be explained by convective transport. Santiago imposes a much shallower vertical extension of the disturbance (less than 2 km) and increases O_3 by a factor of 20% on average, which can be explained by the low occurrence of moist convection. Buenos Aires shows a similar pattern to São Paulo except that the disturbance is confined to below 4 km. Noticeable is the deep vertical disturbance on tropospheric ozone imposed by Bogotá. Its urban emissions strongly affect the troposphere from surface up to 10 km. The most plausible explanation for this feature is the intense convective regime in this area. These findings point to a high potential impact on regional and global scales by increasing emissions on urban areas located in the Northwestern part of South America.

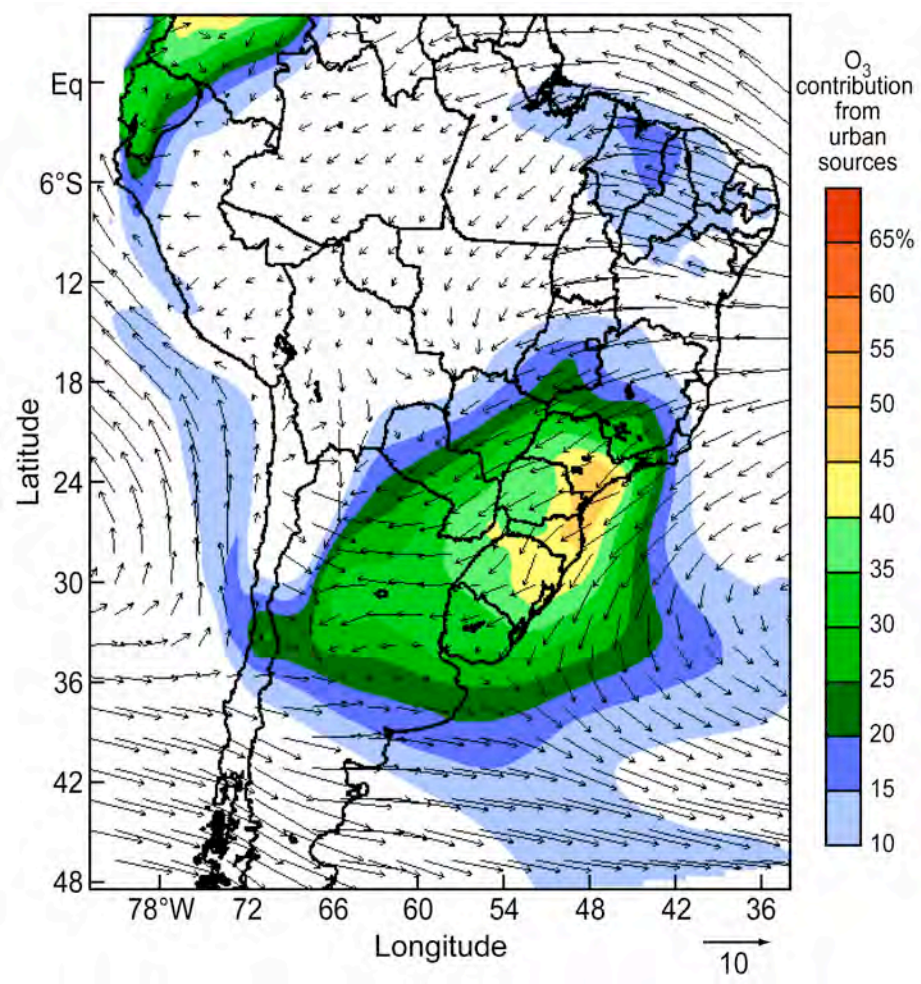

Figure 4 - Percentage O_3 contribution from urban sources elucidating the South American megacities footprint and average wind fields within the first 1 km height above local surface.
Average for October 2007

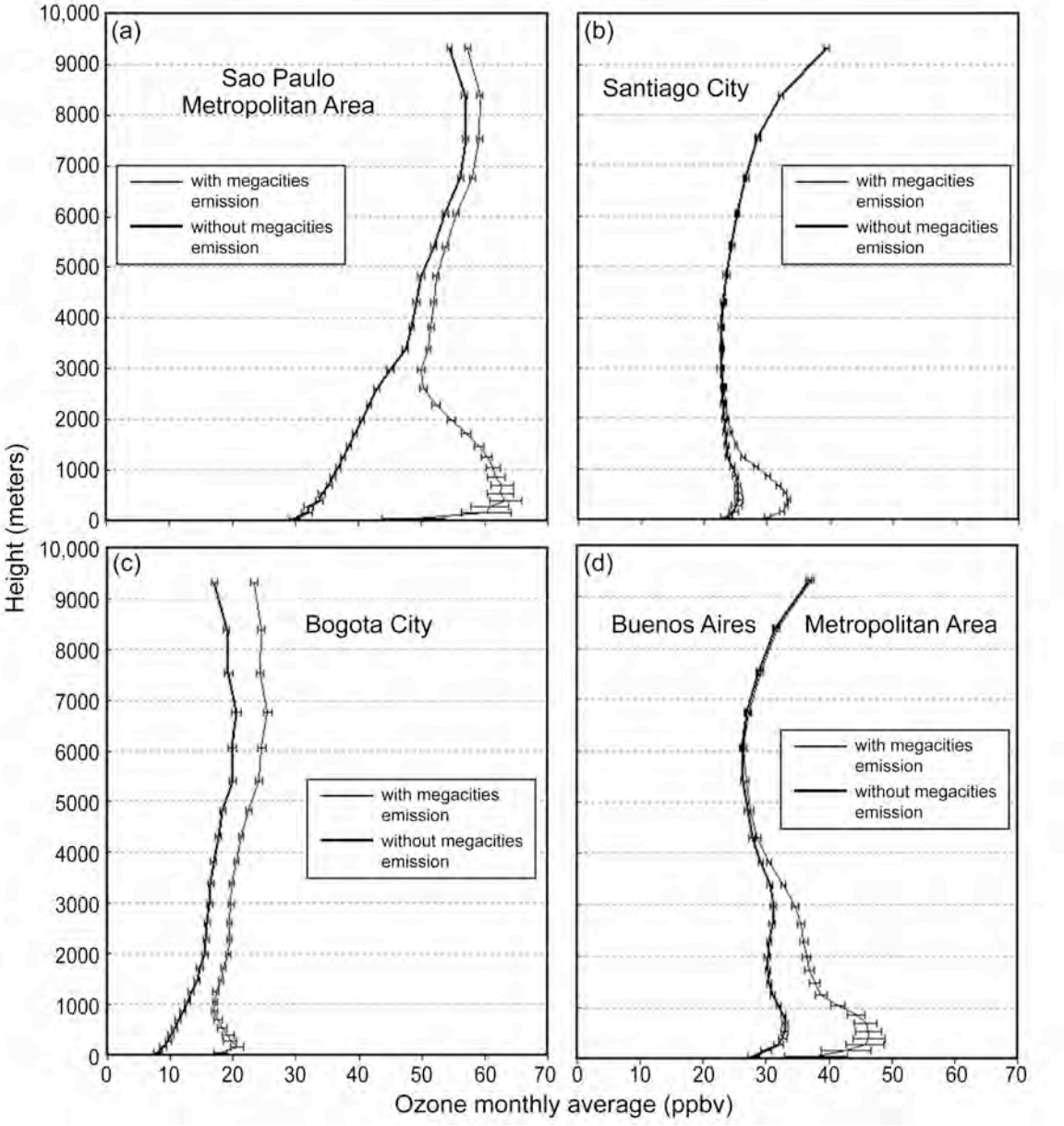

Figure 5 - Monthly average vertical profile of O_3 concentration over (A) São Paulo, (B) Santiago, (C) Buenos Aires and (D) Bogotá cities considering simulation with and without urban emissions

4.2 BOGOTÁ

Bogotá, Colombia, located at 4˚36'N and 74˚4'W at an altitude of 2625 m.a.s.l., covers an urban area of ~520 km^2 with a population of approximately 8.5 million (http://www.citymayors.com/statistics/largest-cities-density-125.html) as of 2010, making its population density around 13,500/km^2, among the ten highest in the world. Bogotá's growth, however, has been rather chaotic, with around a quarter of a million people arriving in the city between 1995 and 2005 [*Urdinola*, 2001]. Annual population growth was as high as 3% during the 1980s and 1990s, and it is currently around 1.5%.

Bogotá is located on the eastern branch of the Colombian Andes. The city and its Metropolitan area are built on a fertile plateau known as Bogotá's Savanna, bordered to the east, southeast by a mountain range reaching 4200 m.a.s.l. To the west, Bogotá is bordered by another mountain range reaching 3600 m.a.s.l. that goes down to the Magdalena Valley (500 m.a.s.l.). A

third mountain range partially borders Bogotá to the North, limiting the influence of winds and the transport of moisture and pollutants.

Bogota's climate is influenced by rainy and dry seasons. The passage of the Intertropical Convergence Zone (ITCZ) and its trade winds produces two rainy seasons in March, April, May, and another in September, October, and November. The rest of the year is dry. The annual average temperature is 14°C, with daily oscillations between 5°C and 23°C in the dry season and between 9 and 17°C in the rainy season. These large daily oscillations promote the formation of thermal inversions during the night, which are clearly visible as a brown haze over the city in the early morning hours. The inversions are normally break around 8 AM. Bogotá's climate is also under influence of the El Niño/ La Niña phenomenon.

Since the 1990's, transportation plans were developed in Bogotá to reduce air pollution. The transportation plans resulted in the construction of a highly efficient Bus Rapid Transit (BRT) system under the name of *Transmilenio* and an extensive bicycle path network of 340 km known as *Ciclo Ruta*, with the idea of promoting the use of high-quality, sustainable public transportation and reducing the dependence on private cars. The BRT system serves 25% of public bus trips; the remaining 75% are still being served by a disorganized system with undetermined bus stops, aggressive driving, and high-emitting buses. The third phase of the BRT system is now under construction and will represent an additional 15% of total public bus trips by 2012 (http://www.transmilenio.gov.co).

Despite the relatively low share of trips made in private cars, traffic jams are frequent. This is due to several causes including the inconvenient distribution of main avenues, the poor maintenance of many secondary roads, and the inadequate behaviour of public buses, taxis, private cars and motorcycle drivers, among others. Several restrictions on private cars have been implemented to initially reduce the traffic load during rush hours, but have been extended to the whole day (6 AM. to 8 PM.). Restrictions to public buses depend on the environmental performance of the buses, which is controlled according to the opacity level of their exhaust emissions, and are known as the *Programa de autorregulación* which works together with the *Pico y placa ambiental*, a one-day-a-week restriction based on the final plate number of the vehicles. According to the national inspection and maintenance programme, vehicles must undergo an emissions test every year if they are private or every half-a-year if they are public, The inspection is carried out at privately owned centres called *Centros de Diagnóstico Automotor* (CDA).

As the main centre of Colombia's economic activity, Bogotá and its metropolitan area have attracted a number of important industrial facilities across a diverse range of activities. The industrial facilities range from food and drink production to textiles, chemicals, plastics, and rubber to metallurgy and metal products to non-metallic mineral extraction and production and paper, cardboard and derivatives production. Many of these facilities are west of the main urban centre, while others are located in industrial sites in neighbouring towns to the north and west.

During the last decade, several initiatives were undertaken to build an emission inventory. At first, it was based on emission activity indices and foreign emission factors [*Zárate et al.*, 2007]. Recently, a new inventory considering local emission factors has become available [*Behrentz et al.*, 2009]. Traffic related emissions are identified as the main source of particulate matter.

Since 1997, 13 automatic air quality stations (12 stationary and 1 mobile) measure criteria air pollutants. Datasets are available to the public on the website of the city's Secretary of the Environment (http://www.secretariadeambiente.gov.co).

Various studies have concluded that concentrations of NO_X, SO_X and CO are not a significant threat to public health, since they are normally below the Colombian air quality standards of 80 ppb (24-hour), 96 ppb (24-hour) and 8.8 ppm (8-hour), respectively [*Behrentz et al.*, 2009]. However, PM_{10} concentrations show a high number of exceedance-days of the annual air quality standard of 60 $\mu g/m^3$ and the 24-hour air quality standard of 150 $\mu g/m^3$, particularly in the Western bound part of the city. The USEPA 24-hour limit of 20 $\mu g/m^3$ for $PM_{2.5}$ is exceeded most of

the year, whereas the annual standard of 65 μg/m^3 is not exceeded. Ozone is also a regulated air pollutant and its concentrations often surpass the Colombian standard (1-hour averages > 81 ppb). In spite of significant changes in traffic management, an increasing transition from coal to natural gas for industries, and decreases in fuel sulphur content (from over 1000 ppm in 2007 to 50 ppm in 2010), over the last ten years, no significant pollution trends are observed in Bogotá, except for CO. These features are illustrated in Figure 6. The health impacts of air pollution in Bogotá have also been studied [*e.g., Hernández*, 2009].

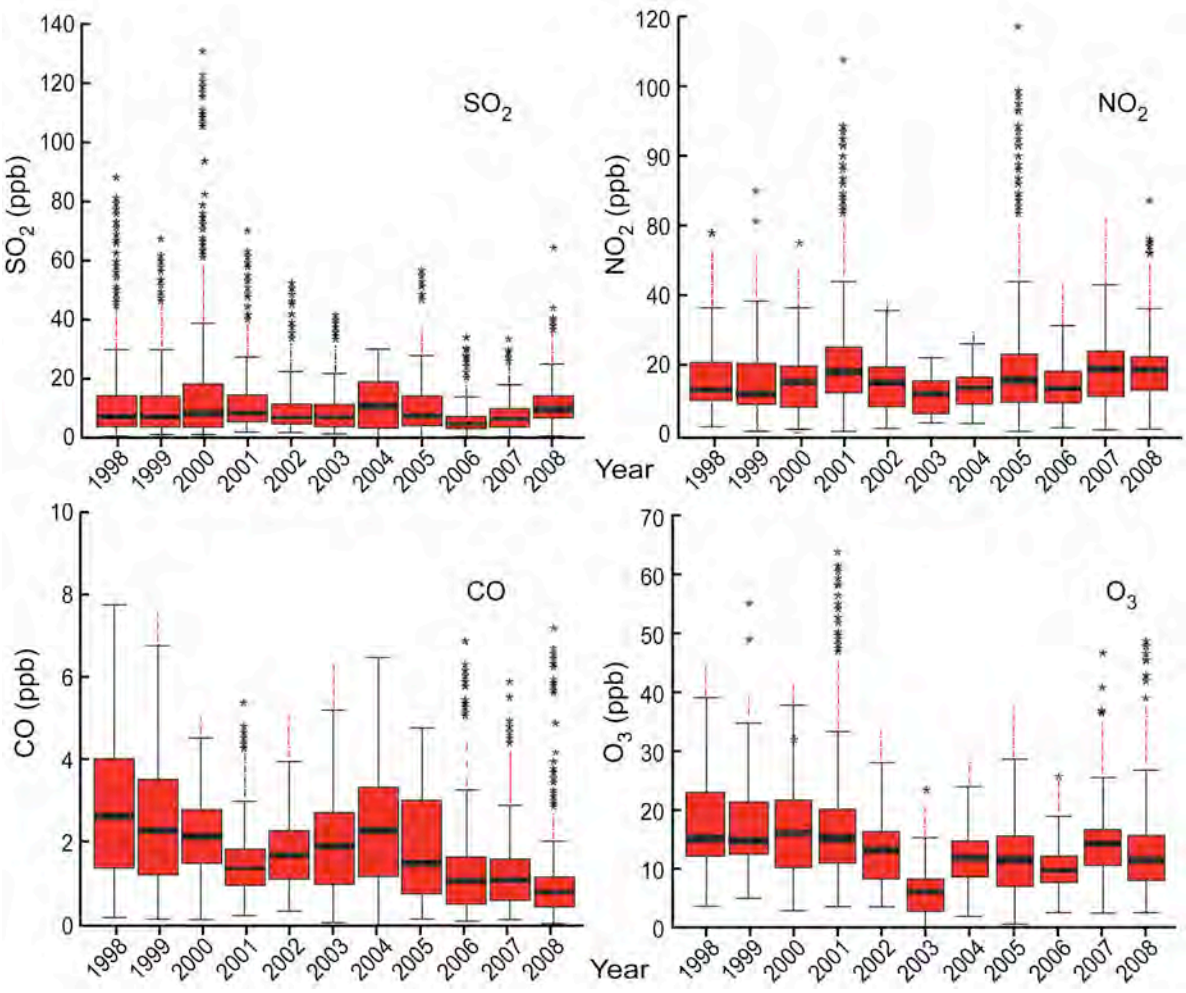

Figure 6 - Evolution of annual average of hourly concentrations of sulfur dioxide, nitrogen dioxide, carbon monoxide, ozone, PM_{10} and $PM_{2,5}$ in Bogotá for the period 1998-2008. Data from all monitoring stations included [*Secretaría Distrital de Ambiente*, 2008]

Although the number and quality of studies in the last decade have resulted in an increased knowledge of air pollution in Bogotá, the sources and impacts of air pollution in Bogotá have increased significantly and further research is needed to reduce uncertainties. Therefore, more effective air pollution research and policy strategies should be applied in the future. Air quality modelling would definitely help in solving these uncertainties and it has been perhaps the least developed air quality management tool in the city. *Zárate et al.* [2007] developed an air quality model for Bogotá. Unfortunately the Secretary of the Environment has not used the model on a continuous basis due to a lack of a capacity building strategy to use the model regularly, but there have been recent developments towards improving and distributing emission inventories with modelling purposes. Another contributing factor to the lack in decreasing pollution trends is the weakness in enforcing regulations.

4.3 BUENOS AIRES, ARGENTINA

The Metropolitan Area of Buenos Aires (MABA), Argentina is located at 34˚35”S and 58˚40”W at an altitude of 20 m.a.s.l, with a population of ~13 million as of 2009 (http://esa.un.org/unpd/wup/unup/). The MABA is situated by the estuary of the La Plata River, resulting in good ventilation and a low frequency (~ 4%) of calm situations. Summer months are characterized by high temperature (24 °C daily average) and humidity values (60-70%), which is associated with the development of thunderstorms during the afternoon hours. Summer precipitation is between 150-250 mm/month. Occasionally, these conditions are disrupted by cold front passages originating in the South. Winter months are characterized by lower temperatures (11 °C daily average) and higher humidity (ca. 80%). Overall, the prevailing atmospheric conditions and MABA’s topography prevent the accumulation of pollutants. At the same time, these conditions favour long-range transport.

During the last twenty years, research institutions have performed air quality measurement activities in the MABA region in an uncoordinated manner producing fragmented and scarce information for MABA. In the City Area since 1985 and for the following 20 years, air quality monitoring was conducted at only one monitoring site (Palermo). Previous to 1985 there is a historical record for Buenos Aires air quality between 1964 and 1983 [*Arrechea*, 1998]. The current authorities in the city of Buenos Aires are installing a new monitoring network composed of four stations that meet international quality standards to measure criteria pollutants (http://www.agenciaambiental.gob.ar). In the literature, one can only find sporadic measurements in connection with a few campaigns [*e.g., Bogo et al.*, 2003; *Reich et al.*, 2006; *Dos Santos et al.*, 2009].

In 2009 the annual average of 1-hour CO concentrations measured at the Palermo monitoring station was 25 % lower than that of 2002 (Figure 7). The key driver for the fall in concentration levels is the decreasing level of emissions associated with the incorporation of new gasoline vehicles with lower emission rates in spite of the increasing number of vehicles. Time series of 1-hour and 8-hour CO concentrations measured by the government of the City of Buenos Aires are, in general, below air quality standards. The levels increase from open to congested areas and reach maximum levels during winter months and minimum levels during summer months. The reported exceedances of City of Buenos Aires standard are normally registered during rush-hour of working days and close to vehicle exhaust emissions generally confined within urban street canyons.

In the period, 2002-2009, annual average of 1-hour NO_X concentrations measured in the Palermo monitoring station exhibit fluctuations (Figure 7) largely associated with NO levels, while NO_2 presents a flatter pattern. Similar to CO concentrations NO levels are higher in winter than in summer. NO_2 concentrations measured by the government of the City of Buenos Aires show a few exceedances of City’s standard in the reported period.

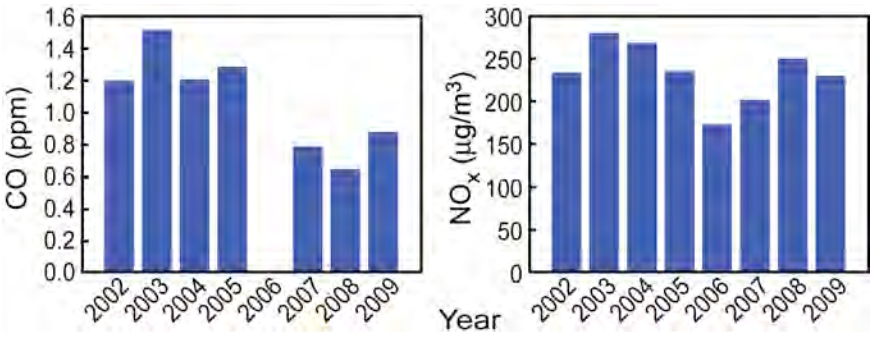

Figure 7 - Annual mean concentrations of CO (ppm) and NO_X (µg m-3) from 2002 to 2009 in the reference monitoring site *[Palermo, http://www.agenciaambiental.gob.ar]*

Maximum SO_2 concentrations measured in the 1970s were 90% lower than the air quality standards. Therefore, SO_2 was considered a pollutant of non-concern in the region and within a minimal budget for air quality monitoring the environmental authorities decided against systematically measuring this pollutant. This decision has remained until present. The available SO_2 concentration data come from different measurement campaigns undertaken mostly in the late 90s [*e.g., Bogo et al.*, 2001]. However the increasing use of liquid fuels in both mobile sources, including ships, and power plants have the potential to revert the present low levels towards increasing values. Therefore, it may be advisable to include SO_2 in future air quality monitoring strategies.

The few measurements of O_3 concentrations in the MABA region have been undertaken by academia. The results show that 1-hour concentrations do not exceed the standards of the City of Buenos Aires. In accordance with ground level ozone chemistry, O_3 and NO concentrations are inversely correlated, with O_3 concentrations being lower in areas with high vehicular traffic. Because of the small size of the sample, the O_3 situation in the region cannot be assessed.

In line with worldwide trends, monitoring of particles in the MABA evolved from measuring total suspended particulate matter (TSP) to PM_{10} and more recently $PM_{2.5}$. Measurements of TSP were carried out mostly by governmental agencies while those of PM_{10} and $PM_{2.5}$ were carried out by research groups that have been interested in a number of issues such as assessing the impact of specific sources, characterizing multi-elemental aerosol composition, and source identification through receptor models. PM_{10} levels are relatively high, although 24-hour concentrations do not typically exceed the air quality standard. The monthly average PM_{10} levels are in some cases close to or slightly higher than 50 $\mu g/m^3$. $PM_{2.5}$ levels are typically high with 24-hour concentrations close to or relatively higher than the air quality standard of the City of Buenos Aires whilst monthly averages are twice the strict limit established by the US air quality standards. Mass concentrations of PM_{10} levels increase from the inner part of the city towards the La Plata river shore while $PM_{2.5}$ is the size fraction more homogeneously distributed. The influence of the adjacent ocean is apparent in terms of sea salt and halogens [*Dos Santos et al.*, 2009].

Lead concentrations in particulate matter were reported in three different studies [*Caridi et al*., 1989; *Ozafrán et al*., 1999; *Smichowski et al*., 2004]. The two latter studies show the decreasing levels of this metal with respect to those measured in the first study, reflecting the phase out of the use of lead alkyl additives to gasoline enforced in Argentina since 1995.

Until recently, no consistent emission inventory was available for Buenos Aires. The first systematic attempt is described in *D'Angiola et al.* [2010]. This inventory refers to on-road mobile sources of MABA and reports emissions for the period 2000-2006 of greenhouse gases (CO_2, CH_4 and N_2O) and criteria pollutants (CO, NOx , NMVOCS, PM and SO_2) disaggregated by district and also presented in a grid of 1x1 km^2 using a distribution algorithm described by *Saide et al.* [2009b]. The same research group is now developing the inventory for stationary sources [*Oreggioni*, 2009]. According to these data, on-road vehicles are the main contributors for almost all of the pollutants in MABA with the only exception of SO_2 and CO_2, for which stationary sources are dominant.

Compared to other cities in South America, the existent relatively high emission rates from MABA are generally not perceived as an environmental threat due to the atmospheric condition that rapidly vent the MABA. Nevertheless, the large population density of Buenos Aires constitutes a risk as more population is exposed to pollution, increasing the probability of health impacts. In addition, the strong ventilation and vertical mixing of the MABA, the tenth largest megapolis of the world, is subject to impact long-range transport of pollution from Buenos Aires.

4.4 LIMA, PERU

Lima, Peru is located at 12˚3'S and 77°3'˚W at an altitude of 110 m.a.s.l with a population of ~9 million inhabitants. The quasi-permanent subtropical Pacific high and the Chilean-Peruvian oceanic current determine largely the climate of Lima, which is arid with an annual precipitation of

less than 10 mm/year. Winter temperatures range between 15°C and 19°C and summer temperatures between 21°C and 29°C with high relative humidity (ca. 80%) year around. The permanent high humidity is associated with the presence of a stratocumulus deck capped by the subsidence inversion. In summer, when the subtropical high is weaker, the cloud deck breaks by mid-morning whereas in winter cloudy skies prevail all day. To the east of Lima there is a mountain range reaching altitudes of about 1000 m.a.s.l. that creates, together with the thermal contrast between land and ocean, up-slope winds in the afternoon and down-slope winds in the night and morning hours [*SENAMHI*, 1988].

In the last few years, and in connection with the establishment of an attainment plan for the Lima region, an emission inventory for stationary sources of criteria pollutants has been compiled [*DIGESA*, 2005]. Preliminary estimates are also available for mobile sources [*MTC*, 2010]. These estimates are not spatially distributed. Only recently, these inventories are being verified and updated under the leadership of the National Hydrological and Meteorological Service of Peru (SENAMHI) within the framework of chemical weather forecasting applications.

The Peruvian Ministry of Health (Dirección General de Salud Ambiental, DIGESA) in 2000 started systematic air quality measurements at five stations. These stations provide information about SO_2, NO_2, $PM_{2.5}$, and PM_{10}. Only part of this information is available on the Internet (http://www.digesa.minsa.gob.pe/DEPA/pral2/lima.asp). Lately, SENAMHI has also compiled air quality data for O_3, NO, NO_2, SO_2, and PM_{10} with automated stations at five sites. These data that have hourly resolution are available on the Internet (http://www.senamhi.gob.pe/?p=0412).

Peruvian air quality standards for NO_2 (hourly 200 $\mu g/m^3$), SO_2 (80$\mu g/m^3$, 24 hours) and $PM_{2.5}$ (50 μg/m3, 24 hours) are often exceeded. NO_X follows traffic activity with maximum concentrations coinciding with the morning and evening rush-hours. PM_{10} concentrations are highest in summer by mid-morning, which suggests the contribution of multiple sources. Concurrently, *Arellano* [2010] analyzed the mixing height over Lima and showed that the mixing layer height reaches 500 m in the summer and 700 m in the winter. High concentrations of particulate matter and sulphur oxides are associated with a very old fleet (>25 years old), very high sulphur content in diesel (5000 ppm), and poor traffic control. The situation is slowly starting to improve as a BRT system is being implemented and sulphur content is expected to decrease to 50 ppm by 2015.

In addition to the obvious health impacts to be expected in a city like Lima, it is also important to address the potential impacts of Lima's pollution on cloud properties. Such impacts have been suggested earlier [*e.g., Kuang and Yung*, 2000] and they might be enhanced by the presence of La Oroya copper smelter some 150 km northeast from Lima that is responsible for an annual emission of sulphur dioxide around 0.16 TgS/year [*Carn et al.*, 2007].

4.5 MEDELLÍN, COLOMBIA

The city of Medellín, Colombia is located at 6˚17'N and 75˚32'W at an altitude of 1500 m.a.s.l. with a population of ~3.5 million. Medellín is situated in the Aburrá Valley between the basins of the Cauca and Magdalena rivers. As in the case of Bogotá, the climate of the area is determined by the passage of the ITCZ with rainy seasons between March and May and between September and November and relatively dry conditions during the rest of the year. The Aburrá Valley is bordered by high mountains (ca. 2500 m.a.s.l.) that strongly influence the wind flow in the region.

A sustained effort regarding emission inventories has taken place since 2000 in the Aburrá Valley (http://www.metropol.gov.co/aire or http://modemed.upb.edu.co). This is a joint venture between environmental authorities and academia. Today's inventory for criteria pollutants and their precursors is a dynamical and automatized database that allows the construction of emission scenarios including changes in emission factors, activity data, etc. Overall, this inventory goes

beyond mere numbers and it provides a more mechanistic approach in which the methodology is transparent to the (authorized) users.

The air quality monitoring network of Aburrá Valley retrieves continuously data collected by automated instruments. Elevated particulate matter concentrations are of concern, often exceeding the new Colombian annual standard of 50 mg/m^3. For instance, in 2008, annually averaged PM_{10} concentrations were above 50 mg/m^3 at 8 out of 10 stations. In addition, annually averaged $PM_{2.5}$ concentrations, measured at four sites, were about 30 mg/m^3. The ozone hourly standard of 61ppbv and 8-hour standard of 41 ppbv standards are also frequently surpassed. Available air quality data shows the close relationship between fuel consumption and vehicle emissions of PM_{10} and $PM_{2.5}$, since the maximum hourly concentrations are occur during rush-hour traffic.

The national air quality network (SISAIRE, http://www.siac.gov.co) is a potentially useful tool for air quality management but it must be fed with reliable data collected in a standardized manner and subject to strict validation and analysis procedures. Such procedures and standards require further work from the community in Colombia.

A notorious decline in sulphur content in diesel fuel took place in 2008 when it was lowered from 3200 ppm to 2127 ppm. In 2010, diesel sulphur content was reduced again to 500 ppm throughout all of Colombia and it is expected to be lowered to 50 ppm by 2013. Given the significant contribution of diesel use to mobile emissions, these measures should result in a lowering of particulate matter levels in the Aburrá Valley, and elsewhere in Colombia.

Observed high concentrations of $PM_{2.5}$ are linked to transportation, mainly diesel vehicles with very old technologies. Ozone precursors are also associated to mobile sources. Hence, a better characterization of mobile emissions is required. This will in turn require locally representative emission factors and better measurement capabilities.

4.6 SANTIAGO, CHILE

Santiago, Chile is located at 33˚27'S and 70˚40'W at an altitude of 500 m.a.s.l with a population of ~6 million inhabitants. The city of Santiago is located in a semi-arid basin (annual rainfall <350 mm) in the central part of Chile bounded by the high Andes (4500 m altitude on average) to the East, a lower parallel mountain range to the West (1500 m altitude on average), and two east-to-west mountain chains to the North and South of the basin respectively. The climate of Santiago is characterized by the quasi-permanent influence of the subtropical Pacific high, and the intrusion of occasional cold fronts, which bring precipitation in wintertime. The South Pacific high determines quasi-stagnant anti-cyclonic conditions that are further intensified, especially in fall and winter by the presence of sub-synoptic features known as coastal-lows [*e.g., Garreaud et al.*, 2002]. There is a characteristic radiatively driven circulation that defines up-slope southwesterly winds in the afternoon and down-slope northeasterly winds in the night and morning hours, more strongly so in the summer.

The regional office of the Ministry of Health is in charge of monitoring Santiago's air quality, which has been measured rather continuously since 1988. A historic record of the data (1988-2008) is available (as in July 2010) on the Internet via the Ministry of Health (http://www.asrm.cl/). The freely available data presented as "validated data" by the Ministry of Health comes in various formats, and, unfortunately, quality control procedures are not readily available nor accompanying the data set. The species measured are so-called criteria pollutants: CO, SO_2, O_3, NO, NO_X, PM_{10} and $PM_{2.5}$. Wind velocity, temperature, relative humidity are also continuously measured at the monitoring stations but these data are not available on the Internet, limiting the assessment and interpretation of the monitoring data. In general, measuring instruments used follow recommendations from the Environmental Agency of the United States of America (USEPA).

Figures 8 and 9 show the evolution of daily averages at Las Condes and Pudahuel stations respectively for the period 1997-2008. Las Condes is in high-income area in the eastern bound

part of Santiago where the highest ozone mixing ratios (>100 ppbv) are usually measured. Pudahuel is in a low-income area in the western side of the city. Pudahuel is the station that usually triggers pollution events by inhalable particle matter during the nighttime. Only CO, SO_2 and PM_{10} show at both stations statistically significant decreases since 1997. At Las Condes a marginal decrease in ozone is observed, coinciding with roughly unchanged or slowly increasing NO_X levels since 2000. Carbon monoxide median values decrease moderately and rather homogenously at all stations, with a slightly steeper slope before 2000. For $PM_{2.5}$, there is no statistically significant trend since 2000. PM_{10} shows up to 35 mg/m^3 and a 40 mg/m^3 decline in median and mean daily averages, respectively, over 12 years. Still, except for Las Condes, annual mean daily averages exceeded 75 mg/m^3 in 2008.

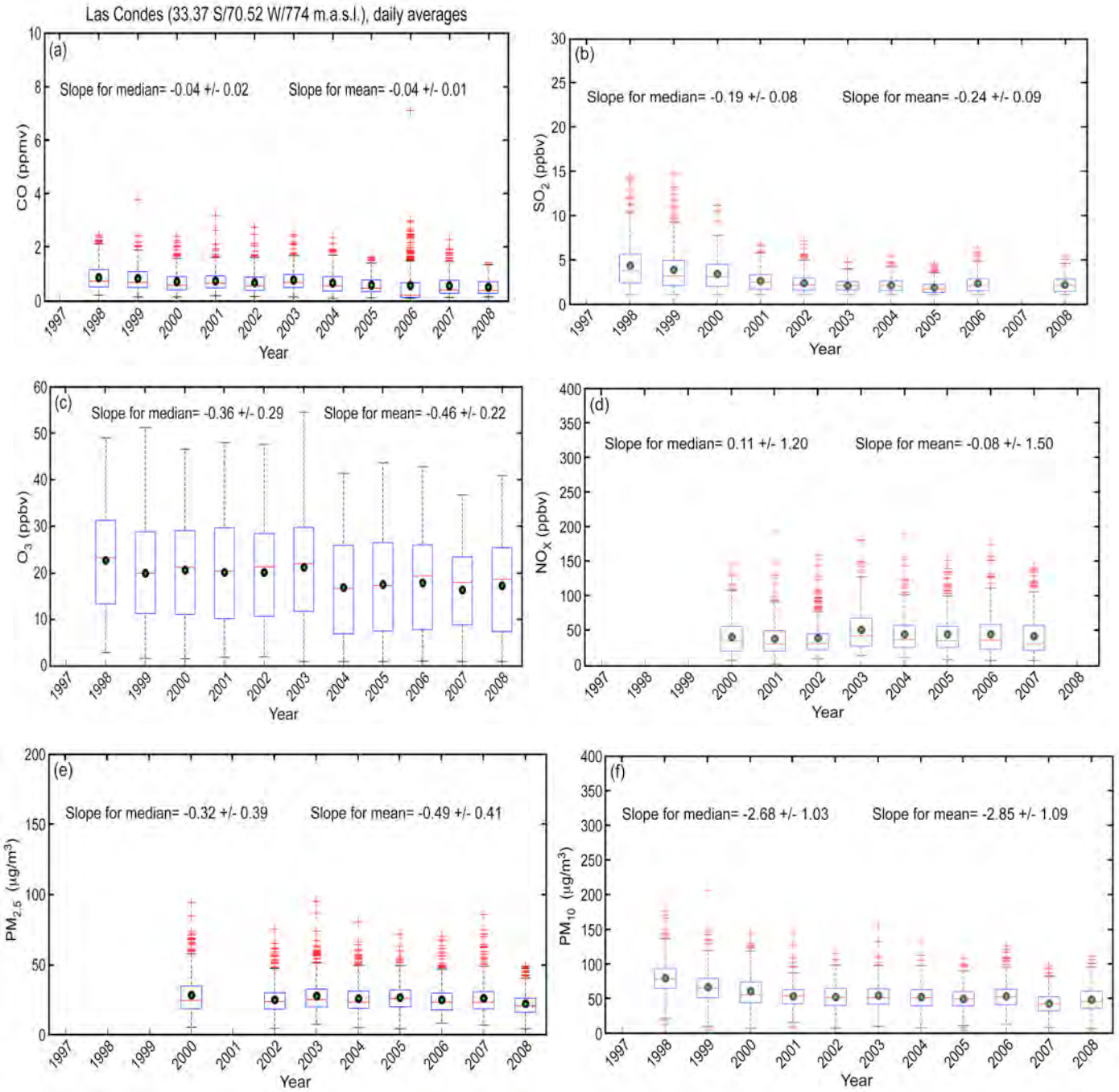

Figure 8 - Air quality data collected at station Las Condes since mid 1997 up to 2008. These are box plots of daily averages. Lower quartile, median, and upper quartile values are shown by the boxes. The whiskers indicate the extent of the rest of the data. Outliers are illustrated with red crosses. The linear annual trends (slopes of linear regressions), and the corresponding 95% confidence levels, of median and mean values are also indicated. Arithmetic means are indicated by green dots

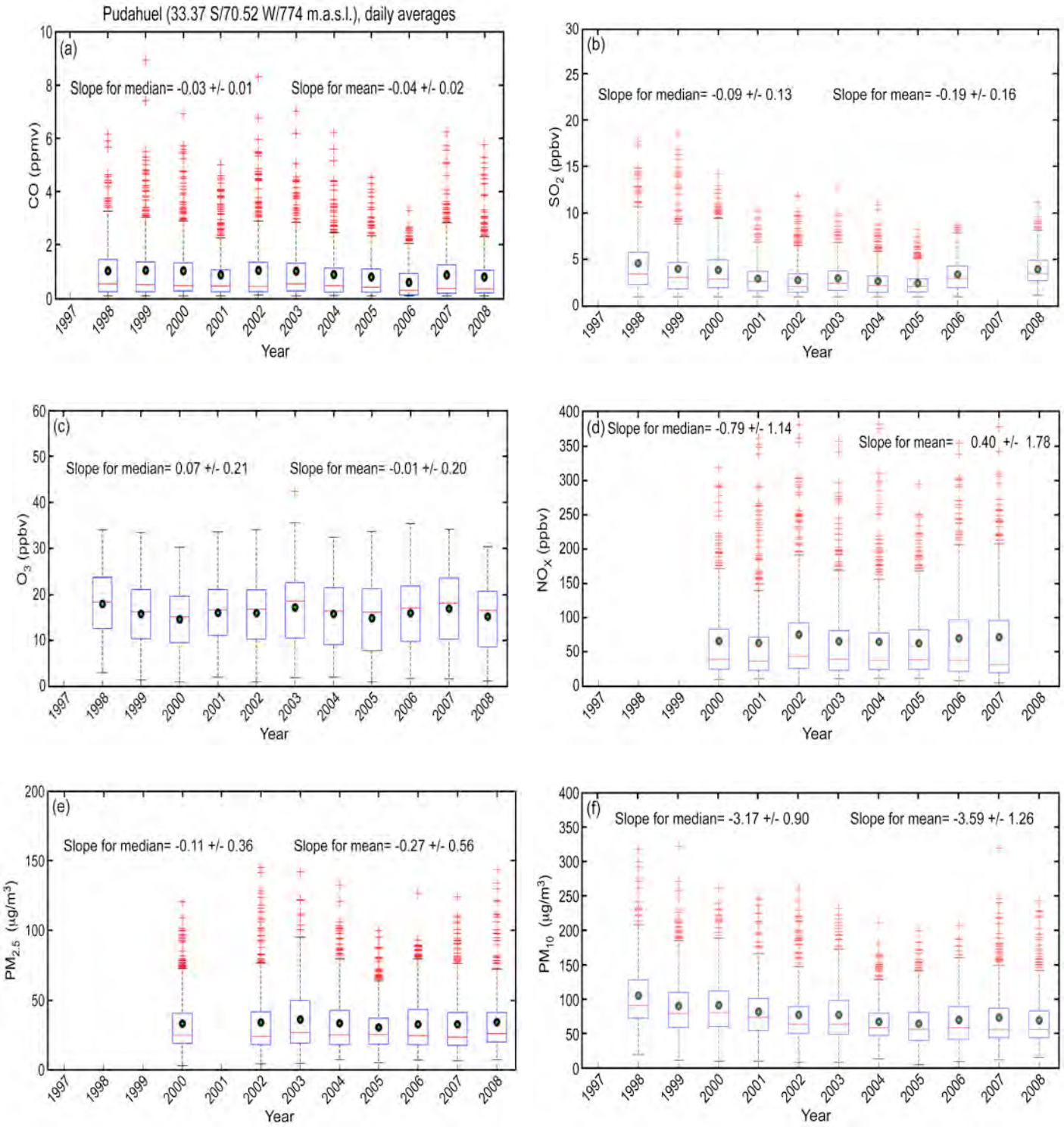

Figure 9 - Air quality data collected at station Pudahuel since mid 1997 up to 2008. These are box plots of daily averages. Lower quartile, median, and upper quartile values are shown by the boxes. The whiskers indicate the extent of the rest of the data. Outliers are illustrated with red crosses. The linear annual trends (slopes of linear regressions), and the corresponding 95% confidence levels, of median and mean values are also indicated. Arithmetic means are indicated by green dots

The National Commission for the Environment (CONAMA, now Ministry for the Environment) has over the years driven various campaigns to assess the composition of particles [*e.g., Artaxo et al.*, 1999; *Didyk et al.*, 2000; *Kavouras et al.*, 1999; *Gramsch et al.*, 2009] and to address speciation of volatile organic compounds and photochemical products [*e.g., Rappenglück et al.*, 2000; *Rubio et al.*, 2004; *Rappenglück et al.*, 2005]. Other studies have been developed within a pure academic framework addressing, again, particles and photochemistry [*e.g., Sienra et al.,* 2005; *Sienra and Rosazza*, 2006; *Rubio et al.*, 2006; *Richter et al.*, 2007; *Morata et al.*, 2008; *Elshorbany et al.*, 2009a; 2009b; *Seguel et al.*, 2009].

In addition to air quality stations, a meteorological network was put in place in 1997. It consisted of 22 stations and it was designed to capture mesoscale meteorological features induced by complex topography in the area. Today, only 10 stations are operational. Vertical soundings have been sporadic, and only since 2007, a ceilometer located in downtown Santiago is providing a record of daytime mixed layer for cloud free days [*Muñoz and Undurraga*, 2010]. In 2010, a backscatter LIDAR was installed by the Chilean Weather Office that is expected to provide unprecedented information about atmospheric stability and aerosol properties.

Regarding impacts of air pollution, health issues have received the greatest attention, particularly the association between particulate matter and morbidity and mortality statistics [e.g., *Adonis and Gil,* 1993; *Ostro et al*., 1996; *Ostro et al.*, 1999; *Ilabaca et al.*, 1999; *Cifuentes et al.*, 2001a; 2001b]. A few studies have approached health issues within the framework of climate change scenarios and mitigation measures [*e.g., Cifuentes et al.*, 2001a; 2001b]. Less attention has been paid to effects on vegetation [*e.g., García-Huidobro et al.*, 2001]. Only recently, the potential impacts of air pollution on the stratus deck downwind from Santiago are beginning to be studied [*e.g., Mena et al.*, 2009; *Spak et al.*, 2010; *Saide et al.,* 2012; *Heinrichs et al.,* 2012].

Since the implementation of the attainment plan for Santiago, CONAMA has supported formulation of emission inventories for base years 1997, 2000, and 2005, and a projection for 2010 [*CENMA*, 1997; *CENMA*, 2000; *DICTUC,* 2007]. Unfortunately, over time this endeavour has been contracted under consultancies of different organizations giving room for differences between methodologies and even lack of transparency in some results.

Mobile emissions were estimated in the CENMA inventories (1997, 2000) according to a bottom-up methodology [*Corvalán et al.*, 2002] considering official traffic modelling results, comprehensive traffic counts, analysis of databases for vehicle technology distribution and emission factors from COPERT III model [*Ntziachristos and Samaras*, 2000] and local measurements. Later, subsequent contractors have modified this methodology but it is unclear what the modifications are. Industrial and biogenic emissions are estimated following USEPA methodologies.

In the first version of the attainment plan (1997), curbing measures considered the introduction of natural gas in the industrial sector, a reduction in sulphur content in diesel (from 5000 ppm in 1989 to 1000 ppm in 1997, and 300 ppm in 2001), introducing emission controls for vehicles and phasing out 3000 old buses, etc. These measures explain the relatively fast reduction in PM_{10} and SO_2 between 1997 and 2000. A second revision of the attainment plan in 2004 emphasized emission control for vehicles, including a reduction in diesel sulphur content to 50 ppm, and the introduction of an ambitious public transportation system called Transantiago (http://www.transantiagoinforma.cl/). Transantiago intended to completely overhaul the existing transportation system with new EURO III diesel buses, and reducing the total amount of buses from more than 8000 to ~4000. Soon after its implementation in early 2006, the collapse of the system due to increased demand required that 2500 old buses (which had been removed from circulation) to be reintroduced, bringing the current total to 6500 units. The contracts required that these old buses would use retrofitted particulate filters, as their emissions are roughly 10 times higher than the new Euro III buses. To date these buses continue to circulate without particulate filters, making the success of Transantiago's emission reductions less apparent.

Between 2000 and 2006, a toll based urban highway was built, which allowed sprawling suburbs to connect with the city in substantially less time. The combination of a slow public transportation system with an efficient highway system may have led previous users of the public transportation system to buy cars and motorcycles. From 2000 to 2008, the number of cars in Santiago grew by 42% to 1.2 million. This growth in private transportation probably accounts for the lack of substantial reductions in pollutant levels despite the improvements in fuel and vehicle technology over the same period. In addition, after 2004, imports of natural gas from Argentina were restricted forcing industry to reconvert to other liquid fuels, such as petroleum products or diesel. This caused increased SO_2 and NO_X emissions, ultimately leading to increased $PM_{2.5}$ yearly means for 2007 and 2008. It is hoped that the recent installation of a Liquefied Natural Gas terminal will allow natural gas use to be re-established in Chile. The latest revision of the air quality pollution prevention programme (2009) includes retrofitting particulate filters in new and old buses and trucks, a scrapping programme for older gasoline and diesel vehicles, introducing a more stringent emission standard for wood burning heaters, banning agricultural burns, and a cap and trade system for SO_2 and NO_X emissions from industry.

According to *Fuenzalida et al.* [2006] who provided the first set of regional present and future climate scenarios using dynamical downscaling, the central part of Chile, where Santiago is located, will see a 40% decline in precipitation, an increase in surface temperatures, and a southward expansion of the subtropical high. These changes may in turn affect stability and ventilation but these aspects were not addressed in this study. One can only speculate that the probable southward expansion of the arid regime could intensify the radiatively driven circulation in the Santiago basin, generating more summer like conditions, which are very favourable for photochemical pollution. If the degree of centralization persists, the expected demographics combined with a warmer climate may result in increased vulnerability [*e.g., Bell et al*., 2008].

Santiago's air quality has been subject to multiple studies since, at least, the early 1980's. Over the years, the amount and the complexity of such studies have increased notoriously. Like in many other cities in the world, research and management initiatives in Santiago were triggered by acute air pollution problems, in this case very high concentrations of inhalable particles (800 mg/m^3 hourly averages in the late 1980's) and associated respiratory problems that followed from uncontrolled traffic and urban growth in the mid 70's [*e.g., Romero et al*., 1999]. Therefore, the accent of these initiatives has focused first on reducing extreme pollution events that are responsible for acute effects and short-term air quality standards. Measures have focused on large emitters: industries, non-catalytic cars, buses and trucks [*e.g., Emmelin et al*., 2007; *Morales et al*., 2006]. Such curbing measures could be identified based on relatively imprecise emission inventories for criteria pollutants and simple receptor modelling approaches. However, as the attainment objectives become more ambitious (e.g., long-term air quality standards for dealing with accumulative effects), the need of determining more subtle cost-effective measures and more precise tools increases. This, in turn, requires coordinated efforts to provide more of a systemic approach [e.g., *Molina and Molina*, 2004].

The number of active, highly qualified (PhD) researchers that can contribute to address air quality and climate issues is increasing but it is still insufficient. For instance, according to the Chilean Academy of Science, in 2005 there were around 50 active scientists in the area of analytical and environmental chemistry, twice as many as in 1997 [*Allende et al*., 2005]. In Atmospheric Science, there are less than 20 active researchers at PhD level, and this number has probably tripled over the last decade. Up to now, the connection between policy making and research has been made based on short-term consultancies, which by construction hampers the establishment of necessary synergies and the study of more complex issues. Hence, it appears necessary to create a research consortium able to convey scientists from different disciplines, combining natural and social science, which, in addition to scientific knowledge, is able to provide independent opinions to environmental authorities. Such consortium could provide a platform for international collaboration and capacity building.

4.7 SÃO PAULO, BRAZIL

The Metropolitan Area of São Paulo, Brazil (MASP) is located at 23˚31'S and 46˚37'W at an altitude of 720 m.a.s.l. with a population of ~20 million inhabitants. The MASP is situated 60 km northwest of the South Atlantic coast and in a valley bounded by mountain ranges on the northwest side and to the south and southeast by the "Serra do Mar" scarp (ca. 1000 m.a.s.l.). MASP is one of the largest urban regions in the world. It covers a total area over 8000 km^2 and 19 % of this area is highly urbanized. Additionally, MASP is the most industrialized area in South America and is different from other cities in South America due to its unconventional mix of vehicle types. The vehicle fleet consists of approximately 9.7 million vehicles that burn on a variety of gasoline blends, including oxygenated gasoline, as well as ethanol and bio-diesel. The high levels of pollution in the MASP constitute a critical health problem in the region [*Lin et al*., 1999; *Braga et al.*, 2001; *Martins et al.*, 2002; *CETESB*, 2010].

The climate of São Paulo is characterized by a dry winter during June-August and a wet summer during December-March. The minimum values of daily monthly-averaged temperature and relative humidity occur in July and August (16 °C and 74%, respectively), and the minimum monthly-accumulated precipitation occurs in August (35 mm). Combined effects of the geographic location and relative intensity of the semi-stationary South-Atlantic Anticyclone and continental low-pressure systems control the seasonal variation of surface winds in São Paulo. They induce surface winds from NE-E during the winter and N-NE during the summer. In addition, cold fronts frequently affect this pattern in winter, as well as sea-breeze fronts [*Oliveira et al*., 2003].

MASP has since 1981 a systematic air quality monitoring programme run by the State Environmental Agency (CETESB, http://www.cetesb.sp.gov.br). Nowadays 21 automatic stations are in operation in MASP in addition to 40 automatic and 47 manual station operating throughout the state of São Paulo. In addition, measurement campaigns of other tracers have been performed in MASP. Results of these measurements can be found in the literature [*e.g., Castanho and Artaxo,* 2001; *Miranda et al*., 2002; *Andrade et al*., 2004; *Ynoue and Andrade*, 2004; *Miranda and Andrade*, 2007; *Martins et al.,* 2006; *Sánchez-Ccyollo et al*., 2009].

The concentration of pollutants in São Paulo in the last thirteen years shows, in spite of the vehicular fleet increase, a sharp drop in the concentrations of CO, NO_X, PM_{10} and SO_2 over the MASP (Figure 10). The decline in concentrations of these pollutants is mostly due to the Brazilian Vehicular Emission Control Programme (PROCONVE), described elsewhere [*Szwarcfiter et al*., 2005; *Sánchez-Ccoyllo et al*., 2007]. In 1979, the Brazilian government started the Alcohol National Programme (PROALCOOL), which led to new and important changes in the fuel composition of light-duty vehicles (LDV). In 2005, Petrobras (Brazilian Oil Company) introduced the S500 diesel with sulphur limit of 500 ppm replacing the 2000 ppm diesel at metropolitan areas. There was also a programme to control industrial emissions starting in the late 80's, when many fuel boilers were switched to electrical power or natural gas. Only in the case of O_3 concentrations, a declining trend is not observed. The maximum hourly Brazilian national air quality standard for O_3 is frequently violated in MASP [*Freitas et al.,* 2005b; *Martins and Andrade*, 2008a; *Sánchez-Ccoyllo et al*, 2009].

MASP has an official emission inventory, which is annually updated [*e.g., CETESB*, 2010]. However, this inventory is not geographically distributed. A great effort has been made to improve the emission inventory for MASP [*Ynoue and Andrade*, 2004; *Vivanco and Andrade*, 2006; *Martins et al*., 2006; *Sánchez-Ccyollo et al*., 2007; *Sánchez-Ccyollo et al.,* 2009]. Like in other urban areas, vehicles in MASP contribute greatly to emissions. According to the official emission inventory mobile sources are responsible for 97% of carbon monoxide (CO) emissions, 97% of hydrocarbons emissions (HC), 96% of nitrogen oxides (NO_X), 32% of sulphur oxides (SO_X), and 40% of particulate matter [*CETESB*, 2010].

To improve the mobile emission inventory for São Paulo, measurements inside road tunnels were performed [*Vasconcellos et al.,* 2005; *Martins et al*., 2006; *Sánchez-Ccoyllo et al*., 2009]. The mean contributions of heavy-duty vehicles (HDV) to emissions of BC, PM_{10}, $PM_{2.5–10}$,

and $PM_{2.5}$ were 29, 4, 6, and 6 times higher than were those of LDVs. The main constituent of diesel exhaust particles was found to be black carbon (BC). The calculated emission factors were used in air quality models to estimate the impact the vehicle fleet has on air quality [*Martins and Andrade*, 2008a; 2008b; *Sánchez-Ccoyllo et al*., 2007].

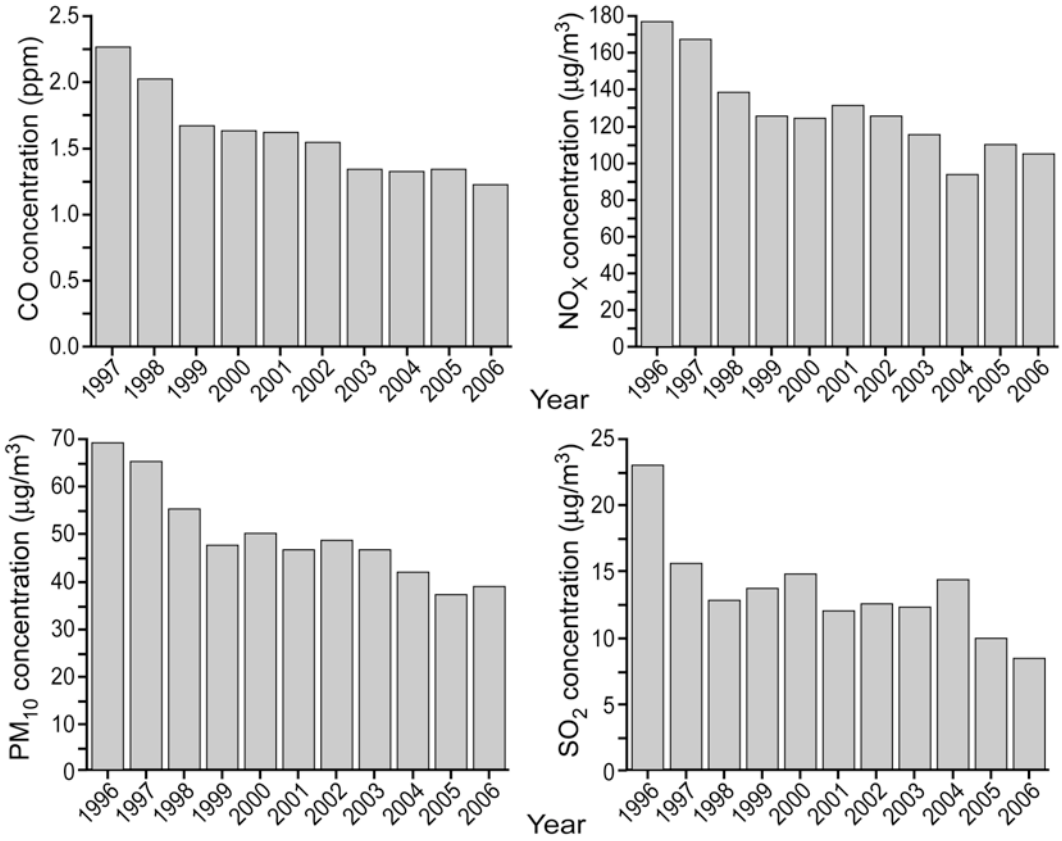

Figure 10 - Annual mean concentrations of CO (ppm), NO_2 (µg m^{-3}), PM_{10} (µg m^{-3}) and SO_2 (µg m^{-3}) from 1996 to 2006

Air pollution exposure has shown to cause increased respiratory mortality and morbidity [*Gouveia and Fletcher*, 2000; *Martins et al*., 2002; *Vermylen et al*., 2005; *Bourotte et al*., 2007]. Also, some studies in São Paulo have shown the impacts of air pollutants in mice [Silva et al., 2008] and the mutagenic activity of air organic matter extracts [*e.g., Umbuzeiro et al*., 2008]. In addition, *Martins et al.* [2010] investigated the relationships among air quality, aerosol size distribution, and meteorological conditions using data from aerosol number size distribution (9.82-414 nm) and numerical modelling for São Paulo. They suggested that the period characterized as clean, based on PM_{10} measurements, could not be considered a period presenting low health impacts, when using ultrafine particles concentrations as criteria.

Winter atmospheric measurements of gaseous carbonyl and carboxylic acids were carried out in urban sites in the city of São Paulo [*e.g., Montero et al.*, 2001]. High values of formic to acetic ratios were found suggesting that photochemical production was the predominant source of these acids during the afternoon. Higher average mixing ratios of acetaldehyde and formaldehyde were found in the morning and gradually decreased from midday to evening. In the morning, vehicle direct emission seemed to be the primary source of formaldehyde and acetaldehyde, whereas at midday and evening, these compounds appeared to be mainly formed by photochemistry [Montero et al., 2001]. In recent work, mutagenic and carcinogenic organic compounds were investigated [*Vasconcellos et al*., 2008]. Most nitro-PAH (polycyclic aromatic

hydrocarbons) were related to diesel powered vehicle emission. The exceptions are 2-nitrofluoranthene and 2-nitropyrene, photochemical compounds that appear as the most abundant in airborne particles, which seems to be associated with sugarcane emissions, suggesting that agricultural activities released the precursors of ambient nitro-PAH identified at all sites.

A tentative identification of some of the mechanisms that influence the concentrations of trace metals and water-soluble ions in different sites in São Paulo State was done [*Vasconcellos et al*., 2007]. Sugarcane burning and industrial activities are most strongly implicated. In addition, the burning of solid waste and biomass is responsible for the high chloride ion concentrations. In all sites the abundance of nitrate and sulphate suggested vehicle emissions. Remote sources also contribute to the concentrations of aerosol produced by fossil fuel combustion, soil resuspension, and biomass burning.

The characterization of particles in the atmosphere of São Paulo started in the late 70s [*Orsini and Boueres*, 1977]. The first large experiment in Brazil was performed to characterize the composition of particles at important Brazilian cities; the results are presented in *Orsini et al*. [1986]. In the 80´s, receptor models to identify the sources of the particles came into use. Principal Component analysis and Absolute Principal Component Analysis were applied with the identification of mobile emissions as the principal source of fine particles and soil resuspension for the coarse mode for São Paulo Metropolitan Area [*Andrade et al*., 1994]. *Castanho & Artaxo* [2001] showed that a significant fraction of fine particulate matter is constituted by organic carbon (OC) representing approximately 40% of the mass concentration. The BC mean concentration represented 21% of $PM_{2.5}$. In total, 60% of the fine particles mass concentration is explained by carbon compounds, mainly due to vehicular emissions, being the heavy-duty diesel the main source of fine particles [*Sanchez-Ccoyllo et al*., 2009]. Recent studies regarding the size distribution and composition of the São Paulo aerosol using Scanning Mobility Particle Size (SMPS-TSI) have been performed. Studies considering trace metals, and chemical speciation for addressing secondary aerosol formation have also been recently undertaken.

There is concern in the scientific community and government that air quality standards should be reviewed mainly for particulate matter. In Brazil, there is no air quality standard for $PM_{2.5}$, although there are many studies showing its importance. There is also the concept that it would be important to measure VOCs, mainly those originated from the emission of alcohol fuelled vehicular motors. In addition, also continuous measurements of greenhouse gases, at least CO_2, CH_4 and N_2O should be done for São Paulo. Brazil has nowadays only one station to measure these gases and has to create a greenhouse gases inventory for its enlarged urban areas.

4.8 RIO DE JANEIRO, BRAZIL

The Metropolitan Region of Rio de Janeiro (MARJ) is located at 22˚54'S and 43˚13'W at an altitude of 30 m.a.s.l. with a population of ~12 million. The MARJ is situated on the Atlantic Coast of Brazil and occupies an area of 4825 km2 [*CIDE*, 2008; *Godoy et al*., 2009]. The uncontrolled growth of the region becomes clear from the assessment of population density that exceeds 1900 inhabitants per km^2 and is the highest in Brazil [*Corrêa and Arbilla*, 2007; *Godoy et al*., 2009; *Gioda et al*., 2011].

The climate of the region is characterized by a well-defined dry season during the Southern Hemisphere winter and a rainy season in summer, associated with the occurrence of convective rainfall. Synoptic scale phenomena such as the Convergence Zone of the South Atlantic, cold fronts, the position of Anticyclone Subtropical South Atlantic, and the upper air cyclonic vortex are important to the rainfall in the region. *Corrêa et al*. [2003] also points out that rainfall rates are influenced by factors related to geographical features of the region.

Air quality monitoring in the MARJ began in the late 60's with intermittent measurements of total suspended particles (TSP). This manual sampling network is still in operation and nowadays performs measurements of TSP and inhalable particulate matter (PM_{10}) every six days. Since the

late 90's, automatic air quality monitoring stations were acquired and the manual sampling network was expanded throughout the region. Nevertheless, the existent monitoring network does not cover most of the Metropolitan area.

According to air quality annual reports [e.*g., FEEMA*, 2008], the highest pollutant concentration levels measured in the region are particulate matter and ozone. High particulate matter levels have been persistently observed in the MARJ since the beginning of air quality monitoring [*Trindade et al*., 1980; *Kretzschmar*, 1994; *Campos et al*., 1999; *Quiterio et al*., 2004; *Quiterio et al*, 2008]. Although values are still higher than the Brazilian standards, *Kretzschmar* [1994] identified a decreased trend in average and extreme levels of TSP concentrations registered in Rio. However, annual mean values still are higher than the national air quality standards established in 1990 (80µg.m-3 and 50µg.m-3 for TSP and PM_{10}, respectively).

A few epidemiological studies discuss the health effects of these high particulate matter levels in the MARJ [*Penna and Duchiade*, 1991; *Daumas et al*., 2004; *Gouveia et al*., 2003]. *Quiterio et al.* [2008] showed that Zn, Cu, Cd and Pb present in particulate matter samples from the area are higher in comparison with other urban and industrial areas. Ozone concentrations levels that exceed the Brazilian standard of 160 µg.m-3 (1 hour) have been registered since 2004. These high values are mostly measured in sites located in near the area of a petrochemical complex [*FEEMA*, 2008]. It is important to emphasize that the extension of the ozone problem in the MARJ is not yet fully known since the existent network does not cover most parts of the region. MARJ has the second largest number of emission sources in Brazil, mainly distributed between seven industrial districts and in several traffic routes where more than 2.8 million vehicles circulate [*CIDE*, 2008].

In 2004, the first inventory of atmospheric pollutant emissions from stationary and mobile sources in the region was completed [*FEEMA*, 2004]. The vehicular fleet in MARJ runs off of gasohol (64.7%), ethanol (11.7%), compressed natural gas - CNG (12.3%), and diesel (8.8%) [*Machado et al*., 2009]. Major industrial emissions in the area are due to petrochemical and energy generation industries [*FEEMA*, 2004]. Industrial sources are the largest contributor to sulphur dioxide emissions (SO_2) at 88% and to particulate matter (PM_{10}) at 58%. Vehicular emission of carbon monoxide (CO) is significantly higher as expected and represents 98% of the total. Of the total, approximately 67% of emissions of nitrogen oxides (NO_X) and hydrocarbons are also due to vehicular emissions [*FEEMA*, 2004]. Lately, sophisticated photochemical measurements have been undertaken in Rio de Janeiro (*Correa et al*, 2010).

4.9 SUMMARY AND OUTLOOK

Air quality is an issue of general concern in many South American capitals (e.g., Bogotá, Lima, Santiago, and Sao Paulo). In addition, it is a growing problem in medium-size cities (e.g., Medellín.). Attainment plans have been developed for these urban agglomerations generally reducing the most acute problems over the last decades. Curbing measures have been identified based on relatively simple approaches involving more or less accurate emission estimates and emission-receptor models, and of course adopting experiences from the developed world. Typically, such measures considered the replacement of fossil fuel by natural gas, ethanol, bio-diesel, etc., and a reduction in fuel sulphur content. However, when objectives become more ambitious and cost-effective measures less obvious, there is an increased need of local knowledge and articulated long-term planning. Unfortunately, except in Brazil, material and human resources for atmospheric research in South America on a country-by-country basis are far too small to allow a significant contribution to international programmes and to produce sustained impacts on local development. Thus, international linkage among our countries is critical. Further, it is essential that a significant fraction of our resources be dedicated to capacity building at various levels.

Coordinated activities should focus on key steps such as reconciling global inventories with, when available, regional, country and city-specific data, and putting in to place validation tools, including inverse modelling, which in turn requires better monitoring. It is very important to note

that such activities in the region require establishing a network involving not only members of academia but also technical staff from governmental agencies who are in charge of emission inventories to assure the interests of both groups are met while harmonizing the different approaches. All this is needed in order to be able to accurately evaluate the efficiency of policies and to improve our diagnostic and prognostic abilities, not only in the short-term (days to few years) but also in the long-term (years to decades).

There is a growing need of an integrated monitoring network that, in addition to solely address the attainment of health related standards, is able to address the many processes responsible for those impacts as well as the impacts on vegetation and climate. This will require, in addition to stronger coordination and planning efforts, the integrated use of modelling tools. Also, as there is an increasing need for and cost in instrument calibration and maintenance, more intensive use and sharing of local laboratory and analytical facilities is badly needed.

Often, air quality studies in South American cities remain largely isolated from one another, short-term, and lack a scientific synthesis. Moreover, scientific research has not sufficiently permeated into environmental policies. The establishment of scientific consortia and networks is one possibility, as it may be more cost-effective manner to promote the exchange and sharing of expertise, laboratory and computational resources, and human resources. Such a community, which is already emerging, would be able to deal with the particular “flavours” of our cities (e.g., sugarcane, ethanol fuel, biomass burning, smelting, power plants), in their particular human and natural geography.

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CHAPTER 5 - NORTH AMERICA

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North American megacities include Mexico City, perhaps the second largest metropolitan area in the world, and Los Angeles and New York City in the United States, two of the ten largest metropolitan areas. Figure 1 shows the population density of North America. The megacities of Los Angeles, New York, and Mexico City are visible, as well as several other high-density urban areas such as Houston, Texas. Summertime photochemical smog was first recognized as a severe environmental problem in Los Angeles, and has been the subject of extensive studies there since the 1940's [e.g., *Haagen-Smit*, 1952]. Differences in topography, meteorological conditions, and anthropogenic emissions lead to marked differences in the air quality considerations in these North American megacities as well as their impacts on the larger troposphere. Figure 2 shows emissions of CO, NO_X, and SO_2 for the year 2000 throughout North America. Probably the greatest difference in emissions between the megacities in North America is their SO_2 emissions, which depend greatly on the type of coal (low- versus high-sulphur) used in energy production. This chapter discusses these differences, as well as the similarities. The greater Houston, Texas urban area in the United States is discussed as an additional contrast; with a population approaching 6 million, this area is not generally considered a megacity, but it is a large urban centre of particular interest since it is home to a large fraction of the petrochemical industrial facilities of the US, which leads to a unique mix of anthropogenic emissions.

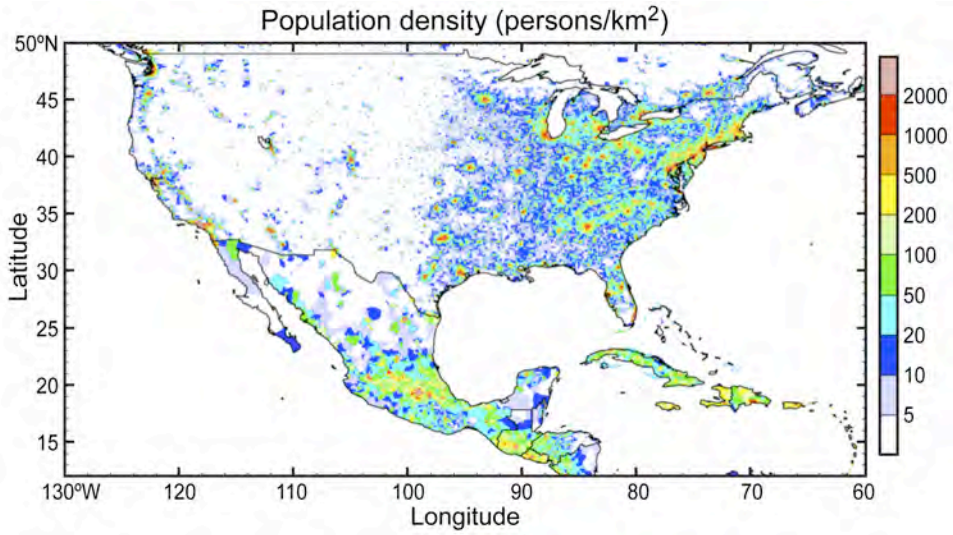

Figure 1 - Map of the population density in North America (persons per km²), based on 0.25° gridded data for 2000 from the Center for International Earth Science Information Network (CIESIN) at Columbia University *[http://sedac.ciesin.columbia.edu/gpw/]*

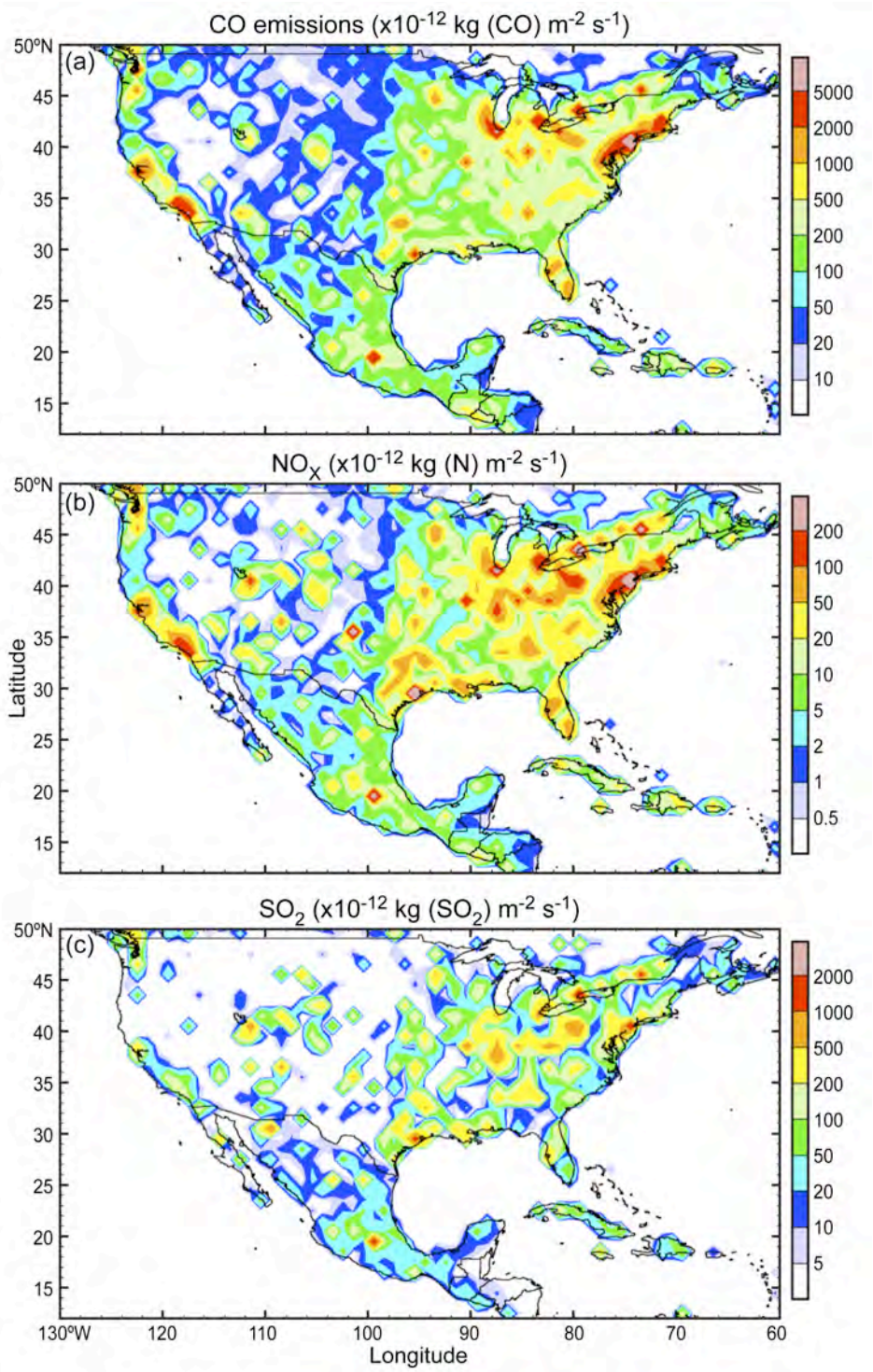

Figure 2 - (a) Emissions of CO; (b) NO_X; (c) SO_2 for North America in kg/m^{-2}s^{-1}. The figures were generated from the EDGAR emission inventory
[EC-JRC/PBL. EDGAR version 4.1. http://edgar.jrc.ec.europa.eu/, 2010]

5.1 LOS ANGELES: AN ENVIRONMENTAL SUCCESS STORY

The Los Angeles Megacity (LAMC), here defined as the South Coast Air Basin (SoCAB) shown in Figure 3, is located at 34˚3'N and 118˚14'W with a population approaching 17 million inhabitants. LAMC is an excellent example for the evolution of emission control strategies in a growing megacity. By several measures LAMC is one of the greatest environmental success

stories found anywhere in the world. Between 1970 and the present, SoCAB VOC and NO_X emissions have declined markedly despite a substantial increase in commerce and vehicle traffic [*Cox et al.,* 2009]. Peak O_3 levels that exceeded 600 ppbv in the 1960's have not reached 200 ppbv since 1998.

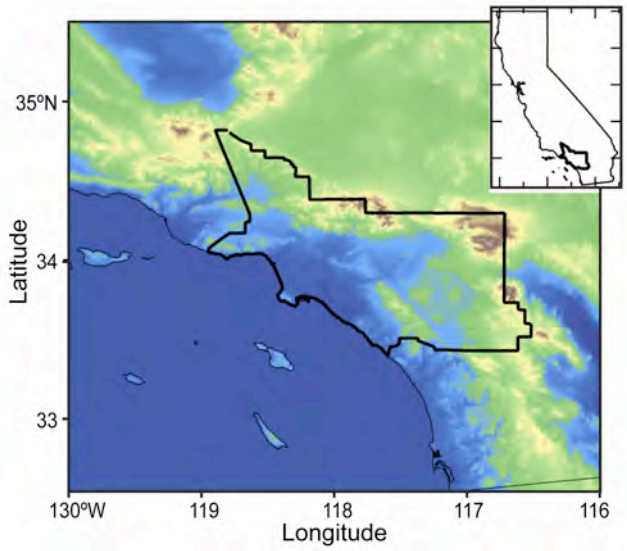

Figure 3 - The Los Angeles Megacity defined as the South Coast Air Basin (outline). The colours indicate the topography of the basin (blues for low elevation, brown for high elevation). The Pacific Ocean lies to the south and west

This improvement in air quality has been accomplished despite several unfavourable circumstances that make SoCAB particularly susceptible to high air pollution concentrations. The large population lives in a basin bounded by the Pacific Ocean on the west and by mountains on the other three sides, which prevent efficient horizontal ventilation of the area. Low inversion heights associated with a persistent high pressure system and the adjacent Pacific marine environment limits the vertical mixing within the basin, and the land-sea breeze system re-circulates polluted air within the basin. These topographic and meteorological features allow emissions to accumulate over several days during episodes of relatively stagnant airflow. During the summer ozone season, May through October, clear skies and high temperatures dominate, which speed photochemical production of O_3 and other photochemical products. Private automobiles on extensive freeway systems provide the primary transportation in the area. This transportation system accounts for a large fraction of the emissions in the urban area.

Over the past three decades ambient concentrations of key pollutants in the SoCAB region have decreased substantially despite a doubling of the population and tripling of vehicle use. Figure 4 compares the temporal trends of four pollutant concentrations. These data are presented in units that correspond to US EPA National Ambient Air Quality Standards (NAAQS) (http://www.epa.gov/air/criteria.html), which are presently: O_3 - 75 ppbv (8-hr); CO - 9 ppmv (8-hr); NO_2 - 53 ppbv (1-yr); $PM_{2.5}$ - 35 μg/m^3 (24-hr). The numbers in parentheses give the averaging period. The O_3 data are 3-yr averages of the 4th highest annual maxima, the CO data are annual maxima, and the $PM_{2.5}$ are annual 98th percentiles. Ozone concentrations apparently peaked in the late 1960s. It is evident that there has been an impressive decline in ozone concentrations as well as other air pollutants since then. Although it still violates the NAAQS for O_3 and $PM_{2.5}$, the Los Angeles basin is in compliance with the NAAQS for nitrogen dioxide and carbon monoxide, as well as sulphur dioxide and lead. It is fair to say that this megacity has gone from being one of the most polluted in the world 50 years ago to presently one of the "least polluted" cities of its size. Estimates are that many thousands of lives have been saved in LAMC from improvements in air quality [*Hall et al.,* 2008].

The relative temporal trends of the primary pollutants, NO_2 and CO, reflect the history of the air quality control strategy adopted in the United States. Initially, the control focus was upon VOCs and CO, notably including the introduction of catalytic convertors on automobiles in the mid-1970s. The focus later shifted to include NO_X emission controls. Figure 4 shows that this control emphasis has led to a significantly larger decrease in ambient CO concentrations (factor of 5.2) than that for the ambient NO_2 concentrations (factor of 2.3) from 1980 to 2008. SO_2 emissions have also decreased substantially over the last three decades, primarily due to reduced sulphur content of fuels utilized in mobile (as of 1 June 2006 15 ppm for on-road diesel) and point sources, and to scrubbing of sulphur from flue gases emitted by point sources.

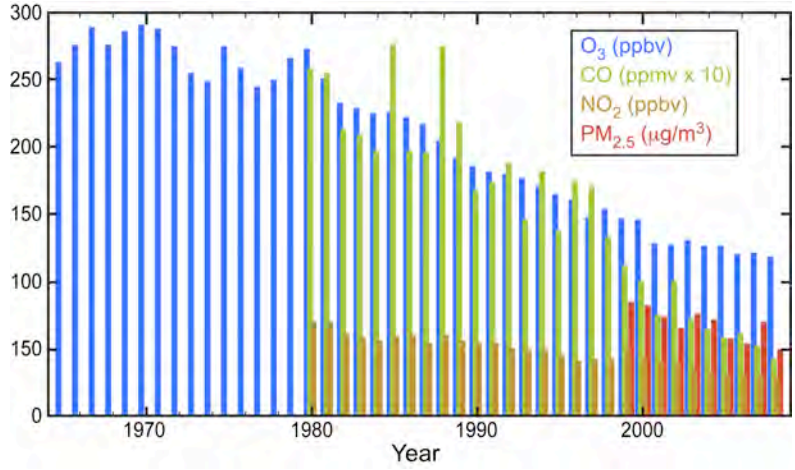

Figure 4 - Air quality trends over the South Coast Air Basin of California. National Air Quality Data derived from monitoring stations in the SoCAB region
[*Alexis et al.,* 1999; *Cox et al.,* 2009; http://www.arb.ca.gov/adam/cgi-bin/db2www/polltrendsb.d2w/Branch].
The O_3 trends have some caveats: since the earliest years there have been changes in the measurement methods, the quality control procedures initially were limited and the monitoring network was smaller, and many site locations have changed

It is important to recognize that there are still significant problems remaining. The region still violates the ozone standard and indeed despite continued emission reductions little improvement in O_3 air quality has been observed since 2000. During 2005-2008 the 8-h O_3 exceeded the NAAQS on 110-120 days each year. The current peak O_3 levels are roughly double the accepted levels set to protect the most vulnerable populations. Several difficulties exist in achieving air quality goals in LAMC. Due to extremely non-linear nature of VOC-NO_X-O_3 chemical system, it is possible that VOC/NO_X ratios over time have shifted to a regime where further VOC reductions are only minimally effective. Controlling emissions from heavy-duty diesel trucks has been far more difficult than passenger cars as the turnover time for this fleet (25-30 years) greatly exceeds that for the passenger fleet (7-10 years). A relatively small fraction of the total motor vehicle fleet, currently 10 million vehicles in the Los Angeles basin, accounts for a very large fraction of the total mobile source emissions. A related problem is the emergence of the Ports of Los Angeles and Long Beach as dominant point sources of diesel-related pollution in the Los Angeles Basin due to a tripling of goods movement from Asia through these ports over the past 15 years. Future progress is anticipated from a greater use of plug-in hybrids, electric cars, alternate fuels and better control technology. Current targets call for on-road emission reductions of VOC, NO_X, SO_X, and $PM_{2.5}$ by respectively 70%, 70%, 50% and 12% between 2007 and 2020

Megacities contribute significantly to the burden of GHGs in the atmosphere. In the past, air quality control strategies have been based largely on health implications with little consideration for the associated climate change consequences. An added complication for future control strategy development is the need to mitigate climate change impacts while improving air quality [*Bell et al*., 2007]. These interactions are often nonlinear and require a better understanding of the feedback processes between air pollution and climate change [*Ramanathan and Feng*, 2009; *Jacob and Winner*, 2009 and references there in]. California is presently embarking on strategies

that improve air quality as well as mitigate the impacts of local and global climate change. Recent studies [*Diffenbaugh et al.*, 2008] indicate that LA basin has extremely high sensitivity to future climate change (Figure 5). A warmer climate, with increasingly hotter days, is also likely to be deleterious to air quality [*Steiner et al.*, 2006; *Murazaki and Hess*, 2006]. In California, the first task in controlling CO_2 and other GHG emissions (CH_4, N_2O, CFCs) has begun with inventorying their baseline emissions and atmospheric abundances so future progress can be measured against a reference point expected to be 2010.

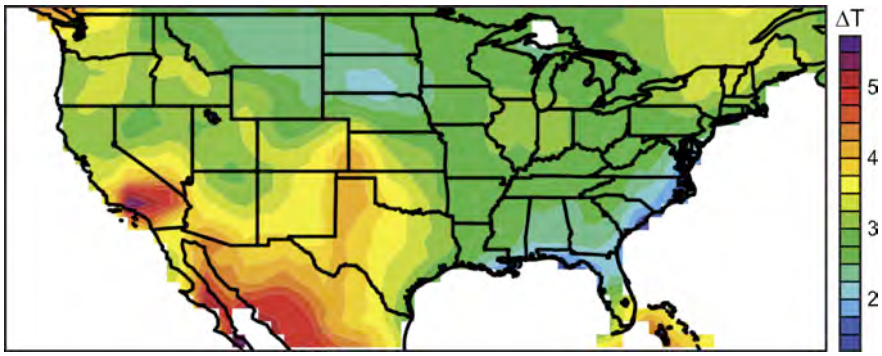

Figure 5 - Future climate change hot spots over North America. The colour scale gives aggregate climate change scores, which take into account long-term mean and variability of warm- and cold-season temperature and precipitation [*Diffenbaugh et al.*, 2008]

Over the last several decades, there have been great advances in measurement technology and it is now possible to measure a large number of precursors and intermediate species (carbonyls, free radicals, aerosol composition etc.) to test and validate models that simulate air quality and climate change interactions. Complementing the models have been several intensive field campaigns that use aircraft platforms to acquire data in three dimensions and provide boundary conditions necessary for model development and forecasts of air quality and climate change. Some of the recent intensive campaigns that have provided unique observational data from SoCAB have been the 1997 South Coast Ozone Study [*Croes and Fujita*, 2003], the 2002 Intercontinental Transport and Chemical Transformation study [*Parrish et al.*, 2004], and the 2008 ARCTAS-CARB study (http://www.espo.nasa.gov/arctas/; unpublished). These campaigns used aircraft platforms to assess emission sources, atmospheric composition, and boundary conditions applicable to this region. Figure 6 shows that emission signatures of different sources can be identified in such measurements. In the upper panel the anthropogenic sources in southern California have relatively small CO to CO_2 emission ratios compared to biomass burning (BB) plumes encountered in Canadian boreal fires as well as wildfires in southern California. In the lower panel, the CH_4 to CO emission ratios are seen to be quite high over agricultural regions of California, lower in urban emissions and much lower in BB plumes. In this lower panel, elevated benzene concentrations (colour scale) are found in the urban emissions and BB plumes, while acetonitrile (CH_3CN) is strongly elevated only in the BB plumes (inset).

More recently, satellites have been able to provide a new dimension to the repertoire of surface based atmospheric observations [*Martin*, 2008]. Although less precise than in-situ techniques, satellite observations extend over several years and provide data on scales much larger than aircraft campaigns. Also, satellites generally provide columns and relating these to surface observations still remains a challenge. Figure 7 shows tropospheric NO_2 columns over urban areas in the western US. The highest concentrations are seen over the Los Angeles basin. Generally all urban areas show strong weekday-weekend differences in emissions. Such satellite measurements provide tests of NO_2 emission inventories [*Boersma et al.*, 2008; *Kim et al.*, 2009]. Similar data are available for aerosols and attempts are being made to relate these to $PM_{2.5}$ measured by ground networks [*Engel-Cox et al.*, 2006]. In the future column measurements of GHGs (e.g. methane) are likely to provide new information on their sources and sinks.

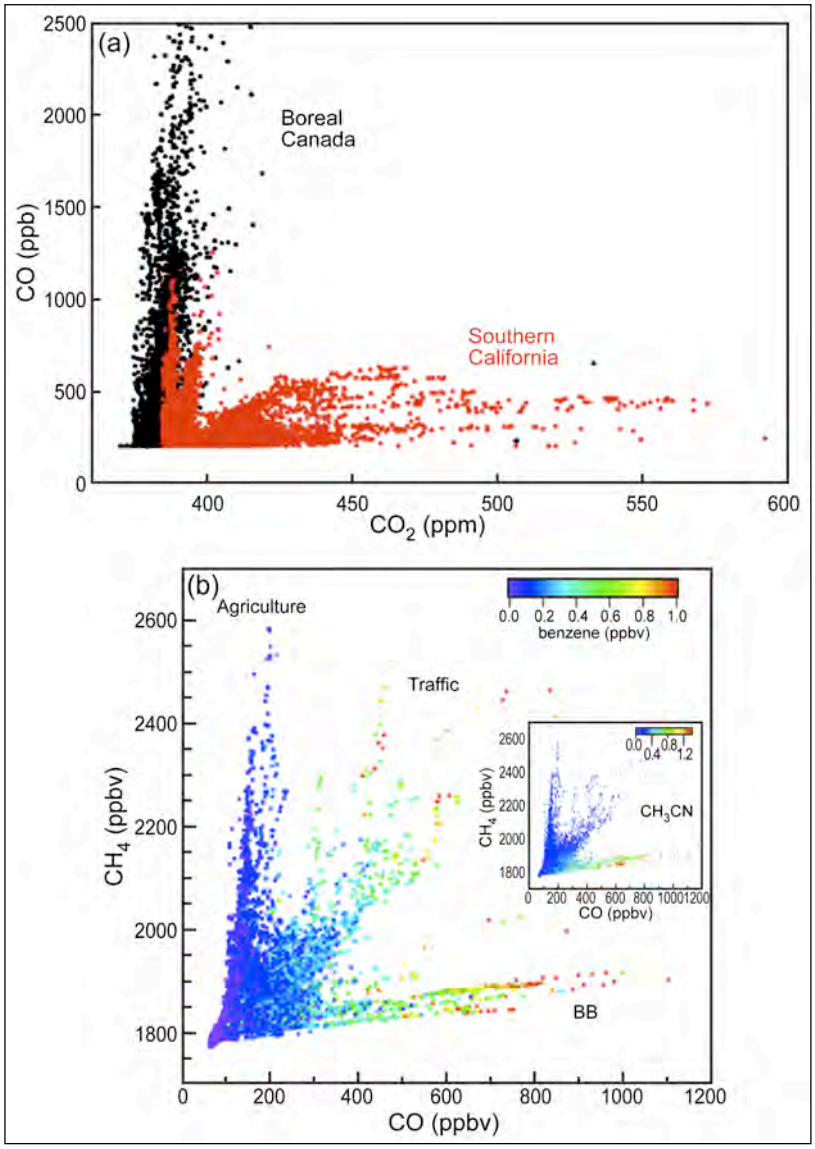

Figure 6 - Relationships between selected trace constituents over Southern California measured from June 17-26, 2008. The upper panel compares the California data with later measurements in forest fire plumes in Canada. The lower panel compares measurements over different California regions: agricultural areas, urban (traffic) and in forest fire plumes (BB). Data were collected aboard the NASA DC-8 by multiple investigators of the ARCTAS campaign [*Jacob et al.*, 2009]

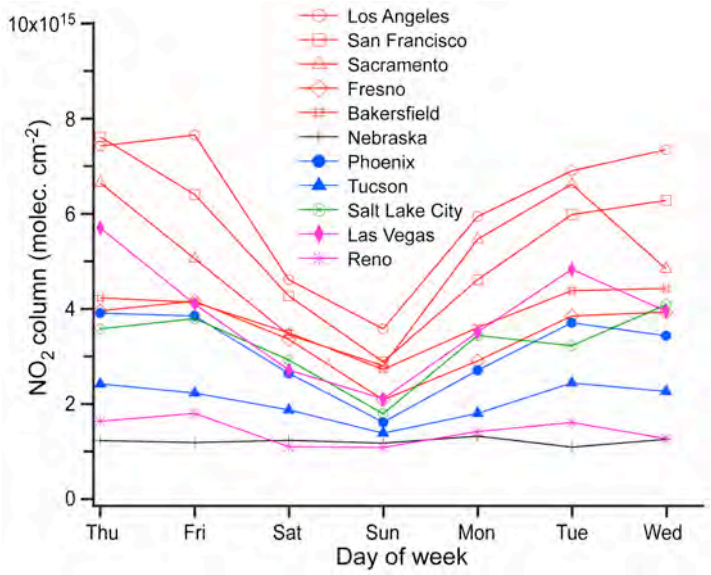

Figure 7 - SCIAMACHY satellite derived tropospheric NO_2 columns over urban areas in the western US for 2003 – 2007 May - September averages [after *Kim et al.*, 2009]

5.2 THE US NORTHEAST URBAN CORRIDOR AND HOUSTON, TEXAS: CONTRASTING US URBAN AREAS

Located at 40˚42'N and 74˚W near sea level, by its widest definition, the New York City region includes 22 million people in parts of three states: New York, New Jersey and Connecticut. It is embedded in the US Northeast urban corridor (Figure 8), which extends from Boston to Washington D.C. and has a population of about 55 million people. In many respects New York City has a more favourable situation from an air quality perspective than does Los Angeles. The Northeast urban corridor in general and New York City in particular has lower per capita emissions of ozone and aerosol precursors and greenhouse gases than western US cities like Los Angeles and Houston. The greater population density that allows much greater use of public transportation accounts for this difference. With no significant topographic barriers to air flow there is generally much better ventilation of the northeast urban areas allowing winds to sweep pollutants away from the point of emission. With the prevailing winds from the continent, the daytime convective boundary layer is generally deeper allowing also greater vertical mixing of emissions. On the eastern coast of the US the summertime period is not so dominated by clear skies. Consequently the New York City area has not experienced the very high O_3 concentrations observed in Los Angeles.

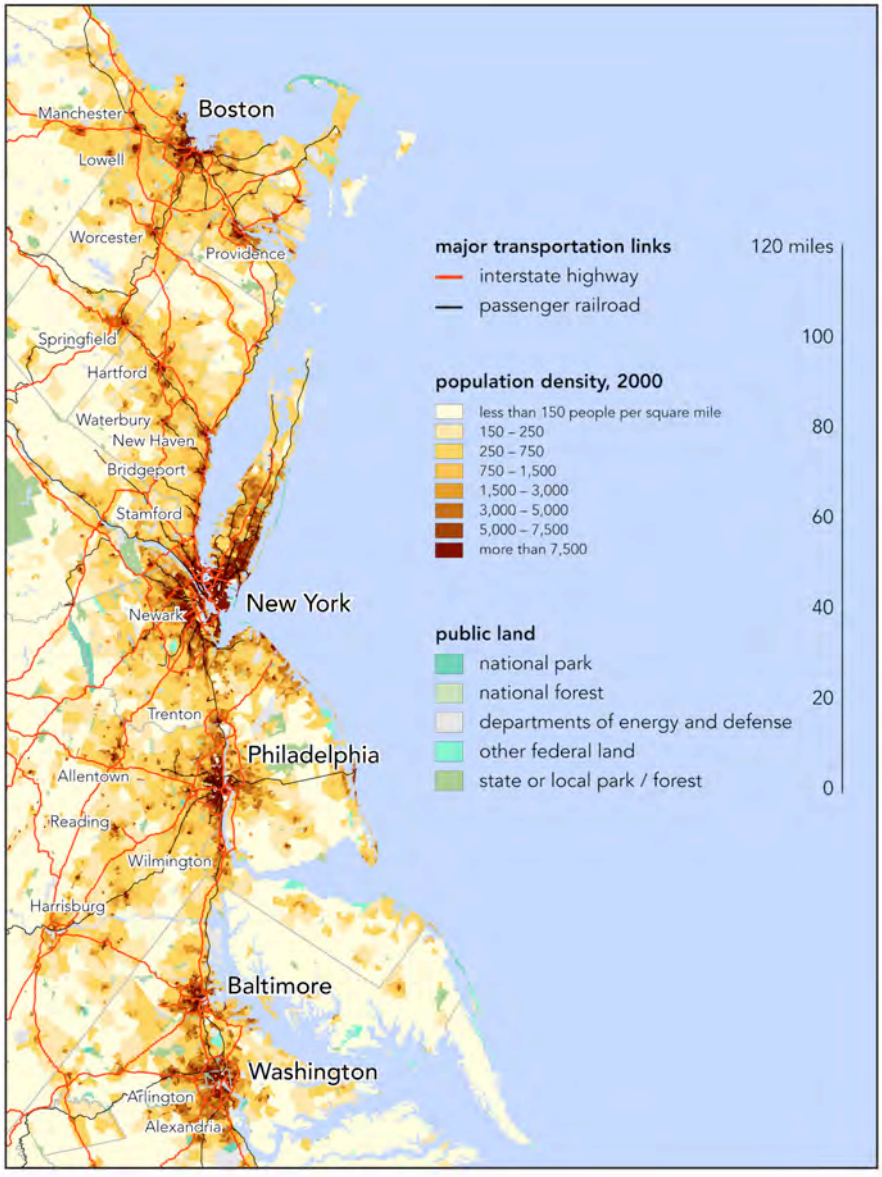

Figure 8 - The US Northeast urban corridor including New York City coloured according to greater population density [*Rankin*, 2009] *(http://en.wikipedia.org/wiki/File:Boswash.png)*

The Houston, Texas urban area located at 29˚45'N and 95˚21'W along the Gulf of Mexico coast has a much smaller population approaching ~6 million. However, Houston has much greater industrial emissions than either the Los Angeles or New York City area. Figure 9 shows the location of these facilities relative to the urban area. Unfortunately, a large fraction of the facilities are located within the urban area along the Houston Ship Channel, which extends from Galveston Bay nearly to the centre of the city. These emissions have a profound effect on photochemical O_3 production [*Ryerson et al.,* 2003], which causes the ambient O_3 concentrations in Houston to be comparable to those found in the two larger US megacities. In fact, in most summers from 1999-2004 the nation's highest 1-hr average O_3 concentrations were found in Houston. In addition to the exceptional industrial emissions, Houston suffers some of the same disadvantages as Los Angeles. It is a coastal city, so is subject to the shallow boundary layers and recirculation associated with the land-sea breeze circulation. It also experiences very hot, sunny and stagnant summertime meteorological conditions.

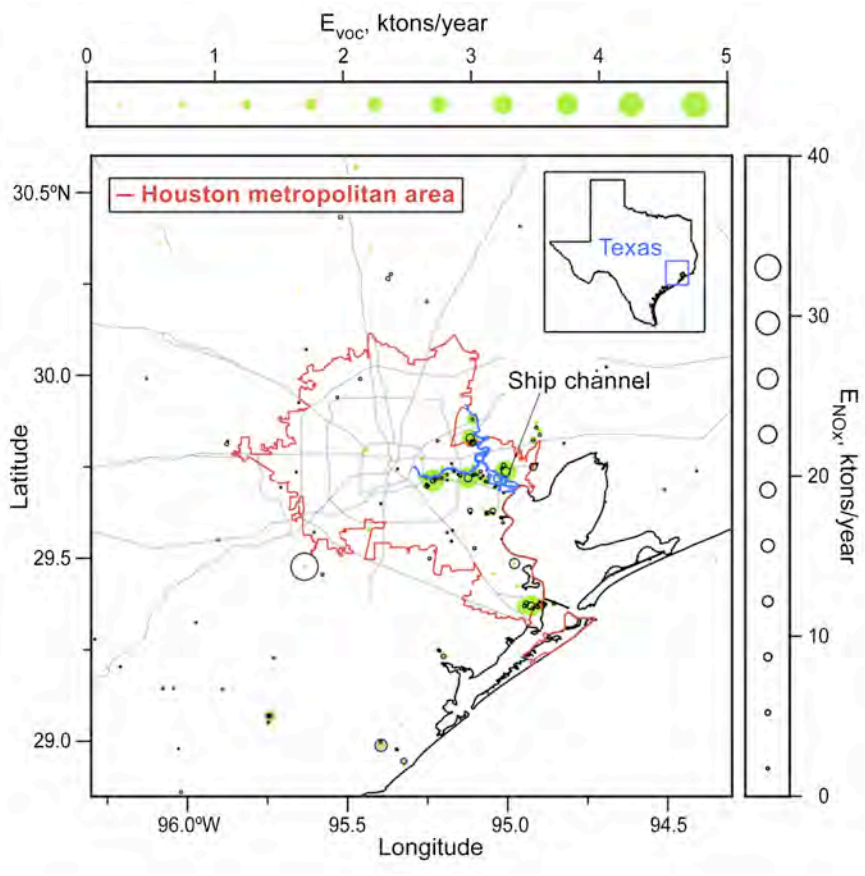

Figure 9 - Location of point emission sources of VOC and NO_X within the greater Houston, Texas region. The Gulf of Mexico is in the lower right corner, and Galveston Bay lies east of the metropolitan area [after *Ryerson et al.,* 2003]

Recent temporal trends of ambient concentrations of O_3 and $PM_{2.5}$ in New York City and Houston approximately parallel those in Los Angeles (Figure 10). In general, Los Angeles experiences the highest concentrations of both O_3 and $PM_{2.5}$, while New York and Houston exhibit similar concentrations, with New York City slightly lower in O_3, but higher in $PM_{2.5}$. Only slow improvement is evident for either pollutant in any of the three urban areas for the time period plotted.

Sulphur dioxide (SO_2) emissions differ markedly between the western and eastern US, and this difference is clearly reflected in differences in aerosol composition (Figure 11) and acidity of precipitation (Figure 12). Sulphur emissions are much higher in the eastern US due to both more concentrated emission sources from the greater density of electrical power generation facilities and the greater utilization of high sulphur coal. Figure 11 shows that sulphate makes smaller fractional aerosol contribution in Los Angeles compared to the northeast US. The lower sulphate

contribution allows accommodation of a larger contribution from nitrate. In both regions, organics also account for a major aerosol fraction. The high density of primarily SO_2 and secondarily NO_X emissions in the eastern US has led to greatly increased acidity of precipitation in that region. However, due to the implementation of the US EPA Acid Rain programme in 1990, the average ambient sulphate and nitrate concentrations in the Northeastern US have decreased by 49% and 44%, respectively, from 1989-1991 to 2007-2009 [http://www.epa.gov/airmarkt/progress/ARP09_3.html]. This decrease has led to a decrease in precipitation acidity (i.e. higher pH) of 43% in the Eastern United States that is clearly illustrated in Figure 12.

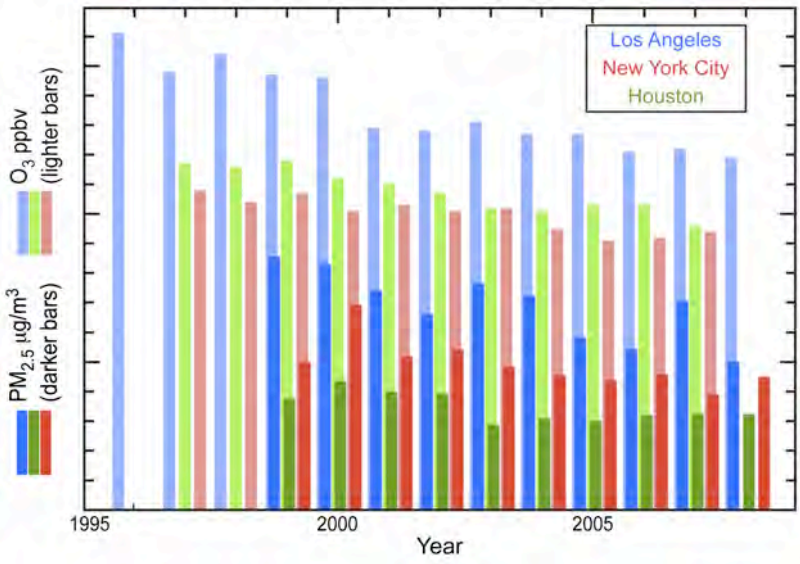

Figure 10 - Comparison of air quality trends for three mega-cities for O_3 (lighter bars, in ppbv) and $PM_{2.5}$ (darker bars, in µg/m³). The data are averaged and derived, as in Figure 4 *[http://www.epa.gov/air/airtrends/values.html]*

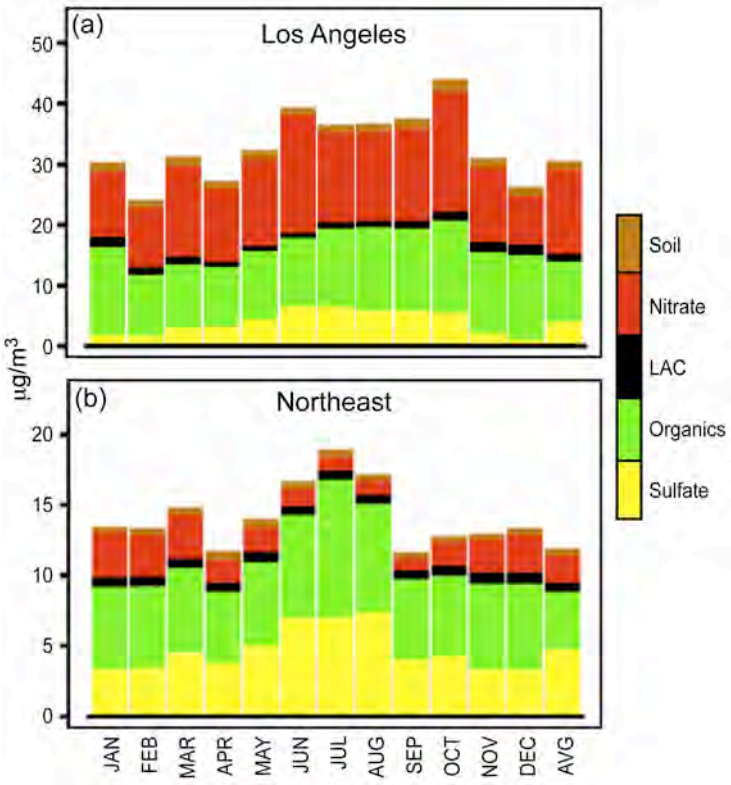

Figure 11 - Comparison of fine aerosol composition (units: µg/m³) in (a) Los Angeles (b) Northeast [after *DeBell et al.*, 2003] as a function of month of the year. The two minor component bars are soil (brown) and light absorbing carbon (black). Data are from US EPA's Speciated Trend Network

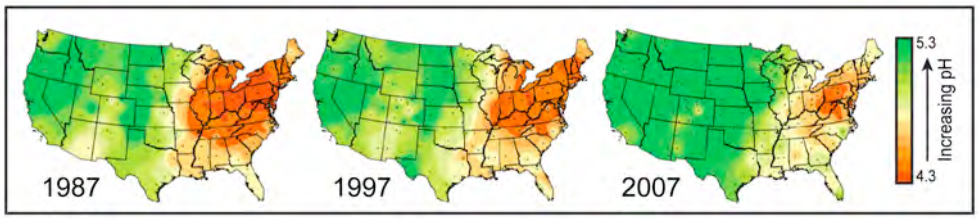

Figure 12 - The spatial distribution of annual precipitation-weighted average concentration of hydrogen ion (pH units) for three *years [after http://nadp.sws.uiuc.edu/lib/data/97as.pdf].*
Data are from US National Atmospheric Deposition Programme

5.3 MEXICO CITY: NORTH AMERICA'S MOST POPULOUS AND RAPIDLY DEVELOPING MEGACITY

The Mexico City Metropolitan Area (MCMA) 19°25'N latitude and 99°10'W longitude lies in an elevated basin at an altitude of 2240 m. The nearly flat basin floor covers about 5000 km^2 of the Mexican Plateau, and is confined on three sides (east, south and west) by mountain ridges but with a broad opening to the north and a narrower gap to the south-southeast (Figure 13). Two major volcanoes, Popocatépetl and Ixtaccíhuatl are on the mountain ridge southeast of the basin. The metropolitan area covers about 1500 km^2 on the southwest side of the basin. During the twentieth century Mexico City increased rapidly in urbanized area and population, from fewer than 3 million people in 1950 to about 20 million presently. MCMA has a very high population density as well as a high concentration of industrial and commercial activities [*Molina and Molina, 2002]*.

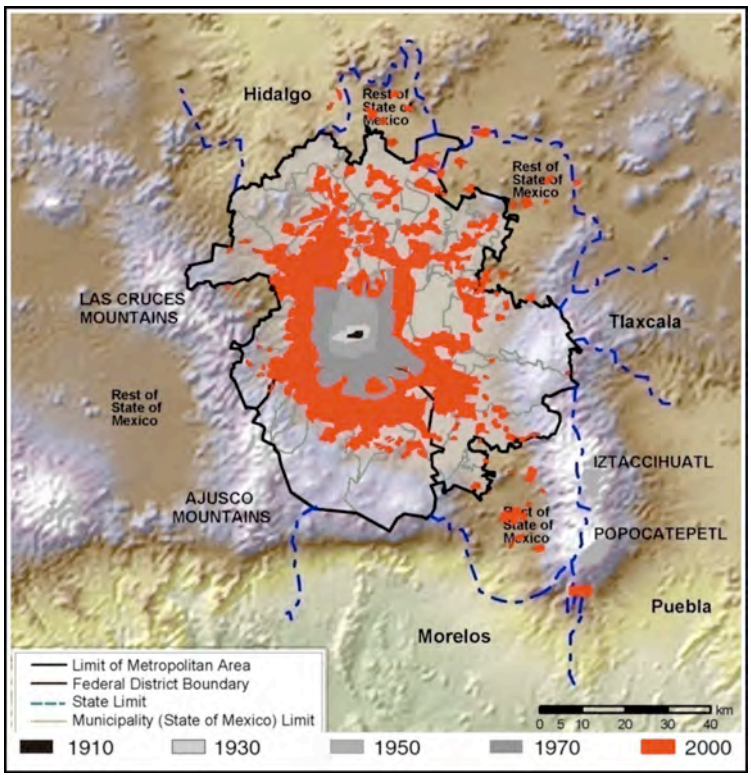

Figure 13 - Topographical map of the Mexico City Metropolitan Area showing urban expansion [*Molina and Molina*, 2002]

The topography and meteorology of the MCMA markedly affect air quality. A cool dry season from November to February is followed by a warm dry season until May and a rainy

season from June to October. Synoptic forcing is generally weak and air transport is strongly influenced by the mountain-valley winds in the basin. Weak winds and strong temperature inversions at night lead to high primary pollutant concentrations that persist into the morning, followed by very rapid boundary layer growth to maximum heights of 2 to 4 km in the early afternoon. There is relatively little recirculation and day-to-day carry-over of pollutants within the basin [*de Foy et al.*, 2006; 2008]. The cool season has stronger surface inversions and higher morning concentrations of primary pollutants. The warm dry season has more intense solar radiation with faster photochemical oxidant formation, as well as increased aerosol loading due to dust and biomass burning. The rainy season has lower PM and CO but continues to have high ozone due to intense photochemistry occurring before the afternoon showers. Air pollution is therefore a year round concern.

Despite the rapid growth and development in the MCMA, air quality has markedly improved during the past two decades (Figure 14) and the rate of increase of greenhouse gas emissions has been reduced. Comprehensive air quality management programmes have been implemented [*Molina and Molina*, 2002]. In the late 1980s, the newly established automatic air quality monitoring network (RAMA, *Red Automática de Monitoreo Atmosférico*) revealed high concentrations of all criteria pollutants: lead, carbon monoxide, nitrogen oxides, sulphur dioxide, ozone, and particulate matter (PM). Ozone exceeded the air quality standards of 110ppbv (1-hr) more than 90% of the days, and peaked above 300 ppbv 40-50 days a year, among the worst in the world [*Molina and Molina*, 2002]. Specific actions included removal of lead from gasoline, installation of catalytic converters in automobiles, reduction of sulphur content in diesel fuel; substitution of natural gas for fuel oil in industry and power plants, reformulation of liquefied petroleum gas for heating and cooking, strengthening the vehicle inspection and maintenance programme, and modernizing the driving restriction programme (*Hoy no circula*). The concentrations of criteria pollutants have decreased substantially over the past decade despite the continuing increase in population and economic activity. Figure 14 indicates that in the early 1990s the ozone concentrations in MCMA may have been larger than those in Los Angeles. Ozone has decreased significantly in both cities, but even more rapidly in MCMA, and in recent years the MCMA concentrations have been lower than in Los Angeles. The MCMA represents around 20% of Mexico's population, but only 9% of its greenhouse gas emissions. Policies focused on greenhouse gas emissions include biogas capture and waste management projects, improved public transportation, fleet renewal projects for taxis and medium-capacity buses, and sustainable housing development projects [*Molina et al.*, 2009].

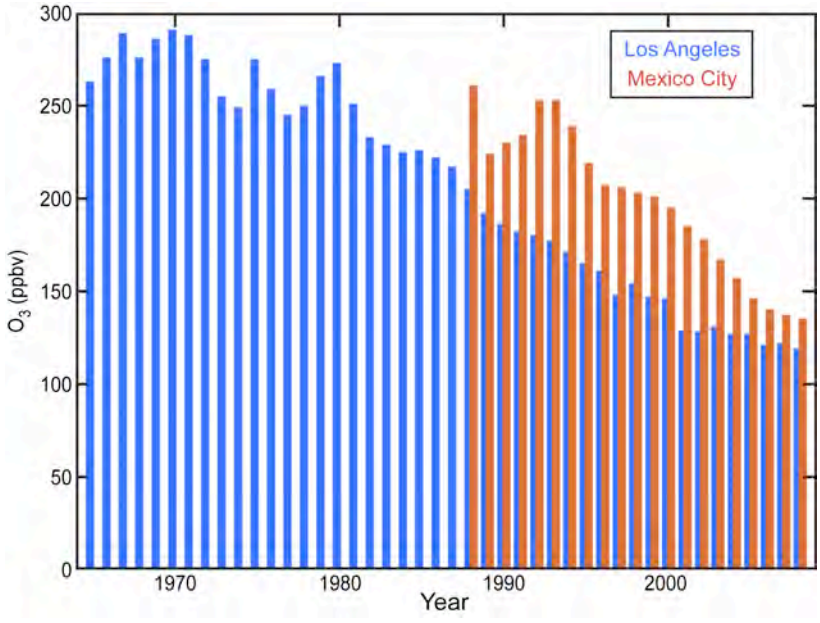

Figure 14 - Comparison of air quality trends for O_3 in Mexico City and Los Angeles. The data are 3-yr averages of the 4th highest annual maxima. The data are derived from Fig. 4 for Los Angeles and from Sistema de Monitoreo Atmosférico de la Ciudad de México (SIMAT) *[http://www.sma.df.gob.mx/simat/]* for MCMA

Emission inventories have been developed in the MCMA since 1986. There were large uncertainties in the early inventories, especially for VOC emissions [*Molina and Molina*, 2002]. Figure 15 presents the 2006 MCMA emissions inventory for PM_{10}, $PM_{2.5}$, VOC and NO_X [*SMA-GDF*, 2008]. Mobile emission sources represent a significant fraction of the total anthropogenic emissions of NO_X and $PM_{2.5}$ (76% and 62%, respectively) but only a relatively small fraction (34%) of VOC. However, ambient VOC measurements suggest that emissions associated with gasoline emissions dominate in MCMA as well as some other mega-cities [*Parrish et al.*, 2009a]. As in all current emission inventories throughout the world, a great deal more effort is required for emission inventory testing and improvement.

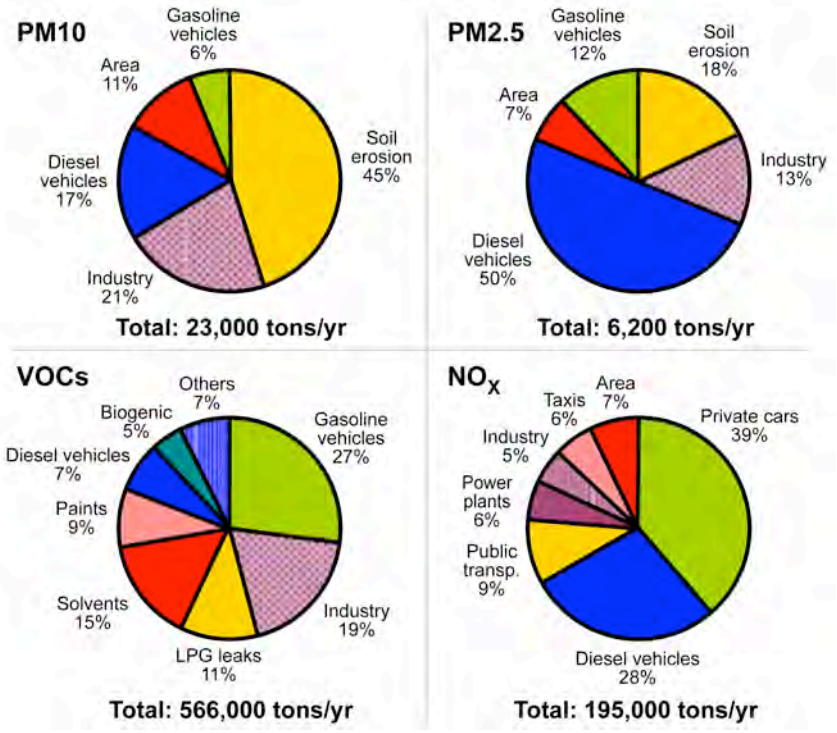

Figure 15 - Emissions inventory for the Year 2006 [*SMA-GDF,* 2008]

The combination of population, topography, meteorology, and multi-pollutant emission density of the MCMA has attracted a number of field studies. The Mexico City Air Quality Research Initiative (MARI) project gathered surface and vertical profile observations of meteorology and pollutants during 1990-1994 [*Streit and Guzman*, 1996]. The IMADA-AVER (*Investigación sobre Materia Particulada y Deterioro Atmosférico*, Aerosol and Visibility Evaluation Research) campaign in February - March 1997 yielded comprehensive meteorological measurements in the basin, and provided insights into particulate composition [*Doran et al.*, 1998; *Edgerton et al.*, 1999]. The MCMA-2002/2003 campaigns in February 2002 and April 2003 provided detailed measurements of many oxidant precursors and photochemical intermediates including radicals, as well as meteorology and emissions [*Molina et al.*, 2007]. The largest study to date, MILAGRO (Megacity Initiative: Local and Global Research Observations), took place in March 2006 and included a wide range of instruments at ground sites, on aircraft, and satellites that provided detailed measurements of gas and aerosol chemistry, aerosol microphysics and optics, radiation and meteorology [*Molina et al.*, 2010; *Singh et al.*, 2009; and *references therein*].

The intensive field campaigns have provided a wealth of information on the emission, dispersion and transformation of species emitted to the MCMA atmosphere and their urban, regional and hemispheric impacts. Motor vehicles produce abundant amounts of VOC and NO_X (ozone and aerosol precursors), primary PM, elemental carbon, particle-bound polycyclic aromatic hydrocarbons, CO and a wide range of air toxics, including formaldehyde, acetaldehyde, benzene,

toluene, and xylenes [*Molina et al*., 2007]. VOC/CO emission ratios are notably higher than in the US (Figure 16) and aldehydes emissions are significant [*Garcia et al*., 2006; *Lei et al*., 2009]. VOC flux measurements have been made with fast-response sensors coupled with eddy covariance techniques, from both tower and aircraft platforms, to provide independent evaluation of emissions inventories [*Velasco et al*. 2005; 2007; 2009; *Karl et al*., 2009]. A mobile laboratory measured on-road vehicle fleet emission indices for various vehicle classes and driving speeds [*Zavala et al*., 2006; 2009]. Figure 17 compares the total annual emissions from light-duty gasoline vehicles estimated from fuel-based emission factors from on-road and remote sensing measurements combined with 2006 fuel sales records against the corresponding mobile emissions estimates in the official emissions inventory [*SMA-GDF*, 2008]. The comparison indicates agreement to better than 30% for CO and NO, but an inventory under estimate by a factor of 1.4 to 1.7 for VOCs and a probable severe under estimate by a factor > 4 for primary $PM_{2.5}$ emissions [*Zavala et al*., 2009].

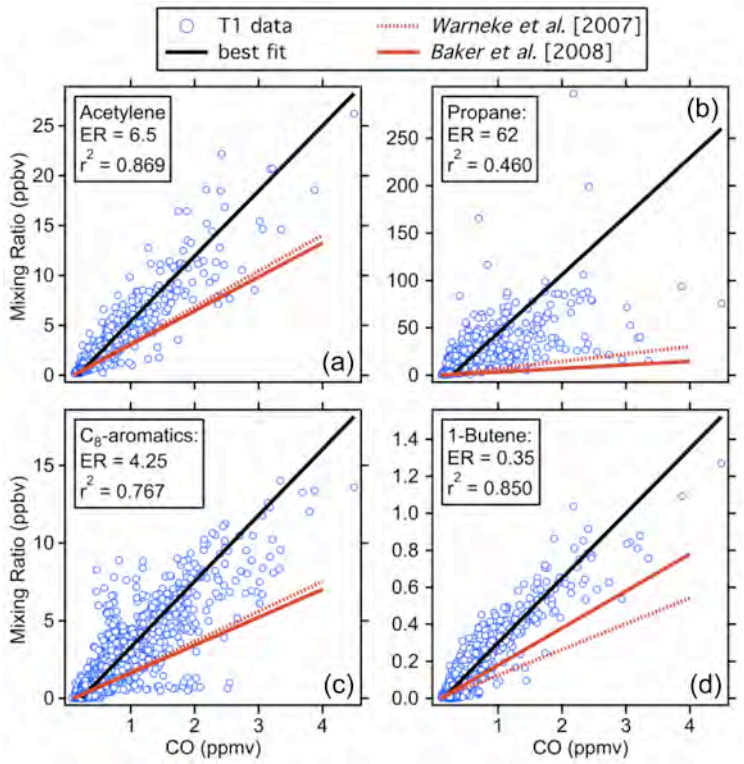

Figure 16 - Correlations between hydrocarbons (a) Acetylene; (b) Propane; (c) C_8-aromatics; (d) 1-Butene, and carbon monoxide measured in Mexico City (blue points) compared to fits obtained in US cities (red lines) [*deGouw et al*., 2009]. Here ER represents emission ratio, which is equal to the best-fit slopes

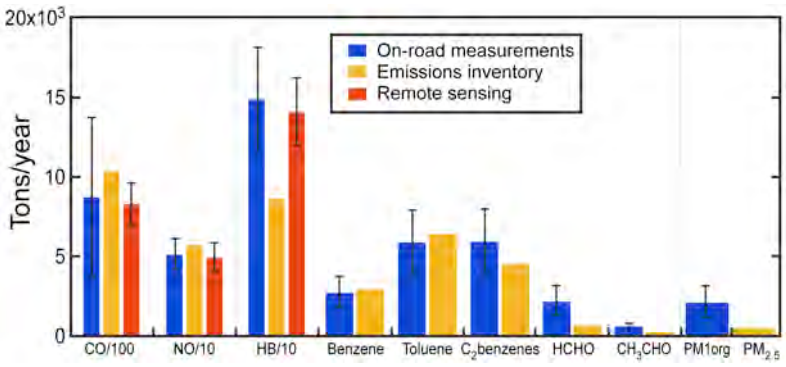

Figure 17 - Evaluation of the Mobile Emissions inventory for light-duty gasoline vehicles [*Zavala et al*., 2009]

The response of urban O_3 to VOC and NO_X precursor emissions remains a topic of interest, and may have shifted toward more VOC-sensitive conditions in recent years. Recent modelling studies [*Lei et al.*, 2006; 2008; *Tie et al*., 2007] suggest that O_3 production is VOC-limited. This is supported by studies of radical budgets showing significant chain termination by NO_X chemistry [*Volkamer et al*., 2010; *Sheehy et al*., 2010 *Dusanter et al*., 2009], and by the weekend effect showing large reductions in NO_X and CO but not O_3 on Saturdays and Sundays relative to weekdays [*Stephens et al*., 2008]. However, definitive policy recommendations must be based upon more extensive modelling studies with emission inventories tested and improved as discussed above.

High aerosol concentrations were observed both at ground sites and from all aircraft during MILAGRO. These aerosol particles included a large fraction of organics, but black carbon, crustal matter, sulphate and nitrate were also significant contributors. Figure 18 shows the average submicron PM composition within the MCMA basin during two different studies. Secondary organic aerosols (SOA) dominate the organic fraction in the city and their origin is still under study [*Volkamer et al.* 2006; *Dzepina et al*., 2009]. Biomass burning (agricultural, forest, wood cooking and trash burning) also contributes to the urban and regional pollution in the Mexico Basin [*Salcedo et al*., 2006; *Johnson et al*., 2006; *Yokelson et al*., 2007; 2009; *Moffet et al*., 2008; *Stone et al*., 2008; *Querol et al*., 2008; *Crounse et al*., 2009; *Aiken et al*., 2009; *Christian et al*., 2009].

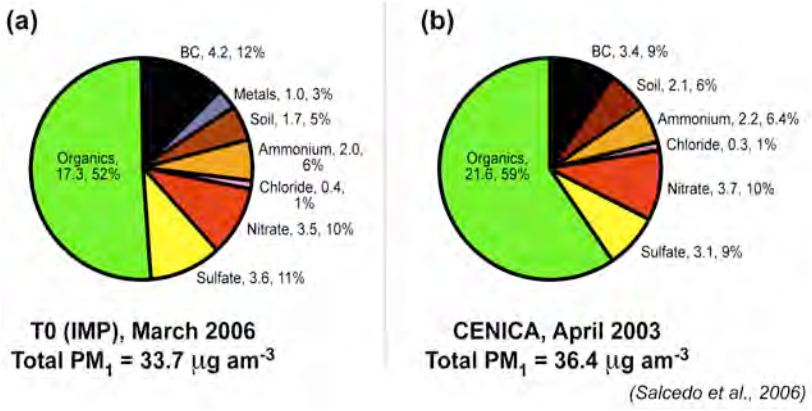

Figure 18 - Submicron PM composition (mass and percent) measured during (a) March 2006; (b) April 2003 field campaigns at surface sites in the Mexico City basin.
[From *Aiken et al*., 2009]

5.4 POLLUTION TRANSPORT IN NORTH AMERICA

By the 1980s it had become apparent that local air quality was not simply due to local emissions, but had significant contributions due to pollutant transport from upwind sources. The Ozone Transport Assessment Group (OTAG) studied this transport contribution in the eastern US; Figure 19 summarizes the important transport regimes found during high ozone events. Within the boundary layer, but above about 800 m, flow is controlled by synoptic meteorological systems, and is generally from the west to northwest during pollution episodes. Channelled flows below the ridge heights of 200 to 800 m follow important terrain features, and often transport air from the southwest along the northeast US urban corridor. These channelled flows include low-level nocturnal jets that are apparently particularly important for transporting pollutants preceding episodes. Finally, near surface flows (below 200m) during episodes are typically light during night and morning allowing accumulation of emissions. Fresh emissions as well as aged urban plumes move downwind and react during daytime, while O_3 aloft and aged precursors are entrained as the mixing layer deepens. This low-level transport is typically to the north through east along the urban corridor for 50-250 km by evening. In summary, meteorological processes interact at large

and small scales to determine local O_3 concentrations. The relative contribution from each scale, from local to inter-regional, can vary widely between episodes. This understanding has led to coordinated multi-state abatement strategies within the US.

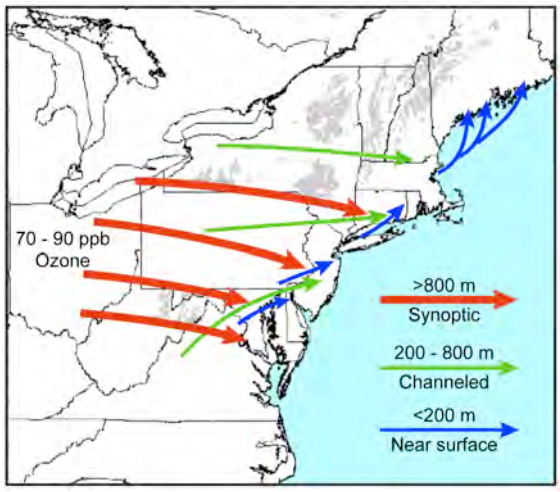

Figure 19 - Conceptual diagram illustrating the multiple, concurrent scales of transport winds typically observed during high ozone events in the Northeast US [after *Blumenthal et al.,* 1997]

The proximate location of urban areas in the northeast urban corridor has made the importance of transport processes particularly obvious in that region, but even in the relatively sparsely populated western US, regional transport makes significant contributions to the urban ozone concentrations. For example, recent studies in Texas demonstrate that the transport of ozone from upwind regions can predominate over in situ production within the Houston urban area, even during O_3 exceedance conditions [see *Parrish et al.*, 2009b, and references cited therein]. Indeed, the transported ozone in eastern Texas, which represents the minimum ozone concentration that is likely achievable through only local controls, can approach or exceed the current NAAQS.

Pollutant transport also is important on intercontinental scales. A large fraction of North American emissions, as well as the resulting ozone and aerosol produced over the continent, is transported beyond the national borders, primarily to the Gulf of Mexico and North Atlantic regions and potentially on to Europe. Significant recent research efforts have been directed toward quantifying the importance of this pollutant export from the Northeastern US [see *Fehsenfeld et al.*, 2006 and references cited therein.] The pollution plume from Mexico City also can be observed several hundreds of kilometres downwind. Figure 20 compares the increase in O_3 relative to CO in pollution plumes exported from these two regions. Air masses close to urban areas exhibit shallow slopes that steepen during downwind transport. Interestingly, quite similar slopes are seen in the aged plumes transported from Mexico City (0.35) and New York City (0.38). Aircraft-based measurements show ongoing production of SOA [*Kleinman et al*., 2008; *DeCarlo et al*., 2008] as well as ozone for several days downwind, with active photochemistry sustained by aldehydes [*Tie et al*., 2009] and nitrogen oxides from the thermal decomposition of peroxyacyl nitrates and photolysis and OH oxidation of nitric acid [*Neuman et al.*, 2006].

Intercontinental transport into North America gives an added complexity to Los Angeles air quality issues. There is strong evidence that the background O_3 concentration transported into California is increasing [*Parrish et al.*, 2009c], possibly in response to increasing Asian emissions of O_3 precursors. This background O_3 increase may be occurring throughout the northern mid-latitude troposphere [e.g. *Parrish et al.*, 2009c], and this increase may be negating many of the benefits from local pollution control measures in California [*Jacob et al.,* 1999; *Lin et al.,* 2008]. The impact of transport of background ozone on surface air quality is a matter of considerable

debate [*Lefohn et al.,* 2008; *Oltmans et al.*, 2008]. Presently no strategies exist to mitigate the impact of increasing background O_3 upon local or regional air quality. It is likely that all northern mid-latitude emissions contribute to this background.

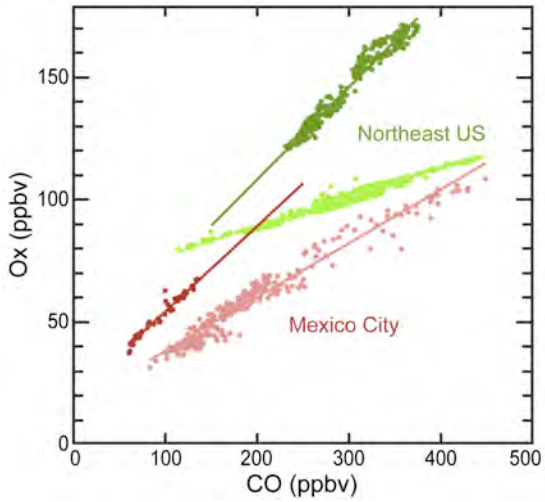

Figure 20 - Odd oxygen ($O_X = O_3 + NO_2$) and CO measured in the vicinity of Mexico City on 18 March 2006 (light red), in the same airmass a day later and about 500 km downwind (dark red), near Boston on 21 July 2004 (light green), and on the same day about 500 km downwind from New York City (dark green). The steeper slopes defined by the latter, dark coloured data result from regional O_X production during transport.
[Mexico City data derived from NCAR C-130 and DOE G1 flights reported by *Zaveri et al*., 2007; northeast US data derived from NOAA WP-3D flights reported by *Neuman et al*., 2006]

5.5 CONCLUSIONS

The experiences in North American megacities demonstrate that urban and industrial development can proceed simultaneously with air quality improvement. Results from past and future field studies will continue to contribute to a fuller understanding of air pollution and its impacts on human health, ecosystem viability, and climate change. The integration of air quality information from old and new studies of megacities in different settings will improve significantly the scientific basis that decision makers in megacities around the world will need to craft effective environmental policies.

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CHAPTER 6 - EUROPE

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6.1 EUROPEAN MEGACITIES: GENERAL AND COMPARATIVE CHARACTERISTICS

6.1.1 Population and Geography

This chapter gives an overview of the megacities and major population centres (MPCs) of Europe, namely London, Paris, Moscow, the Benelux/Rhine-Ruhr region, the Po Valley, and the Eastern Mediterranean including Istanbul.

A summary of the populations and location of the European MPCs is given in Table 1. In contrast to MPCs in several other parts of the world, e.g., Asia and Africa, the populations of the European MPCs, with the exception of Istanbul, have been relatively stable over the past several decades, and are predicted to remain so for at least the next couple decades.

Table 1 - Populations and Characteristics of European Megacities and Major Population Centres

Megacity / MPC	Population[1] (Million)	Latitude	Longitude
Paris, France	10.4	49.4	1.9
London, England	8.6	51.3	0.0
Po Valley, Italy	ca.20	~45-46	~11-12
Ruhrgebiet, Germany and Benelux Region (Belgium, Netherlands, Luxemburg)	28 (BeNeLux) + 5.7[3] (Ruhrgebiet)	~49-54	~2-8
Moscow, Russia	10.5	55.0	37.5
Istanbul, Turkey	10.4	40.1	28.1

[1] *Population Source (unless otherwise noted): Population Division of the Department of Economic and Social Affairs of the United Nations Secretariat World Urbanization Prospects: The 2009 Revision, Web: http://esa.un.org/unpd/wup/index.htm*

Figure 1 shows a map of the population density of Europe. For Europe, the bottom-up statistical and census-based population data plotted here are generally very consistent with top-down estimates based on analyses of stable night lights; alternate representations of the population are available on the internet, e.g., a mapping by district complied by the IIASA European Rural Development (ERD) Project (http://www.iiasa.ac.at/Research/ERD/DB/mapdb/map_9.htm). The map shows that each of the six European MPCs listed in Table 1 is characterized by a large central region with a population density exceeding 1000 persons/km^2. There are also a handful of other cities within Europe with population densities as high as this, e.g., Berlin and Madrid, but the total populations of these cities are not large enough to generally be classified as Megacities or MPCs for the sake of this overview. Beyond the commonality of a dense core region, however, there are substantial differences in the geographical locations and urban and suburban structure of the European MPCs. One interesting feature is that while a large fraction of the MPCs worldwide are either coastal or close to large bodies of water, only one of the European MPCs, Istanbul, is really a coastal city. London is also relatively close to the coast, while Paris, the Po Valley and the Benelux/Rhine-Ruhr region are all several hundred km away from the nearest coast, and Moscow is the most land-locked of all megacities worldwide. Further differences in the demographical and geographical characteristics are elucidated in the discussions of the individual cities, below.

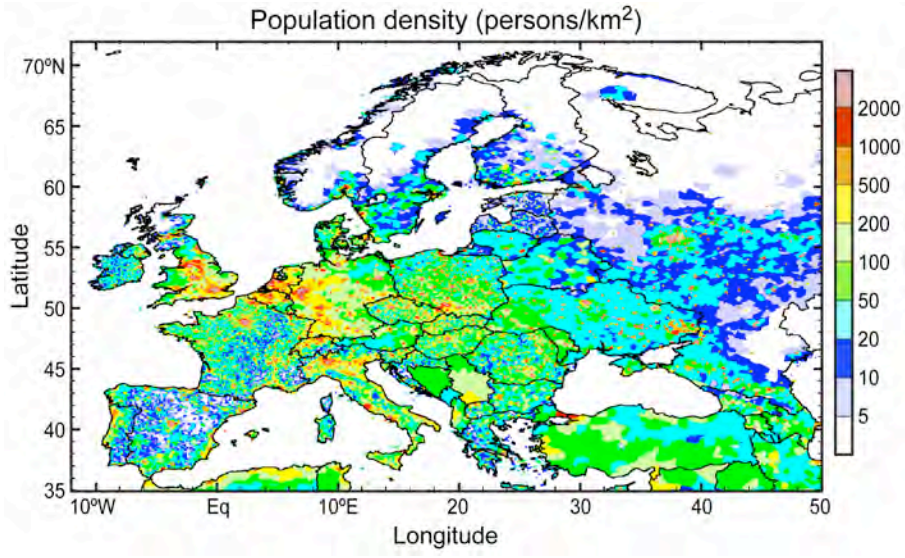

Figure 1 - Map of the population density in Europe and western Asia (persons per km^2), based on 0.25° gridded data for 2000 from the Centre for International Earth Science Information Network (CIESIN) at Columbia University *[http://sedac.ciesin.columbia.edu/gpw/]*

6.1.2 Emissions

Exemplary emissions of key pollutant gases in Europe are depicted in Figure 2. Research on emissions from megacities worldwide has included several studies of city-specific emissions. For Europe in particular, a significant recent effort has gone into developing a collection of inventories for European cities within the City Delta project [*Cuvelier et al.*, 2007]. Four of the European MPCs were also included in the study of *Butler et al.* [2008)], which contrasted the emissions corresponding to 32 MPCs in three widely-used global emissions datasets (EDGAR, RETRO and IIASA) with each other and with city-specific emissions, where available. The comparison shows frequently large differences (often a factor of two or more) between the emissions for individual cities within the global datasets, and normally large underestimates compared to the city-specific datasets. This applies especially to Paris, for which the CO emissions in the global datasets range over nearly a factor of two, from 263 Gg(CO)/yr (EDGAR) to 490 Gg(CO)/yr (RETRO), and are a factor of 4-7 less than the 1907 Gg(CO)/yr estimated in the City Delta project. The discrepancy determined for London is smaller, but still exceeding a factor of 2, with a range of 800-1043 Gg(CO)/yr from the global inventories versus 1993 Gg(CO)/yr from the City Delta inventory. Interestingly, a much better agreement is found for Moscow, with a range of 979-1249 Gg(CO)/yr

from the global inventories versus 1324 Gg(CO)/yr from the study of *Gurjar et al*. [2008]. A much smaller but still highly uncertain CO source is estimated for Istanbul (244-602 Gg(CO)/yr from the global inventories, with no city-specific inventory available). The discrepancies between the datasets for NO_X and NMHCs tend to be even larger. This large uncertainty – even for European Megacities, which should be among the best characterized around the world – points towards the large difficulty which will be inherent in assessing the role of megacities in regional and global atmospheric pollution and climate change, and ascertaining effective mitigation strategies. On the other hand, despite the large differences in the totals, there is a notable similarity in the characteristics of the relative importance of various sectors for different gases. In particular, similar to most OECD nations, the emissions of CO are generally dominated by road transport, and of NO_X and NMHC by a combination of road transport and industrial processes, contrasted with the large role of residential biofuel use for emissions from non-OECD countries.

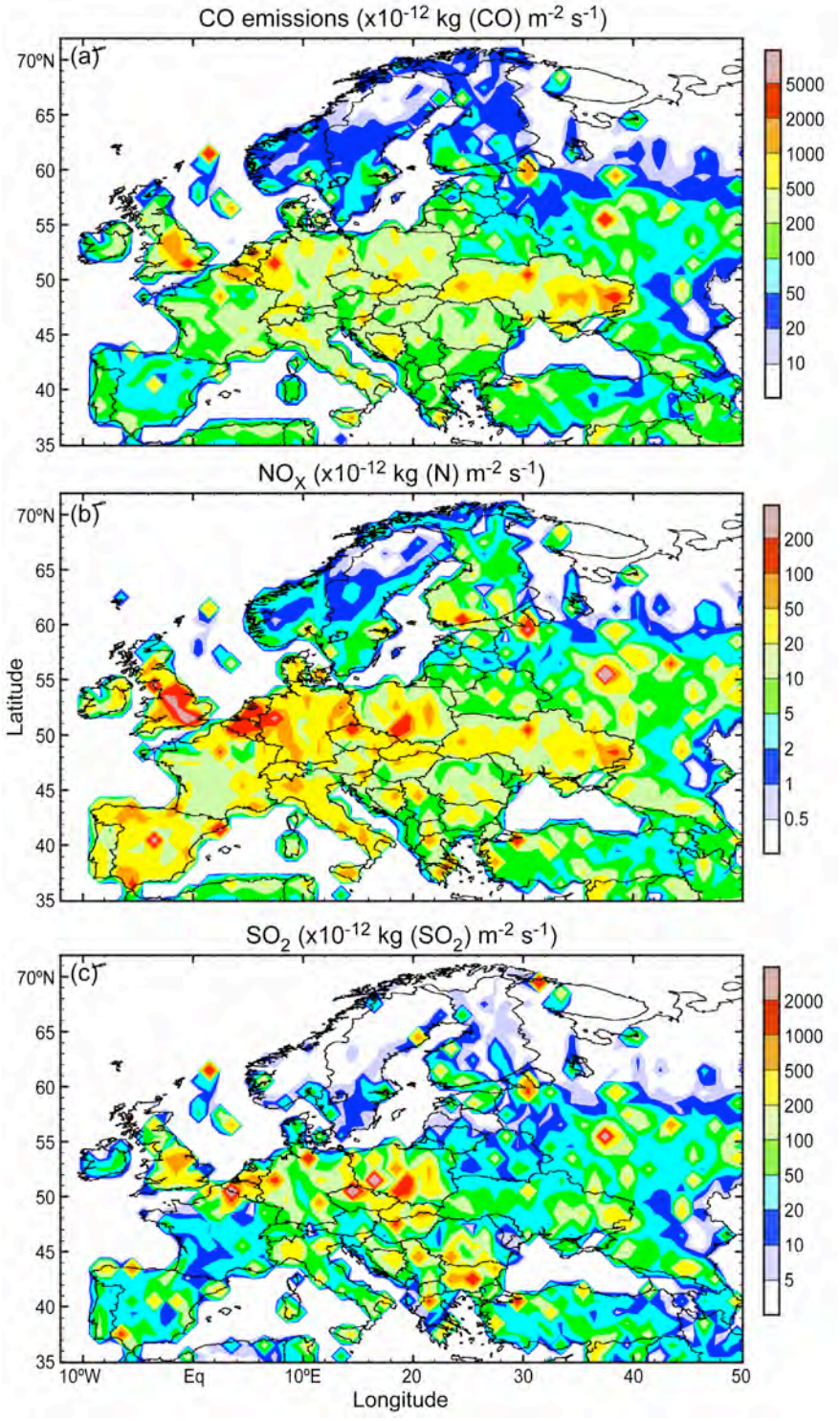

Figure 2 - (a) CO emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (b) NO_X emissions for the year 2000 based on the EDGARv3.2 FT2000 database. (c) SO_2 emissions for the year 2000 based on the EDGARv3.2 FT2000 database

6.1.3 Pollution levels

The air pollution in the European MPCs which results from the emissions discussed in the previous section, as part of the broader topic of general urban air quality, has a very long history of slowly increasing recognition and research. In particular, London has a very important historical role in this respect, with the "London Smog" events dating back at least to the 17th century, culminating in the weeklong tragedy in December, 1952, which caused about 12,000 deaths. Many other large cities have been renowned for their poor air quality, as well as for the substantial clean-up efforts over more recent years, e.g., the reduction of acid rain throughout Europe in the latter part of the 20th century.

6.1.4 Outflow characteristics and effects on regional ozone-related atmospheric chemistry

To our knowledge, only one published study thus far [*Butler and Lawrence*, 2009] has examined the impact of megacities worldwide on regional and global ozone-related atmospheric chemistry, and none have done so yet for aerosol and climate impacts. A few further studies have examined this for specific regions [e.g., for Asia in *Guttikunda et al*., 2005], but none to our knowledge specifically for the European MPCs. Some studies on the impacts of individual cities will be discussed in the following sections.

Butler and Lawrence [2009] used a zeroth-order approach, the so-called "annihilation scenario", to examine the megacity effects by removing their emissions from the corresponding gridcells in a global emissions inventory (at 1^{o} horizontal resolution) before interpolating to the global model grid, and comparing these results with a simulation including the normal total emissions. The results show that the overall impact of megacities on the major ozone precursor gases NO_X and CO are of the order of 5-10%, corresponding to the relative contribution of megacities to the global total emissions of the precursor gases, while the impact on global O_3 is much smaller, around 1%. The change in July mean surface ozone is shown for four scenarios in Figure 3. The exact dependence of the regional chemistry on the emissions varies as a function of geographical location, and corresponds particularly strongly with the latitudes of the MPCs. In the individual grid cells containing megacities, the response for European megacities is similar to most other extratropical megacities, with a reduction in ozone year-round, and often an increase in ozone in the downwind grid cells, particularly in summertime; in tropical megacity grid cells, on the other hand, ozone generally increases year-round. The influence is found to change for various future scenarios. Under a future scenario with a maximum feasible reduction of emissions, the influence of megacities is generally reduced, while under a high-emission future scenario, although the local influence of megacities is increased, the geographical extent of the influence becomes smaller. One note worth making about these results is that the tendency for global emissions datasets to underestimate the emissions compared to city-specific datasets, as discussed above, also means that the impacts of megacities are probably somewhat underestimated in these simulations.

Finally, a special characteristic of the European MPCs has been pointed out and quantified by *Lawrence et al*. [2007], who examined results of simulations with generic, gas-phase tracers with three different representative lifetimes (1, 10 and 100 days) emitted from 36 MPCs distributed globally. Using metrics to rank different outflow characteristics ("regional pollution potentials") of the MPCs, it was found that the MPCs in this region tend to be ranked highest amongst all global regions in terms of the tendency for pollutants to both accumulate locally in the surface layer of the region immediately surrounding each MPC, as well as to be transported extensive distances (e.g. larger than 1000 km) downwind, while still remaining in the boundary layer. Conversely, the emissions from European MPCs are least effectively transported into the upper troposphere compared to other world regions. Two major open issues for follow-up studies are currently being investigated: the same kind of simulations with aerosol tracers (including sedimentation, scavenging and deposition) are being performed for comparison to the gas-phase tracer results [*D. Kunkel et al*., 2011], and comparable regional model simulations are being set up for the European region to determine the impact of using a considerably higher resolution in a non-hydrostatic model on the pollutant dispersion characteristics [*I. Coll et al*., *LISA*, pers. comm.].

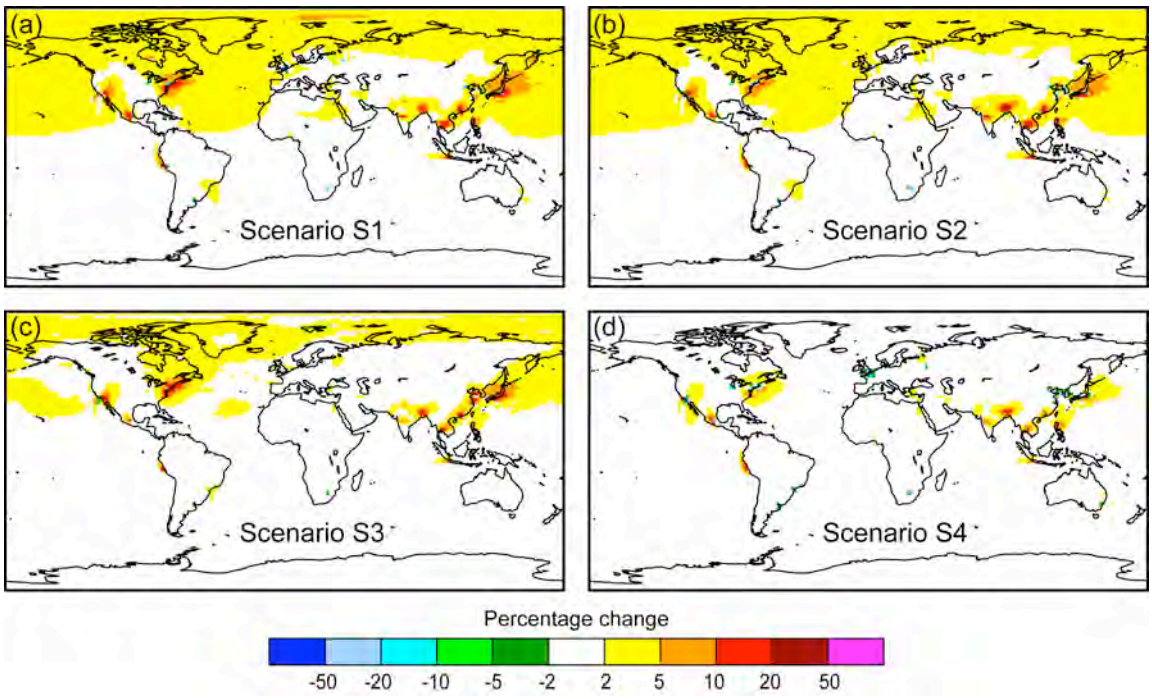

Figure 3 - The percentage change in the global surface July O_3 mixing ratio due to megacity emissions under four scenarios [*Butler and Lawrence*, 2009]

6.2 LONDON

6.2.1 City introduction

London is located in the southeast of England, UK. Greater London, which includes the City of London as well as 32 London boroughs in the surrounding area has a population of approximately 7.5 million (as of mid-2005). The entire London metropolitan area is just under 14 million people. Greater London covers an area of 1,570 sq km and sits on either side of the Thames River, approximately 40 km west of the coast of the North Sea. The minimum and maximum temperatures range from 2.4 to 13.7°C and 7.2 to 22.3°C in January and July, respectively.

6.2.2 Emissions sources, trends and data

The large majority of nitrogen oxides (NO_X), carbon monoxide (CO), particulate matter with an aerodynamic diameter less than or equal to 10 μm (PM_{10}) and certain volatile organic compound (VOC) emissions in London are from mobile sources [*Mattai and Hutchinson*, 2008]. In an estimate from 1997, road emissions of NO_X, CO, and PM_{10} were found to contribute 76%, 97%, and 77% to total emissions for all of London, respectively [*Crabbe et al.*, 2000]. Correspondingly, a significant number of studies monitoring roadside pollution throughout London have been conducted. The trends in NO_2 and NO_X from 1997 to 2003 from 36 sites in London have been documented by [*Carslaw*, 2005] as shown in Figure 4. While NO_X concentrations decreased, the NO_2 concentrations have remained basically the same from 1997 to 2003. These may have been due to changes in the vehicle fleet composition, changes in traffic management in London and/or control technologies applied to diesel vehicles [*Carslaw*, 2005]. Changes in the concentration of PM in London have also been linked to traffic. Annual mean PM_{10} from 1994 to 2004 at different types of locations throughout London were documented by [*Fuller and Green*, 2006], and are shown in Figure 5. [*Fuller and Green*, 2006] found that secondary and natural sources of PM_{10} declined from 1997 to 2003, whereas primary sources increased from 1998 to 2003; the largest increases in primary PM were observed at roadside sites. It should also be noted that long-term transport of air masses, primarily from mainland Europe, can contribute significantly to PM concentrations in the London metropolitan area, especially episodes of elevated concentrations [*Charron et al.*, 2007].

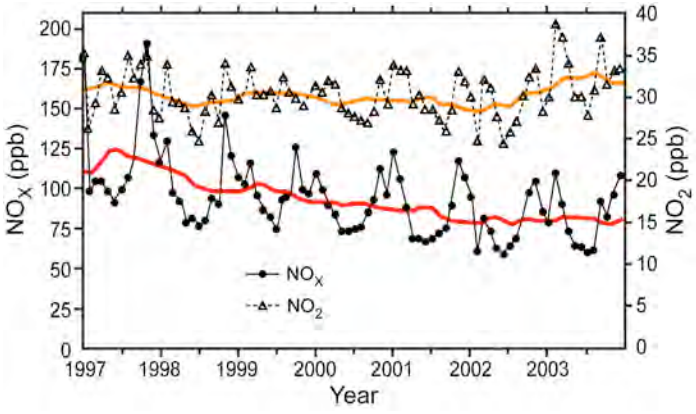

Figure 4 - [*Carslaw*, 2005] Trends in monthly mean concentrations of NO_X and NO_2 averaged across 36 air pollution monitoring sites in London (1997-2003). A 12-month moving average trend has been fitted to both data sets

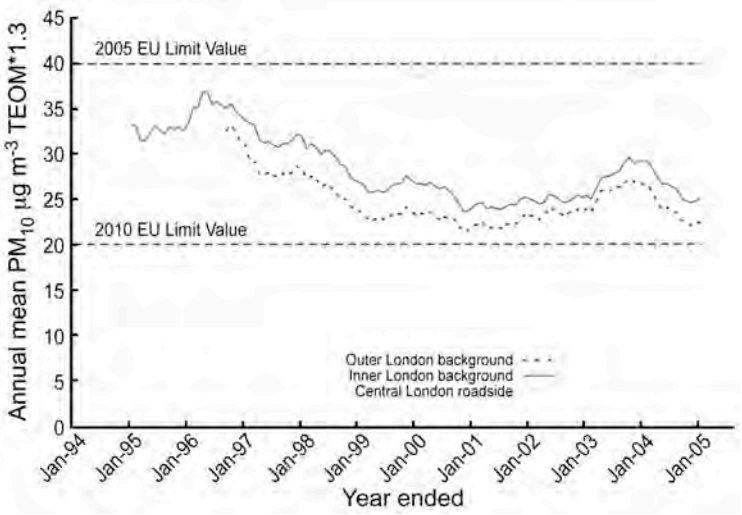

Figure 5 - Running annual mean PM_{10} in London from 1994 to 2004 showing outer London background concentrations (mean of 3 sites), inner London background concentrations (mean of 3 sites), and central London roadside concentrations (1 site). Measurements made during 2004 are provisional [*Fuller and Green*, 2006]

Although there are no extensive ambient VOC monitoring sites across Europe, there have been a number of studies done in the UK, with monitoring sites in London [*Dollard et al.*, 2007; *Field et al.*, 1994]. Data from *Field et al.*, [1994] showed that generally, VOC concentrations declined from 1986 to 1992, especially those associated with motor vehicle exhaust. Later monitoring by *Dollard et al*. [2007] throughout the UK from 1993 to 2004 also found decreasing VOC concentrations. The concentrations were observed to be highest at curbside monitoring stations and lowest in rural locations; decreases were attributed to the implementation of exhaust gas catalyst and evaporative canister control technologies on gasoline motor vehicles [*Dollard et al.*, 2007]. Annual mean concentrations in 2000 ranged from 1.3 to 28.8 µg m^{-3} for toluene, 0.6 to 14.6 µg m^{-3} for ethylene, and 0.4 to 7.2 µg m^{-3} for propylene at rural and curbside locations, respectively [*Dollard et al.*, 2007].

Over the past decade, reductions of carbon monoxide and sulphur dioxide concentrations in London have been successful. For both, significant decreases in concentration have been observed since the late 1990s. Carbon monoxide was decreased by 56% from 1997 to 2007, and has been in attainment of the EU limit of 10 mg m^{-3} (max daily 8h mean) since 2000 [*Fuller et al.*, 2007]. The highest concentrations of CO are found near roadways, reflecting the fact that motor vehicles are the main source. Sulphur dioxide concentrations in London decreased by 78% from 1997 to 2007

and have been within the UK air quality guidelines of 125 µg m^{-3} daily mean (350 µg m^{-3} hourly mean; 266 µg m^{-3} 15min mean) since 1998 [*Fuller et al.*, 2007]. Unfortunately, episodes of elevated SO_2 still occur in London. A plot of the annual mean index value has been reproduced here as Figure 6 from the London Air Quality Report to show the trends from 1997 to 2008.

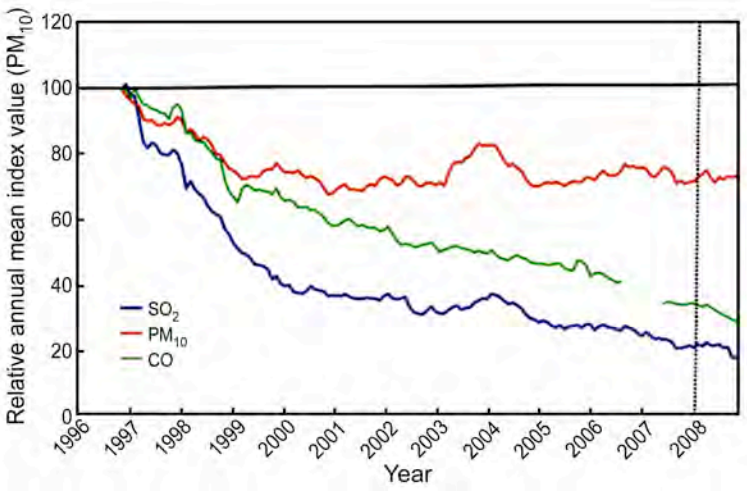

Figure 6 - London Air Quality Network annual mean index values for CO, PM_{10}, and SO_2. Measurements following the dashed line were provisional [*LAQN www.londonair.org.uk*]

6.2.3 Air quality regulations

The set of air quality guidelines most recently published in the UK are from July 2007. The standards are set to protect human health and the environment, based on the most current science, and all those discussed here can be found at www.airquality.co.uk. The air pollutants regulated include benzene, 1,3-butadiene, carbon monoxide, lead, nitrogen dioxide, particulate matter (both $PM_{2.5}$ and PM_{10}), sulphur dioxide, polycyclic aromatic hydrocarbons (PAHs), and ozone. Currently PM_{10} is regulated at an annual mean of 40 µg m^{-3}; $PM_{2.5}$ regulations are not in effect yet, but have an annual mean target of 25 µg m^{-3} to be reached by 2020. The annual mean standard is also not yet in effect for benzene (5.00 µg m^{-3}) or PAHs (0.25 ng m^{-3}), which were to be reached by the end 2010. All other compounds have annual mean values as follows, 1,3-butadiene (2.25 µg m^{-3}), lead (0.25 µg m^{-3}), nitrogen dioxide (40 µg m^{-3}), and PM_{10} (40 µg m^{-3}). Carbon monoxide and ozone are both regulated by 8 hour running means, with standards of 10.0 mg m^{-3} and 100 µg m^{-3}, respectively. The 24 hour mean for sulphur dioxide is 125 µg m^{-3}. Many of these compounds in addition have a limited number of exceedences allowed per year, as well as standards for shorter time periods. For more detail, please see the website listed above. In addition to the UK air quality standards, there are some issued by the European Commission (EC) for ozone, nitrogen dioxide, and sulphur dioxide. The time frames for the EC standards do not correspond to those from the UK. Overall, the UK standards are similar to or lower than those issued by the EC.

An additional measure taken in Greater London in an attempt to combat the air pollution, specifically that from motor vehicles (the main cause of air pollution for London), was the creation of a Low Emissions Zone on certain roads and motorways throughout Greater London. In addition, a Congestion Charging Zone (CCZ) was set up in central London in an attempt to reduce traffic congestion (and emissions) and encourage other methods of transport in the centre of the city. The CCZ was initially started in February 2003 and was expanded into West London on 19 February 2007, and is applicable only on weekdays. The low emission zone (LEZ) regulations went into effect on February 4, 2008, and are being updated with more stringent emissions standards in 2012. Currently, the LEZ does not apply to cars or motorcycles, but does apply to diesel-engined lorries, buses, coaches, motor caravans, motorized horsebox, large vans, minibuses, and other specialist vehicles [*tfl.gov.uk*]. In order not to be charged for driving in the LEZ, vehicles must meet the Euro III standards for particulate matter. Figure 7 shows the Greater London area with the LEZ and CCZ demarcated.

Figure 7 - Greater London Map with the Low Emission Zone and the Congestion Charging Zone marked *[tfl.gov.uk/lezlondon]*

6.2.4 Monitoring network

The London Air Quality Network (LAQN) was set up in 1993 by the Environmental Research Group (ERG) at King's College in London in cooperation with the London Boroughs and Regional Health Authorities. There are over 160 continuous monitoring sites throughout Greater London, and up to date information about air pollution levels is available on their website www.londonair.org.uk.

6.3 PARIS

6.3.1 Population and geography

Among the 22 regions of French mainland, the *Ile de France* region has the highest population, with about 11.7 millions inhabitants, and the highest population density (nearly 1000 inhabitants per km^2), over an area of about 12000 km^2. It hosts about 19% of the French population, but generates 28.6 % of the gross domestic product of France (excluding overseas departments) [source: *INSEE*, Institut National de la Statistique et des Etudes Economiques].

Most of the *Ile de France* population is living in an urbanised area of about 30 – 40 km diameter, around Paris city. In these areas, population density is between 3000 and more than 30000 inhabitants per km^2. The agglomeration is also surrounded by rural areas, mainly agricultural land and forest areas (Figure 8). The *Ile de France* region is mainly flat, altitudes varying from 11 to 221 meters above sea level.

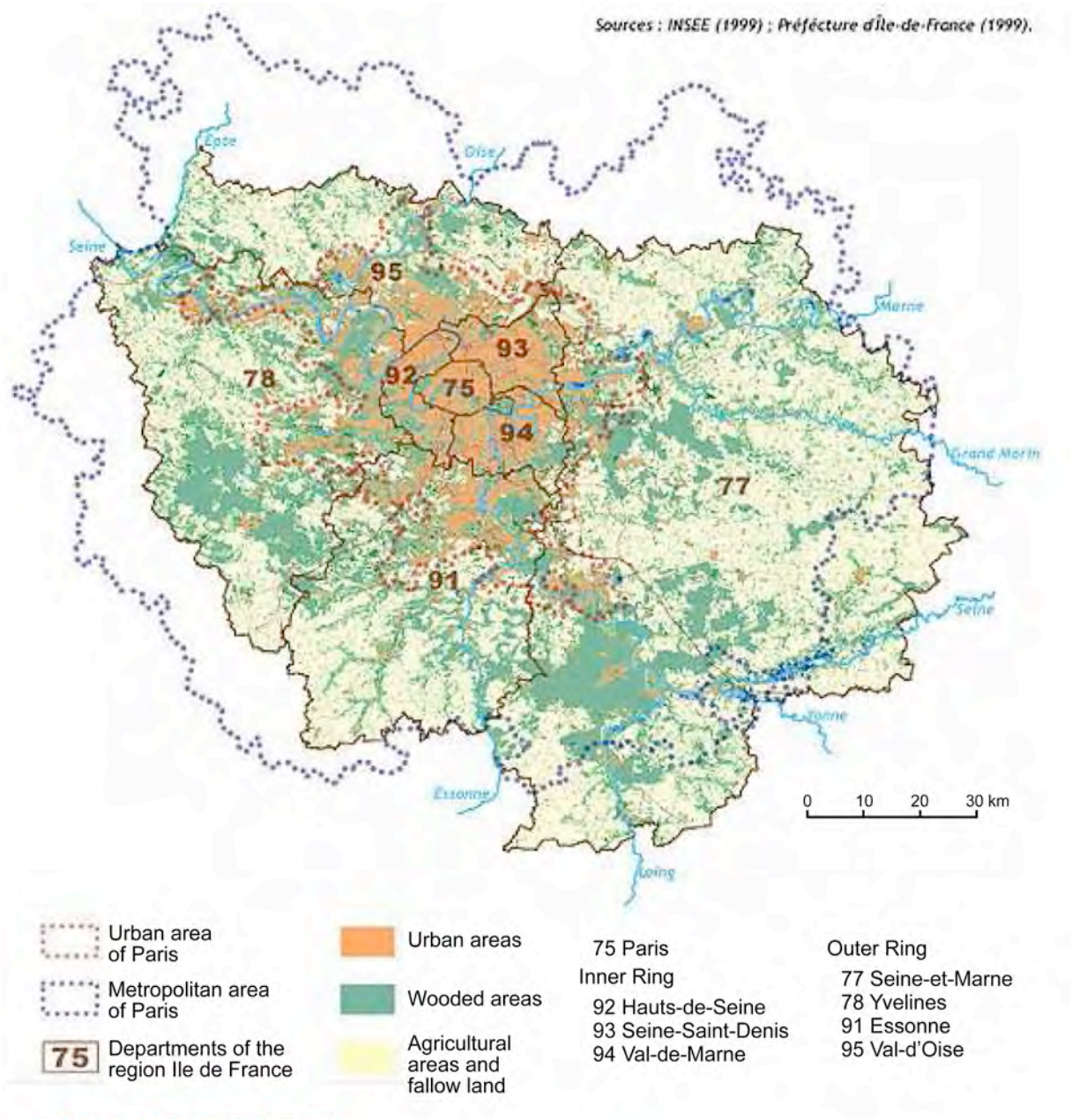

Figure 8 - Land use in the *Ile de France* region. Red: urban areas; green: forested areas; yellow: agricultural areas. The administrative borders of different departments are also indicated.
[http://www.iau-idf.fr/fileadmin/user_upload/SIG/cartes_telecharge/thema/Densipop_1999.pdf]

6.3.2 Pollutant emissions

Pollutant emissions from the *Ile de France* region represent approximately 10 % of the national French emissions for NO_X, SO_2 and CO_2, and approximately 5% for CO, NMVOC, and PM_{10} [AIRPARIF, http://www.airparif.asso.fr/pages/emissions/emisidf]. For NO_X and CO, traffic is the dominant source (Figure 9). For NMVOC and PM_{10}, both traffic and industry are dominant sources followed by the residential / tertiary sector. For SO_2, industry is the dominant source.

Over the *Ile de France* area, emissions are mainly intense over the city of Paris (red colour in Figure 10), the agglomeration, over a larger urbanised area of about 40 km diameter, and along major highways. This is shown in Figure 10 for the example of NO_X emissions.

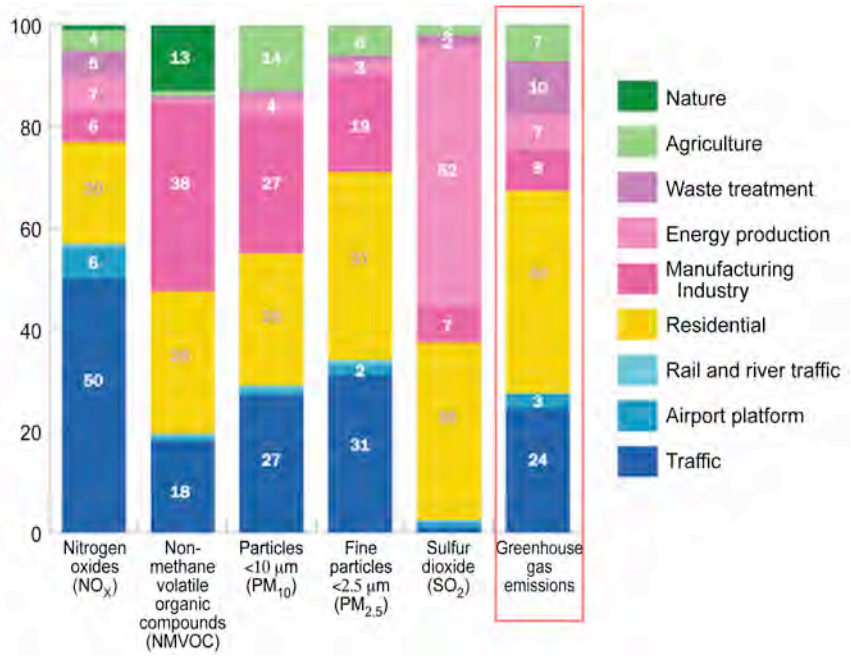

Figure 9 - Partitioning of major pollutant emissions in the *Ile de France* region among major activities, for the year 2005 (in percent); from left to right: nitrogen oxides, non-methane volatile organic compounds, particulate matter PM_{10}, fine particulate matter $PM_{2.5}$, sulphur dioxide, climate active gases (CO_2, CH_4, N_2O in CO_2 equivalents, from top to bottom: nature (dark green), agriculture (light green), waste treatment (dark magenta), energy production (light magenta), industry (medium magenta), residential and tertiary (yellow), rail and ship traffic (light blue), airports (medium blue), road traffic (dark blue)
Source*: [AIRPARIF http://www.airparif.asso.fr/pages/emissions/emisidf*]

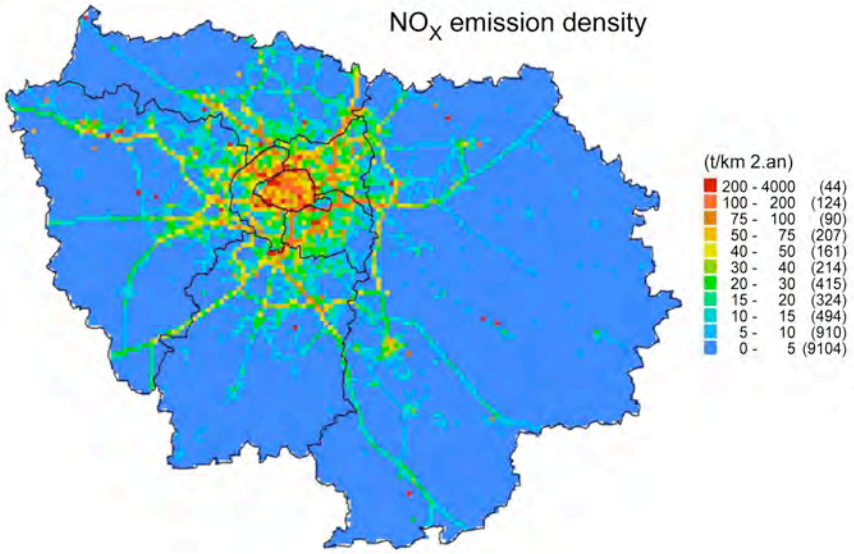

Figure 10 - Density of annual NO_X emissions over the *Ile de France* area, in [tonnes/km²/yr], for year 2005;
Source: [*AIRPARIF http://www.airparif.asso.fr/pages/emissions/emisidf*]

6.3.3 Meteorology

Most of the time Paris benefits from relatively sustained winds from south-west to west, advecting relatively clean oceanic air masses to the region, and allowing for good dispersion of local pollution sources. Under anticyclonic conditions, sunny weather and weaker northerly to easterly winds allow for local pollution build-up, its photochemical processing, and for advection of continental air masses to the area (see example in Figure 11).

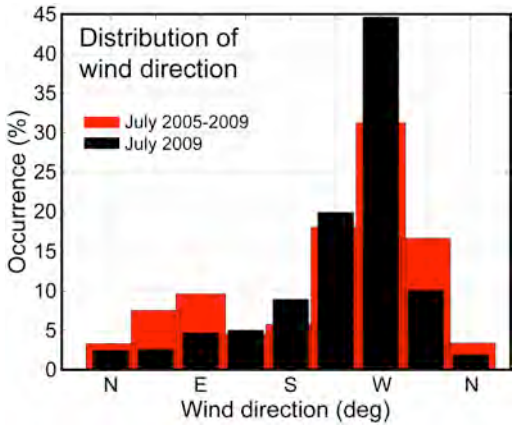

Figure 11 - Wind direction at the SIRTA/IPSL site at Ecole Polytechnique, 20 km southwest of downtown Paris. Red bars: July 2005 – 2009, black bars: particular situation during the MEGAPOLI campaign in July 2009 [Courtesy of *M. Haeffelin*, *SIRTA / IPSL*]

6.3.4 General air pollution situation

In the *Ile de France* region, the Air Quality survey network AirParif (http://www.airparif.asso.fr/) has a dense routine measurement network with nearly 50 automated sites located within the Paris agglomeration and neighbouring rural areas, giving hourly concentrations of target pollutants. Nearly 15 years of gathered data allow for a climatological analysis of the evolution of urban and peri-urban pollution.

Figure 12 illustrates that yearly averages of primary pollutants such as benzene and NO, or SO_2 (of dominant industrial origin) show a negative trend at urban background sites over the last ten years. Ozone shows an upward trend, probably due to decreasing titration with fresh NO emissions. NO_2 and PM_{10} do not show a significant trend. Both are due to primary emissions and secondary chemical transformation.

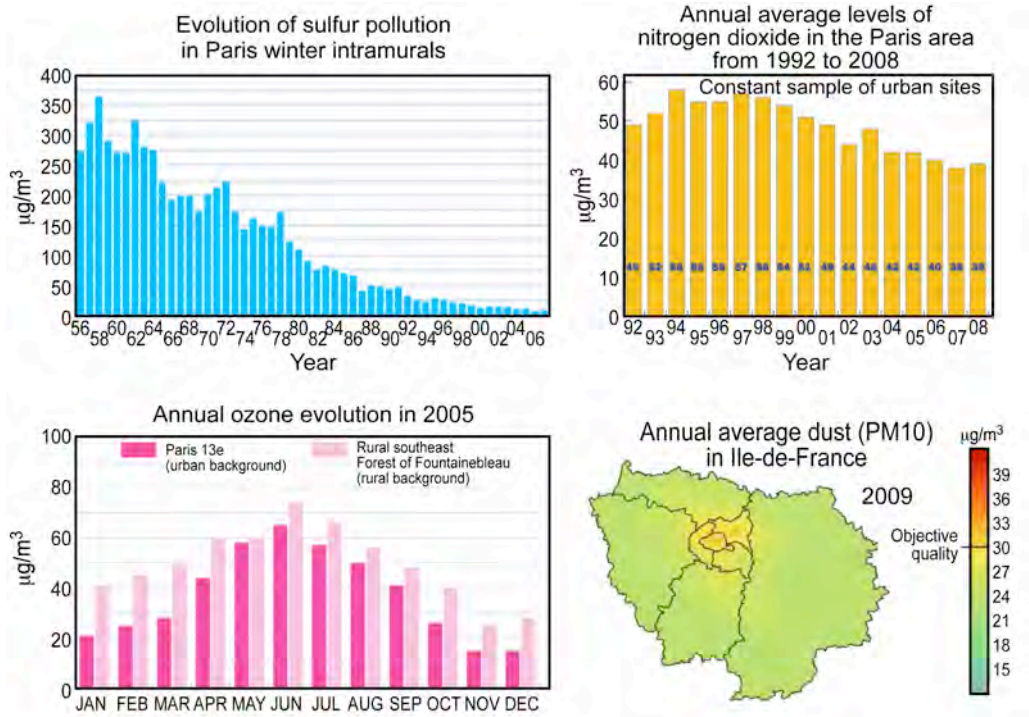

Figure 12 - Trends of major atmospheric pollutants: SO_2, (top left), of major industrial origin, NO_X (top right), of major traffic origin for urban background sites from the AirParif network; ozone monthly averages for year 2005(bottom, left) for an urban background site (dark purple) and an rural site (light purple), PM_{10} annual distribution for year 2009 showing larger values in Paris downtown (inner circle)
[AIRPARIF http://www.airparif.asso.fr/pages/polluants/evolution]

From a regulatory point of view, enhanced urban NO_2 levels present the biggest problem. During the last few years, the annual average limit value of 40 µg/m^3 as of 2010 was violated over a large part of the Paris agglomeration. For ozone, the national air quality standard for protection of human health in 2010 limits exceedances of the daily maximum 8 h average of 120 µg/m^3 at 25 times per year, averaged over three years. According to measurements in recent years, the new ozone standard is achievable, except for a year with exceptionally hot and anticyclonic conditions like during the summer of 2003. The information threshold of 180 µg/m^3 (hourly mean) for ozone is exceeded several times a year. For PM_{10}, the objective of an annual average below 30 µg/m^3 is violated in the central region of the agglomeration during some years (e.g. 2003 and 2007).

6.3.5 Specific scientific campaigns

In addition to the operational air quality monitoring, a large number of field campaigns and scientific studies have been performed in the last ten years to study the processes of air pollution build-up in the Paris agglomeration. A comprehensive gas phase chemistry experiment was performed as part of the ESQUIF campaign during the summers 1998 and 1999. Particulate matter formation in the region has been recently addressed during the very comprehensive MEGAPOLI experiment campaigns in July 2009 and January-February 2010.

A) Photo-oxidant pollution ESQUIF campaign and later studies

The **ESQUIF** campaign (IPSL and LISA, Research Centre Jülich, Météo-France, AirParif, Laboratoire d'Aérologie) has allowed a detailed documentation of major gaseous pollutants (O_3, NO_X, NOy, VOC) and a first characterisation of particulate matter (chemical, size distribution) within the Paris agglomeration and its plume during 13 multiday IOP's (Intensive Observation Periods) between 1998 and 2000 [*Menut et al*., 2000; *Vautard et al*., 2003a]. An integrated observation network including ground based in-situ and remote sensing instruments as well as airborne measurements were set up. Major outcomes from this campaign are presented in the following paragraphs.

Emission uncertainty

Airborne NOy, CO, and VOC measurements from the ESQUIF campaign were used in combination with the air quality photochemical model CHIMERE in order to diagnose uncertainties in the current emission inventory from AirParif for the year 2000 [*Vautard et al*., 2003b]. There is reasonable consistency between simulated and measured concentrations. NO_y simulations agree with measured concentrations to within 35%. There are significant underestimations and overestimations in some individual primary hydrocarbons. However, the total mass and reactivity of the measured hydrocarbon mixture, which accounts for only about half of the total emitted mass, agree with modelled values to within an estimated uncertainty of 40%. These values give direct constraints for the emission uncertainty, given that uncertainty in model parameters (for example boundary layer height) is lower than that of the uncertainty for emissions.

This work was later extended by inverse modelling work assessing the uncertainty in regional emission inventories [*Pison et al*., 2006; *Deguillaume et al*., 2007]. The latter authors applied a Bayesian Monte Carlo analysis using urban and surrounding air quality observations to correct average Paris agglomeration with respect to the initial inventory and to derive their uncertainty: for VOC emissions the result was 0 ± 20 % (1 sigma), and for VOC emissions +16 ± 30 %.

Ozone plume characteristics

Average photochemical ozone build-up in the Paris agglomeration plume during summers 1998 and 1999 was about 15 ppb [*Deguillaume et al*., 2008]. During the ESQUIF IOP days, this value was often several tenths of ppb (Figure 13). During summers 1998 and 1999, plumes were most often encountered in the north-eastern sector, consistent with the climatology of the region of predominantly south-westerly winds. However, the plumes with the highest pollution levels were encountered in the south-western sector, corresponding to anticyclonic situations with stagnant north-easterly winds.

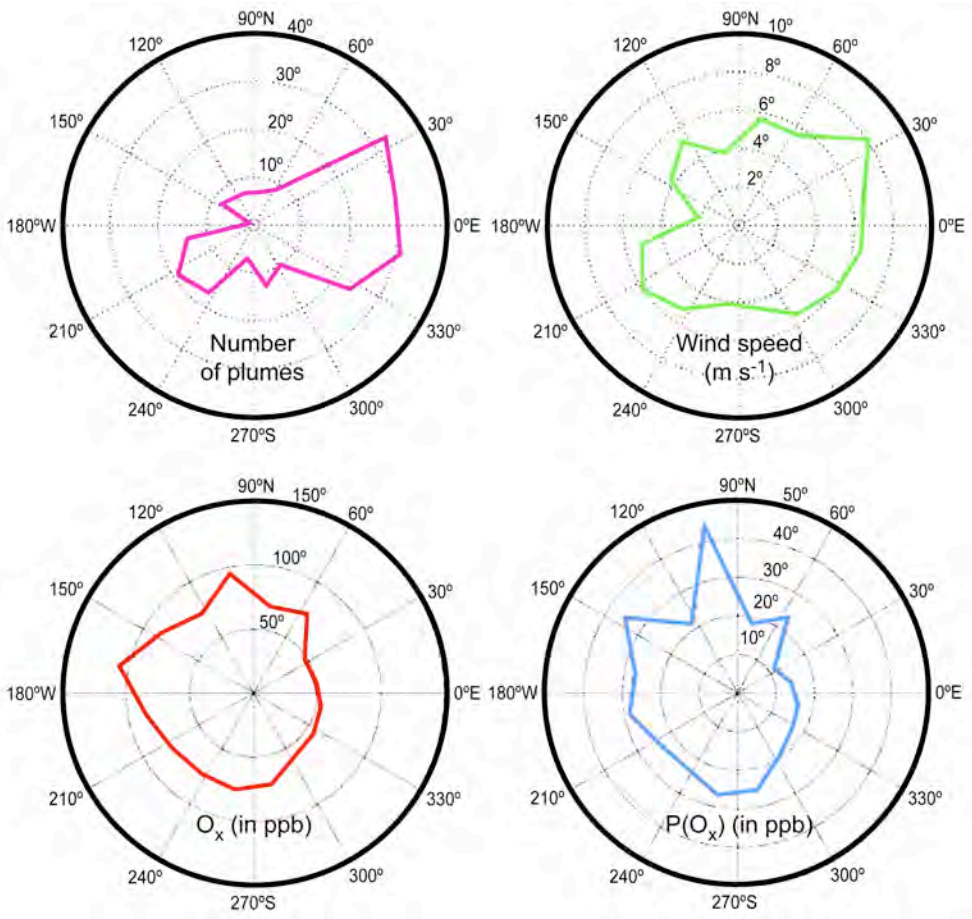

Figure 13 - Number of ozone plumes during summers 1998 and 1999. An ozone plume is defined for days with a photochemical O_X ($=O_3+NO_2$) production of more than 10 ppb [*Derognat*, 2002]

Major factors of photooxidant pollution and chemical regimes

From an observational based approach, using ozone, NOy, and VOC measurements in the plume in conjunction with simulations, *Sillman et al*. [2003] derived either a VOC or NO_X sensitive chemical regime ozone build-up depending on the meteorological conditions for particular days. Using Monte Carlo analysis with an observational constraint, *Beekmann and Derognat* [2003] showed that using the ESQUIF data set could reduce the uncertainty by a factor of 1.5 to 3 for different days. Extending this type of analysis to summers 1998 and 1999, *Deguillaume et al*. [2008] showed that the photochemical ozone build-up in the plume was in general VOC sensitive, and that the result of an average VOC sensitive regime was robust with respect to model uncertainty. *Derognat et al*. [2003] showed that biogenic VOC emissions (mainly isoprene) significantly contribute to photochemical ozone build-up in the region, 9 ppb on the median for ESQUIF IOP days (a selection of more polluted days), and up to 40 ppb for an exceptionally hot and polluted day. Using the CHIMERE adjoint for sensitivity analysis, *Menut et al*. [2003] tested the sensitivity of ozone build-up due to a large variety of parameters and found them mainly driven by traffic and solvent surface emissions and meteorological parameters such as temperature. On average, only about a quarter of the ozone present in the Paris agglomeration plume is formed from local emissions, with the majority of the ozone advected from outside. During high pollution episodes (O_3 > 90 ppb), the local fraction is more than 40% [*Derognat*, 2003]. In extension of results of the ESQUIF campaign, the LISAIR campaign (LSCE /IPSL, AirParif) has gathered a large amount of urban VOC measurements (C2 – C12) during May 2005 in conjunction with aerosol lidar measurements (boundary layer height evolution) in order to address the spatial and diurnal VOC variability for different urban sites [*Gros et al*., 2007].

B) Particulate matter pollution

Besides from air quality network measurements, climatological information about aerosol loads over the Paris region is available from optical measurements. Aerosol optical density (AOD) is measured by a sunphotometer part of the AERONET network 20 km SW of Paris town centre. During clear sky days, AOD is mostly comprised between 0.1 and 0.4, with a median value of 0.17 [*Chazette et al*., 2005]. The average single scattering ratio (at 532 nm) of 0.9 is typical for urban aerosol [*Raut and Chazette*, 2007]. From comparisons of aerosol backscatter lidar measurements at the same site and model simulations, it has been inferred that secondary organic aerosol was probably underestimated in the CHIMERE model simulations [*Hodzic et al*., 2004].

Regional / continental origin of aerosols in the Paris region

Hourly concentrations of inorganic salts (ions) and carbonaceous material in fine aerosols (aerodynamic diameter, A.D. < 2.5µm) have been determined from fast measurements performed for a 3-week period during the spring of 2007 at an urban site within the city of Paris [*Sciare et al*., 2010, submitted]. The sum of these two chemical components (ions and carbonaceous aerosols) has shown to account for most of the fine aerosol mass ($PM_{2.5}$) in this area. This time-resolved dataset allowed investigation of the factors controlling the levels of $PM_{2.5}$ and showed that polluted periods with $PM_{2.5}$ >40µg/m^3 (during periods I and III in Figure 14) were characterized by air masses of continental (European) origin and a chemical composition made up of 75% of ions. By contrast, clean marine air masses show the lowest $PM_{2.5}$ concentrations (typically of about 10µg/m^3, period II in Figure 14) with carbonaceous aerosols contributing most of the mass (typically 75%). The rather stable levels of carbonaceous aerosols observed during this study suggest that the region of Paris is a major contributor to this fraction.

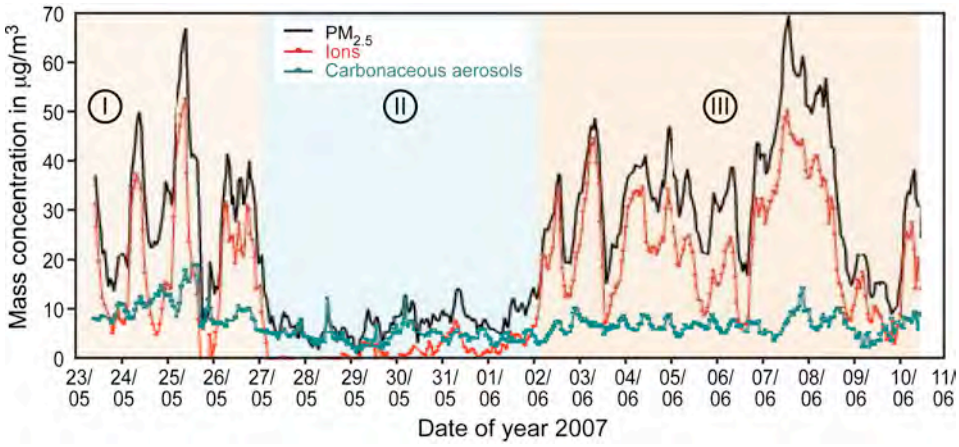

Figure 14 - Time resolved PM measurements in downtown Paris in May/June 2007. The periods I and III correspond to a predominance of inorganic ions of mainly continental origin in Paris downtown $PM_{2.5}$, the period II corresponds to a predominance of carbonaceous aerosol of mainly local origin
[*Sciare et al*., 2010]

By contrast, long-range transport from Europe is proposed as the main contributor for ions measured in Paris during the springtime. Further studies need to address if these results are valid on a longer climatological time scale.

C) The MEGAPOLI campaign

The campaign, performed in the framework of the EC FP7 project MEGAPOLI, aimed at a better quantification of primary and secondary organic aerosol sources and their relation to gaseous precursors, at the example of a big European Megacity, the Paris region. Indeed, these aerosol fractions make up an important contribution to urban fine particle matter and their sources are among the most uncertain. The campaign design included three primary and four secondary fixed ground measurement sites, an aircraft, and five mobile platforms (Figure 15). Fixed sites were distributed over urban and peri-urban areas. Mobile platforms allowed sampling the pollution plume and background conditions. For many sites, complete instrumentation was set-up comprising aerosol chemistry and physical properties as well as components of gas phase chemistry. The summer part of the campaign took place in July 2009 while the winter part took place from January 15 to February 15, 2010. More than 25 laboratories participated (funded by EC, from French national funding, or from own means).

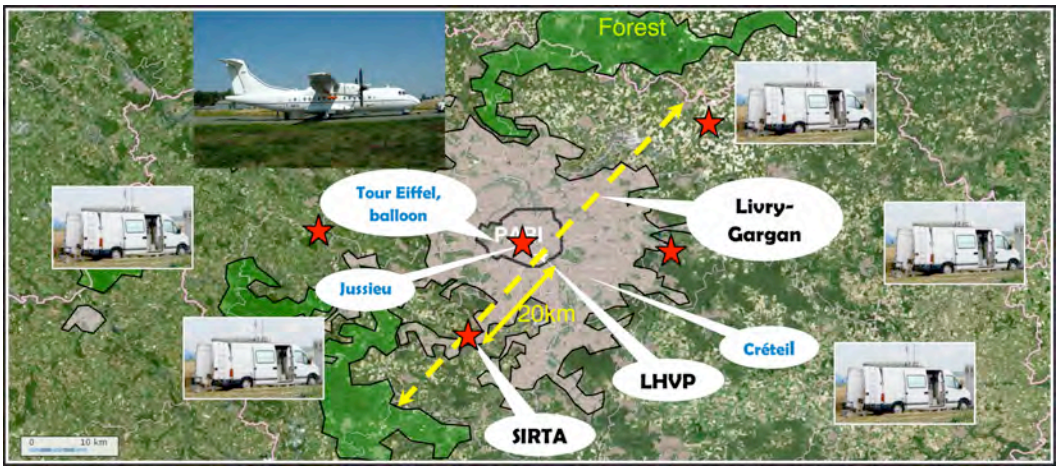

Figure 15 - MEGAPOLI campaign set-up

3 primary sites	=> full *in situ* measurements (LHVP, SIRTA, Livry_Gargan / + meteo at SIRTA
3 secondary sites	=> lidar and spectrospcopic measurements / or in some situ
3 mobile labs	=> in situ measurements, aerosol characterisation
1 mobile lab	=> lidar measurements
1 mobile lab	=> MAXDOAS
1 aircraft ATR-42	=> full in situ measurements (SAFIRE, CNRS, MPI)

Additional lidar network in winter (red stars)

The campaign was clearly a success, with measurement coverage above 90%. At the moment, measurements are being analyzed and quality checked by partners. Measurement quick-looks have already been submitted to the campaign database at LISA (http://megapoli.lisa.univ-paris12.fr/). First interesting results include:

- From airborne primary pollutant measurements, the pollution plume was still well defined at more than one hundred kilometres downwind from the agglomeration.
- Very preliminary attribution of organic aerosol (OA) from AMS mass spectrometer urban and peri-urban measurements during the summer campaign shows a large fraction of oxidised organic aerosol (OOA) of secondary origin and a smaller fraction of unoxidised organic aerosol (HOA) of primary origin. At the urban site, about half of OA is water soluble, corresponding probably to classical secondary organic aerosol, another half is water insoluble, corresponding probably to primary and chemically processed primary OA.
- Significant new particle formation events were observed in the area during the whole month of the summer campaign. These events were assisted by the relatively low particulate matter concentration levels and resulting low surface area during most of July 2009.

- During the winter campaign, wood burning was a significant source of organic aerosol. Both local emissions and continental advection were responsible for aerosol pollution build-up.

6.3.6 Conclusion and outlook

In conclusion the Paris agglomeration is a major population hotspot with high pollutant emissions from traffic, the residential/tertiary sector, and industry. Good dispersive conditions limit the impact on air quality of these emissions. With respect to valid air quality regulation (or valid in near future), NO_2 (annual mean) is the most critical pollutant, followed by fine particles and ozone.

Several intense studies have allowed to draw a rather coherent picture of photo-oxidant pollution in the agglomeration and its plume, as well as a quantification of precursor emissions (NO_X, CO and VOC) and their respective impact on photo-oxidant levels. For particulate matter, the observations, simulations, and source apportionment of carbonaceous aerosol is still uncertain to highly uncertain. The recent MEGAPOLI campaign intends to close this knowledge gap. Detailed data sets on aerosols and their radiative and hygroscopic properties from this campaign should also help to better quantify the aerosol impact on regional climate in the region.

6.4 MOSCOW

6.4.1 Population, demographics, geography, and urban structure

Moscow is the capital and the largest city of Russia. It is also the largest metropolitan area in Europe, and is the seventh largest megacity in the world. According to the 2002 census the Moscow population was estimated at 10,382,754. However, this figure only takes into account legal residents. Substantial numbers of internal migrants mean that Moscow's population is still increasing, whereas the population of many other Russian cities is in decline. The city encompasses an area of 1035 km^2. The Moscow River flows through the centre of the city and the Kremlin lies in the direct centre. In Figure 16 the blue-gray pixels in this false-colour image are urban areas of Moscow. The light green areas surrounding the city are farms and the brown regions are more sparsely vegetated areas. This image of Moscow was acquired by the Enhanced Thematic Mapper plus (ETM+), flying aboard the Landsat 7 satellite on July 23, 2002. Average elevation of the city is 156 m. The highest point is Teplostanskaya highland at 255 m. The lowest point, 110 m, is the Moscow river bank in the South-eastern part of the city. Green areas make up 30% of the territory, which is a fairly high value for a megacity.

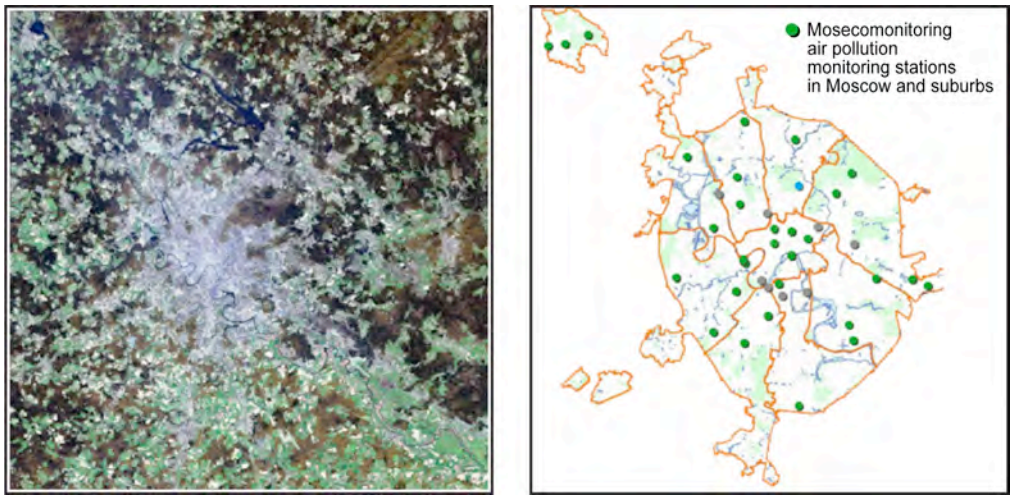

Figure 16 - Left: Satellite image of Moscow acquired by the Enhanced Thematic Mapper plus (ETM+), flying aboard the Landsat 7 satellite. July 23, 2002 (*NASA, http://visibleearth.nasa.gov/view_rec.php?id=3434*)
Right: Map of the Mosecomonitoring air pollution monitoring stations in Moscow and suburbs

Moscow has a humid continental climate with warm, somewhat humid summers and long, cold winters. Typical high temperatures in the warm months of June-August are around 23 °C, but during heat waves daytime temperature highs often top 30 °C. In the winter, temperatures normally drop to approximately −10 °C, though there can be periods of warmth with temperatures rising above 0 °C. Snow cover is formed at the end of November and melts in mid-March, but in recent years snow cover has melted earlier than usual. For example, in the winter of 2006-2007 there was almost no snow cover up to February and during the winter of 2007-2008 the snow cover melted at the end of February. The winter of 2009-2010 was very unusual for the last decade – a 'real' Russian winter with lots of snow, minus temperatures, and winter sunshine. Monthly rainfall totals vary minimally throughout the year (575 mm a year), although the precipitation levels tend to be higher during summer vs. winter. On average Moscow has 1731 hours of sunshine per year.

Urban features influence meteorological conditions and microclimate of the megacity. The estimated anthropogenic heat flux for Moscow gives 55.9 W/m^2 and surface air temperature in the city is on 2-5 degrees higher than outside Moscow.

Overall data show a complex environmental situation in Moscow. The average population density is 8,900 people/km^2, and the city is growing rapidly, with extensive emissions: 46 kg of pollutants per year are emitted per capita in Moscow [*Report of the Moscow Environment*, 2002]. While hundreds of thousands of sources emit pollutants into the air, only 60% of the enterprises have implemented pollution control measures. Cars provide a significant amount of the pollution and the majority of them do not meet typically European air pollution standards. In addition to extensive atmospheric gaseous and particulate pollutants, motor vehicles also contribute to highly toxic heavy metals: vehicular exhaust is the largest contributor of lead, while zinc comes from tire wear and diesel engines release cadmium into the environment. Industrial enterprises produce large amounts of dust, nitrogen oxides, iron, calcium, magnesium, and silicon. These compounds contribute to the haze over the city, increasing fog and precipitation and reducing solar radiation that reaches the surface.

The Moscow environment is closely associated with the background pollution, the regional natural conditions, and the climate of Eastern Europe. The dominance of westerlies is of key importance for air pollution: the western and northwestern districts of the city tend to receive more fresh air, due to the forests west of the Moscow region. Polluted air from the western parts of the city is transported to the eastern parts. During periods of easterly and southeasterly winds Moscow gets less fresh air, since the south-east area is only 25-30% forested, the land is largely plowed for agriculture, and more industries are located in this area.

6.4.2 Overview of emissions estimates

According to the *Report of the Moscow Environment* [2002; 2009] the main emission sources in Moscow are the following: 31,000 industrial and construction sites (including 2,500 vehicles companies), 14 central thermo-electrical factories and branches, 71 district thermal stations, 110 small heating plants, as well as about 3.5 million vehicles.

Every year about one million tons of pollutants are emitted from Moscow into the atmosphere (Figure 17). The main source of pollution is motor vehicles (from 83% to 92% of the total emission based on different estimates) followed by emissions from industrial stationary sources (up to 8%), and thermal power generation facilities (up to 4%). Thus, vehicular exhaust has a decisive role in shaping the level of air pollution in the surface layer of the urban Moscow atmosphere. Average vehicle emissions for the Moscow transport sector are shown in Figure 18. Specific urban pollution episodes can also be caused by emissions from outside the city: e.g. from forest fires, industrial emissions, dust from soil, etc.

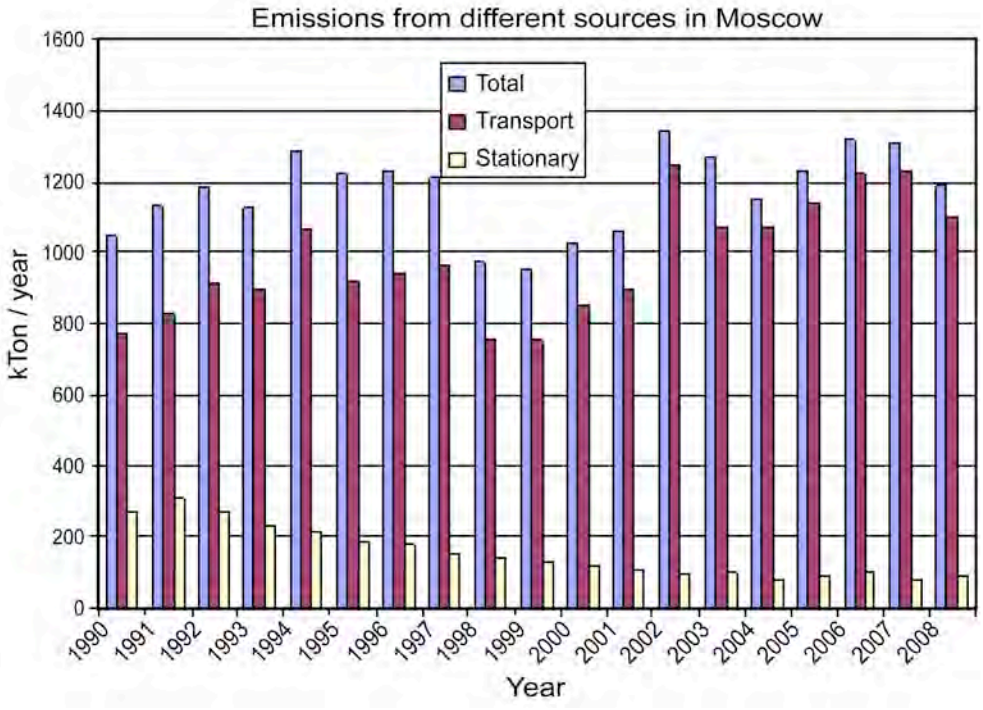

Figure 17 - Total emissions from different sources in Moscow during 1990 – 2008 [*Report, 2002; 2009; Kasimov et al.,* 2004]

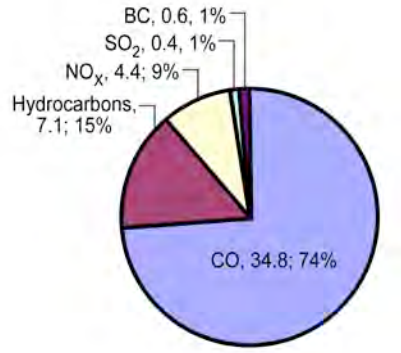

Figure 18 - Average vehicle fleet emissions for the Moscow transport sector in g/km and % of the total emissions [*Atlas,* 2000]

6.4.3 Overview of pollution levels

Urban and in the surrounding suburb regions

Air pollution in Moscow is very inhomogeneous (Figure 19). Hotspots are the roads and their surrounding areas. In residential areas, the pollutant concentrations are about 15-30% less than in the centre of Moscow and 30-50% less than in the vicinity of highways (Table 2). There is also some variability over time, though the overall integrated level of pollution is relatively stable, as can be seen in the integrated air pollution index (API), which calculated based on the concentrations of 5 major pollutants (CO, NO_2, NO, O_3, and formaldehyde) relative to the national guideline concentrations for each pollutant, and a scaling factor which indicates the relative toxicity of each pollutant, so that a dimensionless comparable quantity indicating the overall pollution level is determined. The API was 6.2 in 2008, 6.3 in 2007, 6.4 in 2006, 6.1 in 2005, and 6.2 in 2004.

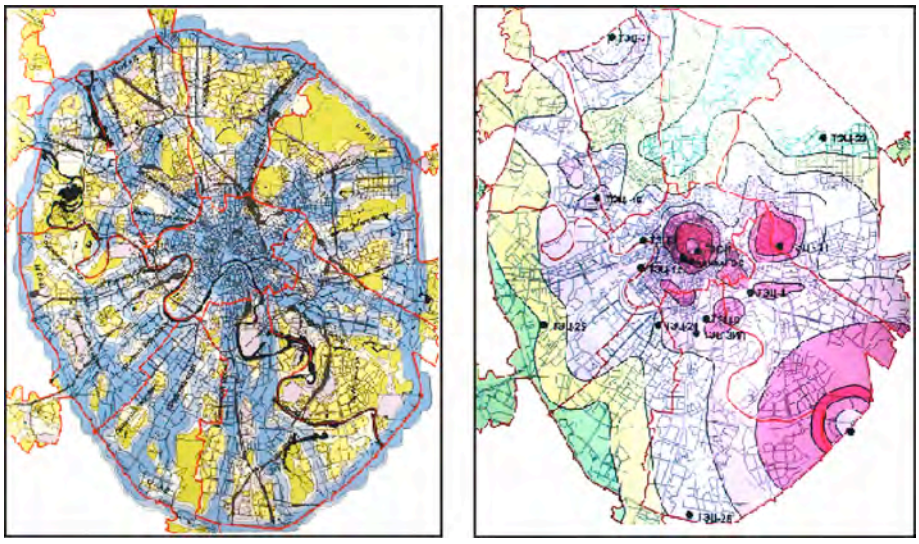

Figure 19 - Moscow NO_2 air pollution in threshold limit values, TLV (0.04 mg/m^3): Left: from Moscow transport, (> 2.0 - deep blue, 1.0-2.0 - blue, 0.5-1.0 – light blue); Right: from industrial and energy production sources (> 2.5 - deep red, < 0.5 - blue) [*Atlas*, 2000]

Table 2 - Annual average concentrations of the main pollutants (mg/m^3) in years 2006 – 2008 for different areas of the Moscow megacity and national threshold limit values (TLV) [based on *Report*, 2009]

		Average for city			Close to highways			City centre			Residence areas		
	TLV *	2006	2007	2008	2006	2007	2008	2006	2007	2008	2006	2007	2008
CO	3.0	0.8	0.7	0.57	1	1	0.71	0.9	0.8	0.59	0.7	0.6	0.51
NO_2	0.04	0.046	0.042	0.036	0.05	0.051	0.044	0.05	0.044	0.035	0.043	0.039	0.031
NO	0.06	0.048	0.046	0.038	0.067	0.057	0.053	0.055	0.054	0.035	0.039	0.041	0.026
SO_2	0.05	0.006	0.006	0.003	0.007	0.007	0.004	0.006	0.006	0.004	0.006	0.007	0.002
PM_{10}	(0.15)	0.033	0.035	0.037	0.045	0.048	0.046	0.035	0.035	0.039	0.031	0.032	0.038
O_3	0.03	0.028	0.032	0.032	0.031	0.039	0.037	0.025	0.031	0.03	0.026	0.028	0.032
API **	-	6.4	6.3	6.2	7.3	7.1	7	6.3	6.5	6.3	6.1	6.1	6

* *national threshold limit values for daily average concentrations (TLV)*

** *integrated air pollution index (API) calculated on 5 major pollutants: CO, NO_2, NO, O_3, and formaldehyde*

High levels of air pollution are observed near large highways and industrial zones, especially in eastern and south-eastern parts of the city. The highest air pollution levels are observed in areas of Kapotnya, Lyublino, and Maryino due to the location of an oil refinery. The minimum level of pollution is observed in the city districts of Krylatsky and Silver Bor.

The annual O_3 concentration in 2008 of 0.032 mg/m^3 exceeded the national threshold limit values (TLV, see Table 2). In the city, average O_3 concentrations varied from 0.023-0.027 mg/m^3 (0.8-0.9 TLV) in the central part of the city, while close to highways the O_3 concentration is up to 0.036-0.038 mg/m^3 (1.2-1.3 TLV). The highest O_3 concentrations are observed in May (1.5 TLV), June, and July (1.3-1.4 TLV). The lowest O_3 concentrations are observed during winter (0.5-0.6 TLV). Daily mean concentrations were above the TLV 30 to 70% of the time.

Contamination of the surface air layer, to a large extent, depends on meteorological conditions. On average, the air pollution potential is low directly in the Moscow region, due to a good dispersion potential. This leads in turn to a large regional pollution potential, which affects the

air quality in the surrounding areas; Moscow is computed to be the most effective megacity worldwide in exporting pollutants to the boundary layer of regions more than 1000 km downwind [*Lawrence et al*., 2007]. In certain periods, when meteorological conditions trigger accumulation of harmful substances in the surface layer, the pollution concentrations may be drastically increased, leading to high pollution episodes. Both summer and winter episodes with high concentrations occur frequently in Moscow. One of highest summer pollution episodes in Moscow occurred in September 2002 and was caused by peat fires in the Moscow region [*Chubarova et al*., 2009]. The highest winter pollution episode occurred during February 2006 [*Report of the Moscow Environment*, 2007] when from 3 to 9 February 2006 a combination of weak winds and a capping inversion layer increased pollutant concentration levels to the highest yet seen. Elevated levels of air pollution led to the continuous growth of allergic and asthmatic diseases for children and high mortality among elderly population during the summer smog events.

Historical trends, connection to political regulations, future prognosis

MosEkoMonitoring has established a multiyear database of pollutant concentrations in Moscow that gives the possibility to analyze trends, develop forecasts, and to inform the decision making process. Analysis of air quality monitoring data shows (Figure 20) that in recent years air quality has generally remained at an approximately constant mean level. During the last 10 years the level of air pollution was highest during 2002 (an anomalous year with long-term adverse weather conditions when Moscow was polluted from forest and wood smoke-peat fires in the surrounding areas). There are modest negative trends in the air concentrations of SO_2 and NO_X, while the CO concentrations have been decreasing continuously and significantly since 2002.

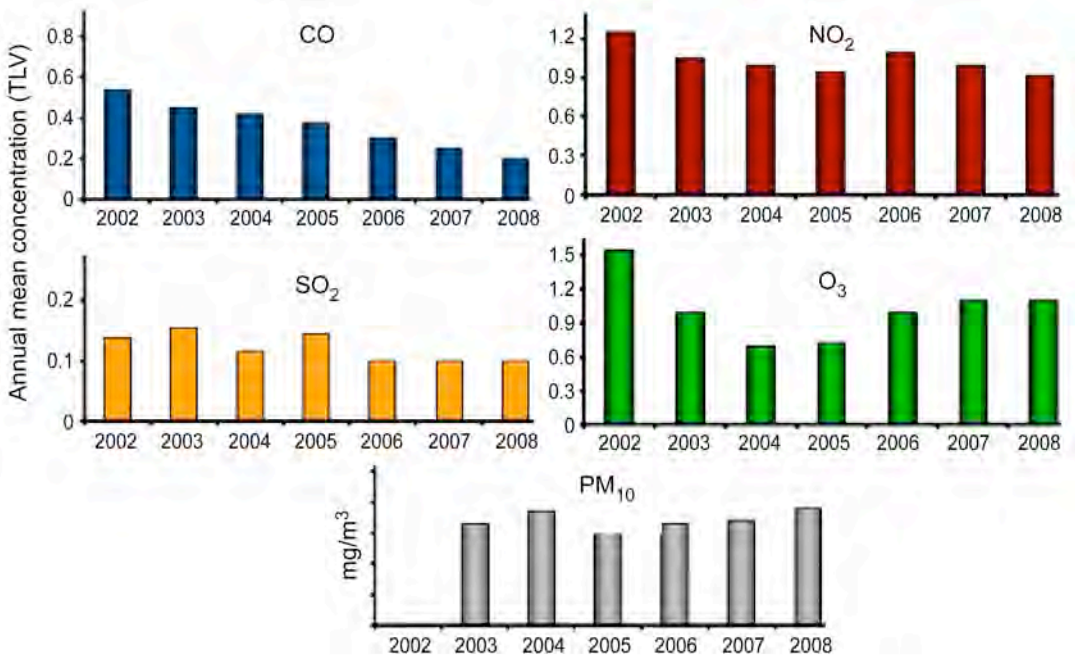

Figure 20 - Temporal annual mean concentrations of the main pollutants in Moscow during the years 2002-2008; in threshold limit values (TLV) for CO, NO_2, SO_2, O_3 and in mg/m^3 for PM_{10} [*Report*, 2009]

The concentration of total hydrocarbons has remained approximately unchanged since 2004 at 1.6-1.7 mg/m^3 for residential areas outside the direct impact of road transport and industrial enterprises and 1.7-1.9 mg/m^3 for areas exposed to vehicle emissions. SO_2, as an indicator of the use of reserve fuel types by thermal power plants, in recent years has remained stably low in all areas of the city (average concentrations amounted 0,006 - 0,007 mg/m^3). Average concentrations of PM_{10} remained almost unchanged, except for a notable interannual variability. Since 2004 PM_{10}

concentrations have remained stable at 0.037-0.036 mg/m^3 in areas away from highways, where the major sources of pollution are due to long-range atmospheric transport and soil dust from neighbourhoods. Close to major highways a slight (within 6%) but steady annual increase in the maximum observed annual average concentrations of PM_{10} (0.045 mg/m^3 in 2004, 0.046 mg/m^3 in 2005, and 0.049 mg/m^3 in 2006) is observed. Since 2006 a trend of slow but steady decrease in NO_2 pollution has been observed, but the average concentration is still very high (1.1 national TLV for the central part and 0.8 TLV – for peripheral residential areas). The average concentration of ground-level O_3, which is linked strongly to meteorological conditions, varies substantially over the years, from a minimum of 0.6 mg/m^3 (2004) up to a maximum of 1.5 mg/m^3 (2002).

The CO_2 concentration in Moscow is measured at 3 stations situated within residential areas as well as on the TV tower Ostankino. The mean annual CO_2 mixing ratio in the surface air varies from 423 to 451 ppm and does not exceed the EC norms. Vertical profiles CO_2 measured at the TV tower give the following average values: at 130 m – 390 ppm; at 250 m – 400 ppm; and at 350 m – 370 ppm. The main cause of maximum at 250 m could be hot air (with CO_2) emissions from smokestacks of the city's power plants.

6.4.4 Field campaigns in Moscow

The Moscow air quality monitoring system started in 1996 by the decision of the Government of Moscow. Over the years the system is continually evolving and improving. At the present time it includes a network of 28 automatic stations and 2 mobile laboratories that measure 18 priority pollutants. Near real time monitoring data is transferred to a joint information-analytical centre of «Mosekomonitoring».

In March of 1998, the Moscow government, the Fund Programme Management Branch of UNEP, the UN Centre for Human Settlements (Habitat), and the Centre for International Projects (CIP) signed an agreement to carry out the Moscow sustainable cities project in the framework of the world Sustainable Cities Programme (SCP). The Moscow sustainable cities project was aimed at conducting municipal, national, and international activities in the framework of the SCP and preparing and publishing the local options of the Habitat agenda for Moscow [*Ginzburg*, 2000].

The Moscow government initiated in 2000 a WMO GURME Pilot Project on "Meteorological Servicing for Sustainable Development of the Moscow Megalopolis" [*Vasiliev and Liakhov*, 2000]. The project focused on atmospheric pollution and urban meteorology and in particular the urban heat island (UHI) effect measured by microwave remote sensing data [*Kuznetsova et al*., 2004]. Continuous temperature observations in the PBL provided unique data to investigate spatial and temporal features of temperature fields over the Moscow megalopolis. The results showed that over the Moscow megalopolis, meteorological conditions are more favourable for self-cleaning of air then in suburban areas. Temperature inversions block vertical exchange of air in the Moscow megalopolis less often then in suburban areas and they are less powerful. There are two seasonally-varying types of UHI over the megalopolis: (i) the high dome (up to 600 m) and (ii) low dome (up to 200-300 m).

One of the most interesting field campaigns was on the long-term transport of megacity plumes and observations of the atmospheric composition over Russia. This field campaign began in 1995 and is called "TROICA" (TRans-Siberian Observations Into the Chemistry of the Atmosphere). Long-range transport is monitored using the TROICA mobile observatory, which is implemented in a wagon in the trans-siberian railway train [*Elansky*, 2006]. Scientists from across the world joined forces in a number of TROICA measurement field campaigns to gather the necessary information for better understanding the chemistry of the atmosphere [*Crutzen et al*., 1998]. The TROICA campaigns have been carried out over different regions of Russia, including sampling urban atmospheric pollution in Moscow and in other large Russian cities. In addition to primary pollutants, greenhouse gases, and volatile organic compounds, the chemical composition and characteristics of different aerosols in size bins ranging from 0.4 to 1000 nm were also measured. The results will help understand the scale of possible urban pollution effects from Russian megacities like Moscow, Novosibirsk, and Omsk. Several further research organizations are also doing field research and studies of Moscow air pollution, including IFA RAS, Geographical

faculty of MSU, IGCE RAS, AeroCosmos, HydroMetCenter, etc. More information on these and other programmes is given at the end of the next section.

As examples of the other kinds of international cooperation air pollution projects, two bilateral Russian-British projects during the last decade were:

1. Project “Creation and distribution of collected volumes for air contamination in Moscow” [*Moscow city government*, *British Council and DEFRA*, 2004; 2005]. The project indentifies target groups among organizations and the public interested in environmental information and creating mechanisms of gathering, analysis, presenting, and disseminating environmental information targeted to these groups using UK experience.

2. Project “Air quality management in Moscow and London” (2006-2007)

The aims of this project were to compare air quality in both cities, share knowledge of modelling air pollution using the ADMS-Urban dispersion model, share experience in relation to policies to reduce pollution, and establish a long-term biological monitoring network using lichen. A review of the pollutants measured, physicochemical equipment used, and characteristics of the sites monitored has been carried out for each city and the results have been compared. The information is related to objectives defined by the EU Air Quality Directive for the protection of human health and sensitive vegetation and ecosystems and the Russian health objectives. Air quality policies in each city have been reviewed with particular interest in schemes such as traffic management and alternative transport modes. This part of the project is led by Prof. Frank Kelly of King's College London, UK.

6.4.5 Current and planned major activities focusing on the city's air pollution

In Moscow, which was a very polluted city a few decades ago, there has been some reduction in urban pollution due, in many cases, to the economical crisis and industry degradation in 1990s after the USSR collapse. However, nowadays Moscow pays much more attention to air quality management programmes and emissions reduction strategies, so that the air quality is approaching the standards of West European cities.

The current Master Plan for the City of Moscow up to 2025 (approved by the Government of Moscow in 2005) defines the main priorities of urban development trends and practices to ensure environmental safety as well as a combination of economic, environmental, and urban development priorities. The main measures in air pollution mitigations include: (i) reducing the negative impacts of road transport system by tightening the requirements for environmental performance of transport (the transition to Euro-IV and Euro-V); (ii) reduction of total stationary source emissions by 25% due to the use of advanced environmental technologies and abatement equipment.

An important factor in improving the ecosystems of the city is the preservation and development of gardens, parks, and trees in yards, which have suffered in recent years due to a focus on building. The forest-park protective belt surrounding Moscow has decreased by 7% since 1997 due to the expansion of Moscow and the remaining suburban forests are rapidly losing their ecological protective functions [*Report*, 2002]. According to the ecological requirements the forest-park belt area around a megacity shall exceed the city area by not less than 5 times, and in Moscow it is just 1.5 times (in comparison with about 10 times for London, Paris, and Washington).

Several research organizations are doing research and studies of Moscow air pollution, including:

1. Institute of Atmospheric Physics of the Russian Academy of Sciences (IFA RAS), http://www.ifaran.ru.
2. Moscow State University (MSU), especially the Faculty of Geography, Department of Meteorology and Climatology, http://www.geogr.msu.ru/cafedra/meteo/.
3. Scientific Center of Aerospace Monitoring «Aerokosmos», http://www.aerocosmos.info.
4. Institute of Global Climate and Ecology, http://www.igce.ru/.
5. Mosekomonitoring, http://www.mosecom.ru/, which is responsible for organization and

implementation of the state environmental monitoring system for the city of Moscow.
6. Institute of Applied Geophysics of Roshydromet, SPA "Typhoon", and Institute of Experimental Meteorology (Obninsk, Kaluga reg.), http://www.typhoon.obninsk.ru/.
7. State Institution «Moscow Center for Hydrometeorology and Environmental Monitoring with regional functions» (PG Moscow ITF-R), as specifically authorized by the territorial authority Roshydromet, monitor pollution in the territory of Moscow and Moscow region.
8. Department of natural resources and environmental protection in Moscow (Moscow Government), http://www.moseco.ru/moscow-ecology/monitoring/air/. It monitors gas component of pollution (mainly CO, NO, NO_2, O_3, CH_4, CH) using the gas and aerosol (PM_{10}) in the network of automatic inspection of quality of atmospheric air (approximately 30 positions) and the Ostankino TV tower.
9. Research and design and survey institute of ecology city (Department of Atmospheric Environment), http://www.ecocity.ru/catalog/eco_mod/s_lowzagr.

According to a joint agreement between the EC DG Research and the Russian Ministry of High Education and Science, a new research call for a collaborative project with the EC FP7 project MEGAPOLI was opened in the year 2008 by the Russian Agency of Science and Innovations. This partnership project, complementary to MEGAPOLI and within the scope of the Federal Framework Programme of the Federal Science and Innovations Agency / Ministry of Education and Science of the Russian Federation, is focusing on development of integrated technologies of aerospace and ground-based monitoring of the urban-conglomerate and megacity (first of all Moscow) environments. The “AEROCOSMOS” Scientific Centre for Aerospace Monitoring, Moscow is coordinating the project with a Russian research consortium including MSU, IFA RAS and Roshydromet centre.

6.5 BENELUX/RHINE-RUHR

6.5.1 Population, demographics, geography, urban structure of major population centres

The Benelux/Rhine-Ruhr area is a strongly industrialized region located in Central Europe with high population density and about 40 million inhabitants. The BeNeLux area consists of Belgium (10.7 million inhabitants, area: 30.528 km^2), the Netherlands (16.5 million inhabitants, area: 41.526 km^2) and Luxembourg (0.5 Million inhabitants, area: 2.586 km^2). Overall within Benelux 27.7 million inhabitants are living on an area of 74.640 km^2. Additionally, the Rhine-Ruhr area (12 million inhabitants, area: about 7.000 km^2) within the German state of North-Rhine-Westphalia (NRW) is part of this emission hot spot. High emissions due to traffic and industrial activities make Benelux/Rhine-Ruhr a hot spot area with respect to air pollution in Europe.

Three major European metropolitan areas are located within Benelux/Rhine-Ruhr. The largest one is the urban agglomeration of Rhine-Ruhr itself with 12 million inhabitants living in an area of about 7.000 km^2, which has a megacity character with respect to population density, traffic, industry, and environmental issues. The main centre of European steel production and the biggest inland port in the world is located in Duisburg, one of the major cities in the Rhine-Ruhr area (major cities are: Cologne, Düsseldorf, Duisburg, Essen, Dortmund, Bochum). The Randstad is a conurbation in the Netherlands consisting of the four largest Dutch cities (Amsterdam, Rotterdam, The Hague, and Utrecht) with 6.7 million inhabitants in total, about 40% of the Netherlands, with Rotterdam as one of the most important sea harbours of the world. Within Belgium the Brussels-Antwerp region with 4 million inhabitants (about 40% of Belgium) again forms a conurbation with metropolitan character and an important sea harbour (Antwerp).

Ship traffic along the coast and toward the harbours, including ships going along the river Rhine, is an important source of air pollution to this area (see *Dalørsen et al*., 2009 for a general global picture of ship emissions). Population density weighted fine particle concentrations ($PM_{2.5}$) in general are the highest in Europe, in particular the contribution of ships has been found to be high in the Benelux/Rhine-Ruhr area according to recent modelling studies [*Andersson et al*., 2009]. CO, SO_2 and benzene are no longer a big issue in air quality within the Benelux/Rhine-Ruhr area due to the strong efforts for improvement of air quality in Europe during the last decades [*Vestreng et al*.,

2007; *Andersson et al.*, 2007]. Harmonized regulations within the European Community are aiming at further improvement of air quality in Europe (e.g. air quality directive 2008/50/EC and the emission ceiling directives). Major problems within urban agglomerations remain with respect to ozone, which can still exceed 300 µg/m^3 during summer conditions (e.g. in 2003 and 2006), to PM_{10}, which still exceeds the 24h average limit value of 50 µg/m^3 more than 35 times in a calendar year, and to NO_2 which still has a tendency to increase and exceeds the annual limit value of 40 µg/m^3 considerably, especially in street canyons.

Nearby metropolitan areas are London, Paris, and Frankfurt. Due to atmospheric transport processes, the air quality in the Benelux/Rhine-Ruhr area might be influenced from the transport of pollutants from Paris, London, and/or Frankfurt depending on atmospheric conditions. In particular, compounds formed during transport or with a long lifetime might contribute from outside to the air quality within Benelux/Rhine-Ruhr. On the other hand compounds emitted within Benelux/Rhine-Ruhr might contribute considerably to air pollution outside this hot spot due to long-lived primary constituents as well as secondary formed air pollutants such as aerosols or ozone. Therefore it is interesting to investigate the import/export budgets for air pollutants under different meteorological conditions relevant for the area considered.

Meteorological features in Central Europe are characterized by predominant westerly flow, and, therefore, quite often, clean air is transported with moderate or high wind speeds from the Atlantic towards the Benelux/Rhine-Ruhr area. Meteorological patterns leading to air quality problems often are related to high pressure systems over Central Europe. Anticyclone conditions, due to low wind speeds, favour the accumulation of primary emitted gases or particulate matter, in particular during winter or fall with limited vertical exchange (temperature inversions). Temperatures may reach 40°C during summer, as for example during the summer heat wave in August 2003 or in July 2006 summer smog conditions [*Vautard et al*., 2006]. Episodes with high particle concentrations during winter or fall are quite often correlated with easterly winds over Central Europe leading to transport of polluted air masses towards areas located west of Benelux/Rhine-Ruhr. This can cause additional impact from Benelux/Rhine-Ruhr to France and/or the UK. The quite high ship emissions in interaction with specific meteorological (or other) conditions in coastal areas (e.g. land-sea breeze, boundary layer height, deposition, sea spray) may lead to particular events with respect to air quality. Other large-scale factors, which might influence the air quality in Central Europe, include transport of dust from the Sahara desert [*Bruckmann et al*., 2008] and from the Ukraine by resuspended soil from dried-out farmlands [*Birmili et al*., 2008].

6.5.2 Overview of emission estimates

The Benelux/area is heavily burdened with air pollutants like ozone, NO_2, and PM_{10}. In addition, large amounts of ammonia, an important gaseous precursor for secondary particles formation, are emitted in areas of the Benelux/NRW regions that are characterized by agricultural use. Table 3a shows the emissions for the Benelux/NRW area for 2004 as a reference year. Emission density in the area is about twice as high as the German average, similar to the population density. Table 3b shows the emission trends based on EMEP data from 1980 till 2020. Figure 21 shows as an example the nitrogen oxide emissions for the region [*Thiruchittampalam et al*., 2008]. The metropolitan areas Randstadt (Amsterdam, Rotterdam), Brussels-Antwerp and Rhine-Ruhr are clearly visible as well as the strongly burdened cities of London and Paris. The relations between 1980 and 2004, upper part of Table 3b, show the considerable success of the efforts to decrease emissions in the Benelux area (and Europe) during the last decades. In particular sulphur emissions have been diminished considerably. Future emission abatement strategies are assumed to be successful with respect to NO_X, SO_X, CO, PM_{10} and NMVOC but not for NH_3, 95% of which is due to agricultural activities.

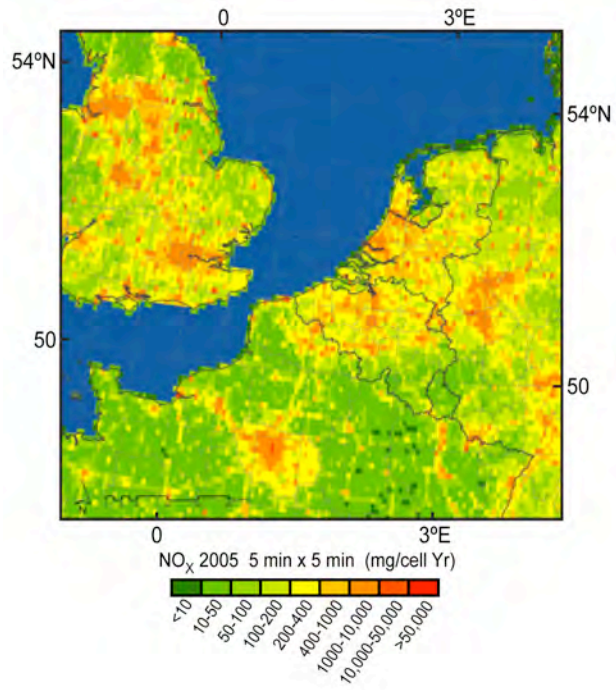

Figure 21 - NO_X emissions in Central Europe, including the Benelux-Rhine-Ruhr area. Ship emissions are not included on this scale, but they contribute considerably to emissions in the domain shown here [*Thiruchittampalam et al.*, 2008]

Table 3a - Area, population and annual emissions for the Benelux/NRW/Rhine-Ruhr area. Emission data for the Benelux and Germany are for the year 2004 and based on EMEP *(www.emep.int)*, expert emissions as used in EMEP models (Vestreng, 2006a; W-06emis04-V7 (2006-08-29), emission data for NRW are taken from the LANUV web site *(www.lanuv.nrw.de)*

Year: 2004	Area (km2)	Population (Mill.)	NO_X (kt as NO_2)	SO_2 (kt as SO_2)	CO (kt)	NMVOC (kt)	NH3 (kt)	PM_{10} (kt)
Netherlands	41.526	16,50	360,00	66,00	623,00	216,00	134,00	41,00
Belgium	30.528	10,70	298,00	154,00	972,00	167,00	74,00	62,00
Luxembourg	2.586	0,50	29,00	4,00	48,00	10,00	7,00	4,00
Benelux	**74.640**	**27,70**	**687,00**	**224,00**	**1.643,00**	**393,00**	**215,00**	**107,00**
NRW	34.084	18,00	418,00	167,00	1.506,00	273,00	73,00	32,00
Benelux/NRW	**108.724**	**45,70**	**1.105,00**	**391,00**	**3.149,00**	**666,00**	**288,00**	**139,00**
E/P-density (to/km2)		420,33	10,16	3,60	28,96	6,13	2,65	1,28
Germany	357.104	82,00	1.554,00	559,00	4.095,00	1.268,00	641,00	173,00
E/P-density (to/km2)		229,62	4,35	1,57	11,47	3,55	1,79	0,48
Benelux+NRW/GER		1,83	2,34	2,30	2,53	1,73	1,48	2,64

Table 3b - Past and future emissions in the Benelux area for the years 1980 and 2020. Data are taken from EMEP *(www.emep.int)* for 1980 for all pollutants. The CO emissions for 2020 based on expert estimates as used in EMEP models (Vestreng, 2004: 'W-04emis20-BL-V3' (2004-07-15)). The projected emissions for all other pollutants based on the 'NEC_6_Current legislation' scenario from the GAINS model *(http://www.iiasa.ac.at/rains/gains-online.html?sb=9* based on *Amann et al.,* 2008)

Year: 1980	NO_X (kt as NO2)	SO_X (kt as SO2)	CO (kt)	NMVOC (kt)	NH3 (kt)	PM_{10} (kt)
Netherlands	583,00	490,00	1.530,00	579,00	234,00	
Belgium	442,00	828,00	1.285,00	274,00	89,00	
Luxembourg	23,00	24,00	193,00	15,00	7,00	
Benelux	**1.048,00**	**1.342,00**	**3.008,00**	**868,00**	**330,00**	
1980/2004	**1,53**	**5,99**	**1,83**	**2,21**	**1,53**	
Year: 2020	**NO_X (kt as NO_2)**	**SO_X (kt as SO_2)**	**CO (kt)**	**NMVOC (kt)**	**NH3 (kt)**	**PM_{10} (kt)**
Netherlands	196,00	50,00	678,00	161,00	130,00	39,00
Belgium	158,00	88,00	286,00	127,00	77,00	48,00
Luxembourg	14,00	1,00	37,00	7,00	6,00	3,00
Benelux	**368,00**	**139,00**	**1.001,00**	**295,00**	**213,00**	**90,00**
2020/2004	0,54	0,62	0,61	0,75	0,99	0,84
1980/2020	2,85	9,66	3,00	2,94	1,55	

6.5.3 Overview of pollution levels

Air pollutants, in particular SO_2, NO_X, CO, ozone, and total suspended matter (TSP) have been measured by the local environmental agencies or responsible institutions for several decades. Due to the success of local and European abatement strategies sulphur and carbon monoxide no longer constitute a major problem in air pollution, in particular in western and Central Europe. Therefore the number of measurement sites has diminished considerably during the last years. On the other hand, measurement networks for PM_{10} and $PM_{2.5}$ have been established during the past 5 – 10 years.

During the last two decades the emissions of air pollutants decreased due to measures undertaken to reduce anthropogenic emissions. This in general leads to a decrease of pollutants concentrations in the region. For example the annual average of SO_2 in NRW decreased considerably from about 60 µg/m^3 in the beginning of the 1980s to about 10 µg/m^3 today; the annual average of NO_2 decreased from 50 µg/m^3 in the beginning of the 1980s to 30 µg/m^3 today for urban background stations, for measurement sites near streets, however, it remains between 40 and 50 µg/m^3 since 1989 with small variations only (see: www.lanuv.nrw.de/luft/immissionen/ber_trend/konti_trend_2008.pdf). Minor variations might be attributed to interannual variations due to meteorology. The decrease of SO_2 is directly related to the strong decrease of SO_2 emissions, whereas the decrease of NO_2-emissions is less pronounced. In particular the primary NO_2 emissions due to traffic have a tendency to increase even if the total NO_X emissions are decreasing. It is expected that the tendency of an increased relative amount of NO_2 in the total NO_X emissions in traffic emissions in the European Union will continue in the coming years from 12.4% in 2004 to 19.6% in 2010 and 32% in 2020 [*Grice et al.,* 2009]. Therefore it is expected that the annual limit values for NO_2 (40 µg/m^3) according to the air quality directive 2008/50/EC will not be fulfilled within the next years.

Fewer problems might occur with respect to the annual averages of PM_{10} (limit value of 40 µg/m^3, implmented in 2010), and $PM_{2.5}$ (limit value of 25 µg/m^3, to be fulfilled in 2015). However, with respect to PM_{10}, problems may arise in the number of exceedances of the daily average of 50 µg/m^3. The daily limit values should not be exceeded more than 35 times a calendar year according to the air quality directive 2008/50/EC. Measurement sites in the Benelux/Rhine-Ruhr area show, as for other sites in Europe, that the requirements of the air quality directive cannot be expected to be

fulfilled for the daily limit values set in 2010. According to model calculations the situation might be improved by 2015 (see Figure 22).

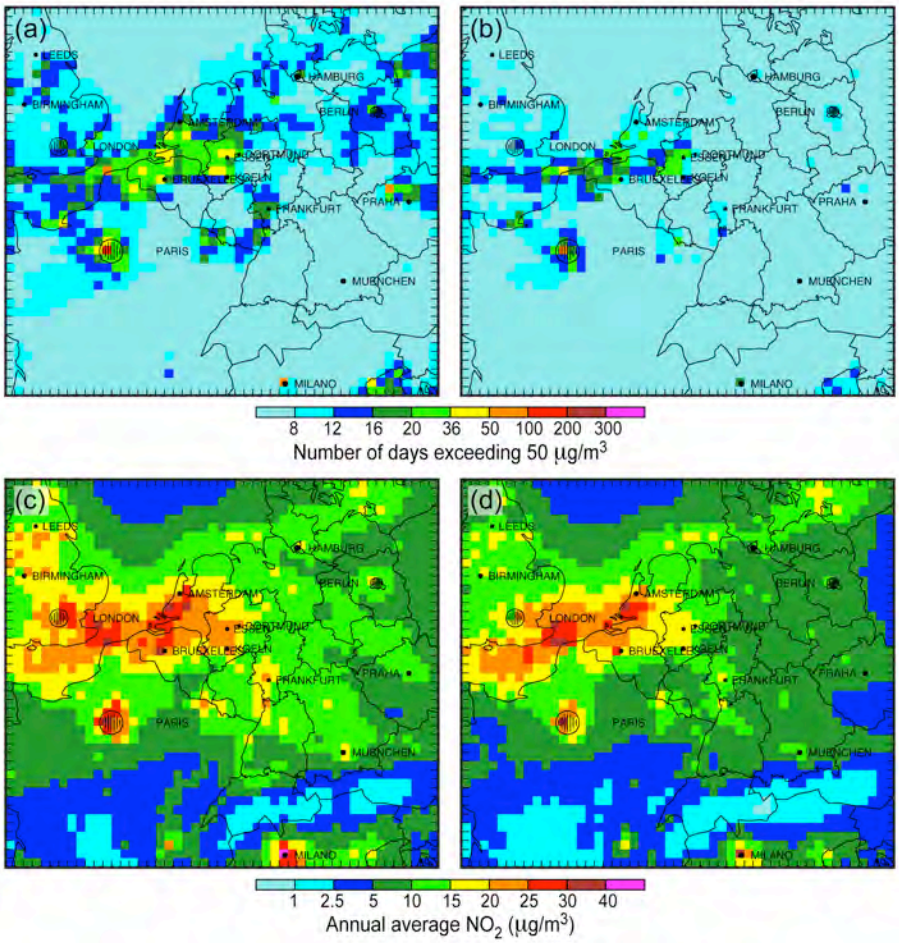

Figure 22 - (a) PM_{10}, number of days with daily average above 50 µg/m³ for the base case, reference year 2006 (b) The scenario 2015 with an emission projection for 2015 (CLE). Based on calculations with the EURAD-model [*Memmesheimer et al.*, 2009]. (c) NO_2, number of days with daily average above 50 µg/m³ for the base case, reference year 2006 (d) The scenario 2015 with an NO_2 emission projection for 2015 (CLE). Based on calculations with the EURAD-model [*Memmesheimer et al.*, 2009]

Annual averages of near surface ozone over Europe show an increasing trend on a decadal basis since the 1990s [*Vautard et al.*, 2006] whereas the number of episodes with high ozone values, e.g. with hourly values above the information or alert level (180 µg/m³, 240 µg/m³), show a tendency to decrease. The decrease of ozone precursors over Europe during the 1990s led to an ozone increase both during urban minima and urban maxima due to less titration (see also Figure 23). Especially during winter, with less photochemistry, one might expect less ozone in the urban areas for the '1990' simulation due to titration effects. However, in addition to the impact of regional precursor emissions, there are several processes which influence the average and high ozone concentrations as e.g. stratospheric intrusion events, stratospheric depletion of ozone, long-range transport, and global increase of ozone precursor emissions.

Health effects of air pollution have been discussed for the European scale within the CAFE programme [*Amann et al.*, 2005]. It was estimated that the losses in statistical life expectancy attributable to the exposure to anthropogenic $PM_{2.5}$ for the EU-25 is 8.1 months. It turned out that this value is considerably higher for Belgium (13.2 months) and the Netherlands (11.8 months), which show the highest value within the EU-25, in contrast to Finland with 2.6 months, which is the lowest value. All numbers are for the reference year 2000. It can be expected that the loss in life

expectancy in NRW, in particular the Rhine-Ruhr area according to the methodology used by the IIASA (Amann et al., 2005) is on the same order as for Belgium and the Netherlands. These findings emphasize the importance of the Benelux/Rhine-Ruhr area as one “hot spot” within Europe for air pollution and related problems, e.g. health. Furthermore it should be mentioned that the calculations performed by IIASA have been undertaken with a horizontal resolution of 50 km. The quite large horizontal gradients for particle mass concentrations from the urban agglomeration to the nearby mountain areas as the Eifel cannot be displayed with appropriate accuracy. Urban agglomeration show annual averages for PM_{10} of 35 – 40 µg/m^3 and the mountain areas show annual averages of about 15 – 20 µg/m^3. The concentration of air pollutants in inhabited areas might even be higher than calculated for a 50 km grid in most cases, which is due to neighbourhoods that have quite high primary emissions in inhabited areas, in particular during weak wind conditions. A coarser resolution might thus underestimate concentrations in inhabited areas and health effects. Local impacts, in particular due to streets, on the prevalence of coronary heart disease have been investigated within the framework of the Heinz Nixdorf RECALL study in the Ruhr area [*Hoffmann et al*., 2007].

6.5.4 Projections, modelling studies

Air quality modelling on regional and local scales is an important tool to investigate the impact of past and future changes in emissions as well as the effect of future climate changes on air quality. Regional modelling efforts have been undertaken during the last decade [*Forkel and Knoche*, 2007; *Cuvelier et al*., 2007; *Vautard et al*., 2005; 2006; 2007; *Stern et al*., 2008; *Memmesheimer et al*., 2004; 2006; 2009] that focus on regional air quality in Europe and on the local urban scale within the Citydelta Initiative [*Cuvelier et al*., 2007; *Vautard et al*., 2007] in major European Cities. Two examples for modelling of PM_{10}, NO_2 and ozone are given in Figures 22 and 23. Figure 22a shows the number of exceedances of the daily limit value of 50 µg/m^3 (as defined in the air quality directive 2008/50/EC) for a reference calculation in the year 2006 and a scenario calculation performed with an emission projection for 2015 [*Memmesheimer et al*., 2009]. The emission projection is based on data from IIASA that is available on the EMEP web site (www.emep.int). It can clearly be seen that a significant decrease of the number of exceedances can be expected by 2015 according to the model calculations. Exceedances of the daily limit value occur predominantly within episodes governed by high-pressure systems during winter and fall. Therefore the number of exceedances may vary from year to year depending on the frequency of meteorological situations that favour their occurrence (high pressure systems during winter and fall, weak winds, temperature inversions). Annual averages of NO_2 are displayed for the same cases, values in Central Europe stay below the yearly limit value of the air quality directive of 40 µg/m^3. However, it should be pointed out that measurements of sites in streets with high traffic density show annual average concentrations of NO_2 of 60 µg/m^3 or even more. And there is currently no trend for decreasing NO_2 concentrations in Europe because the NO_2/NO_X ratio for the emissions is expected to increase considerably [*Grice et al.,* 2009].

To illustrate the impact of emission reduction on average and high ozone values a scenario calculation for the August 2003 heat wave has been calculated using the same meteorological fields (for July and August 2003) as generated by the MM5 model but applying the emissions for 1990 instead of the actual emissions for 2003. In central Europe, anthropogenic NO_X and VOC emissions for 2003 are about 60% of those in 1990. Figure 23a shows average concentrations for ozone as calculated for the episode from July 10 - August 15, 2003. The average ozone concentrations within the Rhine-Ruhr urban agglomeration area show only minor changes between the base case (2003 with 2003 emissions) and the scenario calculation (2003 with 1990 emissions). Outside the urban areas the average ozone concentrations for the scenario case are about 10 µg/m^3 higher as in the base case. For the hourly maxima that are shown in Figure 23b the highest changes occur near the Ruhr area. The hourly maxima increase by about 60 µg/m^3 for the scenario case with emissions from 1990. *Vautard et al*. [2005] found, for the European scale, a significant decrease applying an emission projection for 2010 to the August 2003 heat wave for the information threshold (180 µg/m^3) as well as for the AOT60 index, which is considered to be a relevant health impact index (integral of positive departures from 60 ppbv during 6 – 18 local time, i.e. daytime).

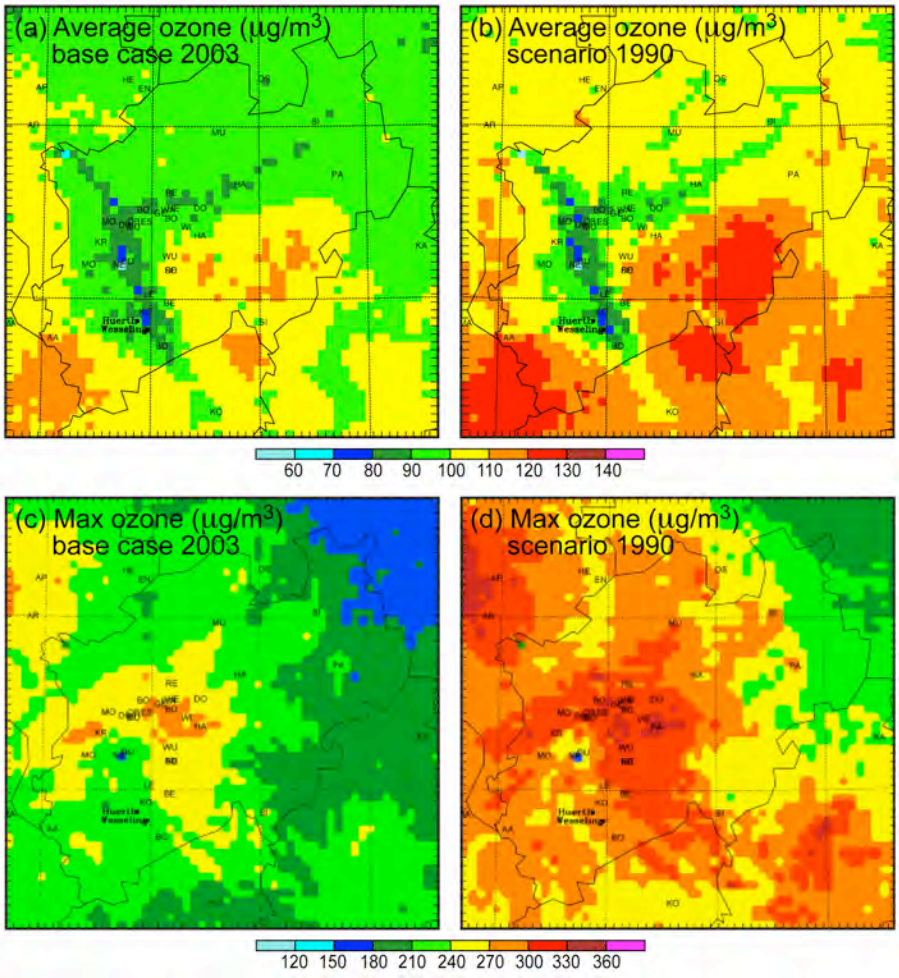

Figure 23 - (a) Ozone (µg/m³) in 2003 (left) vs. (b) 1990 (right); averaged over the episode from July 10 – August 15, 2003. Based on model calculations for North-Rhine-Westphalia. (c) Ozone, 2003 (left) vs. (d) 1990 (right), hourly maximum (µg/m³), based on model calculations for North-Rhine-Westphalia

6.5.5 Effects of climate change on air pollution

Air quality depends on meteorological parameters and land use characteristics. Therefore it is clearly influenced by climatic changes and shows strong interannual variations [*Forkel et al*., 2007; *Andersson et al*., 2007]. On the other hand, changes in air quality by reduction of anthropogenic emissions can lead to changes in radiation, fog and in regional temperature trends as has been shown recently by *Vautard et al*. [2009].

Forkel et al. [2007] performed calculations with a nested regional climate-chemistry model (MCCM) with a horizontal resolution of 20 km for central Europe. Two time slices of about 10 years representing present-day (1990s) and future climate conditions (2030s) were run. Anthropogenic emissions and outer domain boundary conditions were not changed. The model results show higher temperatures and decreased cloud cover and, in particular, increased emissions of isoprene lead to enhanced ozone formation in the troposphere. Daily maxima increase in the range of 2 – 10 ppb depending on the region. Particulate matter has not been considered in this study.

Regional climate-chemistry calculations with nested models, for several decades and with a grid resolution of 1 – 5 km, have not been undertaken until now for the Benelux/Rhine-Ruhr region. The development, application, and validation of these models are still a challenge for modelling activities as discussed, for example, in the recent findings by *Vautard et al*. [2009]. Interactions of air pollution and radiation, or even fog and clouds, have not been considered on this scale for the

time range of several decades. On an episodic basis the impact of particulate matter on radiation and temperatures have been estimated for local scale applications in southern Germany (state of Baden-Württemberg) by *Riemer et al*. [2003] and *Bäumer et al*. [2008].

Vautard et al. [2009] have analysed multidecadal data of horizontal visibility, and found that the frequency of low-visibility conditions, such as fog and mist, has declined in Europe over the past 30 years. They found that this decline is spatially and temporally correlated with trends in sulphur dioxide emissions and suggest a significant contribution of air-quality improvements. Further, using statistical methods to link local visibility changes with temperature variations they estimated a contribution of 10-20% of Europe's recent daytime warming due to low-visibility conditions, and about 50% in Eastern Europe. *Vautard et al*. [2009] discussed that their approach was of statistical nature, and they assumed a causal relationship through radiation processes. However, they emphasized that for a quantitative understanding of the processes a regional modelling approach is required. It is stated that this might be difficult to achieve with the current state-of-the-art regional models and therefore further development will be a challenge for models.

Long-range transport effects might influence the Central Europe region and might also change due to climate change connected at the global scale. There have been two interesting episodes in the recent years leading to high concentrations of atmospheric particles in Central Europe, including the Benelux/Rhine-Ruhr area. *Birmili et al*. [2008] discuss in detail one event, during which high concentrations of PM_{10} between 200 and 1400 $\mu g/m^3$ were measured over Central Europe during (24 March, 2007). Based on analysis of measured data, southern Ukraine could be identified as the source region of the plume. Due to the meteorological situation soil dust could be activated by high wind speeds and transported to Central Europe. *Birmili et al*. [2008] point out that such an event might be an infrequent phenomenon that probably does not occur more often than once in ten years. However, they emphasized that such events might become more frequent in the future due to ongoing anthropogenic desertification processes. Again the quantitative investigation of the atmospheric processes involved in such an event is a challenge for further development and application atmospheric models.

A further episode with high PM_{10} concentrations occurred end of May 2008 [*Bruckmann et al*., 2008]. Analyses of observational data and meteorological circulation patterns show clearly that the origin of the PM_{10}-event can be attributed to the Sahara Desert (Sahara dust event). The daily average of PM_{10}-concentrations on 29 May and 30 May exceeded the European PM_{10} daily limit value of 50 $\mu g/m^3$. As an interesting feature it could be observed that the $PM_{2.5}/PM_{10}$ ratio changed from quite small values of 0.3 – 0.4 in the Alpine region to 0.7 – 0.8 in the Benelux/Rhine-Ruhr area. Measurements of the chemical composition show an increase of secondary formed ammonium nitrate and ammonium sulphate in the Benelux/Rhine-Ruhr area compared to the Alpine region and southern Germany. Secondary particulates therefore seem to be generated from anthropogenic precursor emissions as NO_X, NH_3 and SO_2 during the transport from the Alps towards the Benelux/Rhine-Ruhr area. Part of the mineral dust material seems to be replaced by the secondary formed compounds.

6.5.6 Major past studies or field campaigns examining the city's air pollution

Coordinated studies for the whole Benelux/Rhine-Ruhr area are hampered by political boundaries. Some specific measurements for the Rhine-Ruhr area concerning atmospheric particulate matter have been documented in *Kuhlbusch et al*. [2004]. The EU project CityZen collected measurement data used for evaluation models on different scales from 2008-2011.

6.6 PO VALLEY

6.6.1 Introduction and specific features of the city

The Po valley is neither a city nor an administrative unit. In can be identified by a geographic and pedological point of view as the floodplain enclosed between the Alps on the Western and Northern side, the Apennines chain on the Southern and the Adriatic Sea on the Eastern side. It is roughly located around 45° N latitude and between 7°'30' and 12° 30' E longitude.

The Po River Basin includes six administrative regions, its plains account for 2957 municipal units for a total population of about 20 million people and an average population density of 414 inhabitants/km^2 (Figure 24). The Basin accounts for nearly 50% of Italy's GDP. It is home to 37% of the country's industry, about 55% of livestock, and 35% of the country's agricultural production. The Po valley is therefore exposed to substantial emission loads. The population distribution and its density (persons/km^2) are illustrated in Figure 25 on the basis of municipal land units. The major urban conglomeration of Milan its clearly identified in central-northern part of the plain, while Turin urban area can be recognized towards its western edge and the largely urbanized zones of the Veneto plains and southern Emilia can be noticed near its eastern and southern limits. The population density is clearly correlated with the urbanisation represented in Figure 26 by urban land cover percentage on a 1 km^2 grid resolution. The core of Milan urban area, roughly coincident with its province, accounts for 3.7 millions inhabitants [see e.g. http://www.citypopulation.de/], while the city commuting area has been evaluated to include around 7 million people [*OECD*, 2006]. Turin metropolitan area accounts for more than 1.5 million people, while many cities counting more than 100,000 inhabitants are scattered throughout the plains. During the last decades, the urbanisation of the region surrounding the major cities has increased by the re-settlement of part of the population from the city core to nearby areas.

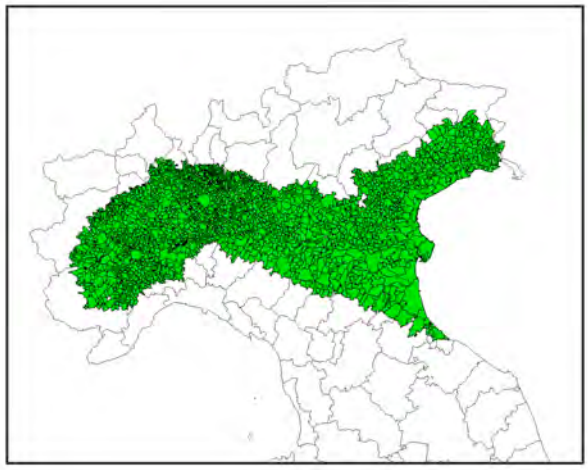

Figure 24 - Po valley municipal units identified through pedological features

Urban agglomerations located within the Po Valley basin suffer air quality conditions worse than those experienced by other European cities like Paris and London. This is mainly due to the concentration of urban and industrial emissions and to the adverse meteorological conditions that often affect the region due to its peculiar topographic and geographic features.

The atmospheric circulation of the Po valley is characterised by the strong modification of synoptic flow due to the high mountains (Alps and Apennines chains) that surround the valley on three sides. The local atmospheric circulation features, dominated by calms and weak winds, favour the development of critical pollution episodes. Meteorological conditions causing winter air pollution episodes are analyzed in *Kukkonen et al*. [2005].

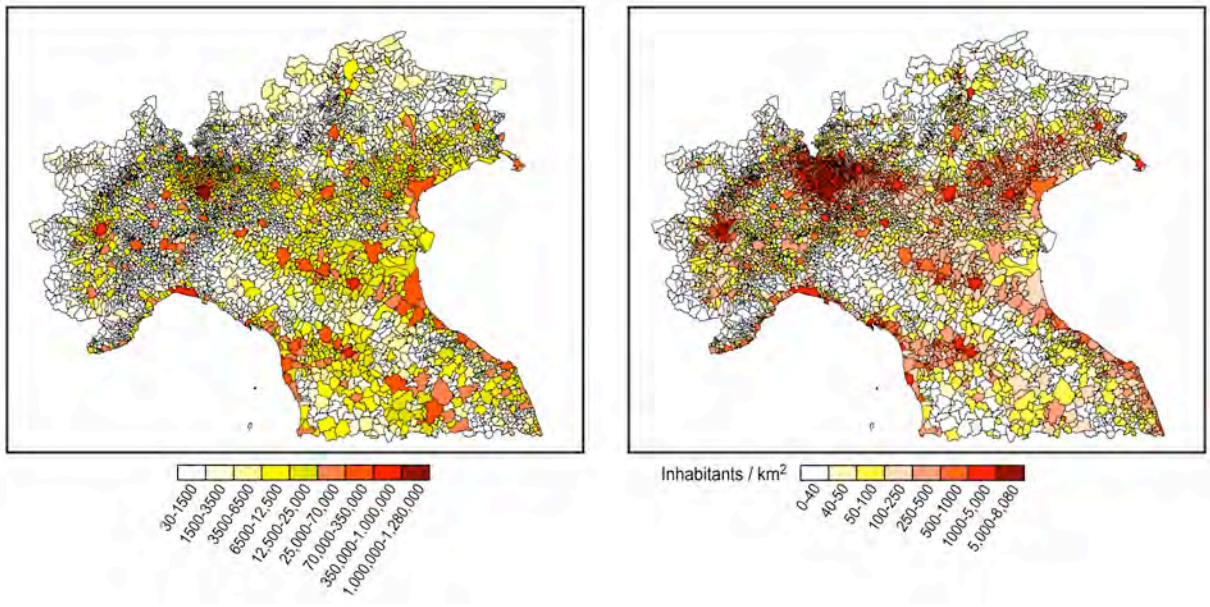

Figure 25 - Inhabitants (left) and population density (right) of the Po valley region. Elaboration on data published by the Italian Institute for Statistics [*www.istat.it*]

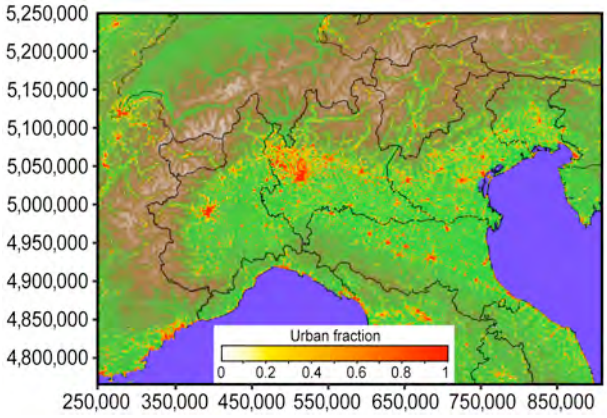

Figure 26 - Po valley region topography and urban density (percentage of urban land cover elaborated from CORINE LandCover 2000)

6.6.2 Emission sources of air pollutants

From an emission point of view, Lombardy Region represents the most important area in the Po Valley basin. About 20% of Italian industry is located here. In particular, the Milan Metropolitan Area is located in the Lomdardy region and accounts for about 8% of the national total of industrial emissions, especially due to the presence of specialized industry districts (textile, wood sector, metal, etc.) (http://sitis.istat.it/sitis/html/index.htm). The other large source of air pollution in this region is road transport due to the large number of vehicles (an average of 59 cars for every 100 inhabitants; estimated by Osservatorio Autopromotec) and the presence of major roads that give rise to important traffic volumes.

Table 4 shows emission from different source sectors in the Lombardy Region. The energy production sector is the dominant source for SO_2 emissions, while residential rombustion produces about 1/3 of CO and PM_{10} emissions. The industrial sector (industry combustion and production processes) is a significant contributor to SO_2 emissions (due to gas oil and residual oil use in industrial boilers) and is the second source, after road transport, of NO_X emissions. The extraction

of fossil fuels and waste treatment sectors are important quite exclusively for CH_4 emissions, accounting for approximately1/4 of total emissions. The solvent use sector (in particular painting, chemical products' synthesis and manufacturing, printing industry, and domestic use of solvents) produce only volatile organic compounds. About half of CO and NO_X emissions are due to the road transport sector, which also produce 1/3 of regional PM_{10} emissions, with different contribution given by urban and non-urban traffic according depending on the pollutant.

Table 4 - Annual emissions (t/year) of Lombardy Region

	SO2	NOX	COV	CH4	CO	N2O	NH3	PM10
Energy Production	12794	14736	1496	952	3509	223	12	551
Comb in Residential	4074	18905	34204	9530	138557	1831	242	6443
Comb.in Industry	8062	27113	4571	1828	13738	740	132	1343
Prod. Processes	3798	3220	13152	125	23753	44	101	1005
Extraction Fossil Fuels	0	0	9179	103611	0	0	0	0
Solvent Use	0	1	115676	0	0	0	6	74
Road Transport	526	89207	59710	2717	201743	667	2575	6790
Other Transport & Mobile Machinery	540	24941	5664	102	17430	711	3	3121
Waste Treatment	199	5290	284	108329	8917	375	166	107
Agriculture Forestry & Land Use Change	92	1184	566	206925	4828	10642	95417	1304
Nature	23	102	28336	4819	3725	1	23	781
TOTAL	30109	184700	272838	438939	416200	15234	98678	21521

If the attention is only focused on the Milan Metropolitan Area (here defined as a group of 188 municipalities that form one of the administrative sub divisions of a region called "province") it is possible to stress some differences in the contribution of different sectors. Table 5 shows annual emissions of Lombardy Region and of the Milan Metropolitan Area. The Milan contribution to SO_2, PM_{10}, N_2O, and CH_4 emissions is between 15 and 20%, whereas for NO_X, VOC, and CO it is between 25 and 30%. However, the Milan Metropolitan area accounts for a small amount of NH_3 emissions (7%) due to the low presence of agricultural activities in the area surrounding Milan.

Table 5 - Annual emissions (t/year) of Lombardy Region

	SO2	NOX	COV	CH4	CO	N2O	NH3	PM10
Lombardy Region	30109	184700	272838	438939	416200	15234	98678	21521
Milan Metropolitan Area	5147	50054	85404	72070	98433	2073	7161	4154

Figure 27 compares sectorial contributions to total emissions in Lombardy Region and Milan Metropolitan Area. Milan has a lower percentage of emissions from energy production due to the fact that large power plants are located elsewhere in the Lombardy Region. As already observed, the contribution from agricultural activities is higher in the whole Region than in Milan area. On the contrary, the very high fraction of urbanized area and the presence of very busy roads make the traffic contribution higher in Milan than in the Lombardy Region.

6.6.3 Data available on air pollutants

The air quality monitoring network in the Lombardy Region is managed by Regional Environmental Protection Agencies (ARPA), which provides information, data, and air quality reports to the public with different methods and approaches. The air pollutants routinely measured include SO_2, NO_2, CO, PM10, PM2.5, O_3, and C_6H_6. Other chemical species as well as chemical composition and size distribution of particulate matter is available for time limited field campaigns. Monitoring network description, recent data, archives, and air quality reports for the whole region and main urban areas can be accessed on the ARPAs or municipalities web sites (all in Italian language):

Piemonte (Turin): http://www.sistemapiemonte.it/ambiente/srqa/
Lombardia (Milan): http://www.arpalombardia.it/qaria
Veneto (Venice): http://www.arpa.veneto.it/aria_new/htm/qualita_aria.asp
Emilia-Romagna (Bologna): http://www.arpa.emr.it/liberiamo/
Friuli Venezia-Giulia (Udine): http://www.arpa.fvg.it/index.php?id=112

Air quality data can also be downloaded from the National Environmental Protection Agency (ISPRA) central database BRACE (http://www.brace.sinanet.apat.it), where data are usually loaded with a 1-2 years delay.

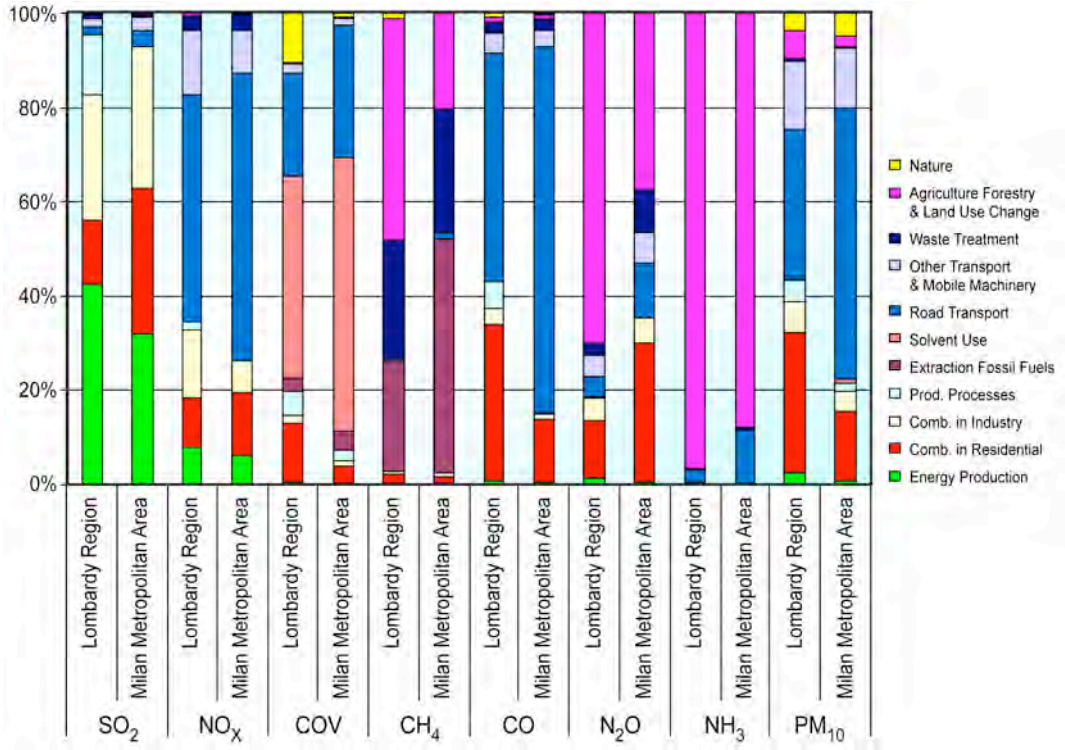

Figure 27 - Comparison between sectors' contribution to total emissions in Lombardy Region and Milan Metropolitan Area

6.6.4 The status and trend of air pollution

The main cities within the Po valley experience similar air quality problems with frequent exceedances of EC directives' limit values for PM_{10} during the winter and for ozone during the summer. Non-attainment conditions are recorded for yearly averaged values of PM_{10} and NO_2. Examples of the air quality conditions experienced during the past decade are shown in Figures 28-31. The yearly averages of NO_2 have values equal to or larger than 50 µg/m^3 with a very weak declining trend (Figure 28) at some stations. PM_{10} concentrations are characterized by yearly average concentrations above the limit threshold of 40 µg/m^3 (Figure 29) and a number of exceedances of the daily average limit, often getting above 100 µg/m^3 (Figure 30). Stations located within urban parks (Parco Lambro) and around the urbanized area (Motta Visconti e Lacchiarella) show a large number of exceedances of the limit values for the ozone daily maximum of the 8 hours running mean (Figure 31). Only stations located within the urban area (Juvara/Pascal) respect the maximum number of exceedances foreseen by EC directives during favourable years (2004-2006). The year 2003 summer heat wave caused much higher O_3 concentrations than all the following years.

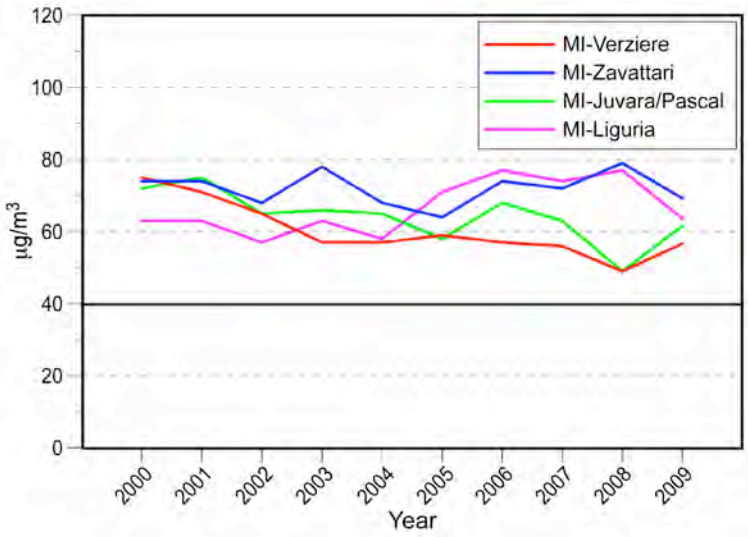

Figure 28 - Yearly average NO_2 concentrations in selected stations of Milan urban area. Solid line indicates the limit value stated by the European Legislation

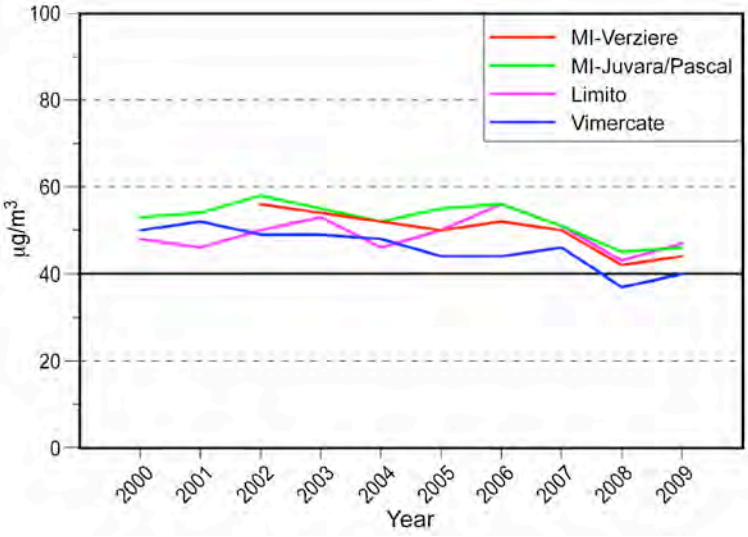

Figure 29 - Yearly average PM_{10} concentrations in selected stations of Milan urban area. Solid line indicates the limit value stated by the European Legislation

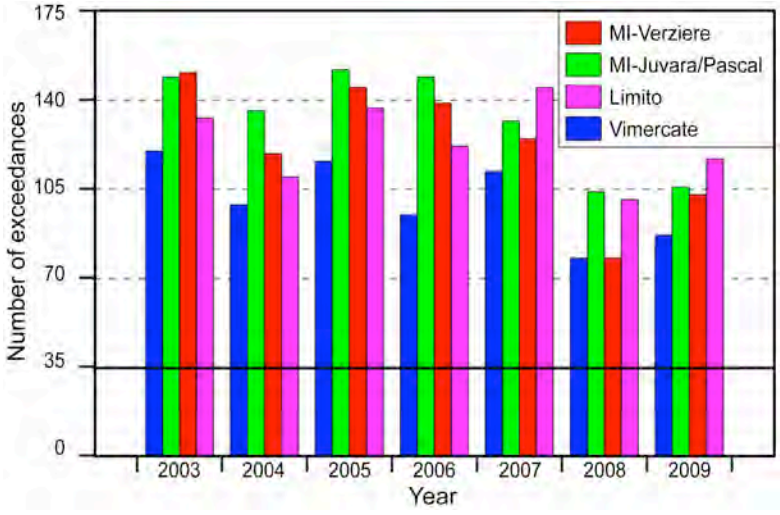

Figure 30 - Number of exceedances of PM_{10} daily average concentration limit in selected stations of Milan urban area. Solid line indicates the limit value stated by the European Legislation

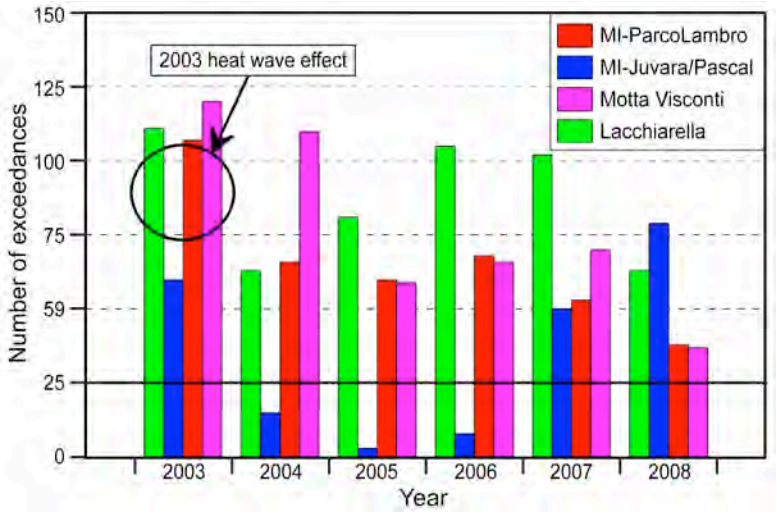

Figure 31 - Number of exceedances of O_3 daily maximum limit for 8 hours running average concentration in selected stations of Milan urban area. Solid line indicates the limit value stated by the European Legislation

Long-term trends of the major pollutants are described in Figure 32-33 for the station having the longest air quality record in Milan (Juvara/Pascal). It has to be mentioned that this station was relocated during 2007 to a nearby site with similar environmental features. The dramatic decrease of the yearly average SO_2 concentrations is the most evident concentration trend, declining from about 100 µg/m^3 to values stably below 10 µg/m^3 over 20 years (Figure 32). The extent of the observed reduction clearly dominates over the inter-annual variability and can be ascribed to the change of fuel for house heating and transport as well as to the closure and/or relocation of large factories that were located in Milan suburbs. The NO_2 trend shows an increase during the eighties reaching values over 100 µg/m^3 followed by a decrease of yearly average concentrations until values reached around 60 µg/m^3 in 2005. Positive effects on air quality due to changes in fuels, the improvement of industrial technologies, and changes in domestic heating have been less effective for NO_2 than for SO_2. In addition, much of the improvement is likely counterbalanced by the growth in traffic volume. The decrease of NO_2 from 1994-95 on is mostly likely due to the introduction of catalytic converters. Ozone yearly average concentrations increased until the end of the 20th century, showing a slight decrease after the 2003 peak and a further increase during recent years. This behaviour can be tentatively explained, within urban areas, by the decrease of NO_X concentrations and the variation of the NO_X/VOC ratio, superposed to the natural variation of the meteorological forcing.

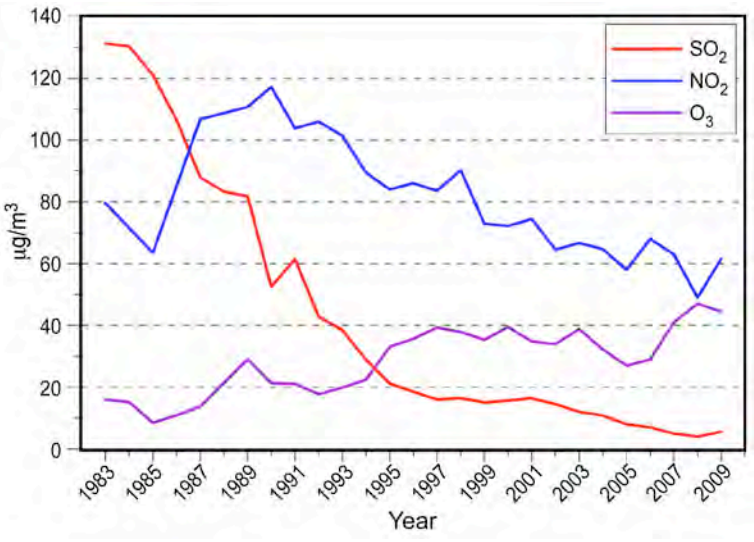

Figure 32 - Yearly average concentrations of SO_2, NO_2 and O_3 measured at the urban background station of Milano Juvara/Pascal

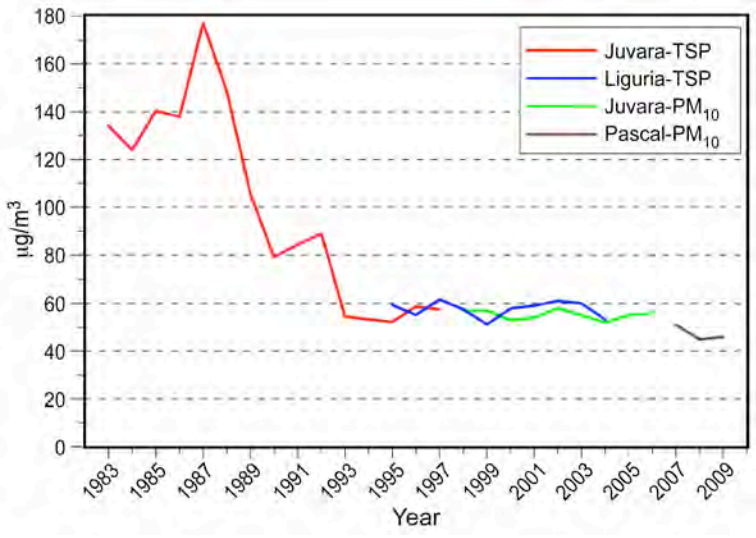

Figure 33 - Yearly average concentrations of TSP and PM_{10} measured at the stations of Milano Juvara, Liguria and Pascal

Total Suspended Particulate (TSP) concentrations increased from 1983 to 1987 followed by a strong reduction from the late eighties to mid-nineties, and have more or less remained stable since (Figure 33). The trend in PM concentration is likely due to previously discussed air pollution reduction measures since PM is emitted by the main pollution sources, but is also produced by chemical transformation of other pollutants (e.g. SO_2 and NO_2) and by car traffic as a non-exhaust product.

The air quality data presented here are from measurements from the Lombardia Region air quality network and were made available by the Lombardia Region Environmental Protection Agency (http://www.arpalombardia.it), see e.g. *Angius et al*. [2009].

6.6.5 Relationships between trends regulations and practices

The administrative regions of the Po Valley are negotiating with the EC about their non attainment conditions in regard to PM_{10} limits imposed by the European Directive 2008/50/EC. Similar difficulties are expected to be faced for NO_2 yearly average limits that are generally exceeded within the major urban conglomerations.

Different mitigation actions have been attempted during the last decade. These mitigation efforts include amongst other: traffic bans during the weekends in case of air pollution episodes (all the main cities), partial traffic ban based on odd-even car plate number (Turin urban area), winter circulation ban for obsolete (EURO 0) vehicles (Lombardy critical areas), and the ECOPASS (a pollution charge experiment in Milan depending on the European vehicle classification). The mitigation efforts, generally managed and applied locally, thus far are not able to produce a clearly detectable reduction in air pollution trends.

During recent years, awareness is growing that air pollution within the Po valley is a basin problem that cannot be fixed by local measures alone that are decided and managed independently. This consciousness carried the Po valley Italian Regions (Emilia-Romagna, Lombardia, Piemonte, Veneto, Trento and Bolzano autonomous Provinces) and the Swiss Canton Ticino to sign in February 2007 an agreement to coordinated action to reduce air pollution over the whole Po Valley. Future common measures are expected to include: public transportation support, progressive introduction of limitation (in 2010) to the circulation of obsolete polluting vehicles, obligatory particulate matter filter for diesel cars, financial support for car park renewal, ban on dense oil for house heating, and enforcement of stricter limits for wood, biomass fired plants, and house heating devices. Negative contributions to PM air pollution increased due to the increase in the number of diesel vehicles that characterize the Italian car market from the late nineties and by the growing trend to move from gas to wood fired stoves for house heating due to the rise of oil and gas prices.

As for the development and implementation of technical instruments, Regions Piemonte, Emilia Romagna, Friuli Venezia Giulia, Veneto, Puglia, Trento and Bolzano autonomous Provinces are collaborating with Regione Lombardia to develop high-resolution emission inventories with the same methodology originally developed by *Caserini et al*. [2004]. Many of the Po Valley ARPAs implemented air quality modelling systems built around chemical transport models (CTM) to perform yearly air quality assessment, air quality forecast, and near real time air quality analyses [see e.g. *Finardi et al*., 2008; *Silibello et al*., 2008; *Stortini et al*., 2007).

6.6.6 Climatic change issues

Milan city [*Tebaldi et al*., 2007] and the Po Valley [*Brunetti et al*., 2006a; 2006b] data show a climatic trend in general agreement with mid latitude European trends. From 1970 a clear increase in temperatures has been observed, with milder winters, warmer springs, hotter and more humid summers, and with a wider seasonal variability. The precipitation regime shows differences among the areas located around the Alps. During the last years, in Milan, it shows stationary behaviour during the winter, a reduction of spring and autumn rains, and a growth of summer precipitation intensity.

Mild winters may have contributed to fewer occurrences of temperature inversion episodes that cause severe air pollution episodes [*Finardi and Pellegrini*, 2004; *Kukkonen et al*., 2005] and to a reduction in the number of NO_2 exceedances of the hourly average limit of 200 µg/m^3, as observed during the last years. The climatic tendency does not show reducing effects on winter PM concentrations. Summer heat waves cause prolonged ozone episodes, as confirmed by both average and peak ozone values recorded during summer 2003.

6.6.7 Major past studies or field campaigns examining the city's air pollution

The Po Valley is probably one of the areas of Europe where more research activities have been focused during the last decades. Several of these research activities are the PIPAPO campaign [*Baltensperger et al*., 2002] for the characterization of ozone episodes, the PARFIL project and campaigns (http://www.disat.unimib.it/chimamb/parfil.htm) to investigate formation and distribution of PM within critical areas of Lombardy Region, and the City Delta project [http://aqm.jrc.ec.europa.eu/citydelta/, *Vautard et al*., 2007] for inter-comparison of model responses to urban-scale emission-reduction scenarios.

6.6.8 Cutting edge research

Ongoing relevant projects include POMI Po-Valley Modelling Intercomparison Exercise (http://aqm.jrc.ec.europa.eu/POMI/) and the EC 7th Framework Programme projects MEGAPOLI (http://megapoli.info/) and CITYZEN (https://wiki.met.no/cityzen/start), both of which are considering the Po Valley as a target area to analyse the effects on air quality and climate at regional and global scales due to pollutants emission from megacities and pollution hot spots areas.

6.6.9 Problems remaining

Despite the large number of scientific investigations that targeted the Po Valley region, the processes driving formation and accumulation of PM within the basin are still far from being completely understood. The region remains one of the EU pollution hotspots recording a large number of exceedances of EC directives air quality limits causing severe concerns on the impacts on human health. Despite emission inventories estimates that cite a measurable reduction of PM emissions, PM_{10} ambient concentrations did not show a decreasing trend during the last decade (1998-2008). The importance of secondary aerosol formation and the uncertain relationship between local PM emission and background atmospheric concentrations make mitigation actions difficult to determine and to evaluate. Moreover, many modelling studies employing different CTMs over the Po Valley basin showed difficulties in reproducing winter PM_{10} peak concentrations and accumulation phenomena.

6.7 EASTERN MEDITERRANEAN AND ISTANBUL MEGACITY

6.7.1 Introduction – location, meteorological patterns and air pollution levels

6.7.1.1 Meteorological patterns

The Mediterranean region consists of a landmass surrounding a body of saline water, the Mediterranean sea, which does not exchange very rapidly with the rest of the oceans. The typical Mediterranean climate is characterized by hot, dry summers and mild, rainy winters. Evaporation greatly exceeds precipitation and river runoff in this region, a fact that is central to the water circulation within the basin.

The Mediterranean, and particularly its East basin, is at a crossroad of air masses coming from Europe, Asia, and Africa. At this crossroad, anthropogenic emissions, mainly from Europe, Balkans, and the Black sea, meet with natural emissions from the Saharan desert, vegetation, and the Mediterranean Sea as well as from biomass burning, which presents a strong seasonal pattern. The transport of anthropogenic pollutants from North America also exerts a significant influence in the free troposphere [*Lelieveld et al*., 2002]. As a consequence of its unique location and emissions, the Mediterranean region is a climatically sensitive region, often exposed to multiple stresses, such as a simultaneous water shortage and air pollution exposure [*IPCC*, 2007]. Pollution in this region has been extremely high in the last couple of years. Pollution episodes are favoured due to the Mediterranean climate and the likely growth in the future emissions due to rapid urbanization of the region. The meteorology and chemistry of the Mediterranean lead to two “distinct” but interlinked regions, the western and the eastern basins.

Recirculation and regional patterns are more important in the western Mediterranean, which receives more precipitation, is drier, and has more oligotrophic seawater than the eastern basin. Evaporation is especially high in its eastern half, causing the sea water level to decrease and salinity to increase eastward. In contrast to Central and Northern Europe, in the Southern Europe/Mediterranean region, photochemical episodes can also occur since at these latitudes solar radiation is still important for photochemical reactions that favour air pollution.

The Eastern basin of the Mediterranean and the surrounding regions include several megacities such as the Istanbul (>12 million inhabitants, Turkey) and Cairo (~16 million, Egypt) and several large urban centres such as Athens (>4 million) and Thessaloniki (>1 million) in Greece, Izmir (4 million) and Adana (>4 million) in Turkey, Amman (>2 million, Jordan), Beirut (~2 million, Lebanon), Damascus (6 million, Syria), and to the south Alexandria (4 million, Egypt). The regional coverage includes rural (inland Greek and Anatolian peninsulas), maritime (Crete island), and desert (Anatolian plateau, north Africa, Middle East) conditions. Section 6.7.2 is dedicated to the megacity of Istanbul in the East Mediterranean region whereas in Section 6.7.3 we present some characteristics of the Athens extended area as another example of rapidly expanding urban agglomeration. Relevant information for Cairo megacity situated at the South edge of the basin is provided in the Chapter on African megacities.

During the last decades the Mediterranean, following the general trend, has experienced a rapid growth in urbanization, vehicle use, and industrialization, which is reflected in pollutant emissions to the atmosphere. Air pollution is one of the challenging environmental problems in Istanbul and Cairo megacities but also for the whole East Mediterranean region. Ozone and aerosol air quality limits are often exceeded over the entire Mediterranean in particular during the summer months. High ozone and aerosol concentrations are harmful to human health and ecosystems, and they also cause agricultural crop loss and contribute to climate change. The contribution of natural emissions to these exceedances seems significant and remains to be determined.

6.7.1.2 Air pollution in the East Mediterranean

Ozone and its precursors - The Mediterranean, located at the boundary between the tropical and mid-latitudes, is subject to large (about 50%) changes in the total O_3 column [*Ladstaetter-Weissenmayer et al*., 2007] that have been attributed to changes in the location of the

sub-tropical front [*Hudson et al*., 2003]. In summer, the total observed variability in tropospheric O_3 is about 5-10% of the total O_3 column, that is 25 DU [*Ladstaetter-Weissenmayer et al*., 2007]. Because Mediterranean background O_3 levels are high, particularly in the spring and summer, it is difficult to control ozone in urban and industrial areas. This is especially true because the background ozone levels are controlled by meteorological conditions, large-scale atmospheric dynamics, long-range transport, and photochemical formation. Experimental studies over the eastern Mediterranean [*Kourtidis et al.,* 2002*; Kouvarakis et al*., 2000] have demonstrated that transport from the European continent is the main mechanism controlling ozone levels in the eastern Mediterranean, especially in summer (or spring depending on the prevailing air transport patterns) when ozone presents a maximum of about 60±10 ppbv [*Gerasopoulos et al*., 2005]. *Kalabokas et al*. [2007] found that during the summer, high tropospheric ozone values in the eastern basin were confined in the low troposphere whereas in the middle troposphere O_3 was only 5–10% higher than over Central Europe. Enhanced levels of NO_2 pollution over the last decade can be detected by satellites (NO_2, Figure 34, O_3) over East Mediterranean (Istanbul, Athens and Izmir) and over the Middle East, in particular around the main ports of the Persian Gulf, around the Red Sea port of Jedda near Mecca, and around the cities of Riyadh, Cairo, and Tehran [*Van Noije et al*., 2006, *Lelieveld et al*., 2008; *Vrekoussis et al*., 2009].

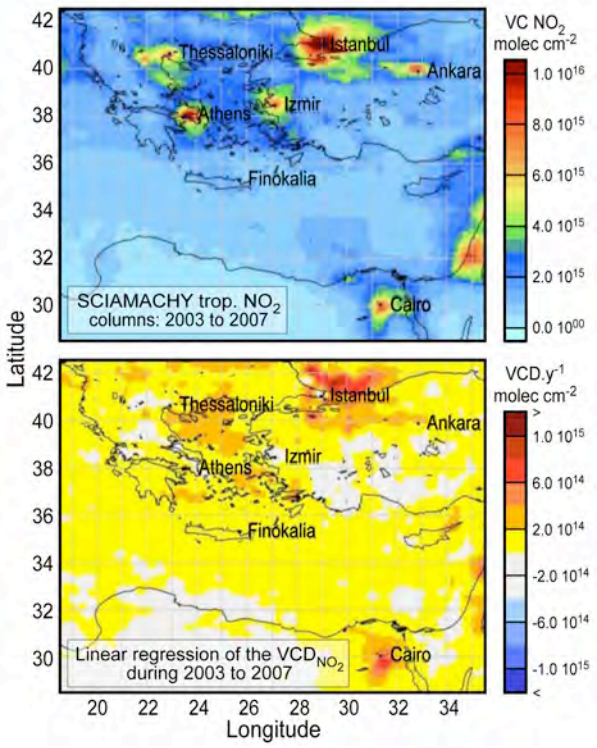

Figure 34 - (left) SCIAMACHY Vertical Column Densities (VCDs) of NO_2 over the eastern Mediterranean basin gridded to 0.125x0.125°. (right) Annual mean changes in the vertical tropospheric column density (VCD) of NO_2 over the East Mediterranean from 2003 to 2007 in molecules $NO_2 \cdot cm^{-2} \cdot y^{-1}$ as derived from SCIAMACHY observations [*Vrekoussis et al*., 2009; © *IUP*, University of Bremen]

Airborne particulate matter - The Mediterranean is one of the areas with the highest aerosol optical depth (AOD) in the world, which can be seen by satellites [*Hatziannastassiou et al*., 2009]. Observations over the area show high concentrations of aerosols, both PM_{10} and $PM_{2.5}$ [*Querol et al*., 2009]. Chemistry-transport models successfully simulate the occurrence of high loadings of aerosols over remote locations in the Mediterranean [*Gangoiti et al*., 2006; *Kallos et al*., 2007; *Kanakidou et al*., 2007]. In the Mediterranean, PM_{10} has a similar seasonal behaviour than $PM_{2.5}$, which is marked by dust emission and transport, particularly in spring and fall in the eastern basin and in February-March and late spring-summer in the western basin (especially at regional background sites), whereas PM_1 behaves differently [*Gerasopoulos et al*., 2007; *Koçak et al*., 2007;

Saliba et al., 2007]. Mineral dust transport events that occur episodically over the area are a major contributor (more than 40%) to the PM_{10} exceedances of the EU limit of 50 μg m^{-3} [*Koçak et al*., 2007; *Gerasopoulos et al*., 2006b; *Mitsakou et al*., 2008]. This is also observed by lidar [*Papayannis et al*., 2008], sun photometer [*Fotiadis et al.,* 2007] networks, and satellite observations [*Papayannis et al*., 2005; *Kalivitis et al*., 2007]. Re-suspension of dust is likewise a significant component of aerosols in the cities [*Rodriguez et al*., 2004]. In industrial and urban areas, most exceedances (around 70-80%) are due almost exclusively to local anthropogenic sources [*Querol et al*., 2009].

In the eastern basin the fine aerosol fraction (<1μm) is mainly dominated by pollution components. About 60% of the total mass is due to ionic mass and 30% to organics [*Lelieveld et al*., 2002; *Bardouki et al*., 2003; *Koulouri et al*., 2008a]. In the coarse mode organics account for about 10% whereas ionic components and dust contribute about 50% and 40%, respectively [*Koulouri et al*., 2008a]. Compared to the colder Central and North Europe, the high temperatures that impose a low thermal stability of ammonium nitrate in summer favour the formation of nitric acid rather than ammonium nitrate in the area [*Querol et al*., 2008; *Mihalopoulos et al*., 1997].

High sulphate background loadings in the East Mediterranean are mostly attributed to the long-range transport of sulphur-dioxide pollution [*Zerefos et al*., 2002]. Significant interactions exist in the Mediterranean between natural and anthropogenic components in the atmosphere both in the gas and aerosol phases that deserve further investigations. Observations and modelling have shown that marine biogenic emissions contribute up to 20% of the total oxidized sulphur production [*Kouvarakis and Mihalopoulos*, 2002] and about 17% of the nitric acid plus particulate nitrate formation and thus the nutrient deposition to the sea [*Vrekoussis et al*., 2006] on a mean yearly basis. Of particular interest for the sulphur abatement strategy is that during summer in the eastern Mediterranean, heterogeneous sulphate production on fine particles appears to be negligible whereas only about 10% of the supermicron nss-sulphate can be explained by condensation of gas-phase sulphuric acid; the rest must be formed via heterogeneous pathways [*Mihalopoulos et al*., 2007].

6.7.1.3 Air pollution and impacts

In the Mediterranean, aerosols reduce the solar radiation absorption by the sea by about 10%, alter the heating profile of the lower troposphere, and exert a cooling effect five times higher than that induced by the greenhouses gases, but opposite in sign [*Lelieveld et al*., 2002; *Vrekoussis et al*., 2005]. As a consequence, evaporation and moisture transport, in particular to North Africa and the Middle East, are suppressed. *Rosenfeld* [2000)] studied satellite observations and found that aerosols caused important perturbations to cloud microstructure and convection, probably decreasing precipitation. *Querol et al.* [2009] analyzed available aerosol data in the Mediterranean and pointed out three very important climate relevant features of the aerosols in the area: the increasing gradient of dust towards the east; the change of hygroscopic behaviour of mineral aerosols (dust) via nitration and sulphation; and the abundance of highly hygroscopic aerosols during high insolation (low cloud formation) periods. Radiative forcing by aerosols also influences the energy budget of the Mediterranean and the Black seas, but the consequences of this are still poorly understood. A changing energy budget and anomalous winds are expected to influence the ocean circulation [*Tragou and Lascaratos*, 2003]. Therefore, aerosols may affect several components of the eastern Mediterranean atmosphere-ocean system including the regional water cycle.

These aerosol-generated effects are already substantial today, even though sulphate from Europe has actually decreased in the past two decades through the abatement of acidification. During summer the persistent northerly winds carry large pollution loads from Europe that can deposit onto the Mediterranean sea, for instance, nitrate and phosphorus containing aerosols, which affect the water quality and could contribute to eutrophication [*Kouvarakis et al*., 2001].

In addition, ageing of aerosols, such as coating of dust by pollution compounds [*Falkovich et al*., 2004] or chemical trapping of nitrogen on pollen particles [*Franze et al*., 2005], can be harmful for human health. *Katsouyanni* [1995] points out adverse health effects of air pollution at levels of

pollutants around or even below the air quality standards set by national and international institutions. Air pollution effects on health, partly determined by specific mixtures of air pollutants, may be altered by other environmental, behavioural, and social patterns. Mediterranean countries have some common characteristics in terms of climate, geography, and population activities that differ from those in the colder Central and Northern Europe. *Katsouyanni* [1995] summarizes the knowledge on the impact of air pollution on health in two major Mediterranean cities, mainly with regard to exposures to sulphur dioxide and black smoke. She points out that the health effects of the interactions between these pollutants and photochemical oxidants can be enhanced in the Mediterranean. The area is also appropriate for the study of synergies between air pollutants, high temperatures and humidity patterns. She stresses that even if the health effects of air pollution only slightly increase the risk to an individual, they are likely to be important for public health because of the ubiquitous exposure of the population. *Zanobetti et al*. [2002] studying 10 European cities among which 4 were in the Mediterranean area, found that the overall effect of PM_{10} per 10 $\mu g \cdot m^{-3}$ is a 1.61% increase in daily deaths, whereas the mean of PM_{10} on the same day and the previous day is associated with only a 0.70% increase in deaths. Their study confirms that the effects observed in daily time-series studies are not due primarily to short-term mortality displacement and that the impact of airborne particles more than doubles when longer-term effects are taken into account.

6.7.2 The Greater Istanbul Area

6.7.2.1 City characteristics, geography, population, meteorology

The city of **Istanbul** located at 41.01°N, 28.58°E, is one of the largest cities (21st in 2009) [*Thomas Brinkhoff: The Principal Agglomerations of the World,* http://www.citypopulation.de/] in the world with 12.5 million inhabitants according to the last population census of 2007 and an annual growth rate to be about 4.5%. The city extends on two continents with the European part of the city being the oldest part, separated from the Asian part by the south part of the Bosporus strait. The Bosporus strait of 30-km length, connects the Marmara Sea at the south with the Black Sea at the north. In the past, the city expanded along the Marmara shore but in recent years a northerly expansion has also occurred (Figure 35, *Ezber et al*., 2007) to move away from the North Anatolia Fault which passes along the south of Istanbul in the Marmara sea and produced the large Earth Quake in Istanbul on August 17, 1999 [*Stein et al*., 1997; *Armijo et al*., 1999; *Hubert-Ferrari et al*., 2002]. The metropolitan region covers 6220 km^2.

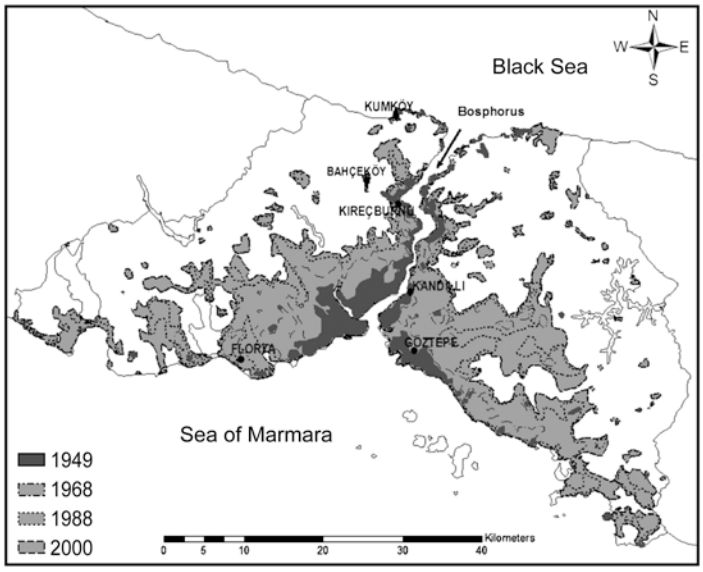

Figure 35 - Istanbul is located on both European and Asian continents, separating the Black Sea from Bosporus [*Ezber et al*., 2007]

Almost 17% of the population of Turkey occupies the Greater Area of Istanbul (GIA). In the GIA, almost the entire population inhabits the urbanized areas while a very small portion occupies the surrounding rural areas. The city's population nearly doubled in the 20 years between 1980 and 2000, the fastest growth period for the population. For the period between 1990 and 2000, the population growth rate of Istanbul was 29.6% for urban parts and 81% for rural parts of the city. Total population growth rate was 33.1% for the same period. To compare, these figures are 26.8, 4.2 and 18.3%, respectively, for the whole of Turkey (Figure 36; *Ezber et al*., 2007; *CIA World Factbook*, 2009]. GIA's population growth rate is slightly over 4% [*OECD*, 2008]. The 2005 Istanbul's population is expected to grow from its present level of 12 million to 16 million in 2017, and to 23 million in 2023. This will result in intensified pressure on industrial and residential uses in the northern part of the metropolitan region, where the natural protection areas and the watersheds are located [*OECD*, 2008].

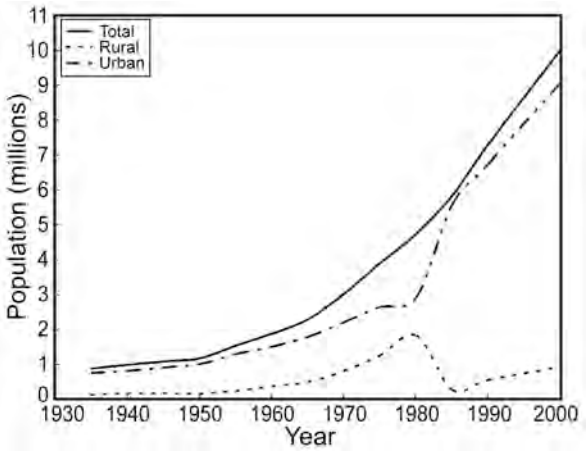

Figure 36 - Population growth in Istanbul from 1935 to 2000 [*Ezber et al*., 2007]

WHO [2006] statistics for the year 2004 report for Turkey a mean life expectance at birth of 71 yr that is lower for males (69 yr) and higher for females (73 yr). Similar numbers are given for 2009 (71.96 yr, 70.12 and 73.89 yr respectively) by the 2009 CIA World Factbook. An increase is reported since 1990 when these numbers were 64 yr for males and 67 yr for females. According to Turkish Statistical Institute [*TÜİK*, 2007], the life expectance at birth was 66 yr in 1990 and 71.7 yr in 2007 (70.1 yr for men and 73.9 yr for women) and is expected to increase to 74.4 yr in 2050.

During the last decades the Mediterranean, following the general trend, has experienced a rapid growth in urbanization that intensify vehicle circulation and industrialization, which are reflected by the increase in pollutant emissions to the atmosphere. Such development affected the city of Istanbul within the last 40 years.

The orography of Istanbul is dominated by seven hills that affect the air circulation in the region. The prevailing wind in Istanbul is northeasterly. The southern part of the GIA, the most urbanized, has Mediterranean type climate. During summer, it experiences sea and land breeze circulation patterns with wind blowing from Marmara sea to Istanbul during day and from the city to the sea during night. Such circulation is influencing pollutants transport [*Im et al*., 2006]. The northern part of GIA is affected by the colder northern air masses and the cooler Black Sea and has slightly cooler temperatures and higher precipitation than the southern part of the GIA. Average seasonal air temperatures in Istanbul are about 28^{o}C in summer and 8^{o}C in winter, and the wind speed is highest in winter and lowest in summer with annual average of about 17 km/h. The humidity is high during all seasons. Average annual total precipitation is around 800mm [*Ezber et al*., 2007]. The heating effect due to urbanization was found to produce two-cell structures during the summer, one on the European and one on the Asian side of the city. The cells extend to about 600–800 m height in the atmosphere over the city and combine aloft [*Ezber et al*., 2007]. In

addition to local pollution, Istanbul is vulnerable to trans-boundary transport of air pollutants from Europe, because its location is on the eastern end of the Continent, where westerly winds prevail [*Kidnap*, 2008].

6.7.2.2 Emissions sources of pollutants and their precursors in the area

Between 1980 and 1990 the consumption ratio of coal to fuel-oil increased from 0.68 (in 1980) to 3.09 in 1990 [*Tayanc et al.,* 2000]. Liquefied petroleum gas (LPG) has been widely used in traffic starting from the beginning of 1998. Through the emissions from the transport sector, a big portion of ozone precursors and aerosols are emitted to the atmosphere. There has been a shift from coal to natural gas for domestic heating purposes starting from the early 1990s, leading to a decrease in the concentrations of primary pollutants such as sulphur oxides (SO_x) but an increase in secondary pollutants such as aerosols and ozone. The region experiences very dense industrial activities, almost the highest in the country. Almost half of Turkish industry is located around the Marmara Sea [*OECD*, 2008]. Based on Istanbul Chamber of Industry reports, 37% of the industrial activities comprise textile industry, 30% metal industry, 21%chemical industry, 5% food industry, and 7% other industries [*Im et al*., 2006]. Low quality solid and liquid fuels with high sulphur content, natural gas, and LPG are the most commonly used fuel types in the industrial activities. In addition, over 2 million cars circulate in Istanbul, most of them are older cars. Under these the density and variety of industrial activities, the region experiences very complex air quality conditions.

Markakis et al. [2009] have developed an emission process kernel aimed at compiling a high spatial and temporal resolution emission inventory for anthropogenic sources for the GIA at a 2 km grid. They considered nitrogen oxides (NO_X = NO + NO2), carbon monoxide (CO), SO_X, ammonia (NH3), non-methane volatile organic compounds (NMVOCs), PM_{10}, and $PM_{2.5}$. They estimated total annual emissions in the GIA of 437 kt for CO, 305 kt for NO_X, 91 kt for SO_X, 77 kt for NMVOCs, 7 kt for NH_3, 61 kt for PM_{10}, and 37 kt for $PM_{2.5}$. Their results indicate that the road transport sector is the main contributor to the emissions in the area, whereas residential and industrial combustion as well as cargo shipping are also important source categories. Industrial combustion (49% of PM_{10} and 24% of $PM_{2.5}$) and road transport (17% of PM_{10} and 29% of $PM_{2.5}$) emissions also play the major role in PM emissions. Particularly, 76 % of the organic portion of PM comes from traffic. Energy and industrial combustion are found to be the main SO_X emitters (36 and 23%, respectively). NMVOCs mainly originate from road transport (45%), solvent use (30%), and waste (20%). The spatial distribution of emissions follows the residential distribution and the seasonal variation shows higher emissions during the winter. Weekend emissions are lower than weekday emissions and diurnal calculations show that the profile fits with the rush hours due to the highest contribution of traffic emissions [*Makrakis et al*., 2009].

6.7.2.3 Air pollutants

Ozone and its precursors - *Im et al.* [2008] reported O_3 observations from two different locations in Istanbul close to the Marmara sea from 2001 to 2005. One location, Saraçhane, is on the European side of the city and the other one location, Kadıköy, is the Asian side of the city. The highest ozone concentrations were observed during hot, sunny summer when maximum temperatures were above 25 °C. These episodes were mainly characterized by southwesterly surface winds during the day and northeasterly surface winds during the night. High VOC-to-NO_X ratios at both stations indicated that NO_X-sensitive chemistry was dominant in the region. However, higher correlations of VOCs in Kadıköy, as compared with those in Saraçhane, indicated that VOCs also have an important contribution to ozone formation. High O_3 days demonstrated a typical diurnal profile with maximum concentrations appearing during afternoon hours and minimum concentrations appearing during rush hours due to NO_X titration from traffic emissions. According to measurements, daily average O_3 concentrations were around 100 $\mu g \cdot m^{-3}$ whereas individual episodic periods had hourly concentrations reaching 180 $\mu g \cdot m^{-3}$ in Saraçhane and 140 $\mu g \cdot m^{-3}$ in Kadıköy [*Im et al*., 2008]. Lower-daily maximum values at the Kadıköy Station resulted from better ventilation due to its location near the sea and from larger titration values of NO_X resulting from close proximity to dense traffic.

Exceptionally high ozone concentrations up to 310$\mu g \cdot m^{-3}$ have been observed in Istanbul in the early morning hours [*Im et al.,* 2006]. The high ozone concentrations can be explained by decreasing inversion heights during the early hours of the day that led to suppression of pollutants close to surface and thus an increase in ozone concentrations [*Im et al.,* 2006]. During this exceptionally high ozone event, high levels of NO_2 from traffic and CO, locally exceeding 100 $\mu g \cdot m^{-3}$ and 1.5 $mg \cdot m^{-3}$, were also reported.

High correlations between NO and NO_X were calculated for both stations suggesting that NO_X emissions originated locally. *Vrekoussis et al.* [2009] have also computed the annual levels of the vertical column densities of NO_2 (VCD_{NO2}) based on SCIAMACHY observations during the period 2003-2007 and over a region of 0.5^ox0.5^o (4 grids) around the city centre. The calculated annual mean NO_2 VCDs over Istanbul are ~$1 \cdot 10^{16}$ $molecules \cdot cm^{-2}$. Satellite observations [*Vrekoussis et al.*, 2009] indicate an increase in the tropospheric columns of NO_2 over recent years (Figure 34).

Airborne particulate matter - Some regions of Istanbul are continuously exposed to high pollution levels during the heating season (November–March) [*Gülsoy et al*., 1999]. Since 1966, several stations run by the Istanbul Municipality monitor PM_{10} concentrations. At the end of the 1980s and the beginning of the 1990s, sulphur dioxide (SO_2) and PM concentrations exceeded the short-term air quality standards[1] on many days [*Tayanç*, 2000]. The levels of SO_2 over Istanbul increased from 1985 to 1991 reflecting the use of low quality fossil fuels during that period. In 1995-1996, there was a considerable decrease in air pollution for 3 reasons: 1) increasing ventilation of the City, 2) switching to natural gas for heating and 3) treatment of coal before its entrance into the city.

Karaca et al. [2005] found that the annual (July 2002-July 2003) arithmetic mean of PM_{10} was 47.1 μg m^{-3}, higher than the European Union air quality annual PM_{10} standard of 40 $\mu g \cdot m^{-3}$. The annual mean concentration of $PM_{2.5}$ (20.8 μg m^{-3}) was also higher than United States EPA annual $PM_{2.5}$ standard of 15 μg m^{-3}. *Ozdemir et al.* [2009] reported average PM_{10} levels of about 66 $\mu g \cdot m^{-3}$ observed at 10 Istanbul municipality stations during the last 10 years with values ranging from 47 $\mu g \cdot m^{-3}$ to 115 $\mu g \cdot m^{-3}$. *Karaca and Camci* [2010] have analyzed the episodes of high PM_{10} levels in Istanbul and attributed 52% of them to distant source contributions, massive anthropogenic activity over all of Europe, and southwestern airflow most likely carrying PM_{10} originating from the Sahara Desert and other global dust generation regions located in the northern part of Africa.

Recently, the first complete chemical characterization measurements of aerosol in GIA became available from the Bogaziçi University sampling station. The station is located at an urban background site in Bosporus straight coast in Istanbul, and 9 different water-soluble ions, water soluble organic carbon (WSOC), organic and elemental carbon (OC, EC) and several trace metals were measured between November 2007 and June 2009. *Theodosi et al*. [2010] found that trace elements related to human activities (as Pb, V, Cd and Ni) reached peak values during winter due to domestic heating, whereas natural origin elements like Al, Fe and Mn peaked during the spring period due to dust transport from Northern Africa. Organic carbon was found to be mostly primary and elemental carbon was strongly linked to fuel oil combustion and traffic. Both OC and EC concentrations increased during winter due to domestic heating, while the higher WSOC to OC ratio during summer can be mostly attributed to the presence of secondary, oxidised and more soluble organics. Source apportionment analysis of these observations using PMF indicates that approximately 80 % of the PM_{10} in Istanbul is anthropogenic in origin [secondary, refuse incineration, fuel oil and solid fuel combustion and traffic, *Koçak et al.,* 2010].

Guelsoy et al. [1999] analyzed precipitation samples in Istanbul from January to October 1996. They observed high sulphate and nitrate concentrations in the precipitation of up to 150 $mg \cdot l^{-1}$ and 70 $mg \cdot l^{-1}$, respectively, during the heating period, associated with high pH values due to neutralization of the acidity by calcium and ammonium observed at sufficient levels in these

[1] *The current Turkish legislation sets SO_2 and PM_{10} limits as follows (Ministry of Environment and Forest, Turkey, 2008): SO_2 hourly limit: 360 ug/m³; SO_2 24-hour limit: 125 ug/m²; annual SO_2 limit: 20 ug/m³. PM_{10} 24-hour limit: 50 ug/m³; annual PM_{10} limit: 40 ug/m³.*

precipitation samples. In general the Istanbul rainwater pH was about 6-7, only 18.6% of the events showed acidic pH below 5.6.

There have also been a limited number of modelling studies that investigate the elevated PM_{10} levels in the area. The results showed that local anthropogenic emissions under unfavourable meteorological conditions can be responsible for up to 90 per cent of PM_{10} levels [*Im et al*., 2010]. Long range transport from Europe can contribute up to 50 per cent to these high concentrations [*Kindap et al*., 2006].

6.7.2.4 Climate effects including Heat island

Karaca et al. [1995] and *Ezber et al.* [1997] have studied the climatic effects of urbanization in Istanbul. *Karaca et al*. [1995] statistically analyzed long-term temperature data (1912- 1992) from stations within and around the Istanbul and reported a warming trend in urban temperatures in southern Istanbul, which was the most densely populated part of the city, whereas in northern GIA a cooling trend was observed. *Ezber et al.* [2007] have focused the analysis for the period from 1951 to 2004 by using both statistical and numerical mesoscale modelling tools. They have shown that the urbanization effect on climate was most pronounced during the summer. Changes in the trends occurred in the 1970s and 1980s when the population growth rate in Istanbul increased dramatically. A significant expansion of the urban heat island in Istanbul has been computed for the period from 1951 to 2004, fairly consistent with the expansion of the city during this period. The maximum reference-level temperature difference between the past and present simulations was found to be around 1^{o}C. The modelling experiment by *Ezber et al*. [2007] also indicated that the velocity of the prevailing northeasterly wind and the water vapour mixing ratio were both reduced over the city.

6.7.3 The Greater Athens Area

6.7.3.1 City characteristics, geography, population, meteorology

Greece has one of the most aged populations in Europe with almost one fifth over 65 years old. In 2004 the annual deaths of 104 000 inhabitants were slightly higher than the 101 000 births. In 2004, the life expectancy of the total population at birth was 79 years. The male population, which constitutes 50% of the national total population, has a lower life expectancy (77 years) than females (82 years), one of the highest in Europe [*WHO*, 2006]. The Greater Athens Area (GAA) currently has a population exceeding 4 million inhabitants, which is about 40% of the total population of Greece, and has experienced a population growth rate of 0.6% per year over the last decade.

The climate of Athens is typically Mediterranean with hot dry summers and wet mild winters. The mean daily summer and winter temperatures are 25.8^{o}C and 10.1^{o}C, respectively. The mean annual total precipitation is about 400 mm, 85% of which occurs from October to March [*Kalabokas et al*., 1999a]. The mean wind pattern in the atmospheric boundary layer in Athens during the warmer part of the year is a persistent northeasterly flow. The Athens basin is exposed to the summer monsoon circulation of the Eastern Mediterranean. The city of Athens is located in a basin on the west coast of the Attica peninsula (Figure 37). It is surrounded by moderately high mountains forming a channel with only one major opening toward the sea to the southwest. The mountains act as physical barriers with only small gaps between them. The most important gap is the channel between Hymmetus and Pendeli leading to the northeast coast of the Attica peninsula that gives the Athens basin access to the Etesians, a system of semi persistent northerly winds. During the appearance of the Etesians, good ventilation of the basin is favoured and thus pollution episodes do not appear. The weakening of the synoptic wind allows the development of local circulation systems, such as sea/land breezes along the axis of the basin (NE to SW) and anabatic/catabatic flows from the surrounding mountains. During such cases, the ventilation of the basin is poor, the boundary layer is shallow, and the air pollution potential increases [*Melas et al.*, 1995 and references therein]. Air pollution episodes may occur in Athens during all seasons of the year but most of these episodes are associated with the development of sea-breeze [*Kallos et al*., 1993].

Figure 37 - Map of Greater Athens Area with altitude contours at 200 m intervals [*Melas et al.*, 1995]

6.7.3.2 Emissions of pollutants and their precursors in the area

The massive number of registered vehicles in circulation (over 2.5 million, growing at a rate of 7% yearly) is allegedly the major cause of air pollution related problems in the area, taking into account that a large proportion of these vehicles are non-catalytic (0.8 million) or are powered by old technology diesel engines (0.2 million). Although the use of natural gas for domestic heating purposes has increased lately, combustion of fuel oil is still primarily used for central heating.

An anthropogenic emissions inventory was compiled for Greece and the Greater Athens Area (GAA) for the reference year 2003 [*Markakis et al.*, 2010a; *Markakis et al.*, 2010b] with a 10 km spatial grid over Greece and a 2 km grid over GAA. The emission inventory has monthly, weekly, and hourly temporal analysis. Total annual emissions for the GAA were estimated to be 473 kt for CO, 78 kt for NO_X, 31 kt for SO_2, 93 kt for NMVOCs, and 20 kt for PM_{10}. Approximately 75% of CO, 70% of NMVOCs, and almost half of NO_X emissions originate from the road transport sector while the most important SO_2 emitter is the industrial sector. The majority of PM emissions stem from the industrial sector. The large industrial complexes are located several kilometres outside the Athens basin where the majority of the population resides. Inside the basin the road transport is again the most important emission source for PM. Almost 20% of PM emissions originate from non-exhaust sources like tire and break wear as well as road abrasion. The annual variation of emissions shows that although the central heating operations do not account for more than a few percent in the annual totals (with an exception of SO_2 – 15% contribution), in the winter months they make a significant contribution. According to Greek national totals Athens is responsible for almost half of the road transport sector CO emissions and 70% of the NO_X emissions. Taking into account that almost half of the county's population inhabits the GAA as well as the fact that Athens experiences very severe congestion with the average speed not to exceed 12 km/h during rush hours, the results are not surprising.

6.7.3.3 Air pollution levels and abatement measures

The GAA has been subject of intensive field campaigns like MECAPHOT-TRACE in summer 1994 [*Ziomas*, 1998 and references therein] and PAUR I and II [in summer 1996; *Zerefos et al.*, 2000 references therein]. For the GAA, the air quality reports have showed that there has been a great improvement in the latest years regarding the pollution levels. This is mostly due the fact that the large industries were reallocated from inside the Athens basin to the greater area while more

strict legislation were enforced with a number of large units to be equipped with filters. In addition, pollution abatement measures taken by the state authorities during the period 1990-1994, consisting in the replacement of the old technology gasoline-powered private cars and the reduction of the sulphur content in diesel oil, seem to be the primary cause of the improvement in air quality in Athens during the recent years.

Kalabokas et al. [1999a; 1999b] analyzed 11 years of observations from the automated local air pollution network operating by the Ministry of Environment. Since 1987, a significant downward trend for almost all primary pollutants in all stations was observed. Comparison between the 3-year periods 1988-1990 and 1995-1997 gives the highest reduction in the centre of GAA of 52%, 34%, 26% and 20% decreases for sulphur dioxide, carbon monoxide, nitrogen oxides and black smoke, respectively, whereas the concentrations of secondary gaseous pollutants (especially Ox = sum of ozone and nitrogen dioxide) appear to have remained essentially at the same levels since 1990. Observations of O_3 prior to 2000 [*Kalabokas and Repapis*, 2004] at three stations in the GAA and the surroundings were found to exhibit characteristic seasonal variation of rural ozone concentrations with lowest concentrations occurring during winter afternoons with values at about 50μg.m^{-3} in December–January and average summer afternoon values at about 120μg.m^{-3} in July–August, indicating that high summer values were observed all over the area.

The latest air quality report of the Ministry of Environment concludes that, except for PM concentrations, the gaseous pollutants are generally below the EU limits. In the GAA, PM still shows large exceedences, in contrast to gaseous pollutants. At 3 urban stations, the daily limit value for PM was exceeded almost half of the year with the average annual value ranging from 48 μg m^{-3} to 57 μg m^{-3}. Based on one year (2005-2006) of PM_{10} observations at two locations in Athens, *Koulouri et al*. [2008b] reported that the 24-h limit value of 50 μg m^{-3} are exceeded 44% of the time at both sites, far exceeding the compliance with the air quality standard that demands a maximum proportion of exceedences at 9.6 % per year. Furthermore, $PM_{2.5}$ concentrations at both stations exceed the long-existing US-EPA limit value of 15 μg m^{-3}.

6.7.4 Open questions for the area

Open questions being addressed by current projects such as the EC FP6 projects CIRCE and SESAME and the FP7 project CityZen are: how air pollution sources affect air quality, human/ecosystem health, climate, and visibility, the effects on water resources and evaporation, and how large are the anthropogenic and natural contributions to deposition to ecosystems (acidification, eutrophication, ocean biological productivity). CityZen also investigates future development and mitigation options applying state-of-the-art atmospheric models.

Under the EU FP7 project CityZen, a number of key observations were performed in the East Mediterranean with a focus on GIA and Athens as pollution sources and Finokalia (Crete) as a pollution receptor location in the area. The new observations in Greater Istanbul Area (GIA) consist of aerosol chemical composition measurements at two sites within GIA and at two sites downwind to identify the main aerosol sources of this Megacity and quantify its role as source of air pollutants in the area. In parallel, air quality data over Istanbul and other hot-spots in the East Mediterranean region for ground based stations and satellite observations are being compiled and analyzed in conjunction with back trajectory analysis and numerical chemistry/transport and climate modelling in order to identify possible trends, the factors that are causing them, and the environmental consequences of human impacts in the area. Cutting edge research in the area is investigating the role of Eastern Mediterranean Megacities and hot-spot areas compared to the transport of pollution on aerosol load and their climatic relevant properties in the Eastern Mediterranean atmosphere.

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CHAPTER 7 - OVERVIEW OF INTERNATIONAL COLLABORATIVE RESEARCH ACTIVITIES

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In this chapter we briefly describe a number of projects, usually international, intended to improve our understanding of air pollution and its impacts, including policy relevant aspects. This review is not exhaustive but reflects current efforts. Figure 1 shows the locations of these research activities.

Figure 1 - The locations of the international collaborative research activities described in this chapter. ADAPTE: Adaptation to health impacts of air pollution and climate extremes in Latin American cities; CalNex 2010: Air Quality and Climate Change Field Study in California in 2010; CAREBeijing: Campaigns of Air Quality Research in Beijing and Surrounding Regions; CityZen: megaCITY–Zoom for the Environment; ICARTT: The International Consortium for Atmospheric Research on Transport and Transformation; IMPACT: Integrated Measurement Programme for Aerosol and oxidant Chemistry in Tokyo; MILAGRO: Megacity Initiative: Local and Global Research Observations, Mexico City; MEGAPOLI: Megacities: Emissions, urban, regional and Global Atmospheric POLlution and climate effects, and Integrated tools for assessment and mitigation; PRIDE-PRD: Programme of Regional Integrated Experiments of Air Quality over Pear River Delta; SAEMC: the South American Emissions Megacities and Climate

7.1 MEGAPOLI

MEGAPOLI (Megacities: Emissions, urban, regional and Global Atmospheric POLlution and climate effects, and Integrated tools for assessment and mitigation), funded by the European Union through Framework Programme 7, brings together leading European research groups, state-of-the-art scientific tools and key players from other countries to investigate the interactions among megacities, air quality and climate. The project was coordinated by Alexander Baklanov (DMI, Denmark), with co-coordinators Spyros Pandis (FORTH, Greece) and Mark Lawrence (MPIC, Germany). A total of 23 groups from 11 countries in Europe were funded by the project. In addition, the MEGAPOLI framework has led to support from individual institutions and/or national funding agencies for further scientific contributions to the project, such as a field campaign in Paris.

The MEGAPOLI project bridges the spatial and temporal scales that connect local emissions, air quality and weather with global atmospheric chemistry and climate. The main objectives were:

1. To assess impacts of megacities and large air-pollution hot-spots on local, regional and global air quality,
2. To quantify feedbacks among megacity air quality, local and regional climate, and global climate change,
3. To develop improved integrated tools for prediction of air pollution in megacities.

In order to achieve these objectives, the partners of the project have undertaken various activities:

- Develop and evaluate integrated methods to improve megacity emission data
- Investigate physical and chemical processes starting from the megacity street level, continuing to the city, regional and global scales
- Assess regional and global air quality impacts of megacity plumes
- Determine the main mechanisms of regional meteorology/climate forcing due to megacity plumes
- Assess global megacity pollutant forcing on climate
- Examine feedback mechanisms including effects of climate change on megacity air quality
- Develop integrated tools for prediction of megacity air quality
- Evaluate these integrated tools and use them in case studies
- Develop a methodology to estimate the impacts of different scenarios of megacity development on human health and climate change
- Propose and assess mitigation options to reduce the impacts of megacity emissions

These tasks were organized into nine WorkPackages, which are depicted in Figure 2, along with their interrelationships. Within MEGAPOLI, a pyramid strategy (see Figure 3) followed by undertaking detailed measurements in one European megacity, Paris, performing detailed analysis for a subset of MPCs selected from 12 potential candidates with existing air quality datasets, and investigating the effects of all megacities on global atmospheric chemistry and climate. The results were disseminated to authorities, the policy community, researchers and the other stakeholders in the corresponding megacities.

The funding for MEGAPOLI ended in September 2011; the analysis of the project results, including model simulations and the Paris field campaign data, is on-going and being published in a final report for the project, and in various journal papers, including two special issues in Atmospheric Chemistry and Physics, one for the "Megapoli-Paris 2009/2010 campaign" [http://www.atmos-chem-phys.net/special_issue248.html], and another titled “Megacities: air quality and climate impacts from local to global scales"[http://www.atmos-chem-phys.net/special_issue229.html], growing out of studies presented in the annual megacity sessions of the EGU general assembly (including results from MEGAPOLI, CITYZEN and MILAGRO, among others discussed below).

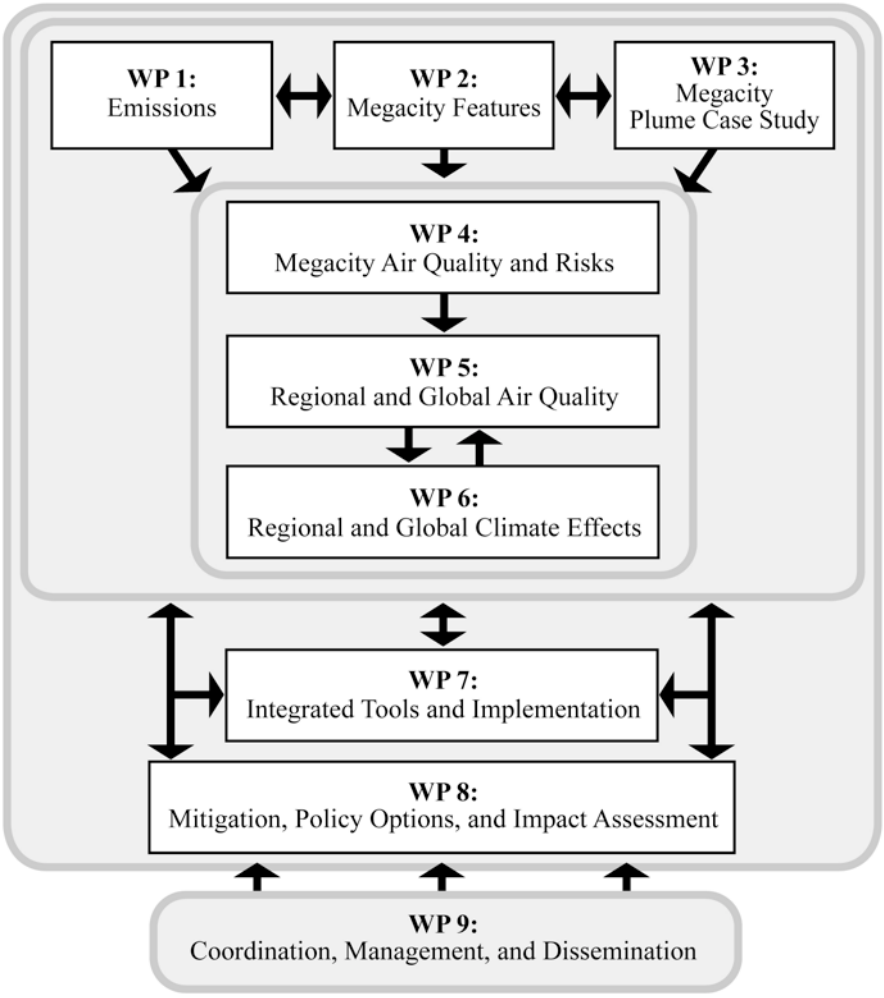

Figure 2 - Depiction of the Work Packages and their organization within MEGAPOLI

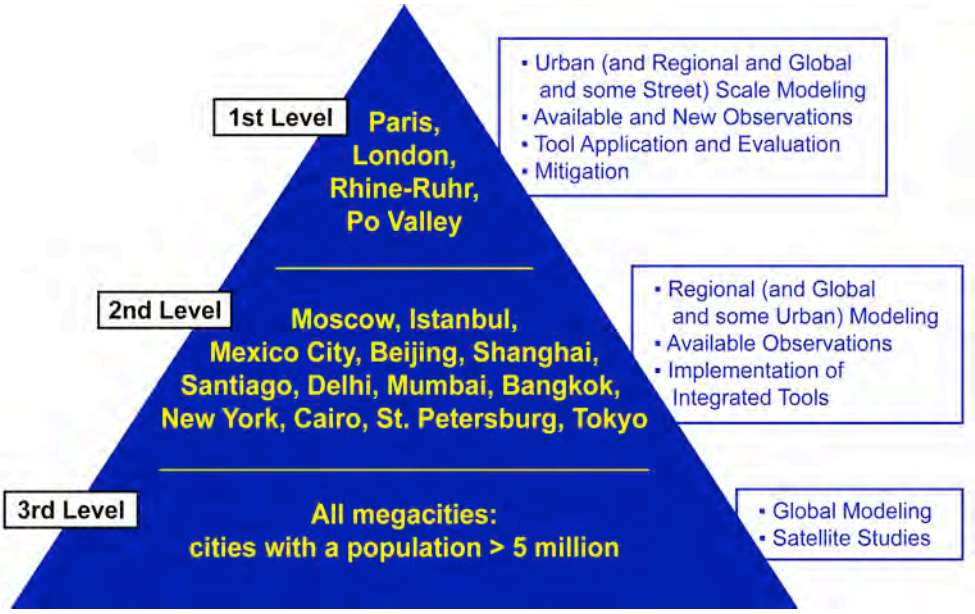

Figure 3 - The pyramid of megacities examined within MEGAPOLI; the project addressed practically all major megacities around the globe at three different levels of detail

7.2 CITYZEN

The CityZen project (megaCITY - Zoom for the ENvironment), funded by the European Union through Framework Programme 7, investigated air pollution distribution and change in and around selected megacities and emission hotspots for the last decade and the 3-year period of the project (2008-2011). Long-term satellite observations, in-situ measurements, and a series of different scale models (local-regional-global) were employed in order to analyze the impacts of air pollution hot spots on regional and global air quality. Potential future changes were studied based on various mitigation and climate change scenarios. CityZen chose four focus areas: 1) The Eastern Mediterranean including Istanbul, Athens and Cairo, 2) the Po Valley including Milan, Genova, and Torino, 3) the Ruhr region in West-Germany together with the BeNeLux area, and 4) the Pearl River Delta with the major cities being Guangzhou, Shenzhen, and Hong Kong. In addition, the hot and polluted European summers of 2003 and 2007 were chosen for intensive case studies.

The project included a total of 16 partners from 11 countries in Europe, Africa, and Asia and was coordinated by the Norwegian Meteorological Institute. Similar to MEGAPOLI, the CityZen project aimed at bridging the spatial and temporal scales that connect local emissions, air quality and weather with global atmospheric chemistry and climate. The main objectives for CityZen were to:

- Quantify and understand current air pollution distribution and development in and around selected megacities/hot spot regions, including the interaction across the different spatial scales
- Estimate the future impact from emission changes with a focus on the effect of rapid growth in the population of megacities/hot spots and the increasing background of pollutants
- Estimate how megacities/hot spots influence climate change
- Estimate how megacities are responding to climate forcing which can influence transport patterns, chemical oxidation and biogenic emissions
- Study mitigation options, e.g. by introducing biofuels, to keep the air pollution load in and around megacities/hot spots within sustainable limits in terms of human health effects and climate impact
- Develop tools to estimate interactions between different spatial scales (megacities to global)
- Use the scientific results and methods developed and applied during the course of the project for technical underpinning of policy work

In order to address these objectives several scientific questions have been addressed in the project:

1. Have megacities and hot spots changed the regional and global distribution of ozone, particulate matter, and their precursors including carbon monoxide CO and other pollutants significantly compared to what would be the case with more evenly distributed emissions?
2. Have megacities affected the radiative budget and aerosol microphysics such that precipitation and the number of sunlit hours and thus temperature and photochemistry have changed significantly both locally and over larger regions? Will this become more significant in the future as megacities and their emissions grow?
3. Will climate change alter weather patterns (winds, temperature, stability, precipitation) and surface properties, which affect air quality in megacities and regional hot spots? If more frequent high pressure situations occur, will episodes with reduced air quality become more frequent?
4. Will climate change induce episodic and permanent changes in the natural and anthropogenic cycles of atmospheric trace chemicals?
5. Will changes in frequency and intensity of forest fires and other biomass burning contribute significantly to air pollution in megacities and hot spots?
6. Can measures be defined that reduce the adverse effects of megacity/hot spot emissions? The adverse effects relate both to air quality (human health) and climate change/weather modification.

The work was organized in four work packages, as illustrated in Figure 4. WP1 focused on analyzing past trends and present distributions, using both models and observations. WP2 dealt with climate-chemistry interactions, while WP3 looked at mitigation options. Both WP2 and WP3 included subtasks for providing emissions inventories. In WP4 we integrated the work of the first three work packages, ensured dissemination and contact with policy workers and other policy-oriented frameworks.

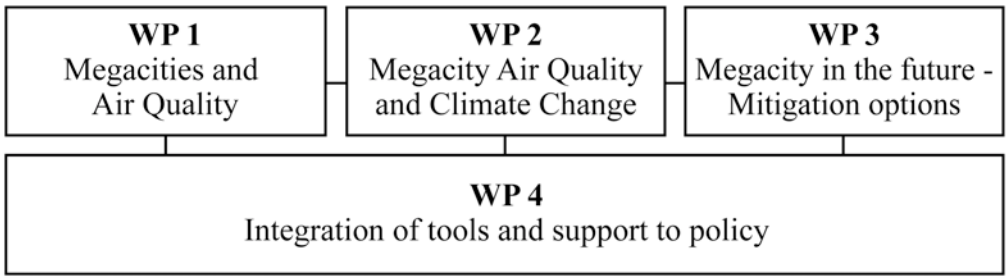

Figure 4 - The organisation of the work in CityZen in work packages

The CityZen consortium combines expertise in the fields of observations, modelling and the provision of emission scenarios. The flow of information is intended to proceed from the data sources (observations, emissions, meteorology, chemical transformation and sink mechanisms) towards the "Core research activities" where all available information and model calculations are blended together into 4-dimensional datasets of a spatial and temporal resolution that are determined by the problem under study. On the basis of these "Core research activities" research is undertaken in work packages 1-3 by a goal-oriented and specialized analysis of the data fields (observed, calculated, assimilated). Figure 5 illustrates the coordinated action of different scientific tools.

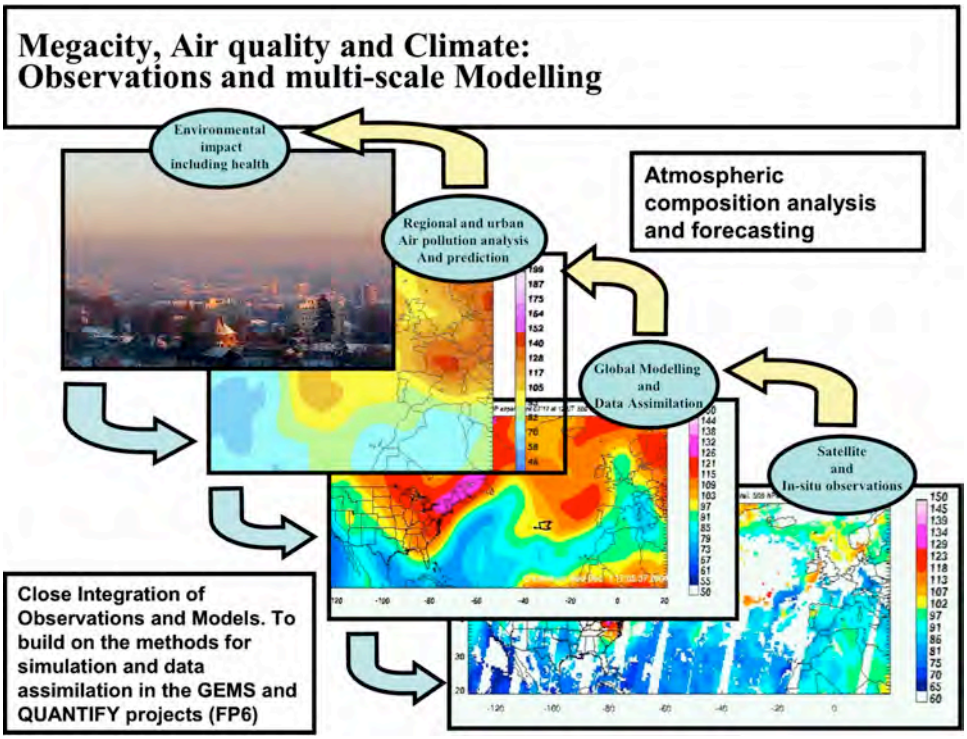

Figure 5 - Model systems bridge processes occurring on different spatial scales spanning from the urban to the global. Air pollution flows between megacities/hotspots, the surrounding regions and globally. The analysis and forecasting of air pollution in hotspots require observations and modelling efforts that take into account the interactions of transport processes and air pollution loads on this range of spatial scales and cover the range of temporal scales determined by the atmospheric life times of the essential pollutants involved

A webpage was established for CityZen, with useful information about the project (http://www.cityzen-project.eu). Results and publication are uploaded to the web site as they become available. The project established collaboration with its sister project MEGAPOLI. Collaboration with partners outside the CityZen and MEGAPOLI consortia was encouraged.

7.3 ICARTT

ICARTT (The International Consortium for Atmospheric Research on Transport and Transformation) coordinated several field studies, each focused on some aspect of climate change and air quality issues over North America, the Atlantic and Europe [*Fehsenfeld et al.*, 2006]. These studies were independently planned for the summer of 2004, and early in the planning process it became evident that coordination between these studies would provide a more effective approach to addressing the common issues. This approach yielded a series of coordinated studies of the emissions of aerosol and ozone precursors, their chemical transformations and removal during transport to and over the North Atlantic, and their impact downwind on the European continent. The primary source of the emissions studied was the US Northeast urban corridor, a major megacity in North America (see Chapter. 5), so in an important sense, ICARTT was focused on the climate and air quality effects of a megacity.

The independently planned research programmes (primary location and sponsor) were NEAQS-ITCT (US - NOAA), INTEX-NA (US - NASA), COBRA (US - NASA), AIRMAP Network (US - NOAA) and CHAiOS (US - NSF, NOAA), PICO-NARE (Azores, Portugal, NOAA, NSF), ITOP (European - NERC, DLR), ITCT-Lagrangian-2K4 Experiment (IGAC), ICARTT Cloud-Aerosol Study (US - NSF, Environment Canada) ICARTT Radiation-Aerosol Study (US - NOAA, NASA). The coordinated programmes conducted extensive measurements from nine aircrafts, a research vessel, balloon-borne sondes launched from several sites, multiple ground stations located in the northeastern United States, Nova Scotia, the Azores, and Western Europe, and several satellite platforms. Figure 6 schematically illustrates the platform deployment.

Figure 6 - Schematic depiction of the locations of platform deployment during ICARTT

The combined research conducted in the programmes that constituted ICARTT focused on three main areas: regional air quality, intercontinental transport, and radiation balance in the atmosphere. *Fehsenfeld et al.* [2006] present an overview of the ICARTT study, and the results are reported in four special sections of *Journal of Geophysical Research-Atmosphere.* Some specific, especially important foci of ICARTT include:

- Role of nitrate radicals and N_2O_5 in the nighttime chemistry of the troposphere
- Effect of reductions in NO_X Emissions from US power plants
- Evaluation of forecasts from air quality models for ozone and particulate matter
- Evolution of the organic component of aerosols in air masses leaving North America
- Relation of CCN activity of aerosols to their composition
- Direct and indirect radiative effects of aerosols
- Transformation of gas and aerosol phase species during transport from North America, over the Atlantic, and to Europe
- Impact of transported pollutants on air quality in Europe

7.4 CalNex 2010

CalNex 2010 is a field study that addressed both climate change and air quality issues in California. The particular focus of the study is the nexus between climate change and air quality issues – hence the name CalNex. Figure 7 illustrates the species involved in this nexus. Though we often treat them as separate issues with separate solutions, climate change and air quality are interrelated in many ways. The major air pollutants - aerosols (including soot) and ozone in the lower atmosphere - are also significant climate change forcing agents. These agents often have the same sources (transportation, industry, agriculture, forests), and they are short-lived in the atmosphere (days or months, rather than decades and longer for other climate gases). The overall goal of CalNex was to provide needed information to develop integrated policies that address these highly interrelated environmental issues together. There are obvious economic and societal benefits to an integrated approach, and managing the short-lived air quality agents can bring early co-benefits for climate (years instead of decades). Importantly, some air quality strategies can be detrimental to climate change mitigation (e.g. control of aerosols removes their climate cooling effect) and some climate strategies can be detrimental to air quality (e.g. introduction of biofuels to reduce carbon dioxide emissions may increase some air pollutant emissions). The goal is to develop policies that simultaneously improve both issues.

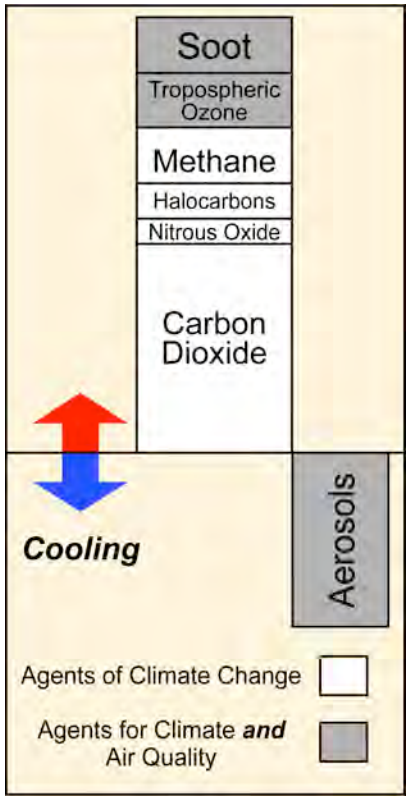

Figure 7 - Depiction of the focus of CalNex 2010 – the nexus of air quality and climate change issues

Many of the measurements were concentrated in the Los Angeles urban area, a major megacity in North America (see Chapter 5), so in an important sense, CalNex also focused on the climate and air quality effects of a megacity. The field measurement phase was conducted in late spring-early summer of 2010, and analyses of the results are ongoing. The study is a collaborative, multiagency effort led by NOAA and the California Air Resources Board (CARB) that conducted measurements from four aircraft, a research vessel, balloon-borne sondes launched from six sites spanning the length of California, two instrumented tall towers, and multiple ground stations located in California and Mexico, the latter under the NSF funded Cal-Mex programme focused in the California-Mexico border region. More information is available from the study websites:
http://esrl.noaa.gov/csd/calnex/
http://www.arb.ca.gov/research/fieldstudy2010/fieldstudy2010.htm

CalNex was planned to address many specific and general science needs that are required to improve policy responses to air quality and climate change issues. These needs can be roughly categorized as emission characterization and quantification (both greenhouse gases and ozone and aerosol precursors), improved understanding of important atmospheric transformation and climate processes, and transport and meteorology. Instrumentation and platforms were deployed to collect the data sets necessary to address these issues. More detailed information is available from the CalNex White Paper (http://esrl.noaa.gov/csd/calnex/whitepaper.pdf) and the NOAA CalNex Science and Implementation Plan (http://esrl.noaa.gov/csd/calnex/scienceplan.pdf).

7.5 MILAGRO

MILAGRO (**M**egacity **I**nitiative: **L**ocal **A**nd **G**lobal **R**esearch **O**bservations) is an international collaborative project to examine the behaviour and export of atmospheric emissions from a megacity. The Mexico City Metropolitan Area (MCMA) – one of the world's largest megacities and North America's most populous city – was selected as the case study to characterize the sources, concentrations, transport, and transformation processes of the gases and fine particles emitted to the MCMA atmosphere and to evaluate the regional and global impacts of these emissions. The findings of this study are relevant to the evolution and impacts of air pollution from many other megacities.

Specific goals of the MILAGRO Campaign included:

i) Quantifying the spatial and temporal extent of the urban plume
ii) Analyzing pollutant chemical and physical transformation in the plume
iii) Quantifying the regional impacts of the plume; and
iv) Examining the interaction of the urban plume with surrounding sources

The MILAGRO campaign was organized under four coordinated components that took place simultaneously during March 2006 and included the participation of over 450 scientists from 150 institutions in 30 countries.

1) The MCMA-2006 (Mexico City Metropolitan Area – 2006 Experiment) examined emissions and surface concentrations within the Mexico City Basin, their transport and transformation in the atmosphere, and the effects on human health. MCMA-2006 was led by the Molina Center for Energy and the Environment (MCE2) with projects sponsored by US-NSF, US-DOE, and several Mexican research agencies, including the Ministry of the Environment and Natural Resources and the Metropolitan Environmental Commission of the Valley of Mexico, as well as European agencies. The overall purpose of MCMA-2006 was to strengthen the scientific base for the design and evaluation of policies to improve the air quality in the MCMA by gathering scientific information that helps to better understand the generation and processing of pollutants in the MCMA. The MCMA-2006 also provided many of the urban measurements needed for understanding the larger scale pollutant evolution which was the focus of its sister campaigns. Two health studies were carried out during the Campaign.

2) MAX-Mex (Megacity Aerosol Experiment: Mexico City) focused on examining how the Mexico megacity aerosol evolves during transport and how the chemical and physical nature of the aerosol affected scattering and absorption of atmospheric radiation. MAX-Mex was conducted by the Atmospheric Science Programme of the US-DOE Climate Change Research Division in collaboration with scientists supported by US-NSF, US-NASA, and Mexican agencies. Measurements were conducted using an airborne lidar, the DOE Gulfstream-1 (G-1) airborne platform that obtained gas and aerosol measurements, and also at the three supersites to examine the aerosol plume evolution.

3) MIRAGE-Mex (Megacity Impacts on Regional and Global Environments - Mexico) examined the chemical/physical transformations of gaseous and particulate pollutants exported from Mexico City, providing a case study of a megacity's effect on regional and global atmospheric

composition and climate. MIRAGE-Mex was led by the US National Center for Atmospheric Research (NCAR) in collaboration with academic researchers under US-NSF sponsorship. The US NCAR/NSF C-130 aircraft sampled air above and at different distances from Mexico City to measure how gases and particles age during transport, specifically tracking the chemical, physical, and optical properties that have the potential to affect air quality, weather, and climate on large geographic scales. The Twin Otter conducted studies of fire emissions and their effect on the local and regional composition of the atmosphere.

4) INTEX-B (Intercontinental Chemical Transport Experiment-B) was an integrated field campaign designed to understand the transport and transformation of gases and aerosols on transcontinental/intercontinental scales and to assess their impact on air quality and climate. Central to achieving this goal was the need to relate space-based observations with those from airborne and surface platforms. The NASA DC-8 was operated from Houston, Texas with research flights over Mexico and the Gulf of Mexico

The measurement phase consisted of using wide-ranging meteorological, gas and aerosols instruments at ground sites and on aircraft and satellites. All of these platforms together can be used to trace the evolution of the urban plume as it merges with the regional background atmosphere (Figure 8). Three supersites, spaced ~ 30 km apart, were set up to examine the evolution of the primary emitted gases and fine particles. The designations “T0” (initial time), “T1” (first time step), and “T2” (second time step) in Figure 9 refer to the timing of transport of the urban plume to different points in space and time. Additional platforms in or near Mexico City included mobile vans containing scientific laboratories and mobile and stationary lidars. Seven research aircraft provided information about the atmosphere over a large region and at various altitudes. Satellite-based instruments provided even larger geographical coverage. The overall campaign was complemented by meteorological forecasting and numerical simulations, as well as an ambient air quality monitoring network operated by the Mexico City government and meteorological measurements were provided by the Mexican National Weather Service. Together, these observations provided a very comprehensive characterization of the MCMA’s urban and regional atmospheric composition and chemistry that will require extensive analyses and use in model improvement to yield maximum benefits.

Figure 8 - MILAGRO Campaign Geographic Coverage. Measurements were performed in the MCMA (see Figure 9). The size of the circle (MAX-Mex, MIRAGE-Mex and INTEX-B) indicates the geographic coverage of the aircraft deployed

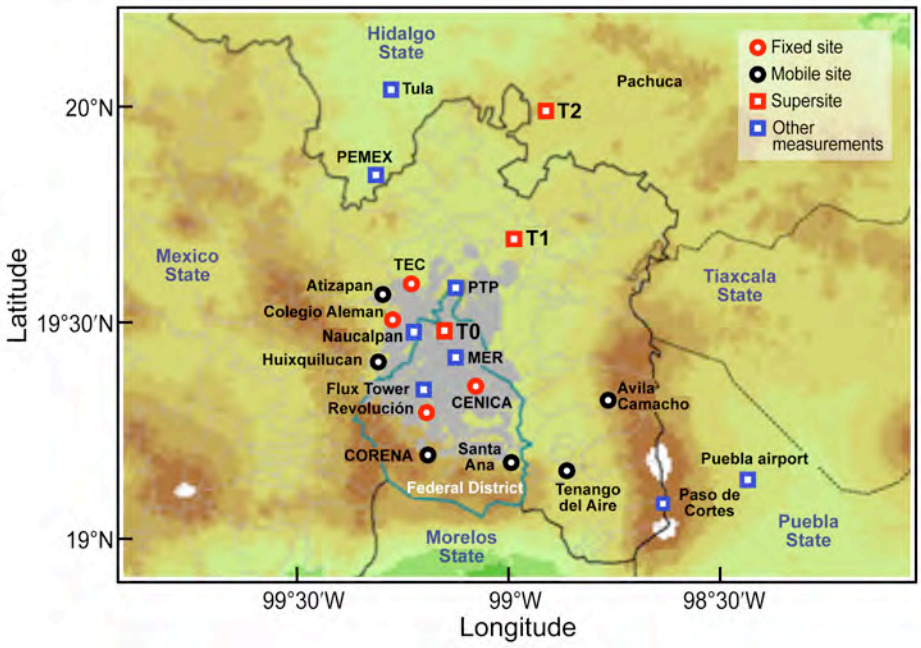

Figure 9 - MILAGRO Campaign: Ground-based Measurement Sites

Major findings are being published in two special issues on MCMA-2003 (http://www.atmos-chem-phys.net/special_issue21.html) and MILAGRO/INTEX-B (http://www.atmos-chem-phys.net/special_issue83.html) in *Atmospheric Chemistry and Physics* as well as in other peer-reviewed journals. An overview article has been published [*Molina et al*., 2010], which reviews over 120 papers resulting from MILAGRO/INTEX-B Campaign that have been published or submitted, as well as additional relevant papers from earlier MCMA-2003 Campaign [*Molina et al*., 2007], with the aim of providing a road map for the scientific community interested in understanding air quality in a megacity such as the MCMA and its local and regional impacts. A complete list of articles is available at: http://mce2.org/publications.html. All data sets are available to the scientific community interested in evaluating the impact of urban emissions on human health, ecosystem viability, and climate change.

The following summarizes some of the key findings:

Meteorology and Dynamics: In order to support the goals of the MILAGRO field campaign, a large range of meteorological measurements were carried out including: surface operational networks, surface supersites, fixed mobile units, radar wind profilers, lidars, rawinsondes, aircraft platforms and pilot, tethered and Controlled Meteorological (CMET) balloons. These were used to characterize the meteorological episodes in terms of emission transport. Numerical modelling during MILAGRO showed that mesoscale models such as WRF and FLEXPART do a sufficiently good job of simulating pollution transport that they can be further used for interpreting atmospheric chemistry measurements. Observations and modelling studies show that under most conditions, pollutant export from the basin is relatively rapid and that pollutant carryover from day to day is not a major factor in the valley's photochemistry. The overall synoptic conditions and boundary layer circulations were similar to those reported by MCMA previous studies and consistent with prior climatology. Meteorological measurements at the surface and aloft coupled with measurements of trace gases and aerosols indicate that the synoptic-scale transport of the Mexico City pollutant plume was predominantly towards the northeast, although regional-scale circulations transported pollutants to the surrounding valleys and basins on some days. Drainage flows at night have a strong impact on air pollution transport and accumulation in the basin leading to high pollutant concentrations. The MCMA has been found to be more similar to Houston, Texas than to Los Angeles, California in that poor air quality days can result early in the day from emissions on a day of weak venting. Fast response actions should therefore be considered with a focus on nighttime and morning emissions.

Emission Measurements: Emissions studies confirmed that motor vehicles play a major role in supplying the NO_X and VOC precursors that fuel MCMA's extremely active photochemistry. Key findings include a vastly improved speciated emissions inventory from on-road vehicles, showing that the MCMA motor vehicles produce abundant amounts of primary PM, elemental carbon, particle-bound polycyclic aromatic hydrocarbons, carbon monoxide and a wide range of air toxics, including formaldehyde, acetaldehyde, benzene, toluene, and xylenes. Measurement and modelling studies suggest that PM emissions may be severely underestimated in the current emissions inventory estimates. Several innovative techniques were developed in 2003 and used again in 2006 to evaluate the official emission inventories used in air quality models. On-road vehicle fleet emission indices in fleet-average mode for various vehicle classes and driving speeds were obtained using a mobile laboratory. The feasibility of using eddy covariance techniques to measure fluxes of volatile organic compounds and aerosols in an urban core was demonstrated, proving a valuable tool for evaluating local emissions inventory. Together, these techniques have allowed the re-evaluation of the emissions inventory used in air quality models with observations of the concentration of many species and fluxes for a few.

Volatile Organic Compounds (VOCs): The studies have provided a much better understanding of the sources and atmospheric loadings of VOCs, including the first spectroscopic detection of glyoxal in the atmosphere and a unique analysis of the high fraction of ambient formaldehyde from primary emission sources. Liquified petroleum gas (LPG) use continues to be an important source of low molecular weight alkanes. Evaporative fuel and industrial emissions are important sources for aromatic VOCs and methanol in the basin. There are also large non-biogenic sources of methanol in the MCMA basin. The two most important measured VOC species in terms of OH reactivity were formaldehyde and acetaldehyde. Aldehydes are major components of the outflow reactivity. They are produced by atmospheric VOC oxidation and some are also emitted directly. In spite of their importance, these compounds are not measured routinely.

Photochemistry: The studies have significantly improved the characterization of ozone formation and its sensitivity to emission changes in VOCs and nitrogen oxides. The results of MILAGRO demonstrate that urban photochemical smog formation in the MCMA is VOC-limited, which is consistent with the results of MCMA-2003; however the degree of sensibility to VOCs is higher during MCMA-2006 due to the lower VOC/NO_X emission ratio and VOC reactivity. Ozone formation in the surrounding mountain/rural area is mostly NO_X-limited, but can be VOC-limited and the range of the NO_X-limited or VOC-limited areas depends on meteorology. Although the formation of the urban photochemical smog is VOC-limited, the reductions of emissions of NO_X can reduce the formation of regional oxidants. Ozone production continues in the outflow for several days, due to the formation of peroxyacetyl nitrates (PANs) that effectively increase the NO_X lifetime.

Ambient Particulate Matter: The studies also provide a much more extensive knowledge of the composition, size distribution and atmospheric mass loadings of both primary and secondary fine PM; and an improved understanding of the evolution and the radiative properties of aerosols. PM_{10} and $PM_{2.5}$ concentrations in the urban area were about double the concentrations in the rural areas surrounding Mexico City. Mineral matter and secondary inorganic ions each made up approximately 25% of the $PM_{2.5}$ in the urban areas, with the remaining $PM_{2.5}$ mass being comprised of largely carbonaceous aerosol. The dominant sources of carbonaceous aerosol were secondary organic aerosol, biomass burning, and vehicle exhaust emissions (soot). The impact of biomass burning on the aerosol outflow from the region was much larger than on the surface concentrations inside the city. During MCMA-2003 and again in 2006, SOA production was observed to grow very rapidly during sunlight hours – far faster than current atmospheric models or laboratory simulation experiments with the expected precursor gases can explain. Results from MILAGRO 2006 indicate that SOA formation from primary semivolatile and intermediate volatility precursors has the potential to close the gap in predicted vs. measured SOA. However these predictions are poorly constrained by the data and more specific measurements are needed in future campaigns.

Aerosol Evolution and Radiative Effects: Results from both ground-based and airborne measurements confirm that the MCMA plumes are significant sources of both primary and

secondary aerosols at the regional scale and black carbon and SOA are contributing to single scattering albedos in the MCMA and downwind that are substantially smaller than in other areas (such as the eastern United States). Studies from MILAGRO have reported significant enhanced UV-Visible absorption from biomass burning, SOA, and aged carbonaceous aerosol components. At the T0 and T1 surface sites single scattering albedos (SSA) were frequently in the 0.7-0.8 range with some early morning values having even lower SSA. This is consistent with high absorbing aerosol concentrations from both fossil and biomass burning sources during MILAGRO. Aerosol contributions from biomass burning sources contained both black carbon and oxidized organics that yielded enhanced UV absorption. This observation indicated biomass burning activities can have important impacts on the absorption or heating by carbonaceous aerosols in megacity (urban) as well as regional scales. Oxidized organics from primary fires and from secondary aerosol formation were also found to have strong absorption in the 300-400 nm region that leads to enhanced optical absorption by these aerosols over that anticipated from black carbon alone.

The results from LIDAR and aircraft operation as well as aerosol mass spectrometers all indicate that there is significant transport of aerosols and that most of this aerosol is in the lower layer of the atmosphere, but can be exported aloft into the free troposphere during venting events. Satellite retrievals of aerosols are being improved by comparisons with measurements of radiation and aerosol properties at the surface and from aircraft. Measurements of surface albedo and reflectance in the MCMA showed that many urban surfaces are more reflective than assumed in common satellite retrieval algorithms, and that use of larger visible surface reflectance in algorithms can produce more accurate retrieved aerosol optical depth (AOD).

As described above, a very large number of instruments were used in the MCMA during MILAGRO for both ground-based and aircraft measurements; and some innovative instruments and measurement techniques were deployed for the first time. The MILAGRO campaign has shown the synergy of using multiple measuring platforms, instrumentation, and data analysis techniques for obtaining an improved understanding of the physical and chemical characteristics of emissions in a megacity. Furthermore, the deployment of a significant number of advanced instruments, many operating with sensitive, fast (~1 s) response times, along with a large number of established air quality monitoring instruments deployed on aircraft and at surface sites, as well as onboard several mobile laboratories, have provided significant opportunities to intercompare and evaluate a number of instruments in a highly polluted environment.

Despite the use of many advanced PM techniques during MILAGRO, some questions remain unanswered or strongly debated and should be the focus of further research. For example, the fraction of dust due to road resuspension vs. natural sources is unclear. The impact of gas-particle reactions is important, for example for nitrate uptake into the coarse dust mode, but needs to be further investigated to reach a quantitative understanding, including through 3D modelling. The identities of industrial sources of metals and organic aerosols and of the urban chloride sources remain unclear. High time-resolution quantitative analyses of dust and metals may yield very useful information for source identification.

The 2006 MCMA emissions inventory underestimates primary $PM_{2.5}$ and needs to be updated with the information arising from MILAGRO and other studies. Forest fire $PM_{2.5}$ appears to be underestimated by an order of magnitude in the official MCMA inventory, but perhaps overestimated about two-fold on a custom satellite-based inventory used in 3D modelling. The impact of some primary organic aerosol sources such as food cooking, biofuel use, and open trash burning may be important, but remains poorly characterized. Some differences in the apportionment of biomass burning PM between different approaches were observed and need further research, as these techniques together represent the state of the art for source apportionment. The differences in the relative oxidation of organic aerosols in urban vs. background samples between different techniques need to be further investigated.

The influence of “hot” sources of radiocarbon in aerosols needs to be further investigated as it could bias assessments of fossil vs. modern carbon. SOA from traditional precursors such as aromatics is much smaller than the observed SOA in the Mexico City urban area, but the dominant

sources of anthropogenic SOA are still poorly characterized. SOA from biomass burning sources, although not dominant in the city, remains poorly characterized and appears to be under predicted by traditional models.

Measurements of aerosol optical absorption in Mexico City and downwind also benefitted from the variety of techniques applied during MILAGRO. However, not all results are in perfect agreement. Future campaigns need increased focus on spatiotemporal coincidence between different techniques, to help resolve these questions. There are also persistent differences among different satellite retrievals of aerosols, as well as between results from satellite and suborbital techniques; this is an area that requires continued effort.

In summary, the MILAGRO Campaign was designed to investigate the extremely vigorous atmospheric photochemistry of Mexico Megacity. Review of the published results has already improved significantly our understanding of the meteorological and photochemical processes contributing to the formation of ozone, secondary aerosols and other pollutants and their transport and transformation. Key findings have been presented at international conferences as well as communicated to Mexican government officials. We anticipate new results from MILAGRO will continue to contribute to our understanding of megacity emissions and their potential impacts on human health, ecosystem viability, and climate change on urban, regional, and even hemispheric scales. This information will provide the scientific knowledge for decision makers in Mexico to design effective policies as well as provide insights to air pollution problems in other megacities around the world.

7.6 SAEMC/ADAPTE

More than 80% of South Americans live today in urban centres. These centres accumulate wealth but also environmental problems, which poses for South Americans a distinct vulnerability in a changing climate. Until recently though, these efforts have been largely decoupled from international efforts. With support from the Inter American Institute for Global Change (IAI, http://www.iai.int), the South American Emissions Megacities and Climate (SAEMC, http://saemc.cmm.uchile.cl) and Adaptation to health impacts of air pollution and climate extremes in Latin American cities (ADAPTE) studies are now underway. Working teams were developed around five research axes: 1) emissions, 2) chemical weather forecasting, 3) modelling tools, including grid and high performance computing, 4) aerosols and 5) health impacts.

Background and rationale

Except in Brazil, Earth System Science capabilities in South America are sparse and oftentimes isolated. This also applies to more specific air pollution science and expertise. Typically, research teams consist of a few scientists and students, who deal with probably too broad a range of subjects and with funding related to short-term consultancies and small scientific grants (<100 kUSD/yr). Policy makers have technical secretaries and personnel who are also overwhelmed by intricate bureaucratic procedures, funding, and political pressures. Therefore, environmental policies oftentimes, in lack of local knowledge tend to mimic experiences from elsewhere, usually from the US Environmental Protection Agency (EPA). At first, this provides some success in improving air quality but it is absolutely insufficient when dealing with less obvious and much more expensive measures, for instance those related to traffic emissions control and urban planning considering a changing climate.

In almost all of our large urban centres, multimillion US dollar investments are made in air quality monitoring, building of emission inventories, aerosol characterization, health and ecological impact assessments, etc. However, databases are difficult to maintain, lack appropriate calibration and reporting procedures, etc. As the connection between policy-making and research is usually made on the basis of short-term consultancies, the establishment of necessary synergies and the study of more complex issues are hampered and at times air pollution scientists are not able to provide independent opinions to environmental authorities.

Again, except in Brazil, material and human resources for global change research in South America on a country-by-country basis are far too small to allow a significant contribution to international programmes and to produce sustained impacts on local development. Nevertheless, global change science cannot be approached solely from a global perspective. On the contrary, turning points are to be found and faced locally. To illustrate this, let's consider the apparently simple issue of conciliating local and global emission inventories. Today's global chemistry-climate models that provide climate change scenarios are based on emission estimates for year 2000 and projections, which are far from reflecting the actual development of large cities that were not relevant for the year 2000. Local inventories are available for most cities in South America but have yet to be included in the emission inventories used in global models. Is it a matter of not sharing information? Yes, to some extent due to too small an overlap of communities, something that is fortunately improving year by year, but also because it is non-trivial combining different scales in a non-linear system [*Alonso et al*., 2010].

All in all, it is clear that there is a mutual benefit for scientists and policy makers in combining efforts and exchanging perspectives, beyond short-term consultancies and perhaps creating consortia that provide a common but independent framework for sound science and policy making. It is also obvious that the understanding of global change requires global but more importantly local knowledge that is not feasible to obtain without the participation and leadership of local scientists. This in turn requires integration of local scientist into global programmes and investment in local capabilities, both human and material. Following is a description of SAEMC/ADAPTE, which demonstrates what can be achieved in four years with less than 1 million dollars.

Project description and highlights

Measurement campaigns in Santiago, Bogotá, São Paulo, Lima and Buenos Aires were performed, which resulted in locally representative emission factors for vehicles, making it possible to create the first consistent inventory for vehicular emissions for Buenos Aires [*D'Angiola et al*., 2010]. Also, tools were developed to obtain disaggregated emissions [*Saide et al.*, 2009]. Past and future emission scenarios were completed for Argentina and have been initiated for other cities. Evaluation tools based on data assimilation techniques were developed and applied at the local and the regional scales [*Hoelzemann et al.*, 2009; *Saide et al.*, 2011]. Reconciliation between global and local inventories was also achieved [*Alonso et al*., 2010]. All of these data have been reviewed and are becoming available for the whole community and are increasingly being integrated into global databases.

At the beginning of the project, there was a well-established regional transport model especially developed for addressing dispersion and impacts of biomass burning in tropical South America [*Freitas et al.*, 2005]. Regional weather services had some experience in numerical prediction for physical weather but none in chemical weather forecasting. Today, there is a fully coupled model that provides operational chemical weather forecasting for South America [*Freitas et al*., 2009; *Longo et al.*, 2009] and produces regionally relevant information for local applications. Also, both at the Chilean and Peruvian Weather Services, there are chemical weather forecasting tools available and operational as well as dedicated teams to run the models. Furthermore, the basis for a community model is now feasible via the use of an already installed but very under-exploited high-speed Internet connection among South American countries and the rest of the world [*D'Almeida et al.*, 2008].

Existing air quality monitoring networks generally provide mass concentration for the different cities. However, to address health and climate impacts, and even identifying emitters or any process understanding, one needs morphological and speciation information. This in turn requires sophisticated instrumentation and analytical capabilities that are very expensive. For a marginal cost the project compiled and evaluated information on the chemical and physical characterization of aerosols from Buenos Aires, São Paulo and Bogotá [*Vasconcellos et al*., 2011]. This was made possible thanks to coordinated sampling and sharing of analytical resources including trace metals (Buenos Aires), organics (São Paulo) and ions (Bogotá). Local and specific studies were also performed [*Dos Santos et al.*, 2009].

Cities and climate are coevolving in a manner that could place more populations at risk from exposure to extreme temperature and air pollution. Urban areas of Latin America are projected to be increasingly affected by heat waves; yet, we do not know how vulnerable the urban population is to the health impacts that may result from this increase. ADAPTE gathered, validated, and analyzed data on temperature, air pollution and vulnerability allowing measuring changes in the Relative Risk of health outcomes such as mortality due to changes in temperature and air pollution and maps of differential patterns of vulnerability within the urban centres [*Romero-Lankao et al*., 2011]. Also, in São Paulo, advances were made to combine air quality models and health statistics [*Martins et al*., 2010].

The project was very successful in connecting South American scientists and students and also at establishing a two-way connection between scientists in South America and "First World" centres. Very importantly, we used the majority of our IAI resources in fellowships for students, some of which are now themselves starting to be leading scientists who can more easily interconnect between countries and between science and policy making. For all cities, we were successful in producing relevant information and implementing tools for scientific understanding and policy making, Bogotá being an outstanding case [*Behrentz et al*., 2009]. We have shown that sharing and using available resources is feasible and that the integrated sum of those is larger than the sum of the individual resources.

Outlook and perspectives for SAEMC/ADAPTE

Continued and future Developments:

- The establishment of community model including shared computing and storage capabilities, which of course does not preclude the use of other tools but puts an emphasis on the establishment and use of local know-how. For instance, which is the adequate aerosol/photochemical model for South American cities where bio-fuel is so prevalent? How do we handle extremely stable boundary layers along the Andes?
- A permanent initiative on emission inventories as a dynamical tool that evolves according to systematic, transparent, and recurrent evaluations.
- Coordination for providing recurrent aerosol and gas phase chemistry characterization for understanding processes and impact assessments. This is especially needed in order to address aerosol-cloud-climate interactions as well as impacts on human health and ecosystems.
- Air quality monitoring is probably among the most resource intensive activities that is needed. Therefore, it is crucial to have tools to facilitate decision making and to actually optimize the design of monitoring networks. Variational and statistical methods have been explored to design and evaluate monitoring networks.
- From the climate change aspects, South American countries need a feasible forecasting tool customized to the complex issues that face South America and relevant associated processes to provide the forecasts to governmental and civil societies. Important questions to be addressed, among others, are: how is global warming and land use/land cover changes affecting the air quality of the densely urbanized areas? What is the impact of aerosols and the urban heat effect on the hydrological cycles on local and regional scales? What is South America's contribution to global changes?

Other projects in South America

There are at least two initiatives that deal with South American cities. One is a pilot project focusing on Santiago but that is expected to be replicated elsewhere in South America ((http://www.risk-habitat-megacity.ufz.de/). Another one is the "Clean Air Initiative in Latin American Cities" (CAI-LAC) driven by the World Bank (http://www.cleanairnet.org). Risk Habitat Megacity (RHM) is a six year (2007-2013) joint initiative between Chilean and German researchers. It is an 'Initiative and Networking Fund' of the Helmholtz-Association. RHM approaches several megacity issues, such as Land use management, Socio-spatial differentiation, Energy, Transportation, Air quality and health, Water resources and services, and Waste management. The programme comprises ten topics: three "cross-cutting concepts" – Sustainable Development, Risk, and

Governance – and seven "fields of application" (Suppan and Schmitz, pers. comm.). CAI-LAC started in 1998, and it was re-organized in 2006. The original purpose was to establish a comprehensive approach to addressing air quality challenges in Latin America. The initial phase supported the development and enhancement of clean air action plans in large urban areas throughout Latin America, enhanced scientific knowledge and understanding of urban air quality and its associated impacts on human health, and provided decision makers with tools for assessing policy options. According to their web page, the restructuring of the original phase of the initiative was envisioned to revitalize efforts by, among other things, creating a forum for strategy and project development, as well as for channelling training, technical assistance, and information exchange at a regional level. Focal points of this initiative reside in environmental agencies and are largely disconnected from academia.

7.7 CAREBEIJING

CAREBEIJING (Campaigns of Air Quality Research in Beijing and Surrounding Regions) was an international collaborative research project to study the regional transport and transformation processes of air pollution that impact air quality in Beijing and to formulate air pollution control strategies for 2008 Beijing Olympics and the long-term strategies for the region. CAREBEIJING was funded by Beijing Council of Science and Technology and coordinated by Tong Zhu of Peking University, it has conducted three campaigns in 2006, 2007, and 2008, with active participation of more than 200 scientists and graduate students from 21 research institutes in Asia (mainland China, Hong Kong, Taiwan, Japan, Korea), Europe (Germany, Italy), and USA.

Beijing is a megacity with a population close to 20 million and air pollution is a serious concern. With 14 stages of air pollution control in the last decade, the air quality in Beijing has been significantly improving. Yet this improvement is undermined by rapid increases in the number of vehicles and energy consumption in Beijing, as well as the regional transport of air pollutants from highly industrialized area surrounding Beijing. The objectives of CAREBEIJING included:

1. To learn about the current environmental conditions of the region, including social and economical factors, air quality, and emission sources.
2. To identify the transport and transformation processes that lead to air pollutants surrounding regions to impact air quality in Beijing.
3. To calculate the impact of the surrounding regions on air quality in Beijing.
4. To formulate policy suggestion for air quality attainment during the 2008 Beijing Olympic game.
5. To propose objectives and strategy of air quality attainment in 2010 in Beijing.
6. To design a regional air quality management framework and propose policy suggestions for regional air quality control.
7. To evaluate the effectiveness of air quality control policies.
8. To evaluate the health impacts of air pollution before and during the 2008 Olympics.

To achieve these objectives, CAREBEIJING conducted intensive field campaigns based on ground, aircraft, and satellite observations. The data was used to validate emission inventories and regional air quality models (Figure 10).

The campaigns in 2006, 2007, and 2008 all served a different purpose:

CAREBEIJING-2006:	To understand the transport and transformation process of regional air pollution
CAREBEIJING-2007:	To evaluate air quality control policies proposed for the 2008 Olympics based on the findings of CAREBEIJING 2006
CAREBEIJING-2008:	To evaluate the effectiveness of the air quality control policies and their impacts on health

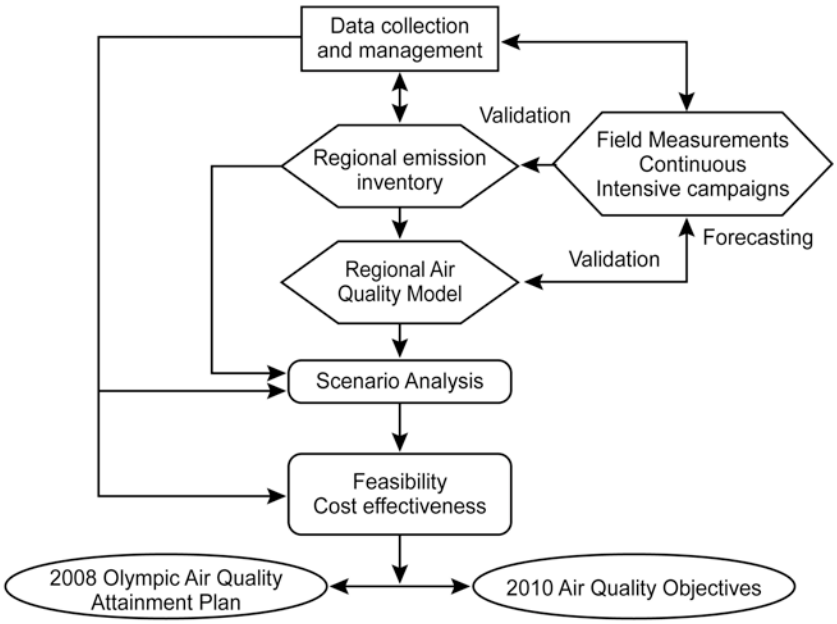

Figure 10 - Research components of CAREBEIJING

Major findings are being published in two special issues in *Journal of Geophysical Research-Atmospheres* (http://www.agu.org/journals/jd/special_sections.shtml?collectionCode=CARBS1&journalCode=JD) and in *Atmospheric Chemistry and Physics* (http://www.atmos-chem-phys.net/special_issue198.html) as well as in other peer-reviewed journals. Based on the findings from CAREBEIJING and other related research projects, air quality control policies were formulated and the final implementation plan was approved by the Chinese central government. A series of aggressive measures to reduce pollutant emissions in Beijing and in surrounding areas were taken before and during the 2008 Beijing Olympics and Paralympics (July 25 – September 17, 2008) in order to ensure the substantially improved ambient air quality. In addition to local efforts in Beijing, five cities and provinces surrounding Beijing also implemented strict air quality control measures on area and point sources. Many heavy-polluting factories were ordered to reduce their operating capacities or completely shut down during the Olympics. Construction activities were all paused. Power plants near Beijing were required to reduce their emissions.

As an important component of CAREBEIJING was a health study with two panels assessing cardiovascular and respiratory responses of susceptible populations (CAREBEIJING-H) to air quality improvements were conduced. Using environmental exposure rates, time series, panel studies, and toxicological experiments, it was shown that due to the improvements in air quality during the Beijing Olympics the number of emergency room visits due to cardiovascular disease as well as the biomarkers of respiratory inflammation were reduced.

During the Olympics, significant reductions of NO_X, SO_2, CO, BC, $PM_{2.5}$, and O_3 were observed. The level of reduction ranged from 10% to 60% depending on the pollutant and what baseline concentration was used. The Beijing Olympics has been widely recognized as one of the most successful and exciting games in Olympic history. Improving air quality of Beijing certainly contributed to the overall success. The experience from the large-scale regional efforts of CAREBEIJING provides a valuable lesson for other megacities facing similar air pollution problems as Beijing.

More information can be found at the web site of CAREBEIJING: http://ceh.pku.edu.cn/carebjindex.html

7.8 IMPACT

1. O_3 and aerosol studies in the Tokyo Metropolitan Area (TMA)

Large amounts of reactive gases and aerosol are emitted from urban areas. Megacities, including the Tokyo Metropolitan Area (TMA), are very large, concentrated sources of these species, which affect O_3 and aerosol levels on local, regional, and global scales [*Molina and Molina*, 2004; *Ramanathan et al.,* 2007]. The uncertainties in emission estimates of these air pollutants are generally large for Asia [*Streets et al*., 2006; *Ohara et al*., 2007] or do not assess the TMA [*Kannari et al.*, 2007].

The increased levels of pollutants have a large impact on regional air quality, nutrient deposition patterns, and climate. In order to assess the impacts of anthropogenic species emitted from these megacities on surrounding areas, we need to understand quantitatively the key processes involved in the oxidation of primary species and the fate of the oxidized species near the source regions (Figure 11). In addition, clusters of megacities lead to accumulation of O_3 and aerosol through large-scale mixing. Reactive species with elevated concentrations in urban outflows can also interact with species emitted from natural sources surrounding the megacities. It should be noted here that O_3 and aerosol are coupled due to similar sources, photochemical interactions (e.g., UV changes by aerosol), and transport.

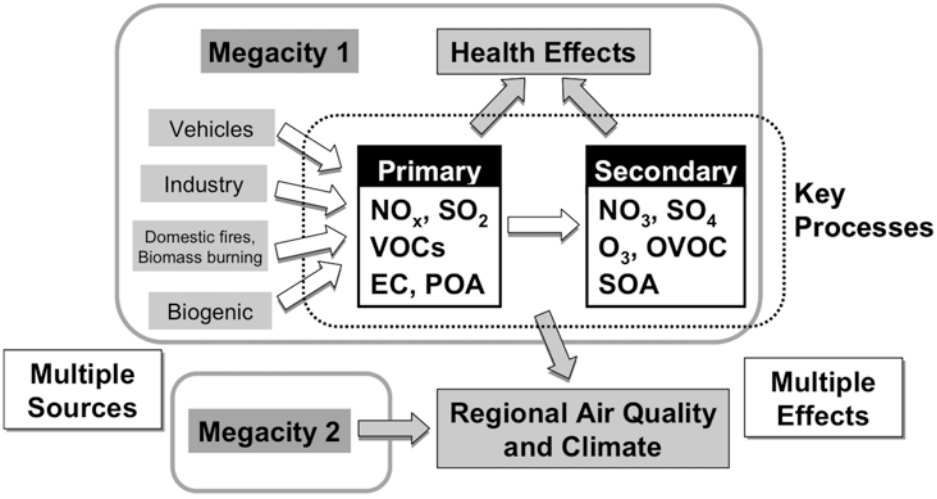

Figure 11 - Schematic diagram of key processes of O_3, aerosols, and their precursor gases near megacity regions

2. Integrated Measurement Programme for Aerosol and oxidant Chemistry in Tokyo (IMPACT)

Studies on characterizing primary emission and secondary formation of aerosols in the Tokyo Metropolitan Area (TMA) have to date been very limited. Further observational studies of aerosols, especially organic aerosol, near source areas are needed in order to improve the understanding of the amounts and chemical composition of aerosols emitted from these large source areas. As a result, the Integrated Measurement Programme for Aerosol and oxidant Chemistry in Tokyo (IMPACT) campaign was conducted. The specific goal of IMPACT was to improve the understanding of atmospheric chemistry in the TMA [*Kondo et al.*, 2010]. IMPACT was conducted within the framework of the International Global Atmospheric Chemistry Project (IGAC), Mega-Cities: Asia. The major objectives of IMPACT were to:

- Characterize the temporal and spatial changes of aerosols, oxidants, and their precursors, primarily through surface measurements near and downwind of urban centres
- Characterize the composition, mixing state, and physical properties of aerosols in urban air
- Evaluate emission inventories of trace gases (e.g., NO_X, SO_2, NH_3, and VOCs) through comparisons of ratios of concentrations of trace species observed in urban air

The instruments used during IMPACT are described in detail elsewhere [*Kondo et al.*, 2006; *Takegawa et al.*, 2006a]. The observation sites were located at the Research Center for Advanced Science and Technology (RCAST), Komaba, Tokyo (35°39'N, 139°40'E), and the Center for Environmental Science in Saitama (CESS), Kisai, Saitama prefecture (36°05'N, 139°33'E), which is located about 50 km north of Tokyo (Figure 12). The locations of the observation sites allowed chemical processes of HO_X radicals and O_3 formation [*Kanaya et al.*, 2007; 2008] and emissions and transformation of primary aerosol, especially BC [*Kondo et al.*, 2006; *Shiraiwa et al.*, 2007], secondary aerosol, cloud condensation nuclei (CCN) activity, and the hygroscopicity of aerosol [*Kuwata et al.*, 2009; *Kuwata and Kondo,* 2009; *Mochida et al.*, 2006; 2008] to be studied. Important findings made by IMPACT are summarized in Table 1.

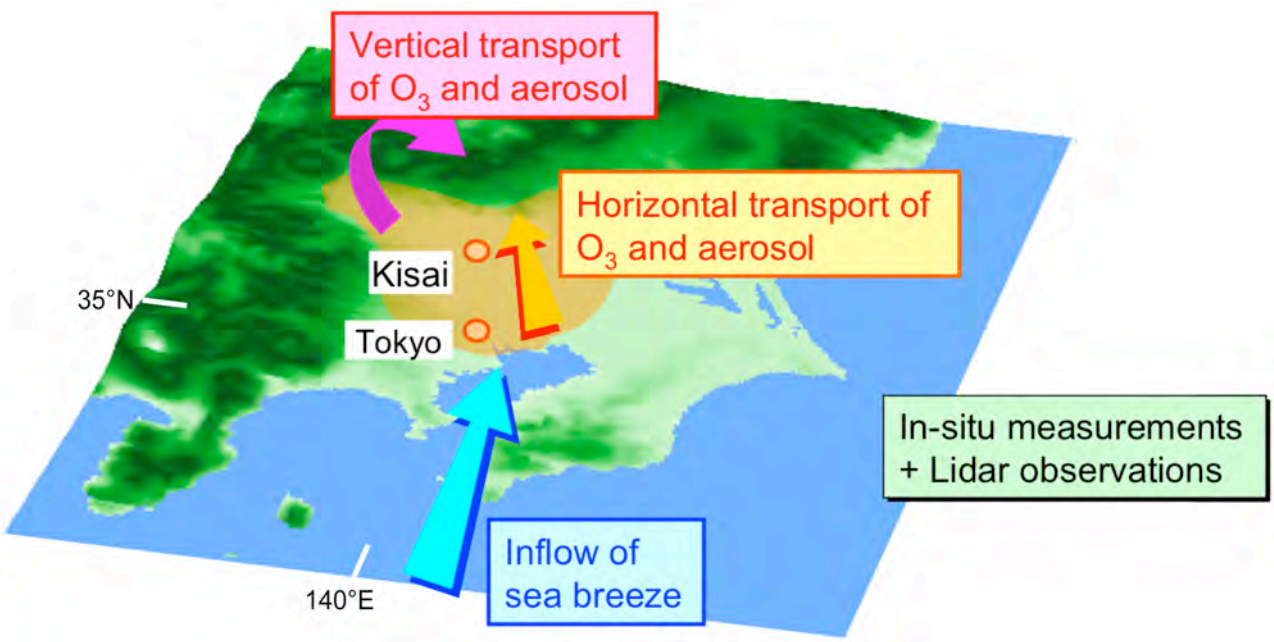

Figure 12 - Transport of pollutants from the urban centre (Tokyo) to suburban areas by the sea breeze

3. Perspectives and future studies

The methodology used for IMPACT is schematically shown in Figure 13 and proved to be very successful for several reasons. First, a combination of a limited number of fully instrumented observational sites (super sites) and monitoring stations of fundamental species proved to be cost-effective for a megacity study. During the IMPACT experiments, two super sites were set up (Figure 12); one was located near the urban centre of Tokyo (RCAST) and the other was about 50 km downwind (CESS). Because of the typical transport time of 3-7 hours, time-resolved measurements provided quite useful information on the chemical evolution of polluted air. Second, instrumentation for vertical sounding, including lidar, were also useful because they provided changes in the boundary layer height that could be related to corresponding changes in the concentrations of various species measured at the surface. Third, regional-scale model calculations were evaluated by the data obtained by the measurement system mentioned above. Model calculations are useful in improving our understanding of chemistry and transport processes over the entire TMA.

The framework of IMPACT and its subsequent scientific findings can be very useful to help understand atmospheric chemistry and transport in other megacities in Asia, especially considering that many of them are also located in coastal areas, i.e., Beijing, Shanghai, Hong Kong, Guangzhou, and Seoul.

Table 1 - Summary of findings by IMPACT

Topic	Major findings	References
Instrumentation	A HNO_3-CIMS with a new calibration/zero system was developed. Performance of an Aerodyne AMS was evaluated based on intercomparison with a PILS-IC and Sunset OC. A new method to quantify BC coating was developed using an SP2.	[*Kita et al.*, 2006; *Takegawa et al.*, 2005; *Moteki and Kondo*, 2007; 2008]
$OH/HO_2/O_3$	Measured OH was reproduced by a box model in winter and summer. Measured HO_2 was underestimated by the model in the high-NO_X regime.	[*Kanaya et al.*, 2007; 2008]
VOCs	Seasonal variation of C_2-C_7 non-methane hydrocarbons was quantified.	[*Shirai et al.*, 2007]
BC	Diesel emissions were identified as a major source of BC in Tokyo.	[*Kondo et al.*, 2006; *Shiraiwa et al.*, 2007]
Nitrate	Nitrate-HNO_3 partitioning in summer was influenced by vertical mixing.	[*Morino et al.*, 2006]
Sulphate	Only a small fraction (3% in winter and 18% in summer) of SO_2 emitted was converted to sulphate.	[*Miyakawa et al.*, 2007]
Organic aerosol	Seasonal and diurnal variation of POA and SOA was quantified. Most SOA (OOA) was found to be water soluble.	[*Takegawa et al.*, 2006a; 2006b; *Miyazaki et al.*, 2006; *Kondo et al.*, 2007; 2008; *Matsui et al.*, 2009]
Hygroscopicity and CCN	Hygroscopicity and CCN activity were perturbed by organics. BC can act as CCN with a small amount of coating material.	[*Mochida et al.*, 2006; 2008 *Kuwata et al.*, 2009; *Kuwata and Kondo*, 2009]
Outflow from Tokyo	Significant formation of organics and alteration of BC mixing state took place within ~0.5 days in summer	[*Takegawa et al.*, 2006b; *Shiraiwa et al.*, 2007; *Miyakawa et al.*, 2008]

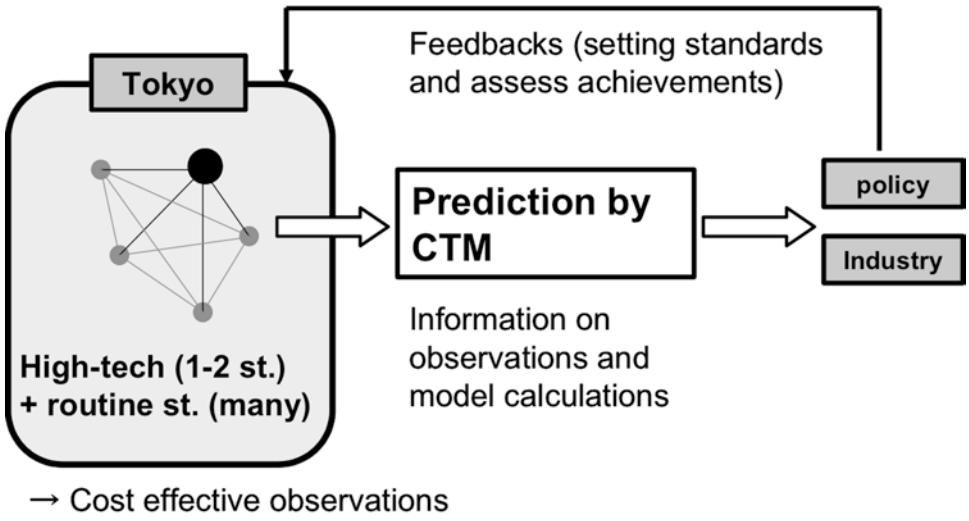

Figure 13 - System for megacity air quality studies: Observations at 1-2 main stations (st.) and routine monitoring stations, modelling, and science-policy feedback

7.9 PRIDE-PRD

Pearl River Delta (PRD) is one of the three city-clusters in China that has experienced extremely fast urbanization and industrialization. The breathtaking speed of economic growth for over three decades in PRD, however, has been accompanied by an increase in air pollution, namely elevated concentrations of ozone O_3 and fine particulates $PM_{2.5}$. The concurrent high concentrations of O_3 and $PM_{2.5}$ together with primary pollutants has led to rather unique pollution characteristics due to interactions between primary emissions and photochemical processes, between gaseous compounds and aerosol phase species, and between local and regional scale processes.

PRIDE-PRD (The Programme of Regional Integrated Experiments on Air Quality over Pearl River Delta of China), sponsored by the Ministry of Science and Technology of China (MOST) during 2003-2008, was developed to investigate in depth the air pollution problem and to improve the understanding of the chemical and radiative processes in PRD. The main objectives were to:

- Characterize the temporal and spatial distribution of the concentrations of aerosol, oxidants, and their precursors by ground based (routine monitoring network and super-sites), airborne and satellite measurements
- Understand chemical composition, size distribution, hygroscopic properties, and optical properties of aerosols
- Quantify the contribution of precursors to the formation of oxidants and secondary aerosols by measurements and modelling
- Study the interactions between aerosols and gases through measurements of precursors of aerosols and oxidants as well as by modelling
- Determine source-receptor relationship across cities within the PRD as well regional contribution to PRD air pollution
- Define the regional mitigation strategies and technical options to keep the air pollution load in PRD within sustainable limits in terms of ecological and human health effects

PRIDE-PRD consisted of two campaigns: PRIDE-PRD2004 during October 1 and November 4, 2004 and PRIDE-PRD2006 during July 4 and July 31, 2006. An international science team from China (mainland, Taipei, and Hong Kong), Germany, Japan, and Korea was involved in both campaigns. Yuanhang Zhang from Peking University and Liuju Zhong from Guangdong Provincial Environmental Monitoring Center coordinated PRIDE-PRD. The campaigns of PRIDE-PRD were truly a team effort, bringing together more than 100 scientists, staff and students from 20 institutes, to operate two well-equipped super-sites, aircraft measurements, meteorological soundings, as well as 16 routine sites of the PRD regional air quality monitoring network (Figure 14).

Figure 14 - Overview of PRIDE-PRD campaign in 2004 and 2006

The successful accomplishment of the PRIDE-PRD campaigns deepened the understanding of regional air pollution and its environmental effects in the PRD. The results were published in a special issue of *Atmospheric Environment* for PRIDE-PRD2004 and in *Atmospheric Chemistry and Physics* for PRIDE-PRD2006 as well as in other peer reviewed journals, i.e., *Science, JGR, and GRL*. The important findings made by PRIDE-PRD are summarized in Table 2.

PRIDE-PRD plays a very important role in the decision making process of the PRD local government to implement regional management on basis of multi-pollutants and multi-objectives of air quality. As one of consequences, Guangdong provincial government recently launched China's first clean air initiative programme in PRD with focus on regional photochemical smog and regional haze. Meanwhile, MOST and Guangdong provincial government set up a major project called "Synthesized Prevention Techniques for Air Pollution Complex and Integrated Demonstration in Key City-Cluster Region (3C-STAR)" to help the PRD authority to build up the capacity of regional air pollution control strategies, regional coordination mechanisms, and related joint actions. A 3-D regional measurement network and real-time ensemble air quality forecasting system will be in use by the end of 2010, providing the basis for regional air quality management and further fundament research on the complex air pollution in this region.

Table 2 - Summary of findings by PRIDE-PRD

Topic	Major findings	References
Overview	Introduction to PRIDE-PRD and major finding in PRIDE-PRD2004 campaign	[*Zhang et al*., 2008a]
Instrumentation	An instrument for measurement of atmospheric peroxy radical by chemical amplification	[*Li et al*., 2009]
Boundary meteorology	Three inversion layers were identified	[*Fan et al*., 2008]
OH/HO_2/O_3	HO_X measurements and unknown recycling Regional high O_3 mainly comes from chemical production O_3 production and its sensitivity to VOCs and NO_X	[*Hofzumahaus, et al*., 2009; *Lou et al*., 2010; *Lu et al*., 2010; *Wang et al*., 2010; *Zhang et al*., 2008b]
VOCs	VOC speciation and source apportionment.	[*Liu Y. et al*., 2008; 2008b; 2008c; *Wang J.L. et al*., 2008]
H_2O_2 and PAN	High concentrations were found in secondary pollutants	[*Hua et al*., 2008; *Wang et al*., 2010]
HONO	High concentration was found during the daytime and emissions from unknown sources with a photo-enhanced mechanism were estimated	[*Qin et al*., 2009; *Su et al*., 2008a; 2008b]
Vertical profiles of air pollution	High loadings of air pollution were found and the boundary layer height was determined by LIDAR	[*Ansmann et al*., 2005; *Mueller et al*., 2006; *Wang W. et al*., 2008; *Wendisch et al*., 2008; *Sugimoto et al*., 2009; *Tesche et al*., 2008; *Tomoaki et al*., 2010]
Aerosol chemical and optical properties	Sulfate and OC/EC were major components of fine aerosol and major contributors to visibility degradation. Relative humidity dependence of aerosol optical properties was identified.	[*Andreae et al*., 2008; *Cheng*, 2008a; 2008b; *Gnauk et al*., 2008; *Garland et al*., 2008; *Jung et al*., 2009; *Liu S et al*., 2008b; *Liu X. et al*., 2008]
Aerosol size distribution	New particle formation phenomenon was found and growth rate was estimated Hygroscopicity of aerosol was measured and numerically simulated	[*Liu S et al*., 2008a; *Gong et al*., 2008; 2010; *Eichler et al*., 2008; *Rose et al*., 2008]
EC	EC mixing state was identified by numerical closure modelling Temporal variation of EC was studied	[*Cheng et al*., 2006; *Verma et al*., 2009]
Aerosol water soluble ions	Nitrate-HNO_3 partitioning in equilibrium was identified. water-soluble organic carbon aerosols were measured	[*Hu et al*., 2008; *Miyazaki et al*., 2009]
Sulfate SOA	Sulfate production in submicron mode can be explained by OH and SO_2 reaction SOA was estimated by PMF and EC tracer method	[*Xiao et al*., 2009; 2011]

7.10 INTEGRATED FOCUS ON WEST AFRICAN CITIES

C. Liousse [1], C. Galy-Lacaux [1], E. Assamoi [1], A. Ndiaye [2], B. Diop [3], H. Cachier [4], T. Doumbia [1, 2], P. Gueye [2], N. Marchand [5], A. Ehgere [5], A. Baeza [6], S. Val [6], I. George [6], V. Yoboué [7], L. Sigha [8], J.P. Lacaux [1], B. Guinot [1], J.F. Léon [1], R. Rosset [1], P. Castéra [1], E. Gardrat [1], C. Zouiten [9], C. Jambert [1], A. Diouf [2], O. Koita [3], I. Annesi-Maesano [10], A. Didier [11], S. Audry [9], A. Konaré [7]

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[3] *Université de Bamako, Mali*
[4] *LSCE, Gif sur Yvette, France*
[5] *LCP, Université d'Aix Marseille, Marseille, France*
[6] *LCTC Université Paris Diderot-Paris 7, France*
[7] *LAPA, Université d'Abidjan, Ivory Coast*
[8] *CRH, Yaoundé, Cameroon*
[9] *LMTG, Toulouse, France*
[10] *Faculté de Médecine Saint-Antoine, Paris, France*
[11] *Hôpital Larrey, Toulouse, France*

Integrated Focus on West African cities (Cotonou, Bamako, Dakar, Ouagadougou, Abidjan, Niamey): Emissions, Air Quality, and Health Impacts of gases and aerosols in the frame of AMMA and POLCA international programmes.

West African cities are largely devoid of dedicated air quality monitoring stations, together with no significant application of regulations. However fossil fuel and biofuel emissions of gases and particles are highly emitted from traffic, transportation, domestic fires, wastes, tailings and industrial activities. These emissions result in very poor still degrading air quality, exacerbated by the weather and climate conditions, as can be readily realized on site. Clearly, this picture is still darkening due to rapid sprawling urban growth into African megacities. Ensuing air quality degradation thus raises severe public health problems such as respiratory and cardiovascular ailments.

The POLlution in the African Capitals (POLCA) programme under the IDAF (IGAC DEBITS Africa) framework jointly with the Laboratoire d'Aérologie (LA) as well as several African laboratories took place in eight African capitals (Abidjan, Dakar, Bamako, Niamey, Ouagadougou, Bangui, Brazzaville, Yaounde). The preliminary results have highlighted the pressing urban pollution problem in West Africa. As shown in Figure 15, measurements showed high concentrations of NO_2, SO_2, etc. that regularly exceed the WHO air quality limits [*Liousse and Galy-Lacaux*, 2010].

Within the African Monsoon Multidisciplinary Analysis (AMMA) and MOUSSON programmes (2005 - 2009), field studies were organized for the short term (May 2005) in Cotonou, Benin and for the long term (2007-2009) in Ouagadougou (Burkina Faso) focusing on gas and aerosol concentration measurements. These programmes have also revealed quite high particulate matter levels. In addition, combustion emission inventories have been developed for African fossil and biofuel emission sources [*Assamoi and Liousse*, 2010] for 2005 and 2030 (Figure 18).

This context has proven to offer a unique opportunity to develop the POLCA programme into a trans-disciplinary study between emissions, air quality and health impacts in West African cities. POLCA is now supported by the CORUS programmes (http://www.ird.fr/fr/science/dsf/corus/english_termes_reference.pdf) under the auspices of the French Ministry of Foreign Affairs (MFA), in partnership with African organizations.

Within this framework, we have constructed a new integrated methodology that includes: (1) combustion emission characterizations; (2) joint experimental determination of gas concentrations (such as NO_2, SO_2, NH_3, HNO_3, O_3, COV), size differentiated aerosol chemistry (total mass) (Figure 16) [*Doumbia et al.*, 2011], organic fraction (PAH), carbonaceous particles, inorganic particles (sulphates, nitrates…) and elements traces (iron, lead…) from ultrafine to coarse size fractions (Figure 17) [*Val et al.*, 2011], as well as in vitro toxicological measurements (oxidative stress

estimations through citokynes measurements). Some representative individuals (cohort) who live in polluted areas and have experienced long-term exposure at the measurement sites have been selected and will be tested for blood and spirometry analyses. Long-term follow up of diseases are also obtained from local hospitals for epidemiological studies (3) integrated environmental-health modelling. Exposures obtained from modelling associated with satellite investigations are related to a new dedicated aerosol/gas module developed for intake, deposition and clearance of gases and particles in the respiratory tract. Dakar and Bamako were first selected to initiate this study, which will soon be extended to Yaoundé (Cameroon) and Cotonou (Benin). Through extension of the long-term IDAF network to correlated human health problems and air pollution, this integrative programme aims at developing for Africa a new emerging multidisciplinary approach linking environmental changes and health.

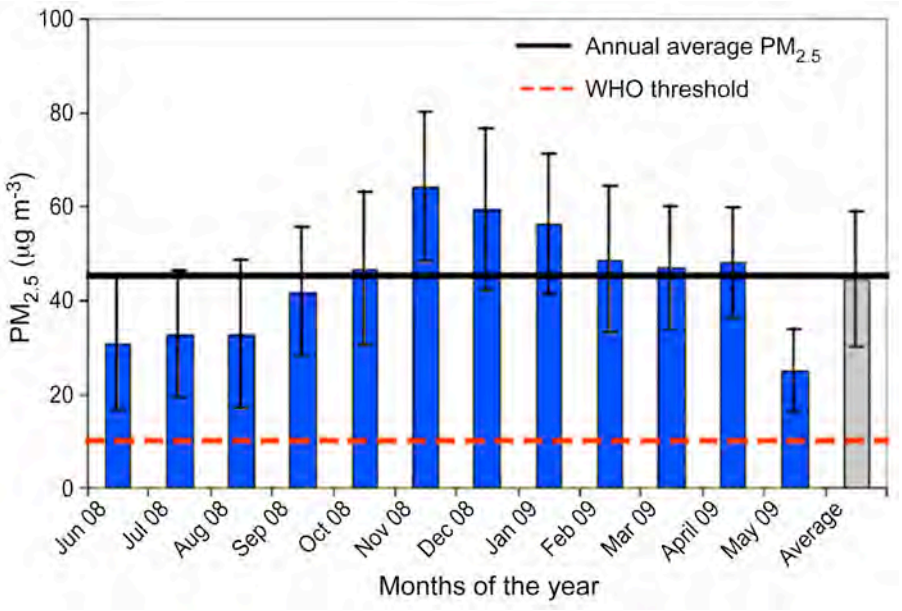

Figure 16 - Monthly $PM_{2.5}$ concentrations in Dakar (from June 2008 to May 2009) [*Doumbia et al.*, 2011]

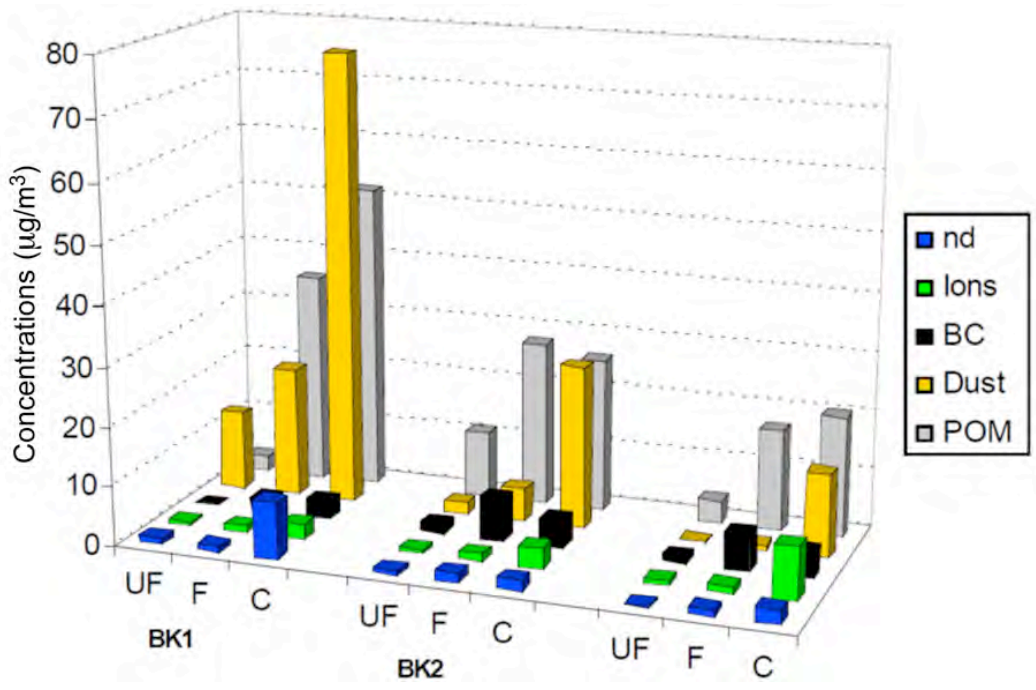

Figure 17 - Size speciated chemical aerosol composition in Dakar and in Bamako during (BK1) and after (BK2) a dust event [*Val et al.*, 2011]

Figure 18 - Picture from Cotonou (Benin)

Keywords: Health-environment-climate; Aerosol and chemistry; traffic; biofuel; health impacts.
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OMP: Observatoire Midi-Pyrénées, IDAF: IGAC/DEBITS/AFrique International Global Atmospheric Chemistry, Deposition of Biogeochemical Trace Species/AFrique, POLCA: POllution of African Capitals (http://www.redgems.org/spip.php?rubrique55), AMMA: African Monsoon and Multidisciplinary Analyses MOUSSON : http://mousson.csregistry.org/tiki-index.php.

7.11 GAW URBAN RESEARCH METEOROLOGY AND ENVIRONMENT (GURME) PROJECT

The main focus of the World Meteorological Organization's GURME project, initiated in 1998, is the important cross-cutting area of urban air quality. GURME addresses the end-to-end aspects of air quality, linking the observational capabilities of GAW with the needs of chemical weather prediction, with the goal of providing decision makers and the general public with enhanced air quality services of appropriate quality. Priority activities include the improvement of observing systems and their integration with urban-scale models and capacity building/training initiatives. Through a training team materials for a basic air quality modelling and forecasting course were developed. These have been modified as per the requirements of each specific training event. GURME provides an international platform for cross-cutting urban air pollution activities, collaborating with other WMO Programmes, international organizations, National Meteorological and Hydrological Services (NMHSs), research institutes, academia and environmental agencies. Participation in MEGAPOLI and several European COST actions has been mutually beneficial. GURME pilot projects have been established for several megacities, below are short descriptions. GURME activities are shown in Figure 19, with the so-called "GURME Figure" in the middle.

GURME activities are guided by the WMO/GAW Scientific Advisory Group (SAG) for GURME, chaired by Prof. Gregory Carmichael.

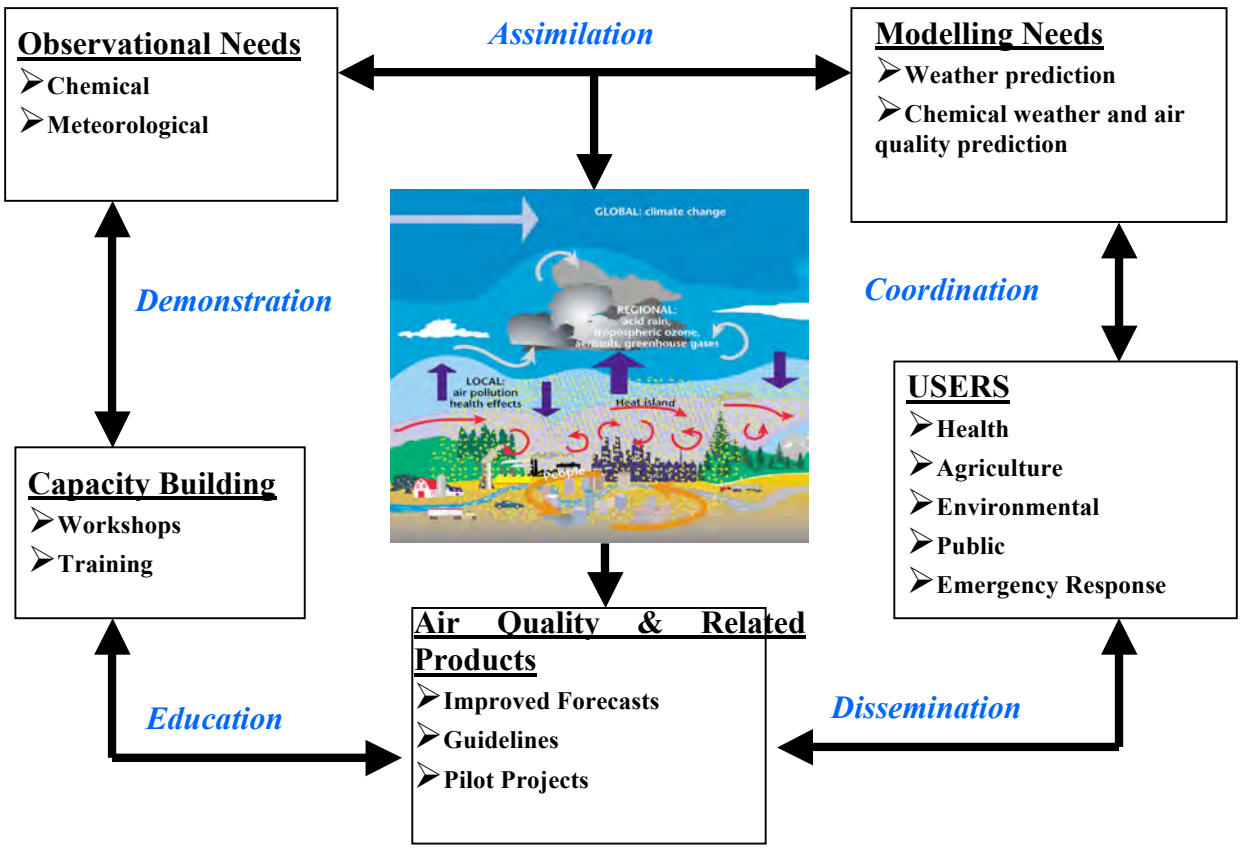

Figure 19 - Main elements and goals of GURME for the strategic planning period 2008-2015

Latin American Cities project

The Latin American Cities project was originally focused on Mexico City, Santiago and Sao Paulo concentrating on capacity development. The project grew out of the GURME Experts Meeting held in Cuernavaca, Mexico in October 2002, which was attended by 25 invited participants from 11 countries. The first Workshop on Air Quality Forecasting in Latin American Cities was held in

Santiago, Chile in October 2003; this was followed by a WRF-Chem and Remote Sensing Workshop in Sao Paolo, Brazil in 2005. The first GURME basic training course on air quality forecasting was delivered in July 2006 in Lima, Peru and included participants from the South American countries, generally from the NMHSs and Environmental Agencies. A comprehensive Training Course on Air Quality Modelling for Latin American Cities Project was held in Mexico City, Mexico in August 2009. Training was held for Central American countries in October 2011. More expert workshops and training courses will be convened in the future as needed. A discussion forum with a network of scientists and groups active in Latin American cities and international experts will be established to facilitate the exchange of information on air quality modelling and forecasting. The main contributors to this project are Luisa Molina, Maria de Fatima Andrade, Paulo Artaxo, Pedro Oyola, Rainer Schmitz and Laura Gallardo.

Beijing project

The Study of Mechanism on Atmospheric Environmental Pollution in Capital Beijing project was established in 1998. Key scientific problems were:

- Environmental and geochemical cycling processes and behaviour of atmospheric pollutants
- The accumulative effect and tolerance of atmospheric pollutants in regional environments
- The prediction theory of the atmospheric environment
- The principles of pollution control on regional scales.

The main topics were:

- The mechanism of the formation of PBL pollution and its effects on the environment in Beijing
- The physical, chemical and ecological function of pollutant between atmosphere and planetary boundary layer
- Formation mechanism of sand-dust and its impact on urban environment in Beijing
- Theory and method of city atmosphere environment pollution monitoring and forecasting
- Atmospheric environment pollution regulation and tackling technique in a comprehensive way.

Shanghai project

Shanghai faces multiple hazards, it is frequently affected by natural hazards such as typhoons, storm surges, heavy fog, heat-waves, and also by atmospheric pollution. Shanghai Expo – successfully launched new air quality and related services as a component of the Expo Multi Hazard Early Warning System (MHEWS) Demonstration Project. The project under the leadership of Xu Tang, Director General, Shanghai Regional Meteorological Center, CMA, is designed to explore the full potentials of urban life in the 21st century. Key elements of the project are to disclose the physical and chemical mechanisms during the formation, transportation and transformation processes of the main air pollutants in the Shanghai area, to establish an air pollution prediction system for the Shanghai area, and to improve the assessment techniques of environmental quality. The observation capacity was increased considerably. Based on various measurements (meteorological and environmental observations, diseases diagnostics) new and comprehensive forecast products to support public health services were developed and demonstrated during Expo. The services include those for air quality, pollen, food poisoning and heat stroke, among others. Through these services effective actions to protect the health of individuals, especially those in sensitive groups, will be designed. The project continues with further development of the public health services focus.

Commonwealth Games New Delhi

The Indian Institute of Tropical Meteorology (IITM), Pune, a constituent under the Ministry of Earth Sciences, Government of India, with Gufran Beig as the co-ordinator, led the country's first major initiative on air quality forecasting, named as "System of Air Quality forecasting and Research (SAFAR)". It has been successfully tested during the commonwealth Games 2010 for National Capital Region Delhi. Currently SAFAR is being spread to other major cities in India.

SAFAR provides location specific information on Air Quality in near real time and it is forecast 24 hours in advance. It is complemented by the weather forecasting system designed by IMD, New Delhi. The ultimate objective is to increase the awareness among the general public regarding the air quality in their city, well in advance, so that appropriate mitigation action and systematic measures can be taken up for the betterment of air quality and related health issues.

More information is available on the GURME webpage at: http://mce2.org/wmogurme/

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CHAPTER 8 - KEY ISSUES AND OUTLOOK

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The population growth, economic and industrial development, and rising standard of living in the world's megacities will bring not only new problems but also new opportunities. There are manifold challenges that accompany megacity growth: providing food, shelter, transportation and other goods and services for an ever-increasing population. Surmounting these challenges brings obvious problems that affect the health and welfare of the urban population as well as societal and ecological environment in areas well beyond the urban centre. At the same time, megacities are often looked upon as the economic engines of the world. As such, they also represent a concentration of resources that provide opportunities to address these challenges more efficiently than possible if the same population were dispersed in smaller cities and rural environments. A special section entitled *Cities* in the 8 February 2008 issue of *Science* explored these issues [http://www.sciencemag.org/content/319/5864.toc]. The goal of this chapter is to highlight some of the key questions and issues that have been identified regarding the challenges and opportunities that accompany the emergence and evolution of megacities. This chapter is not intended to be comprehensive or detailed. Each of the issues will require significant future analysis that will be published elsewhere.

8.1 THE SCALING LAW OF AIR POLLUTION AND HEALTH EFFECTS OF URBAN POPULATION

It is generally recognized that ambient concentrations of air pollutants are higher in large urban areas than in smaller cities. However, given the difficulty of comparing measurements from different cities due to differences in instrument siting, meteorological conditions, and many others, there is little quantitative information regarding how urban pollutant concentrations depend on population. Satellite measurements can now provide such information. Figure 1 shows the vertical column abundance of NO_2 measured by the SCIAMACHY satellite over ten western US cities [*Kim et al*., 2009]. Since the predominant source of NO_2 is emissions from anthropogenic activities, which are predominately located at the Earth's surface, NO_2 is largely confined to the near surface layer of the atmosphere, and the vertical column abundance is proportional to the urban ambient concentrations. In addition to the apparent day-of-week cycle in the figure, it is clear that larger cities (Los Angeles, San Francisco) generally have higher NO_2 columns than less populated cities. Figure 2 is a log-log plot of the relationship between the average weekday NO_2 column as a function of the population of the urban area (population taken for 2005 from a US census bureau web site: http://www.census.gov/popest/metro/CBSA-est2009-annual.html).

The linear relationship shown in Figure 2 suggests that the relationship between urban NO_2 column and population can be approximately captured as a scaling law in the form of a power law function of population, as has been found for many urban relationships [*Bettencourt et al*., 2007]:

$$column\ NO_2 = N^{\beta} \qquad \text{eq 8.1}$$

i.e., many diverse urban properties increase as the population size (N) raised to an exponent b, which is between 0 and 1. The line in Figure 2 indicates the linear least-squares fit to the points

and the slope of that line gives the numerical value of b, which in this case is 0.35. The correlation coefficient for the fit is 0.77 and the y-intercept on that plot allows the calculation of 3.2 x 10^{15} cm^{-2} for the estimated column NO_2 for an average western US city with a population of 1 million. Similar analyses have been done (L. Lamsal, R. Martin, Dalhousie University, private communication) for hundreds of cities in the US, Europe, China, and India. Each of these four regions gives similar results with values of b between 0.34 and 0.61, correlation coefficients between 0.58 and 0.72, and intercepts indicating average satellite-derived surface NO_2 concentration for a city of 1 million inhabitants increasing from India (0.35 ppbv) through China (1.0 ppbv) and Europe (1.2 ppbv) to the US (1.4 ppbv). There is a great deal of scatter about these relationships from a variety of causes, e.g. differences in population density, energy use, topography, meteorology, etc. between cities as well as differences in satellite retrievals over different environments. In addition it is likely that different pollutants may have significantly different scaling exponents, i.e. values of b. Nevertheless, this scaling relationship provides a context in which to consider the population dependence of ambient air pollutant concentrations in different societal environments.

The scaling law in Equation 8.1 allows consideration of the dependence of the total health impact integrated over the population of an urban centre. Since a pollutant concentration increases with population with an exponent b, the total integrated exposure increases faster than proportionally to population. Combining this dependence with the dose/response relationship allows a first order estimate of the integrated health effects. The consideration of the dose/response relationship has not been done here, but these considerations suggest that increasing population exposed to increasing pollutant levels yields health impacts that increase rapidly with population, significantly more rapidly than directly proportional to the population.

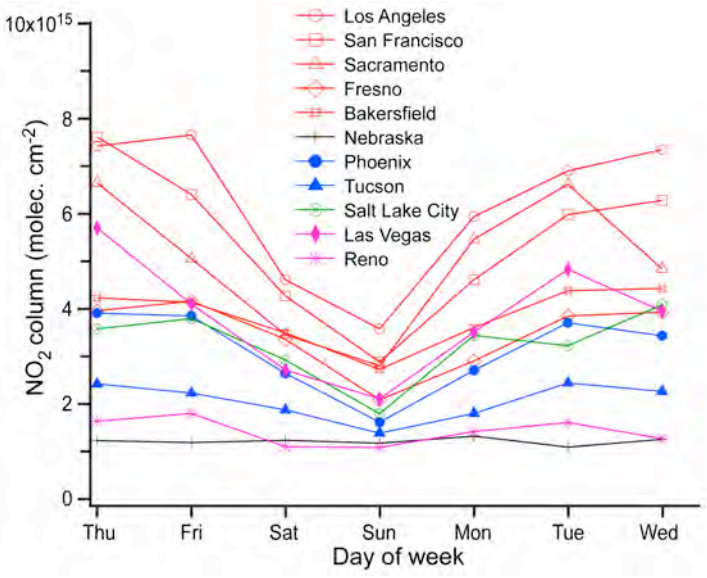

Figure 1: SCIAMACHY satellite derived tropospheric NO_2 columns over urban areas in the western US for 2003 - 2007 May - September averages [*Kim et al.*, 2009] (Also shown in Chapter 5 as Figure 7)

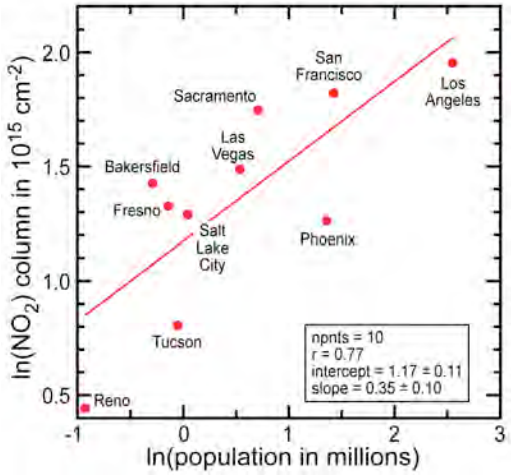

Figure 2 - Log-log plot of the estimated average weekday (Mon – Fri) NO_2 column from the data included in Figure 2 as a function of urban area population

8.2 COMPARISON OF AIR POLLUTANT CONCENTRATIONS, THEIR TEMPORAL EVOLUTION, AND THEIR SOURCES ACROSS DIFFERENT MEGACITIES

Comparing air pollutant concentrations across different megacities, and comparing how the concentrations have evolved with time, provides useful information regarding the sources of the pollutants and the likely effectiveness of control strategies. Figure 3 compares annual mean concentrations of PM_{10} recorded in 26 representative urban areas. Large differences are obvious, with concentrations varying by more than a factor of 10. However, even the most developed cities approach or exceed WHO air quality guidelines. Since PM is the air pollutant with the most significant health effects, understanding the differences between concentrations observed in different cities is critical. *Gurjar et al.* [2008] have proposed a multi-pollutant index that evaluates and ranks megacities in terms of their ambient air quality. This index considers the combined concentrations of PM_{10}, SO_2, and NO_2, and can be used to monitor air quality changes over time, and relate these changes to the—often rapidly—changing state of a megacity. Local meteorology, local sources, and long-range transport all play roles in observed ambient air pollutant levels. The development of a locally tailored control programme that adequately considers all relevant factors is an expensive and lengthy process. Insightful comparisons between cities can certainly provide initial and ultimately cost-effective guides to implementing effective PM control measures before such a programme is ultimately put in place.

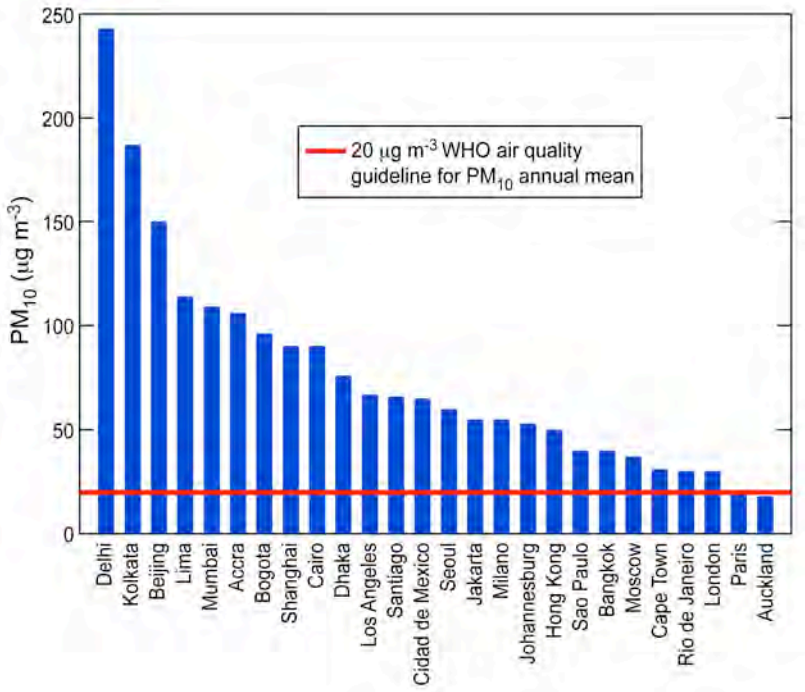

Figure 3 - Comparison of annual mean concentrations of PM_{10} across 26 urban areas. These data were collected from a variety of papers and websites, some of which are cited in other chapters of this book [*L. G. Klenner*, private communication]. It should be recognized that these data were derived from monitoring systems that differ widely in their character, and thus cannot provide definitive comparisons among urban areas

A comparison of chemical composition of PM_1 across different cities provides a rich source of information regarding the sources responsible for the ambient PM loading. As shown in Figure 4, throughout the Northern Hemisphere, sulphate contributes a major portion to the ambient aerosol, but organics generally make an even larger contribution. Nitrate makes a generally smaller, but quite variable contribution, and ammonium is nearly always present in significant amounts. Such data sets are quite informative, but must be interpreted with caution. The results

in Figure 4 were collected through deployments of aerosol mass spectrometers, which measure only non-refractive particles with a diameter less than 1 µm. Consequently, they are insensitive to black carbon, dust, and sea salt and thus overestimate the contributions for the chemical components they do measure. Figure 5 (also shown below) presents a more complete speciation of PM in Mexico City, which can be compared to the Mexico City result in Figure 4 (note the different colour coding in the two figures.)

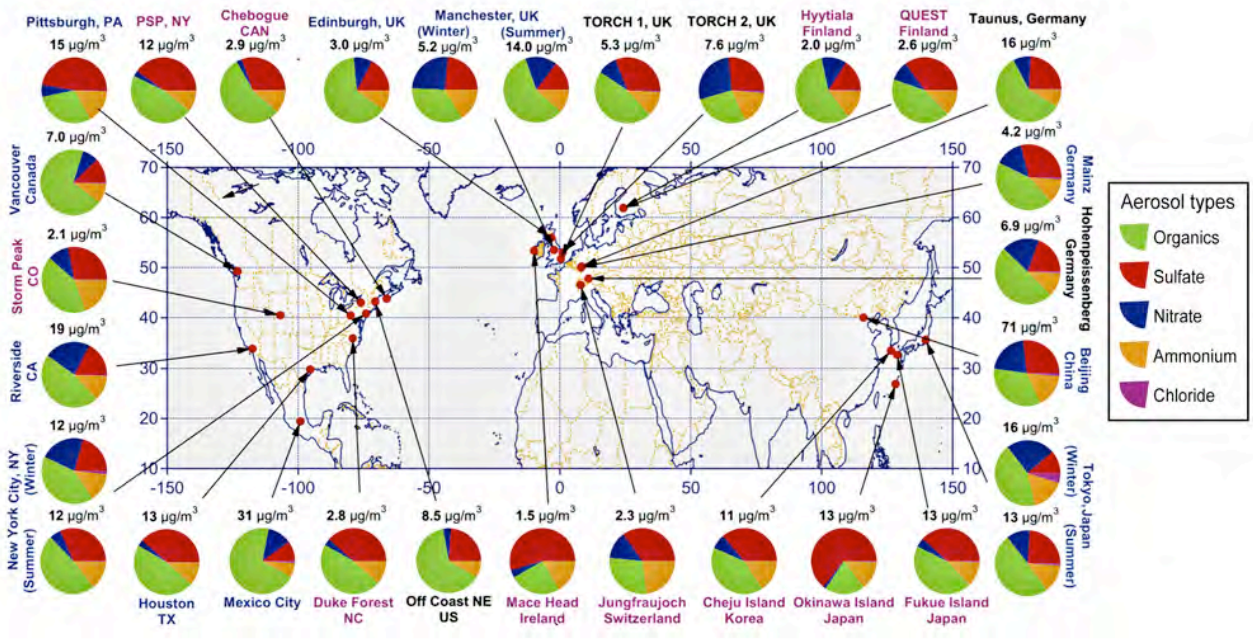

Figure 4 - Comparison of composition of non-refractive PM_1 at many sites in the Northern Hemisphere [*Zhang et al.*, 2007]. Pie charts show the average mass concentration and chemical composition: organics (green), sulphate (red), nitrate (blue), ammonium (orange), and chloride (purple). Colours for the study labels indicate the type of sampling location: urban areas (blue), <100 miles downwind of major cites (black), and rural/remote areas >100 miles downwind (pink). (Also shown in Chapter 1 as Figure 6)

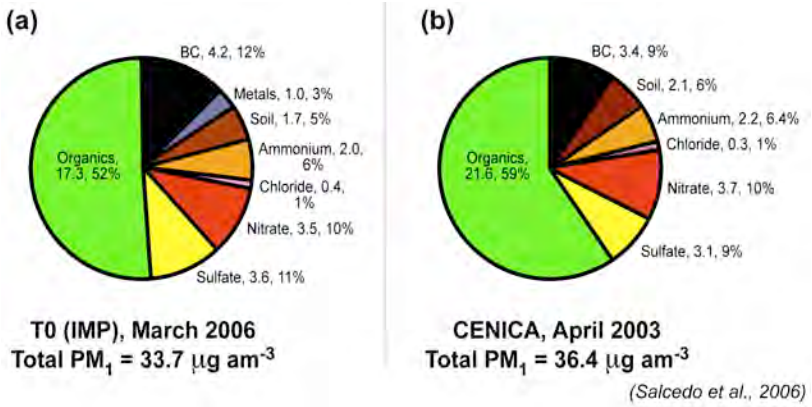

Figure 5 - Submicron PM composition (mass and percent) measured during (a) March 2006; (b) April 2003 field campaigns at surface sites in the Mexico City basin. [From *Aiken et al.*, 2009] (Also show in Chapter 5 as Figure 18)

The sources of organic aerosol (OA) are currently the subject of vigorous debate in the scientific community. The consensus is that most OA in rural areas is not directly emitted, but rather formed in the atmosphere from condensation of oxidized VOCs (secondary organic aerosol or SOA) [*Zhang et al.*, 2007]. Radiocarbon dating of OA typically shows a very high fraction of modern carbon, suggesting that most SOA is formed from biogenic VOCs [*Bench et al.*, 2007]. On the other hand, SOA typically correlates very well with the oxidation products of anthropogenic VOCs, suggesting that urban emissions play an important role in its formation [*de Gouw et al.*, 2005]. Several studies indicated that there is much more SOA in urban air than models can account for [*de Gouw et al.*, 2005; *Volkamer et al.*, 2006]. Figure 6 summarizes some of the recent research aimed at explaining the observed levels of SOA using detailed chemical models. The graph shows that models underestimate SOA by 1-2 orders of magnitude over a wide range of the degree of photochemical processing of an air mass. Given the relatively poor understanding of the sources of OA, it is very difficult to predict how a reduction in the emissions of precursors will affect the levels of OA in the atmosphere. Since OA accounts for such an important contribution to total PM levels, understanding OA sources is a high priority for establishing effective PM control strategies.

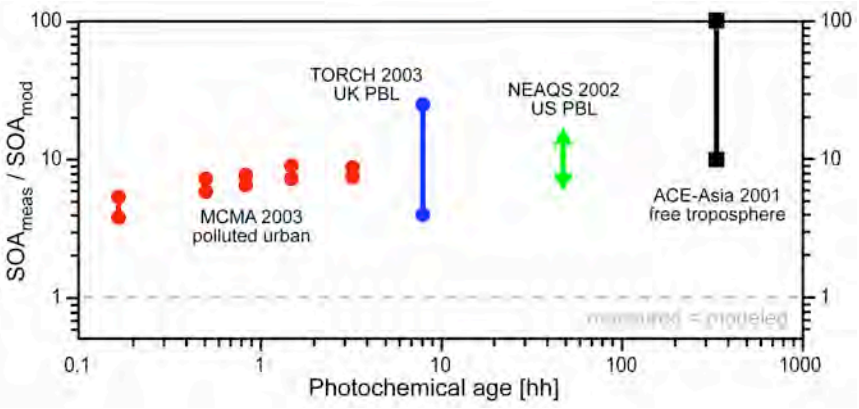

Figure 6 - The ratio between measured and modelled secondary organic aerosol (SOA) as a function of photochemical age, i.e. the degree of photochemical processing, of a polluted air mass [*Volkamer et al.*, 2006]

It is important to note that currently developing megacities can benefit from the experiences of megacities that developed earlier. An excellent example is provided by comparison of the evolution of maximum ozone and PM concentrations in three similarly sized megacities (Figure 7). Ozone concentrations peaked in Los Angeles about 1970 and have decreased over the following four decades. PM concentration measurements started later, but seem to have followed a similar trajectory. In Mexico City, a later developing megacity, the ozone concentrations apparently never reached the peak concentrations observed in Los Angeles and dropped more rapidly, approaching Los Angeles concentrations. PM concentrations in Mexico City have approximately paralleled the ozone concentrations. Evidently Mexico City avoided some to the most severe air pollution problems experienced in Los Angeles through implementation of emission controls before problems became so severe. In contrast, pollutant concentrations in Beijing may be following a different trajectory; PM concentrations are decreasing, but ozone concentrations, which have been low, are evidently rising. Increased attention is being paid to air quality concerns in Beijing, and it will be very enlightening to see how ozone concentrations evolve there in the future.

Correlations between ambient concentrations of air pollutants can yield important information regarding source emissions. For example, *Parrish et al.* [2009a] report ambient measurements of hydrocarbons, carbon monoxide, and nitrogen oxides from three megacities (Beijing, Mexico City, and Tokyo) and compare them with similar measurements from US cities in the mid-1980s and the early 2000s (Figure 8). The common hydrocarbon pattern seen in all data sets indicates that emissions associated with gasoline-fuelled vehicles dominate in all of these cities. This commonality suggests that vehicular emission controls are important to begin as soon

as possible in the growth of vehicle fleets in emerging megacities. Over the three decades covered by the US data sets, the hydrocarbon emissions decreased by a significant factor (approximately an order of magnitude), while the ratios of the individual hydrocarbons remained nearly constant (Figure 8a). Differences in the hydrocarbon patterns between cities provide information regarding secondary sources. The ambient CO to NO_X concentration ratio reported for the three non-US megacities are higher than present US ratios (Figure 8b), but lower than those observed in the 1980s in the US (not shown). The one exception to the preceding statement is the very high concentrations of CO observed in Beijing, which apparently have a large regional contribution.

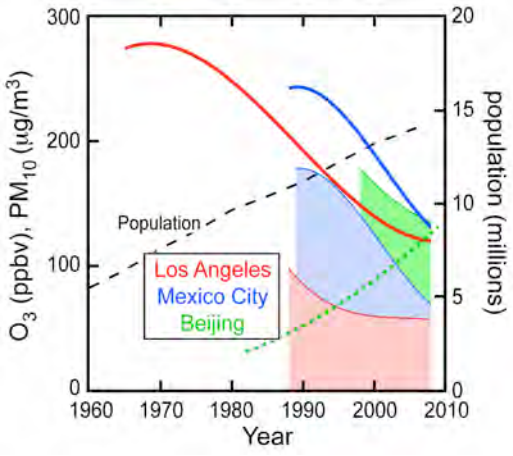

Figure 7 - Comparison of evolution of ambient ozone (solid and dotted heavy lines) and PM_{10} (light lines shaded to zero) concentrations in three megacities of similar populations. Through the time period shown each of the three cities had a population within ± 40% of the 3-city average (dashed black line). The solid lines are polynomial fits to tabulated air quality data. The ozone for each city is the statistic that is used for the US NAAQS (3-year running mean of 4th highest daily maximum 8-hr average.) the dotted line is a fit to the maximum 8-hr average ozone observed in the limited research data sets that are available for Beijing. For Los Angeles PM_{10} is the annual average for the station measuring the highest concentrations; for the other two cities the annual average over all stations in the city's network is given

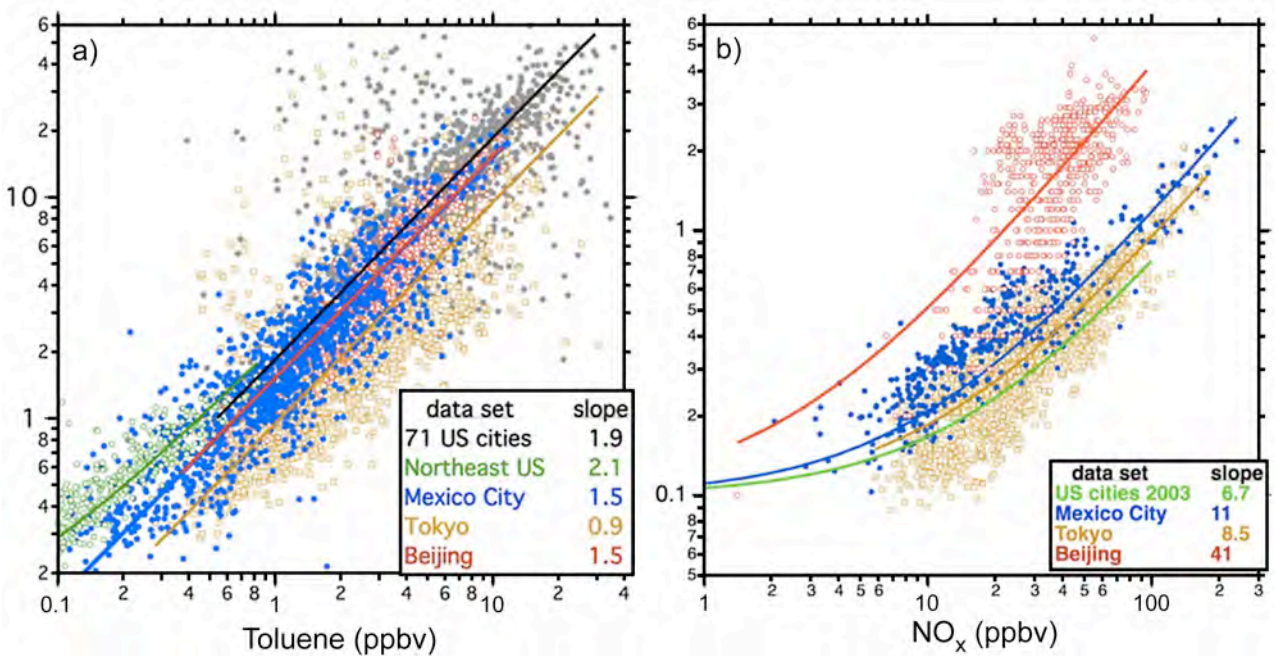

Figure 8 - Comparison of relationships between ambient concentrations of primary pollutants in three megacities. [*Parrish et al.*, 2009a]

During the morning traffic peak, emissions from the on-road vehicle fleet generally dominate ambient concentrations of NO_X, CO, and VOCs in urban areas. Measurements of the ambient concentrations of these species thus provide quantitative information regarding the vehicle emissions. This information can provide "top-down" tests of emission inventories, and useful comparisons of vehicle emissions in different urban areas. For example, Figure 9 compares the CO to NO_X ratio measured in the US (urban average and a specific city in Texas) with results from three urban areas in South America, Paris, Mexico City, and Tokyo.

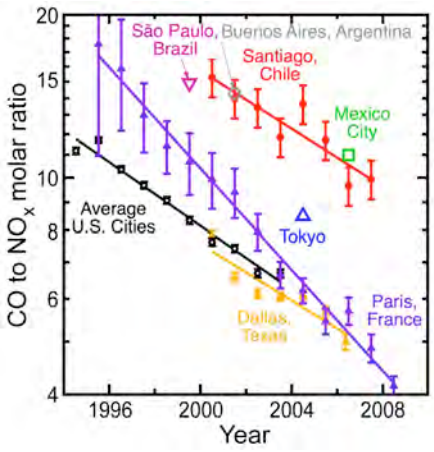

Figure 9 - Evolution of CO to NO_X ratio in several cities. The black points are average US urban data reported to the US Environmental Protection Agency (USEPA), while the other points represent seven example cities throughout the world. The solid lines are linear, least-squares fits to the log-transformed data. USEPA data are from *Parrish* [2006]. Dallas, Texas data are from the Texas Commission on Environmental Quality website (http://www.tceq.state.tx.us/nav/data/aq-data.html). Santiago and Buenos Aires data are from *Gallardo et al.* [2011], and São Paulo data from *Vivanco and Andrade* [2006]. Tokyo and Mexico City data are from *Parrish et al.* [2009a]. Paris data are from the Paris Air Quality Network (AIRPARIF), and were provided by A. Borbon

Decadal scale changes in the CO to NO_X vehicle emission ratio can be followed through the evolution of the measured ambient ratio (linear fits in Figure 9). Interestingly, the ambient ratios in the US data sets, and in Santiago, Chile have decreased at similar rates, i.e. exponential decreases of 6 ± 1% / year, while the decrease in Paris, France has been considerably more rapid (10% / yr). The declining ratio in the US is attributed to the implementation of improved control measures and the replacement of older, higher-emitting vehicles in the fleet [*Parrish*, 2006]. These control measures, especially catalytic converters, are more effective at removing CO than NO_X, so as vehicle fleets are modernized, the CO to NO_X ratio decreases. Likely, a similar evolution of the vehicle fleet has occurred in Paris and Santiago. Importantly, US emission inventories do not capture the observed rapid decrease in the CO:NO_X ratio, so generally modelled emissions are too rich in CO compare to NO_X in urban areas, which are often dominated by these on-road vehicle emissions.

Inter-city differences in the US are attributed to differing average ages of the on-road vehicles. For example the vehicle fleet in El Paso, Texas is significantly older than in Dallas Texas and the US average. The CO/NO_X ratio in El Paso (not shown) follows the same rate of decrease, but the absolute trend is approximately 6 years behind the US average. However, other factors may be important in other cities. The CO to NO_X emission ratio certainly depends on a variety of factors in addition to average vehicle fleet age, including relative fractions of diesel and gasoline fuelled vehicles, extent of ethanol usage (which is particularly high in São Paulo, Brazil), and emission control strategy and schedule of implementation. Importantly, Chilean emission inventories generally underestimate the observed ratios; a preliminary analysis of the data, and results from inverse modelling techniques for CO, suggest that NO_X emissions in Santiago are probably overestimated by a factor 2 to 3 [*Gallardo et al.*, 2011], which accounts for the ambient-inventory ratio differences.

The VOC to NO_X ratio in on-road vehicle emissions is of more critical importance to modelling of photochemical production of ozone and SOA in urban areas than is the CO to NO_X ratio. *Parrish* [2006] showed that VOC emissions from on-road vehicles have decreased at a similar rate to CO emissions. This correspondence is expected because catalytic converters are the principal control measure for both species. Although emission inventories accurately captured the VOC to NO_X ratio in 2000, the subsequent decrease in the ratio was not accurately captured. Consequently, NO_X to VOC emission ratios in US urban areas are likely underestimated by current inventories, which will compromise air quality modelling.

The results presented in Figure 9 represent a limited example of the world's vehicle fleets. It is potentially useful to examine the CO to NO_X emission ratio and its time evolution from vehicle fleets in a variety of megacities. Such an examination promises a wealth of information regarding the emissions from vehicle fleets in those cities, and systematic differences may well provide guidance for developing air quality control strategies in the absence of detailed urban emission inventories.

8.3 IMPORTANCE OF REGIONAL TRANSPORT TO URBAN POLLUTION LEVELS

Presently, many megacities, particularly in Asia, are developing in areas that are densely populated over large regions (e.g. the East China plains and the Indo-Gangetic Plain), and often these megacities are in relatively close proximity to each other. Such cities have a critical new dimension to their air quality concerns – a significant, even dominant, fraction of some pollutants may be transported into the city from the surrounding region, rather than emitted or produced locally. As one example, CO in Beijing behaves very differently compared to other cities. In a typical urban area, CO has a pronounced peak coincident with the morning vehicle traffic maximum, while in Beijing such peaks are much less obvious, and CO concentrations can remain high for days, presumably due to regional accumulation and transport of CO on regional scales. This is reflected in Figure 8b that shows Beijing CO concentrations much higher than observed in other cities at comparable NO_X concentrations. (Regional accumulation and transport are not nearly as important for NO_X, due to its much shorter lifetime.)

Model calculations [*Zhao et al.*, 2009] indicate that under high-pressure systems, which suppress ventilation of pollutants from the boundary layer, elevated ozone concentrations can extend over a major fraction of East China (Figure 10). These results indicate that the high pollutant emissions over the vast stretch of the plains of East China makes the region susceptible to regional accumulation of high ozone concentrations. During the episode illustrated in Figure 10, high ozone extended over an area >1 million km^2, with a population of >800 million people.

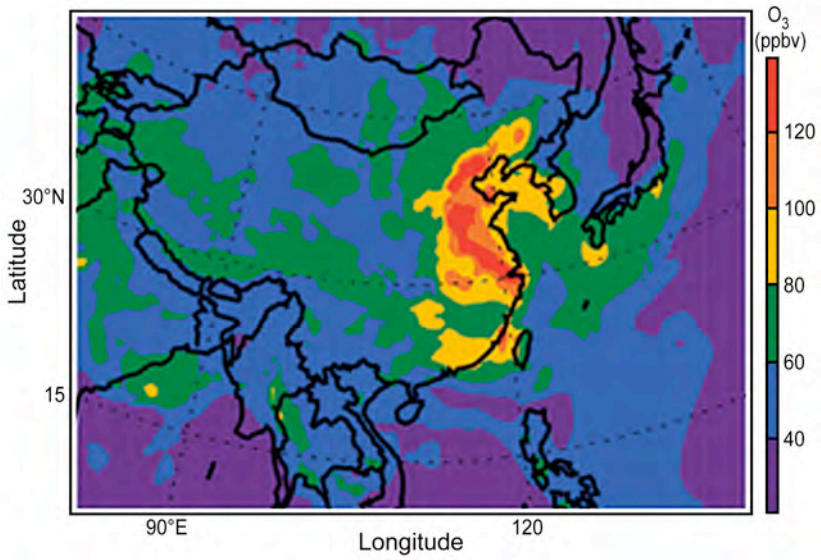

Figure 10 - Modelled ground-level ozone concentrations over East Asia on June 13, 2004 [from *Zhao et al.*, 2009]

It is clear that a single megacity in a large, densely populated region cannot effectively address its air quality issues in isolation; a region wide control strategy is required. The US faced this issue in the 1980s when the Ozone Transport Assessment Group (OTAG) studied this issue. In that country regional transport is significant, but is dwarfed by its significance in Asia. China will have to grapple with its daunting regional transport problem, which will become increasingly important as that country continues to develop, but it may be much more difficult in the Indo-Gangetic Plain, which encompasses much of northern and eastern India, the most populous parts of Pakistan, parts of Nepal, and most of Bangladesh. Mobilizing the required international coordination to develop effective air quality control policies will be difficult in a politically challenging region of the globe.

8.4 AIR QUALITY CONTROL STRATEGIES: URBAN VERSUS DOWNWIND AREAS

VOC and NO_X emitted to the atmosphere are oxidized through photochemical processes that produce both ozone and SOA. The oxidation of anthropogenic emissions of these species is responsible for the presence of elevated concentrations of ozone and SOA in urban areas. Over the past decades, there has been a great deal of debate in the US regarding whether it is cheaper and more effective to control VOC or NO_X emissions. This debate has been largely focused on control of ozone rather than SOA [*NRC*, 1991]. A variety of analyses have argued that VOC controls are more effective and that NO_X control may in fact be counter-productive. These arguments rely upon observationally based analysis, such as weekday-weekend differences [*Blanchard et al*., 2008 and references cited therein], EKMA type analysis [*NRC,* 1991] or even simpler models [*Stedman*, 2004]. Unfortunately, photochemical ozone formation is a complex problem involving emissions from urban cores, transport, distributed anthropogenic as well as biogenic emissions of both VOC and NO_X, etc. It is very likely that the relative effectiveness of VOC and NO_X controls is characteristic of each location, and varies with meteorology.

It is also important to note that the debate described above has generally had a very localized focus – the air quality in the urban area for which emission controls are being considered – and impacts on downwind areas are not considered. If VOC emissions are controlled and NO_X emissions are not, ozone concentrations may well drop in the emission region, but the emitted NO_X may be more efficiently transported to downwind regions, eventually forming a greater amount of ozone than in the absence of the VOC controls. This is an especially important consideration in densely populated regions such as Asia discussed in the preceding section.

To reliably determine the most effective ozone or PM control strategy for a particular urban area or region requires air quality modelling incorporating accurate input from emission inventories, boundary condition characterization and meteorological fields as well as incorporating realistic descriptions of chemical and physical processes [*Song et al.,* 2010]. Arguably, such modelling to the required accuracy is not yet possible for any urban area. Nevertheless, the detrimental health effects of air pollutants in the world's megacities require that air quality improvement efforts must be implemented in lieu of such reliable information. Indeed, such efforts have been implemented and have resulted in great improvement in many cities (Figure 7). Fortunately, reduction of emissions of any important pollutant brings beneficial effects from many perspectives, even if the reductions are not optimized for highest efficiency.

8.5 CONTRIBUTION OF MEGACITIES TO REGIONAL AND GLOBAL CONCENTRATIONS

Studying the impacts of megacities is challenging as they occur on different spatial scales, from the local to the global. Local and regional-scale measurements are crucial to understanding the basic relationships between emissions, current ambient concentrations, and past trends over time. There have been numerous observational studies of air pollution due to megacities on the local to global scale [*Garland et al*., 2008; *Gurjar et al.*, 2008; *Zhang et al.*, 2008]. MILAGRO was a particularly large field programme that coordinated measurements to investigate the impact of emissions from Mexico City on the air quality of the city and its surroundings [*Molina et al.*, 2010,

see Section 7.1]. In addition to the in-situ observations, satellite remote sensing also contributes to megacity research, as is particularly being brought out by the EU CityZen and MEGAPOLI projects. For instance, *Hatzianastassiou et al*. [2009] used MODIS and TOMS satellite data to discriminate between anthropogenic and natural signals in the East Mediterranean, a region influenced by the megacities Istanbul and Cairo (as well as by somewhat smaller cities like Athens, Izmir, and Ankara). Further details of various local-scale in situ observations and satellite retrievals are given in Section 1.5 and Chapters 2-6 of this report.

In order to go beyond our present understanding of the impacts of air pollutants on megacities and to be able to develop future scenarios and information on mitigation strategies, atmospheric models are required. Megacities are a challenging scale problem for models. Due to limited computer power, models that resolve urban air pollution cannot yet be run globally and the provision of consistent boundary conditions for these regional models is not trivial. Global models, on the other hand, can be useful for examining effects such as large-scale transport and the role of large-scale changes in meteorological parameters, but they usually possess too coarse a resolution to be applied in local studies. Although computer power is increasing, the inclusion of ever-increasing complexity in contemporary models largely compensates for the increase in computing power, so that resolution improves only slowly over time.

Numerous modelling studies have examined the local urban-scale effects of megacities; several examples are discussed in the previous chapters. On the other hand, only a few studies have examined the large-scale (regional to global) effects of megacity emissions. The first examination of the effects of megacities on global atmospheric chemistry using a global three-dimensional chemical transport model [*Butler and Lawrence*, 2009] found that the effects on air quality, radiative forcing, and atmospheric oxidation capacity of megacities are disproportionately smaller than the proportion of anthropogenic emissions due to megacities. In contrast, disproportionately large effects of megacities were modelled for reactive nitrogen compounds, in particular PAN (peroxy acetyl nitrate). As an example, Figure 11 shows the effects from

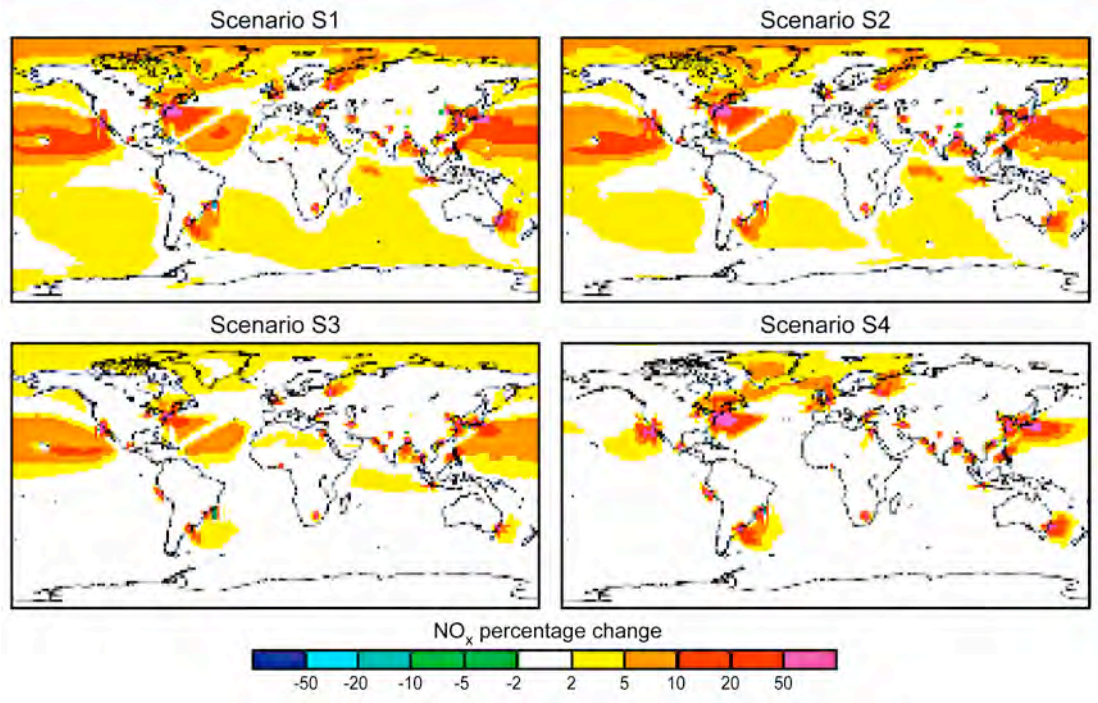

Figure 11 - The percentage change in the global surface NO_X mixing ratio due to megacity emissions, computed by the MATCH-MPIC model [*Butler and Lawrence*, 2009]. Four scenarios were considered, based on the emissions from the simulations for the IPCC-AR4 intercomparison [*Dentener et al*., 2005]: S1 - year 2000 emissions; S2 - projected 2030 emissions based on current emissions control legislation and national expectations of economic growth; S3 - maximum feasible reduction scenario (all currently available emission control technologies); S4 - pessimistic 'worst case' scenario

megacities on global tropospheric NO_X concentrations. Under the low-emission future scenario S2, the influence of megacities is generally reduced, and under the high-emission future scenario S4, although the local influence of megacities is increased, the geographical extent of the influence becomes smaller. In terms of ozone response, the individual model grid cells that contain megacities respond to the megacity emissions differently depending on their latitude. Tropical megacity grid cells generally show increased ozone year-round, while northern extratropical megacities generally show reduced ozone year-round. *Lawrence et al.* [2007] offer another perspective, examining the "regional pollution potential" of emissions from megacities. That study found that the long-range transport characteristics of generic tracers are strongly affected by the megacity location and can be grouped into various regions. Eurasian megacities tend to both retain the most pollution in the region immediately surrounding the city, and also to export the most pollution to the boundary layer more than 1000 km away from the city, while tropical cities, especially from Southeast Asia, tend to primarily export the most pollution to the upper troposphere. Vertical transport, especially by deep cumulus convection, is particularly important for determining the differences between the regions.

8.6 URBAN HEAT ISLAND IN MEGACITIES

Urban structures change the surface of land and sky view factor, and therefore the surface balance of energy, such as the short and long wave radiation, and the sensible and latent heat fluxes. This change together with the anthropogenic heat sources are the main causes of higher temperatures in urban centres than in suburban and rural areas, and hence the formation of urban heat island (UHI).

Urban heat island effect can be described with urban heat island intensity (UHII). UHII is defined as the spatially-averaged temperature difference between an urban area and its surrounding rural area [*Kim and Baik*, 2005]. UHII can be quantified by calculating air or surface temperature differences between an urban area and nearby rural area simultaneously with similar geographic features; satellite imagery has also been used to obtain surface-temperature based UHII over a study area under clear skies. [*Memon et al*., 2009].

High UHII have been reported in studies of megacities, such as 10 °C in Beijing [*Hung et al*., 2005], 8 °C in Tokyo [*Saitoh et al.,* 1995] and Paris [*Lemonsu and Masson*, 2002], and 5 °C in New York [*Gedzelman et al*. 2003]. There is also a trend of temperature increase in large cities. For example, in Osaka the UHII increased from approximately 2.4 °C in 1901 to almost 3 °C after 1981, while the UHIIs of Seoul, Tokyo, and Taipei, have increased by 1 °C to 2 °C [*Kataoka et al.*, 2009].

The higher temperature in urban centres of megacities is associated with higher health risk. It is found that the anthropogenic heat plays an important role for UHII and the boundary layer development, and also has a significant impact on local circulation, such as land sea circulation [*Lin et al.,* 2008]. The formation of an UHI has been shown to have a diurnal variation. For example, over Paris the UHII is stronger at night than during the day thus impacting the structure of the atmospheric boundary layer, which has an important impact on primary and secondary regional air pollutants such as ozone and nitrogen oxide [*Sarrata, et al*., 2006].

With elevated temperatures, UHI has significant impacts on outdoor air quality, especially on the concentration of photochemical oxidants. *Narumi et al*., [2009] used an atmospheric dispersion model to reproduce the observed air pollution conditions for the typical summer day. The results showed that a 1°C increase in temperature leads to an increase of 11% in the maximum photochemical oxidant concentration. A temperature increase of up to 3°C will increase the concentration of photochemical oxidants by 19 ppb. The concentration of photochemical oxidants was 30 ppb higher in the afternoon due to the effect of biogenic VOCs, indicating their strong impact on photochemical oxidant concentrations under UHI condition.

Few studies have discussed the association between UHI and air quality under synoptic patterns. *Lai and Cheng* [2009] analyzed this association in the Taichung metropolis region. The

results show that certain synoptic patterns worsen the air quality and induce the UHI. Under these patterns, the concentrations of air pollutants increase significantly with the UHI intensity. The convergence usually associated with nocturnal UHI causes the accumulation of O_3 precursors, as well as other air pollutants, thereby worsening the air quality that day and also during the following daytime period.

The importance of UHI for the impacts of megacities on climate and air quality and therefore mitigation measures has been well recognized. However, the roles of UHI in the air pollution formation processes and local to regional climate change are not well understood. More studies in this area are needed to formulate policies that have co-benefits of air quality and climate change in megacities.

8.7 CONCLUDING COMMENTS

The preceding chapters have described air quality research results from megacities on five of the world's continents. Notably, there is wide variability of available information - from Los Angeles, certainly the world's most extensively studied city where research has been ongoing for over six decades, to major megacities where very limited ambient measurements or modelling have been conducted. This disparity emphasizes the critical need for both intensive field campaigns and routine monitoring throughout the world.

Nevertheless, experience in earlier developing megacities (e.g. Los Angeles, Mexico City) provides useful guidance for implementing emission control efforts in any megacity, even before extensive measurements and modelling allow formulation of a comprehensive air pollution control programme. Such efforts can yield both immediate and long-term health benefits for the city's population. Initial efforts could include inefficient combustion processes, e.g., open trash burning and cooking or home heating with biofuel and coal. Large industrial facilities and electrical generation plants are important emission sources, and a great deal of general knowledge is available regarding effective control of their emissions and these facilities can also be early targets for air quality improvement. On road motor vehicles are major sources of emissions in all cities, regardless of continent or society, and there is a great deal of similarity between the emissions from vehicle fleets throughout the world. An early introduction of emission controls on new vehicles will sooner begin reducing vehicle emissions through the relatively long process of vehicle fleet turnover.

Effective emission control strategies to improve air quality should also take into account the impact the emission control strategies have on climate. The development of emission control strategies should seek "win-win" solutions for air quality problems that simultaneously benefit climate. Such efforts to optimize the co-benefits of air quality policies for climate will pay dividends at the local, regional and global scales.

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144. Report of the Seventh Session of the EC Panel of Experts/CAS Working Group on Environmental Pollution and Atmospheric Chemistry and the GAW 2001 Workshop (Geneva, Switzerland, 2 to 5 April 2001) (WMO TD No. 1104).

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