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Air Quality
Measurement and Modeling

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AIR QUALITY - MEASUREMENT AND MODELING

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Notice

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Meet the editor

Dr. Philip Sallis is a Professor in Computer Science at the Auckland University of Technology in New Zealand. His research over the past 10 years has predominantly been in the field of GeoComputation with a focus on instrumentation and measurement, particularly in the context of agrometeorological applications of wireless sensor networks for data acquisition and subsequent dynamical systems modelling of micro-climates. In recent years, this work has extended to other areas of environmental sensor applications such as autonomous vehicles, pest control and sensory assistance for children with learning disabilities. With an academic career spanning more than 40 years, he has held university positions in England, Australia and New Zealand, including the senior roles as Head of School, Dean and Deputy Vice Chancellor. He is currently a Pro-Vice Chancellor assisting in the academic leadership of the Auckland University of Technology.

Contents

Preface

It would be to deny the reality of history and perhaps even to be somewhat arrogant to suggest that the planet Earth in our lifetime is somehow in a more special time of its evolution than any other time previously. Epochs of warming and cooling have characterised the longevity of the planet with weather pattern and climate change being a constant. Nonetheless, scientific observations clearly identify a global increase in average monthly temperatures over the past century, which leads us to conclude that the planet is experiencing an evolutionary step change in weather and climate patterns.

What this temperature increase means for climate patterns in the long term is yet to be determined, but the likelihood of consequences such as rising sea levels is likely to be the result of the global warming currently being experienced.

The complexity of relationships between weather variables raises many questions of what to note and how best to model the scientific data now being gathered and analysed. These questions also relate to atmospheric composition and the effect of particulate matter density, especially in the near-ground or lower atmosphere where public health becomes increasingly regarded as at risk from various forms of carcinogenic carbonates present in the air we ingest. Questions that arise from this modern reality are represented in this book as they relate to air quality and without presenting an alarmist perspective and they have at their core the implications of atmospheric composition and the quality of the air we breathe in the context of the contemporary topic of global warming.

All the scientific knowledge we have of this world and other planets in the universe indicates that their physical composition and even their relative geo-positioning is a developing scenario with *change* being an ongoing reality. Twenty-first century trending temperature increases are being observed worldwide with associated factors such as air quality being linked to an overarching scenario of global warming. So is global warming more of a modern phenomenon to be addressed now through policies and practices that can diminish or stop the effects of pollution, or is it an unstoppable trend leading ultimately to devastating consequences for the planet? The authors of the seven chapters in this book have chosen a specific topic for their individual contribution. These chapters have been carefully chosen from the submissions made in order to produce a highly focused collection of topics relating to the precision monitoring and modelling of air quality data. While not addressing every possible dimension of the air quality debate, this book is a comprehensive illustration of the key areas of research in this field of scientific endeavour.

The authors of each chapter have been chosen for their expertise and work in the area of air quality, but not all are environmental scientists and atmospheric or climate scientists. The authors range in academic discipline to include econometricians, medicine, computer sci-

ence and engineering and reflect work on the topic of air quality that is taking place worldwide, reflecting the reality that this is a critical area of concern globally. The book is divided into two sections with the first focusing on methods of air quality monitoring and modelling and the second being more specifically related to case studies in a variety of situations and locations.

This collection of chapters is set in the dialogue milieu of global warming, which at times has led to catastrophisation of possible future implications for the planet and at others, well-reasoned proposals for human intervention and strategic planning for management of the environment and care of the inhabitants of likely affected regions, such as islands and atolls with regard to sea level rises. The history of physical changes to the planet Earth is one characterised by occasional huge events, while at other less dramatic times, changes no less have occurred. The change that came with the so-called Ice Age is an outstanding example of a major climatic shift with its concomitant effect on the terrain, vegetation and lives of the *Homo sapiens* of the time. Many seismic events we know that have varied in effect over time are another obvious examples of the physical change to the structure of life on the planet. Trends in warmer and cooler temperatures, along with other atmospheric and geological shifts, would seem to be what we might think of as the natural course of events in the long time frame of existence within which our world fits as part of the physical universe.

Nonetheless, we need to accept that shifts in the climate of our world are occurring at this time, which means we are experiencing a special time of change in this modern age. Although not happening exponentially, there are incremental increases in temperature. These are having observable effects on the face of the earth such as the melting of snow caps and glaciers, the rising of sea levels and the overheating of land, bringing drought and, consequently, a negative impact on hundreds of thousands of people worldwide, especially those living in the historically more arid parts of it.

Climate change frequently occurs (daily in fact), and on a more topical scale, trends in temperature occur monthly and annually, providing useful comparisons. According to the US National Aeronautics and Space Administration (NASA) for the month of April this year (2016), global land and sea temperatures rose by 1.11C, making them warmer than the average temperature for April during the data recorded period between 1951 and 1980 [1]. Globally, it seems that April was the hottest month on record and the seventh month in a row to have broken global temperature records. NASA predicts that the year 2016 will be the hottest year on record, probably by the largest margin ever recorded.

From a climate science perspective, this data and these trends can relate to numerous factors with perhaps the most significant in 2016 being the effect of El Niño, which in this year is of huge proportions as it pushes hot air and water across the Pacific Ocean.

A report from the Australian Centre of Excellence for Climate System Science indicates that the increasing temperatures are badly affecting ecosystems worldwide [2]. They report that in Australia, 93% of the coral reefs that sit along the Great Barrier Reef have been affected by bleaching, which means that the coral is dying and in the northern parts of the reef, the coral is already dead. An example such as this brings clarity to the debate on what has become known as *global warming*.

Climate change for our world is a historical fact and a feature of nature, but it is the influence on them from the planet's inhabitants that leads to the debate on global warming.

The term *debate* is used here not because there is any doubt that carbon emissions since the Industrial Revolution of the eighteenth century have negatively influenced the atmosphere of the planet, up to and including the troposphere exceeding 10 km above the earth, but the debate is how the world's population deals with the situation.

Recent agreements between governments have seen a determined move towards an international response to the issue of global warming. The United Nations Framework Convention on Climate Change has 197 countries participating in an agreement to set targets for emission reductions [3]. The latest step in this process developed the so-called 2016 Paris Agreement, following on from the earlier 1999 Kyoto Agreement, which segmented countries into groups for the timescale likelihood of emission reduction.

The Paris Agreement seeks to accelerate efforts with each country signing up to a Nationally Determined Contribution to emission reduction. As of 29 April 2016, there were 177 signatories to the Agreement. Of these, 16 countries have also deposited their instruments of ratification for acceptance or approval, which accounts for 0.03% of the total global greenhouse gas emissions. Clearly, there is a long way to go before any significant impact will be observable, but the challenge to reduce global warming has been taken up seriously by the international community of world leaders.

The implications of this challenge for emission reduction are both economic and social. Industries need to find alternatives to fossil fuels for manufacturing and processing; vehicles need to run on other than gasoline or diesel fuel and smoke emitting fires of any kind (household, forest or land burn off) need to be eliminated. The nature of some work will change, and some jobs may go as a result. While the effects of government policies and societal practices may change, the positive impact on the world's atmosphere and, therefore, global warming retardation is likely to be minimal for some time to come. What will not change anytime soon is the need for good science relating to climate, atmosphere and air quality, especially as it relates to pollution.

The news media pays a great deal of attention to global pollution issues and air quality in general. The UK *Guardian* newspaper continues to run specific stories on the topic of pollution especially and its effect on the populace. *The Guardian* began a recent article (18 January 2016) by stating, 'The number of annual deaths caused by pollution around the world is now greater than malaria and HIV combined, according to a recent study, with scientists warning that fatalities could reach 6 million a year by 2050'. [4]. This particular assertion relates to a recent scientific journal article in *Nature* [5].

A later article in the UK *Guardian* newspaper reported that 'Outdoor pollution has risen 8% in five years with fast-growing cities in the developing world worst affected'.[6]. Pollution density maps are used to illustrate statistics such as this one and others relating to the observable decline in air quality worldwide but particularly in those countries using primarily fossil fuels for industry, heating and air conditioning. The decline in air quality is more frequently now linked to public health and well-being.

The World Health Organization (WHO) supports these claims and urges governments to take immediate remedial action to curb what they consider being a pan-epidemic of atmospheric pollution-related illnesses [7]. One example of the data collated by the WHO from another pollution monitoring source is reproduced here in Figure 1, illustrating the intensity of particulate matter (at the level measurement PM10) in the lower atmosphere. The indica-

tion from this map is that those countries burning fossil fuels for heating and industrial operations are producing most of the planet's carbon emissions and, thus, are overtly contributing to the global warming situation. Apparently, this evidence and other similar data are being used to inform the formulation of policy in such agreements as the recent Paris Accord mentioned above.

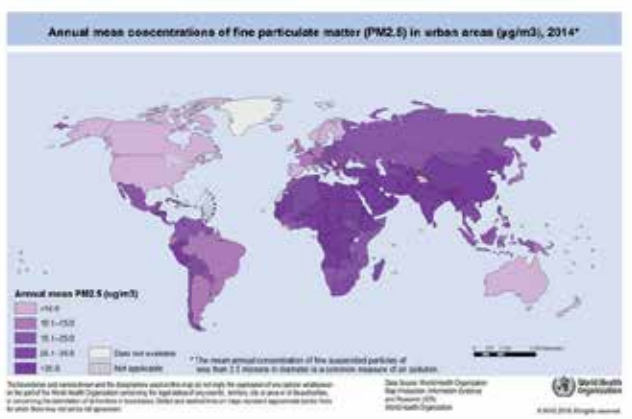

Figure 1. WHO map of PM2.5 concentration in urban areas of the world.

Air quality is the extent (degree) to which the air in a particular location is pollution-free. As a single measure, the ratio of pollutants to clean air in the atmosphere provides a description of how healthy it is for human consumption, leading to how safe it is for humans to inhale it. Climatologists usually label high ratios of pollutants, particularly in urban areas, like *smog*, which is a term coined in origin from the *fog*, itself being dense, moist air visible to the human eye but for *smog*, composed of particulate matter caused by chemical reactions, principally from near-ground ozone.

The primary focus of air quality studies is the measurement and modelling of atmospheric pollution saturation. In this research domain, the topics encompass the effect of pollution on the planet and the methods of data collection, processing and reporting of information concerning air quality trends and treatment.

So generally, the two primary areas of study for air quality research are:

- Air quality—the phenomenon: its nature and impact on the planet
- Atmospheric data—sampling, analysis and reporting methods

In this book, these topics are discussed through studies carried out by the authors and their application of the instruments, methods and reported evidence they have obtained. The subtopics contained in the seven chapters of the book relate to:

- A definitive description of nature and being of air quality
- The nature of pollution and its effect on the planet: plant, vegetable, human and animal
- Air quality in the outdoor atmosphere and indoor environment
- Healthcare and pollution—physical, emotional and social impact

- Vehicle emissions and their impact on air quality
- Econometric impact studies of pollution and the economy
- Government and industry intervention strategies for air quality improvement
- Air quality and climate change
- The influence of weather on particulate matter in the atmosphere
- Climate and atmosphere sensing and data acquisition instruments
- Data modelling methods for trend analysis with result reporting

Although not exhaustive, emerging from discussions here about these topics, the book represents the spectrum of prevalent issues being studied by the scientific community with input from governments worldwide, business and industry, the public health agencies and society in general, all who seek to find ways forward to address the matter of and questions about global warming, a phenomenon of our age.

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References

[1] National Aeronautics and Space Administration, Goddard Institute for Space Studies. Global land-ocean temperature index about base period 1951-1980. http://data.giss.nasa.gov/gistemp/tabledata_v3/GLB.Ts+dSST.txt

[2] Australian Centre of Excellence for Climate Systems Science Centre of Excellence for Coral Reef Studies. Report of the National Coral Bleaching Taskforce. http://www.coralcoe.org.au/person/terry-hughes

[3] United Nations Framework Convention on Climate Change. http://unfccc.int/essential_background/convention/items/6036.php

[4] Damien Carrington. MPs: UK air pollution is a 'public health emergency'. The UK Guardian Newspaper, 18 January 2016. http://www.theguardian.com/environment/2016/apr/27/uk-air-pollution-public-health-emergency-crisis-diesel-cars

[5] Lelieveld, J. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature, 525,367–371 (17 September 2015)

[6] John Vidal. Air pollution rising at an 'alarming rate' in world's cities. The UK Guardian Newspaper, 12th May 2016. http://www.theguardian.com/environment/2016/may/12/air-pollution-rising-at-an-alarming-rate-in-worlds-cities

[7] The World Health Organisation http://www.who.int/mediacentre/news/releases/2016/air-pollution-rising/en/

Section 1

Methods and Analysis

Chapter 1

A Mathematical Approach to Enhance the Performance of Air Pollution Models

El-Said Mamdouh Mahmoud Zahran

Additional information is available at the end of the chapter

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Abstract

The main objective of this chapter is to introduce a mathematical method for enhancing the correctness of the output results of air pollution dispersion models via the calibration of input background concentrations. For developing this method, an air pollution model was set up in ADMS-Roads for a study area in the City of Nottingham in the UK. The method was applied iteratively to the input background concentrations, which effectively reduced the error between calculated and monitored air pollution concentrations on both the annual mean and hourly levels. The inclusion of the traffic flow profiles of the modeled road network reduced further the error between the hourly, but not the annual mean, calculated and monitored concentrations. The application of the calibration approach to the model in ADMS-Roads was compared to the use of grid air pollution sources for the same model in ADMS-Urban. In terms of the accuracy of the model results, the calibration approach was better than using grid sources on the annual mean level and was much better on the hourly level. Compared to the use of grid sources in ADMS-Urban, the use of the calibration approach in either ADMS-Roads or ADMS-Urban can significantly reduce the air pollution model runtime.

Keywords: calibration, validation, background concentrations, modeling, air pollution

1. Introduction

Modeling the air quality is a powerful technique that can be used to assess the ambient air quality against the mandatory air quality standards. In addition, it can be used to assess the effectiveness of the proposed air quality action plans (AQAPs) in improving the air quality within areas in which air pollution exceeds the national air quality standards. This technique

can also be used as a tool to undertake a strategic air quality assessment for a wide range of plans and programs, including local transport plans [1]. As the majority of national air quality standards are in the form of annual mean and hourly objectives [2], this requires accurate annual mean and hourly air quality predictions.

The results of air pollution dispersion modeling should be accurate enough to provide reliable air quality predictions. Recent air pollution dispersion modeling research assesses the validation of air pollution models by the determination of the error between calculated and monitored air pollution concentrations. However, this recent research has not investigated potential sources of this error so that it can be minimized [3–7].

Nottingham City Council compared the monitored annual mean NO_2 concentrations at three continuous monitoring stations to the calculated concentrations by ADMS-Urban. The model overestimated the annual mean of monitored concentrations at the three sites [8]. Therefore, the model results were multiplied by an adjustment factor, the average ratio of monitored to calculated annual mean concentrations at the three monitoring sites, to correct the annual mean results of the model. This might help to improve the annual mean results; however, it did not improve the hourly calculated results of the model.

Ref. [9] used the hourly predictions of ADMS-Urban and the hourly observations for the first half of 1993 to derive a multiplicative adjustment factor. The factor was applied to the air quality predictions for the second half of 1993 and the adjusted predictions were compared to the corresponding observations. This approach improved the long-term results over the second half of 1993; however, it did not show how much improvement was achieved on the short-term level. In addition, Cambridge Environmental Research Consultants (CERC), the developers of ADMS software, have recommended that modelers should avoid the application of such an adjustment factor to the model results [10]. Instead, CERC advised that various details of the model set up, such as input data and modeling options, should be adjusted until the calculated results fit the monitored concentrations.

Ref. [11] stated that the NO_X (not NO_2) concentrations should be verified and adjusted if NO_2 results of the model disagree with the monitored concentrations. It also commented that "The adjustment of NO_X is often carried out on the component derived from local Road Traffic Emissions – the Road Contribution." This is because the source contribution is often small compared with the background contribution. Therefore, Nottingham City Council used this approach to verify the annual mean NO_2 results of ADMS-Urban [12].

ADMS-Urban was used to predict the annual mean road contribution NO_X concentrations. For each monitoring site, the annual mean background $NO_X$was estimated from the national background maps and subtracted from the monitored total NO_X. This resulted in the monitored annual mean road contribution NO_X which was compared to the results of ADMS-Urban for each monitoring site to derive an average adjustment factor. The results of ADMS-Urban were multiplied by this factor, and the adjusted results of NO_X were used, along with the background NO_2 concentrations, to derive the adjusted calculated total annual mean NO_2 concentrations by using the LAQM tools—NO_X to NO_2 spreadsheet [13].

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This approach did not eliminate the error between the calculated and monitored annual mean NO_2 concentrations. This is probably due to inaccuracy in the monitored annual mean road contribution NO_X, caused by inaccuracy in the estimation of the annual mean background NO_X from the national background maps. In addition, the simple NO_X to NO_2 spreadsheet is usually imprecise, and using a chemistry scheme to model the atmospheric chemical reactions of NO_X, and derive the oxidized NO_2 proportion, is recommended [10]. Moreover, this verification approach is only suitable for the calculated annual mean concentrations and is not applicable to the short-term, e.g., hourly, concentrations [10].

Ref. [14] adjusted the air pollution model set-up by the calibration of emission rate inputs to the model through the application of a genetic algorithm. This was helpful to reduce the uncertainties existing in air pollution emission inventories such as those relevant to traffic emission factors [15]. The calibration of input emission rates slightly reduced the error (by 6.46%) between daily calculated and monitored PM_{10} concentrations over 8 days. This implies a nonsignificant reduction in the error between hourly calculated and monitored concentrations over a large time period such as a full meteorological year. Furthermore, no validation was undertaken for the output results of the model, calculated using the calibrated emission rates, against monitored concentrations at monitoring sites independent of the calibration process. This process also required a very expensive computing time, due to the use of a genetic algorithm, which may extend to several weeks on a single PC before the actual running of the air pollution model, which may extend to several days to model the air pollution dispersion in a study area [16, 17].

Therefore, this chapter introduces a mathematical approach for adjusting the model set-up by the calibration of input background concentrations, in order to improve significantly the accuracy of the model results and reduce the computing time. This includes the introduction of four new concepts to the science of air pollution dispersion modeling; namely, macrocalibration, macrovalidation, microcalibration, and microvalidation. The background concentrations are some of the most important input data to the broad variety of air pollution dispersion models [18]. They account for all emission sources that may affect the air quality in a model application area, and are not defined explicitly in the air pollution model. Therefore, a great uncertainty exists in input background concentrations, which may vary for the same model according to the number of explicitly defined air pollution sources. Consequently, the calibration of input background concentrations is necessary to provide the appropriate background concentrations for a certain model set-up. It may also account for the uncertainties existing in input air pollution emission rates.

In the following sections of this chapter, the set-up of the air pollution model of the Dunkirk area in Nottingham is described and the error between calculated and monitored air pollution concentrations is illustrated. Then, the different development stages of the calibration process are discussed, along with the reduction in the error after each stage. The impact of including the traffic profiles of the modeled road network on the error between calculated and monitored concentrations is explained. Finally, the calibration of background concentrations in ADMS-Roads is compared to the use of grid air pollution sources in ADMS-Urban.

2. Set-up of the air pollution model

As a study area, Dunkirk Air Quality Management Area (AQMA) was used to set up an air pollution model in ADMS-Roads version 2.3 for the initial development of the calibration approach. ADMS-Roads was developed by CERC [19]. Dunkirk AQMA is an urban study area in the city of Nottingham, as shown in **Figure 1**, with NO_2 levels exceeding the permissible levels [20]. Therefore, NO_2 was selected as the modeled air pollutant as the majority of the available air pollution monitoring data, required to calibrate and validate the air pollution model, in and around the Dunkirk AQMA was NO_2 data.

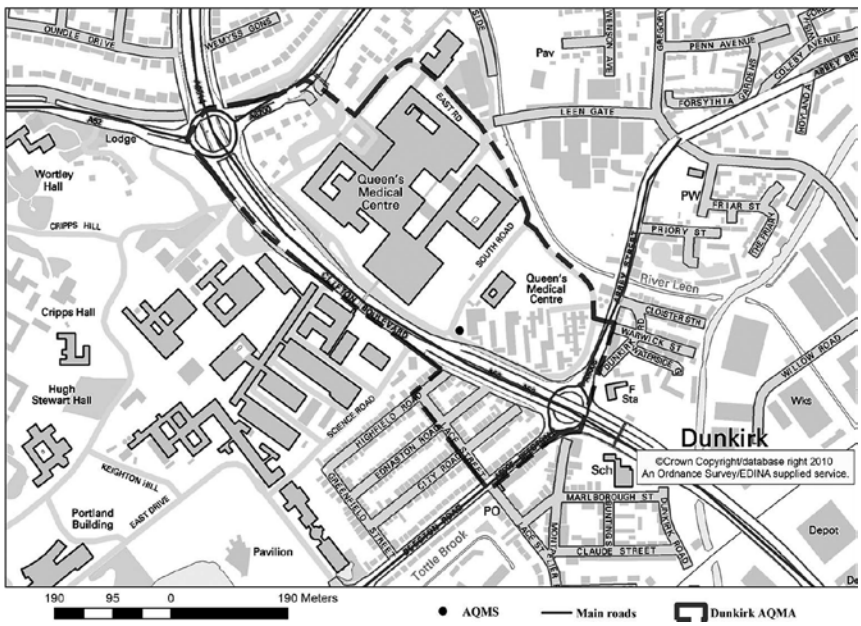

Figure 1. The Dunkirk AQMA.

Note that 2006 was selected as the modeling year of the air pollution model due to data availability for this year. The significant industrial air pollution sources relevant to the Dunkirk AQMA were identified and their emission rates were obtained from Nottingham City Council, which also provided the traffic speed data of the main roads in the Dunkirk AQMA. The emission sources defined explicitly in the air pollution model were the traffic on the main roads within, and close to, the Dunkirk AQMA, as shown in **Figure 1**, and the relevant significant industrial air pollution sources. The Nottingham Watnall Weather Station [21] provided the 2006 hourly sequential meteorological data which included surface temperature, wind speed at 10-m height above the ground surface, wind direction, precipitation, cloud cover, and degree of humidity. The 2006 annual mean and hourly monitored NO_X, NO_2, and O_3 concentrations by the air quality monitoring station (AQMS), located in the Dunkirk AQMA as shown in **Figure 1**, were provided by Nottingham City Council.

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The traffic flow data of the main roads in the Dunkirk AQMA were obtained from Nottingham City Council in the form of the traffic count every 5 min collected automatically using detector loops embedded in the main roads. A visual basic for applications (VBA) computer program was written in MS-Excel in order to calculate automatically the 2006 Annual Average Daily Traffic (AADT) flow and the 2006 hourly and monthly traffic flow profiles from the 5-min traffic counts, using the following mathematics:

For each day, the 5-min flow data was automatically aggregated to yield hourly flow data. Let f_{ijk} be the total traffic flow in both directions in hour i of day j of month k, and let N_k be the number of days in month k, such that $i = 0,...,23$, $j = 1,...,N_k$ (where N_k= 28, 29, 30 or 31 as appropriate), and $k = 1,...,12$.

Therefore,

$$\text{AADT}(\text{vehicles / hour}) = \frac{\sum_{k=1}^{12}\sum_{j=1}^{N_k}\sum_{i=0}^{23} f_{ijk}}{\left[\sum_{k=1}^{12} \text{N}_k\right] \times 24} \tag{1}$$

$$\text{Monthly average}_k\left(\text{vehicles / hour}\right) = \frac{\sum_{j=1}^{N_k}\sum_{i=0}^{23} f_{ijk}}{\text{N}_k \times 24} \;\; \forall k,\; k = 1,\ldots,12 \tag{2}$$

$$\text{Monthly factor}_k = \frac{\text{Monthly average}_k}{\text{AADT}} \;\; \forall k,\; k = 1,\ldots,12 \tag{3}$$

Let p_k, q_k, and r_k be the number of weekdays, Saturdays, and Sundays, respectively, in month k, such that $p_k + q_k + r_k = N_k \;\; \forall k,\; k = 1, ..., 12$. Therefore, the Hourly Average$_i$ (vehicles/hour):

$$\text{For weekdays}\left(\text{if } j \text{ denotes weekdays}\right) = \frac{\sum_{k=1}^{12}\sum_{j=1}^{p_k} f_{ijk}}{\sum_{k=1}^{12} \text{p}_k} \;\; \forall i,\; i = 0,\ldots,23 \tag{4}$$

$$\text{For Saturdays}\left(\text{if } j \text{ denotes Saturdays}\right) = \frac{\sum_{k=1}^{12}\sum_{j=1}^{q_k} f_{ijk}}{\sum_{k=1}^{12} \text{q}_k} \;\; \forall i,\; i = 0,\ldots,23 \tag{5}$$

$$\text{For Sundays}\left(\text{if } j \text{ denotes Sundays}\right) = \frac{\sum_{k=1}^{12}\sum_{j=1}^{r_k} f_{ijk}}{\sum_{k=1}^{12} \text{r}_k} \;\; \forall i,\; i = 0,\ldots,23 \tag{6}$$

Hence,there are 3 × 24 = 72 different day-related hourly average traffic flows; so, correspondingly, there are 72 hourly factors, such that:

$$\text{Hourly factor}_i = \frac{\text{Hourly average}_i}{\text{AADT}} \quad \forall i,\ i = 0,\ldots,23 \tag{7}$$

Therefore, the full traffic flow data processing output for each main road was:

- 24 hourly factors for weekdays, in order, from hour 0 to hour 23.
- 24 hourly factors for Saturdays, in order, from hour 0 to hour 23.
- 24 hourly factors for Sundays, in order, from hour 0 to hour 23.
- 12 monthly factors for the 12 months, in order, from January to December.

Lack of data from some detectors for some time periods during the year 2006 had to be addressed. If the corresponding traffic data was available for another year, then that was used, factored using traffic data from the nearest detectors, for that other year and 2006. Steps were taken in the code to avoid zero division in factoring the traffic data of that other year. If the corresponding traffic data from another year was not available, then 2006 traffic data from the nearest available detectors were used.The traffic flow profiles were compiled to a special text file, a FAC file, which was used in ADMS-Roads to reflect the hourly and monthly variations in the AADT flow on traffic air pollution emissions, so that for each hour, the traffic flow, used in the model to derive the traffic emissions, was the AADT flow × monthly factor × hourly factor. The 2003 DMRB traffic emission factors [22], built-in in ADMS-Roads, were used to derive the traffic emission rates from the traffic flow and speed data.

The chemical reaction scheme (CRS) was used to model the atmospheric conversion of NO_X to NO_2 due to a number of chemical reactions with background O_3 [19]. Modeling these atmospheric reactions was necessary to get accurate NO_2 results, so NO_X and O_3 were modeled in addition to NO_2. However, using this chemical scheme requires inputs for NO_2, NO_X, and O_3 background concentrations. Therefore, Nottingham City Council provided the 2006 hourly sequential NO_2, NO_X, and O_3 concentrations monitored by the Rochester air quality monitoring station. This is a rural monitoring station remote from the Dunkirk AQMA and far from urban air pollution, and hence it was recommended to use its monitoring data as the input background concentrations to avoid double counting [10].

3. Calibration and validation of the background concentrations

An output receptor was defined in the air pollution model at the geographical location of the AQMS. With reference to Run 1 in **Table 1**, the calculated 2006 annual mean NO_X and NO_2 concentrations underestimated the monitored ones by 37.6% and 25.6%, respectively, at the

AQMS. In addition, the calculated 2006 annual mean of O_3 concentrations overestimated the monitored one by 42.7% at the AQMS. This necessitated developing the set-up of the air pollution model by performing two operations. The first operation was the iterative calibration of the rural background concentrations so as to account for the urban background emissions, e.g., residual, poorlydefined, or diffused emissions, from domestic heating sources and minor roads, in the Dunkirk AQMA. The second operation was the validation of the calculated air pollution concentrations after each iteration of the calibration process, in order to decide the final acceptable iteration of this process.

	Δ background	**Calculated concentrations**	**Target concentrations**
Run 1			
NO_2	0	26.25	35.29
NO_X	0	42.19	67.60
O_3	0	44.23	31.00
Run 9			
NO_2	+7.70	37.27	35.29
NO_X	+25.42	67.61	67.60
O_3	-12.60	28.99	31.00
Run 23			
NO_2	+1.48	35.45	35.29
NO_X	+25.42	67.60	67.60
O_3	-5.40	31.01	31.00
Run A			
NO_2	+7.02	36.89	35.29
NO_X	+25.42	67.61	67.60
O_3	-12.40	28.86	31.00
Run B			
NO_2	+10.12	38.73	35.29
NO_X	+25.42	67.61	67.60
O_3	-13.20	29.46	31.00
Run C			
NO_2	+14.55	41.64	35.29
NO_X	+25.42	67.61	67.60
O_3	-15.30	29.18	31.00
Run D			
NO_2	+17.18	43.56	35.29
NO_X	+25.42	67.61	67.60
O_3	-16.71	28.69	31.00

Table 1. Macrocalibration development stages of the rural background concentrations.

3.1. Macrocalibration and macrovalidation

The term macrocalibration in this chapter refers to the adjustment of input background concentrations, so that the error between the annual means of calculated and monitored air pollution concentrations can be effectively reduced. The macrovalidation was undertaken by

the direct comparison between the calculated and monitored annual means of NO_X, NO_2, and O_3 concentrations at the AQMS.

As calculated NO_2 concentrations were linked to calculated NO_X and O_3 concentrations through the atmospheric chemical reactions discussed in Section 2, it was decided to calibrate NO_X and O_3, in addition to NO_2, background concentrations. A trial and error approach was adopted to macrocalibrate the hourly sequential rural background concentrations until the above-mentioned macrocalibration criterion was achieved. This approach comprised 22 runs of the model, and involved changing the background concentrations manually every time. In **Table 1**, the results of an intermediate run (run 9), and the final macro-calibration run (run 23), are shown in order to illustrate the progress of this approach.

For each macrocalibration iteration, the values in the "Δ background" field of **Table 1** were added to every hour of the 2006 NO_2, NO_X, and O_3 rural background concentrations. However, adding these values to the original background concentrations file resulted in having many consecutive hours with a negative O_3 background concentration which raised an error and interrupted the model run. This technical problem was overcome by replacing the negative, invalid, O_3 background concentrations with zero in the macrocalibrated background concentrations file. Another computer logic was applied to this file in order to preserve the fact that NO_X is NO + NO_2. Hence, for every hour in the macrocalibrated background concentrations file, if $NO_2 > NO_X$, then $NO_2 = NO_X$.

After each iteration of the macrocalibration, the macrovalidation was undertaken by comparing the calculated concentrations and the target concentrations in **Table 1**. The calculated concentrations were the 2006 annual means of calculated NO_2, NO_X, and O_3 concentrations and the target concentrations were the 2006 annual means of monitored NO_2, NO_X, and O_3 concentrations at the AQMS. Run 23 in **Table 1** gave the least error between the calculated and target concentrations. Therefore, the background concentrations corresponding to this run were considered the final macrocalibrated background concentrations.

The results of the final macrocalibration run were used to derive Eqs. (8), (9), and (10), which could be used to evaluate directly the background concentration adjustment values, required to macrocalibrate the Dunkirk AQMA air pollution model, without the trial and error approach:

$$\Delta NO_{2\,\text{background}} = \frac{\left(\overline{NO}_{2\,\text{monitored}} - \overline{NO}_{2\,\text{uncalibrated}}\right)}{9.2} \times 1.48, \tag{8}$$

where $\overline{NO}_{2\,\text{monitored}}$ is the annual mean of monitored NO_2 concentrations and $\overline{NO}_{2\,\text{uncalibrated}}$ is the annual mean of calculated NO_2 concentrations using the rural background concentrations.

$$\Delta NO_{x\,\text{background}} = \overline{NO}_{x\,\text{monitored}} - \overline{NO}_{x\,\text{uncalibrated}}, \tag{9}$$

where $\overline{NO}_{x\,\text{monitored}}$ is the annual mean of monitored NO_X concentrations and $\overline{NO}_{x\,\text{uncalibrated}}$ is the annual mean of calculated NO_X concentrations using the rural background concentrations.

$$\Delta O_{3\,\text{background}} = \frac{\left(\bar{O}_{3\,\text{monitored}} - \bar{O}_{3\,\text{uncalibrated}}\right)}{\left(-13.22\right)} \times \left(-5.40\right), \tag{10}$$

where $\bar{O}_{3\,\text{monitored}}$ is the annual mean of monitored O_3 concentrations and $\bar{O}_{3\,\text{uncalibrated}}$ is the annual mean of calculated O_3 concentrations using the rural background concentrations.

3.2. Microcalibration and microvalidation

The term microcalibration in this chapter refers to the adjustment of input background concentrations so that the error between not only the annual means of, but also the hourly, calculated and monitored air pollution concentrations can be effectively reduced. The microcalibration extends the macrocalibration as shown in **Figure 2**. The microvalidation was

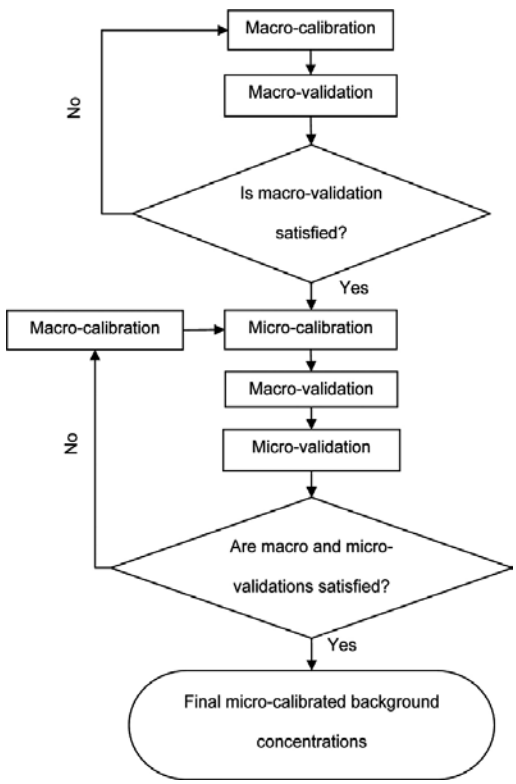

Figure 2. Calibration and validation process for rural background concentrations.

undertaken by comparing statistically two one-dimensional arrays of the 2006 calculated and monitored hourly sequential NO_2 concentrations at the AQMS. The statistical approach to compare these two arrays depended on the definition of them. If these two arrays were to be defined as two samples of two bigger populations, statistical tests would be the best approach to compare statistically the two bigger populations [23]. However, if these two arrays represented the two populations to compare, statistical tests would not be suitable and descriptive statistics would be the convenient statistical approach to compare these two populations.

Therefore, careful consideration was given to define correctly the two arrays of calculated and monitored 2006 hourly NO_2 concentrations at the AQMS, concluding that these two arrays should be defined as two populations, not as two samples. The reason was that these two arrays of concentrations did not comprise NO_2 concentrations from any year other than 2006, or averages over any time period other than an hour. Therefore, a hypothesis that these two arrays are two samples of two bigger populations that may extend over many years of time, or comprise air pollution concentrations calculated or monitored over a diversity of averaging times, was invalid. Consequently, Pearson correlation coefficient (r) and the root mean square error (RMSE) were used to compare the two populations. Further details about these two descriptive statistics are given in [7, 24, 25]. The slope of the regression line through the origin was also used to compare the two populations of hourly calculated and monitored concentrations.

Linear regression through the origin was used because it was already known that the perfect relationship between hourly calculated and monitored concentrations is $y_i = x_i$ without a constant, where y_i and x_i were the calculated and monitored NO_2 concentrations for hour i at the AQMS, respectively. The value of i ranged from 1 to 8760 which was the total number of hours in the year 2006. The linear regression analysis was undertaken for three cases, uncalibrated versus monitored, macrocalibrated versus monitored, and microcalibrated versus monitored, concentrations. In all these three cases, the independent variable was the monitored concentrations.

The comparison between the calculated and monitored hourly NO_2 concentrations at the AQMS was undertaken by the comparison between the slope of the best fit line through the origin and 1.0, the slope of the perfect relationship. The magnitude and sign of the difference between the slope of the best fit line through the origin and 1.0 indicated the tendency of calculated NO_2 concentrations to underestimate or overestimate the 2006 monitored NO_2 concentrations on the micro, hourly, level. Moreover, the slope of the regression best fit line through the origin was used for the graphical representation of the linear approximation of the actual relationship between calculated and monitored hourly NO_2 concentrations at the AQMS, after each stage of the calibration process.

The Dunkirk AQMA air pollution model was run with the uncalibrated rural background concentrations file to output the 2006 calculated hourly NO_2 concentrations at the AQMS. This was carried out for the identification of the initial discrepancy, before any calibration, between the 2006 calculated and monitored hourly NO_2 concentrations at the AQMS, as shown in **Figure 3**. Then, the model was run with the macrocalibrated background concentrations file, corresponding to run 23 in **Table 1**, to output the 2006 calculated hourly NO_2 concentrations

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at the AQMS. This was for the microvalidation after the macrocalibration of the rural background concentrations as shown in **Figure 4**.

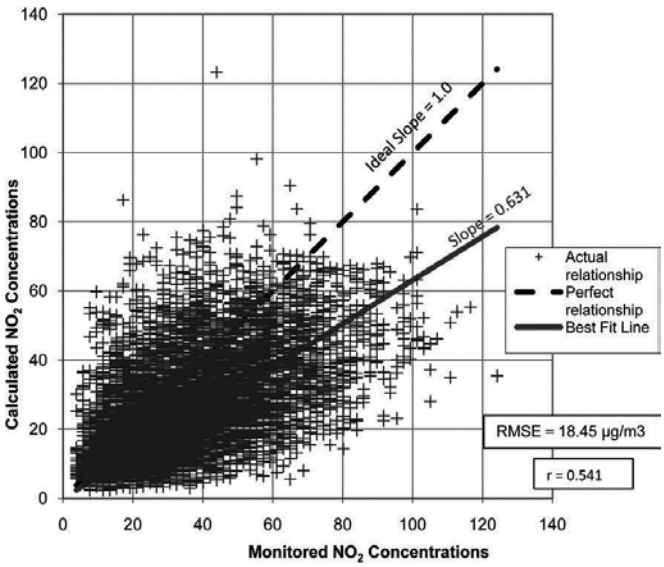

Figure 3. Scatter diagram of hourly NO_2 concentrations at the AQMS before any calibration.

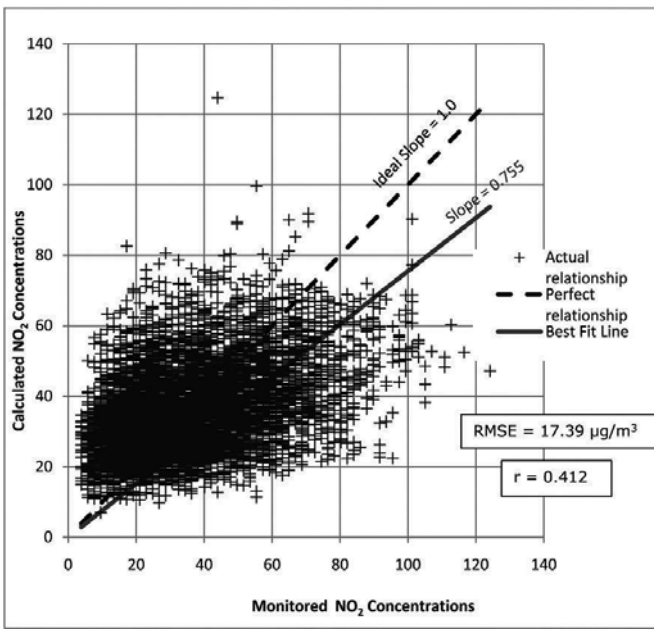

Figure 4. Scatter diagram of hourly NO_2 concentrations at the AQMS after macrocalibration.

Pearson's correlation coefficients were calculated as 0.541 before any calibration, and then as 0.412 after the macrocalibration, as shown in **Figures 3** and **4**. The slight decline in Pearson's correlation coefficient after the macrocalibration implied that the macrocalibration slightly decreased the degree of linearity of the actual relationship between the calculated and monitored hourly NO_2 concentrations at the AQMS. Hence, the macrocalibration slightly increased the drift of the shape of this actual relationship away from the perfect straight-line relationship.

On the other hand, the values of the RMSE were calculated as 18.45 μg/m^3 before the calibration, and then as 17.39 μg/m^3 after the macrocalibration, as shown in **Figures 3** and **4**. The slight decline in the RMSE after the macrocalibration implied that the macrocalibration slightly lowered the difference between the calculated and monitored hourly NO_2 concentrations. Therefore, the macrocalibration not only improved the NO_2 predictions of the model on the macro, annual mean, level but also slightly improved the NO_2 predictions on the micro, hourly, level.

The slope of the best fit line through the origin of the actual relationship between the calculated and monitored hourly NO_2 concentrations at the AQMS was calculated as 0.631 before any calibration, and then as 0.755 after the macrocalibration, as shown in **Figures 3** and **4**. Although the results of the macrocalibration, corresponding to run 23 in **Table 1**, very slightly overestimated the 2006 annual mean of monitored NO_2 concentrations at the AQMS, the slope of the best fit line through the origin after the macrocalibration was less than 1.0. This indicated that, after the macrocalibration, the model generally underestimated the monitored NO_2 concentrations at the AQMS on the micro, hourly, level. However, the slight increase in the slope of the best fit line after the macrocalibration implied that the macrocalibration slightly reduced the tendency of the model to underestimate the monitored hourly NO_2 concentrations at the AQMS. This, together with the reduction in the RMSE after the macrocalibration, confirmed the slight improvement of the NO_2 predictions of the model, after the macrocalibration, on the micro, hourly, level.

To improve further the NO_2 predictions of the model on the micro level, the idea of microcalibration was developed. This idea depended on the modification of Eqs. (8), (9), and (10) in order to generate three one-dimensional arrays for ΔNO_2 background, ΔNO_X background, and ΔO_3 background as follows:

$$\Delta NO_{2\,\text{background}\,i} = \frac{\left(NO_{2\,\text{monitored}\,i} - NO_{2\,\text{uncalibrated}\,i}\right)}{\left(NO_{2\,\text{macro}\,i} - NO_{2\,\text{uncalibrated}\,i}\right)} \times 1.48, \tag{11}$$

where $\Delta hNO_{2\,\text{background}\,i}$ is the adjustment value for the rural NO_2 background concentration for the hour i. $NO_{2\,\text{monitored}\,i}$ is the monitored hourly NO_2 concentration for the hour i. $NO_{2\,\text{uncalibrated}\,i}$ is the calculated hourly NO_2 concentration for the hour i using the uncalibrated rural background concentrations. $NO_{2\,\text{macro}\,i}$ is the calculated hourly NO_2 concentra-

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tion for the hour *i* using the macrocalibrated background concentrations. The value of *i* ranged from 1 to 8760, which was the total number of hours in the year 2006:

$$\Delta NO_{X\ \text{background}\ i} = NO_{x\ \text{monitored}\ i} - NO_{x\ \text{uncalibrated}\ i}, \tag{12}$$

where $\Delta hNO_{X\ \text{background}\ i}$ is the adjustment value for the rural NO_X background concentration for the hour *i*. $NO_{X\ \text{monitored}\ i}$ is the monitored hourly NO_X concentration for the hour *i*. $NO_{X\ \text{uncalibrated}\ i}$ is the calculated hourly NO_X concentration for the hour *i* using the uncalibrated rural background concentrations. The value of *i* ranged from 1 to 8760, which was the total number of hours in the year 2006:

$$\Delta O_{3\ \text{background}\ i} = \frac{\left(O_{3\ \text{monitored}\ i} - O_{3\ uncalibrated\ i}\right)}{\left(O_{3\ \text{macro}\ i} - O_{3\ \text{uncalibrated}\ i}\right)} \times \left(-5.4\right), \tag{13}$$

where $\Delta hO_{3\ \text{background}\ i}$ is the adjustment value for the rural O_3 background concentration for the hour *i*. $O_{3\ \text{monitored}\ i}$ is the monitored hourly O_3 concentration for the hour *i*. $O_{3\ \text{uncalibrated}\ i}$ is the calculated hourly O_3 concentration for the hour *i* using the uncalibrated rural background concentrations. $O_{3\ \text{macro}\ i}$ is the calculated hourly O_3 concentration for the hour *i* using the macrocalibrated background concentrations. The value of *i* ranged from 1 to 8760, which was the total number of hours in the year 2006.

The three one-dimensional arrays of $\Delta NO_{2\ \text{background}}$, $\Delta NO_{X\ \text{background}}$, and $\Delta O_{3\text{background}}$, calculated by Eqs. (11), (12), and (13), were added to the arrays of the uncalibrated hourly sequential rural background concentrations of NO_2, NO_X, and O_3, respectively. Hence the microcalibrated background concentrations file was created based on the above three equations. However, running the model with these microcalibrated background concentrations resulted in the overestimation of the annual means of the monitored NO_2, NO_X, and O_3 concentrations at the AQMS as shown in **Table 2**. In addition, using these microcalibrated background concentrations increased the difference between the calculated and monitored hourly NO_2 concentrations on the micro, hourly, level. This was indicated by the large increase in the RMSE as shown in **Table 2**.

A possible reason for the large increase in the RMSE after the microcalibration based on Eqs. (11), (12), and (13) was the use of the macrocalibrated hourly concentrations in these equations. As discussed before with regard to **Figure 4**, the hourly calculated concentrations of the macrocalibrated model were not precise enough. The macrocalibrated model of the Dunkirk AQMA was validated only on the macro, annual mean, level. Therefore, instead of using $NO_{2\ \text{macro}\ i}$ and $O_{3\ \text{macro}\ i}$, the macrocalibrated calculated hourly NO_2 and O_3 concentrations, it was decided to alter two of the three equations for the microcalibration of the rural background concentrations, using the macrocalibrated annual mean NO_2 and O_3 concentrations, so that:

Case description	Receptor name	Annual mean NO_X		Annual mean NO_2		Annual mean O_3		NO_2 RMSE before calibration	NO_2 RMSE after macrocalibration	NO_2 RMSE after microcalibration
		Calculated	Monitored	Calculated	Monitored	Calculated	Monitored			
Based on equations (11), (12), and (13)	AQMS	73.37	67.60	37.90	35.29	39.43	31.00	18.45	17.39	117.83
Based on equations (14), (12), (15), and run 23	AQMS	68.46	67.60	31.58	35.29	34.99	31.00	18.45	17.39	11.07
Based on equations (14), (12), (15), and run A	AQMS	67.71	67.60	33.03	35.29	33.10	31.00	18.45	17.39	6.63
Based on equations (14), (12), (15), and run B	AQMS	67.55	67.60	33.96	35.29	32.52	31.00	18.45	17.39	5.11
Based on equations (14), (12), (15), and run C	AQMS	67.48	67.60	34.85	35.29	31.47	31.00	18.45	17.39	4.21
Based on equations (14), (12), (15), and run D	AQMS	67.47	67.60	35.19	35.29	30.96	31.00	18.45	17.39	4.09
Based on equations (14), (12), (15), and run D with no FAC file	AQMS	68.65	67.60	35.51	35.29	30.74	31.00	18.45	17.39	5.71

Table 2. Microcalibration development stages of the rural background concentrations.

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$$\Delta NO_{2\ \text{background}\ i} = \frac{\left(NO_{2\ \text{monitored}\ i} - NO_{2\ \text{uncalibrated}\ i}\right)}{\left(\overline{NO}_{2\ \text{macro}} - \overline{NO}_{2\ \text{uncalibrated}}\right)} \times \Delta NO_{2\ \text{macro background}}, \quad (14)$$

where $\Delta NO_{2\ \text{background}\ i}$ is the adjustment value for the rural NO_2 background concentration for the hour i. $NO_{2\ \text{monitored}\ i}$ is the monitored hourly NO_2 concentration for the hour i. $NO_{2\ \text{uncalibrated}\ i}$ is the calculated hourly NO_2 concentration for the hour i using the uncalibrated rural background concentrations. The value of i ranged from 1 to 8760, which was the total number of hours in the year 2006. $\overline{NO}_{2\ \text{macro}}$ is the annual mean NO_2 concentration calculated using the macrocalibrated background concentrations. $\overline{NO}_{2\ \text{uncalibrated}}$ is the annual mean NO_2 concentration calculated using the uncalibrated rural background concentrations. $\Delta cNO_{2\ \text{macro background}}$ is the macrocalibration adjustment value for the rural NO_2 background concentrations, as given in the column headed "Δ background in **Table 1**:

$$\Delta O_{3\ \text{background}\ i} = \frac{\left(O_{3\ \text{monitored}\ i} - O_{3\ \text{uncalibrated}\ i}\right)}{\left(\bar{O}_{3\ \text{macro}} - \bar{O}_{3\ \text{uncalibrated}}\right)} \times \Delta O_{3\ \text{macro background}}, \quad (15)$$

where $\Delta hO_{3\ \text{background}\ i}$ is the adjustment value for the rural O_3 background concentration for the hour i. $O_{3\ \text{monitored}\ i}$ is the monitored hourly O_3 concentration for the hour i. $O_{3\ \text{uncalibrated}\ i}$ is the calculated hourly O_3 concentration for the hour i using the uncalibrated rural background concentrations. The value of i ranged from 1 to 8760, which was the total number of hours in the year 2006. $\bar{O}_{3\ \text{macro}}$ is the annual mean O_3 concentration calculated using the macrocalibrated background concentrations. $\bar{O}_{3\ \text{uncalibrated}}$ is the annual mean O_3 concentration calculated using the uncalibrated rural background concentrations. $\Delta cO_{3\ \text{macro background}}$ is the macrocalibration adjustment value for the rural O_3 background concentrations, as given in the column headed "Δ background" in **Table 1**.

A VBA computer program was written in MS-Excel in order to automate the generation of the three hourly sequential one-dimensional arrays for $\Delta NO_{2\ \text{background}}$, $\Delta NO_{X\ \text{background}}$, and $\Delta O_{3\ \text{background}}$ using Eqs. (14), (12), and (15). For any hour in the year 2006, if either the calculated or monitored hourly concentration was missing, then the equation relevant to the type of missing concentration would not be usable. This was handled in the VBA computer program as follows: $\Delta oNO_{2\ \text{background}\ i} = \Delta\ NO_{2\ \text{macro background}}$ for the hours of missing hourly NO_2 concentrations, $\Delta NO_{X\text{background}\ i} = \Delta NO_{X\text{macro background}}$ for the hours of missing hourly NO_X concentrations, and $\Delta O_{3\text{background}\ i} = \Delta O_{3\text{macrobackground}}$ for the hours of missing hourly O_3 concentrations.

The VBA computer program applied Eqs. (14), (12), and (15) along with the macrocalibration results of run 23 in **Table 1** to generate the microcalibrated background concentrations file. Running the Dunkirk AQMA air pollution model with this background concentrations file

significantly improved the RMSE, *r*, and the slope of the best fit line through the origin as shown in **Table 2** and **Figure 5**. This indicated a significant improvement for NO_2 hourly predictions by the model when using this background concentrations file. However, the model with this background concentrations file underestimated the annual mean of monitored NO_2 concentrations, and overestimated the annual mean of monitored O_3 concentrations, at the AQMS as shown in **Table 2**. Hence, using the trial and error macrocalibration approach, it was necessary to undertake additional runs of ADMS-Roads, beyond run 23, as shown in **Table 1**.

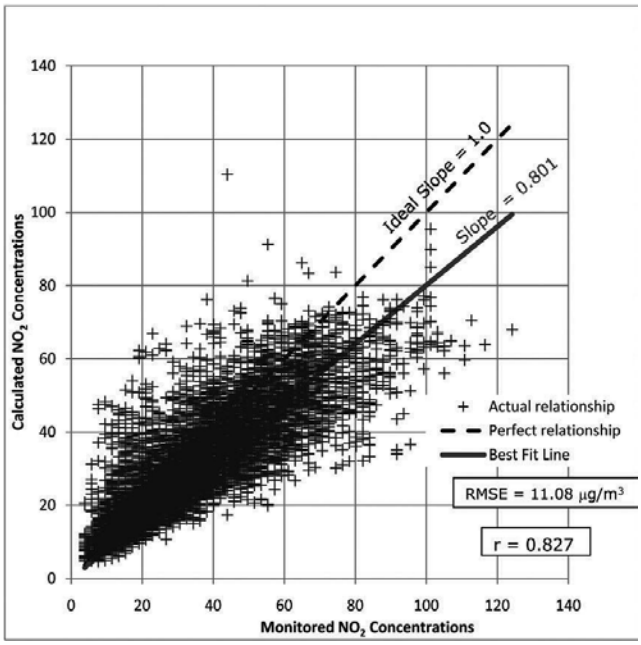

Figure 5. Scatter diagram of hourly NO_2 concentrations at the AQMS after the microcalibration based on run 23.

The background concentrations of these additional macrocalibration runs were modified so that the annual mean of monitored NO_2 concentrations was deliberately overestimated, and the annual mean of monitored O_3 concentrations was deliberately underestimated, by these runs, named A–D in **Table 1**. Consequently, after the "normal" microcalibration underestimation of the annual mean of monitored NO_2 concentrations and the "normal" microcalibration overestimation of the annual mean of monitored O_3 concentrations, the microcalibration runs based on the results of these additional macrocalibration runs gave a good estimate of the annual means of both the monitored NO_2 and O_3 concentrations at the AQMS. This not only improved the results of the microcalibrated model on the macro level, but also further improved the results on the micro level as shown in **Table 2** and **Figure 6**. Therefore, the microcalibrated background concentrations obtained by Eqs. (14), (12), and (15), based on the macrocalibration results of run D, were considered the final microcalibrated background concentrations.

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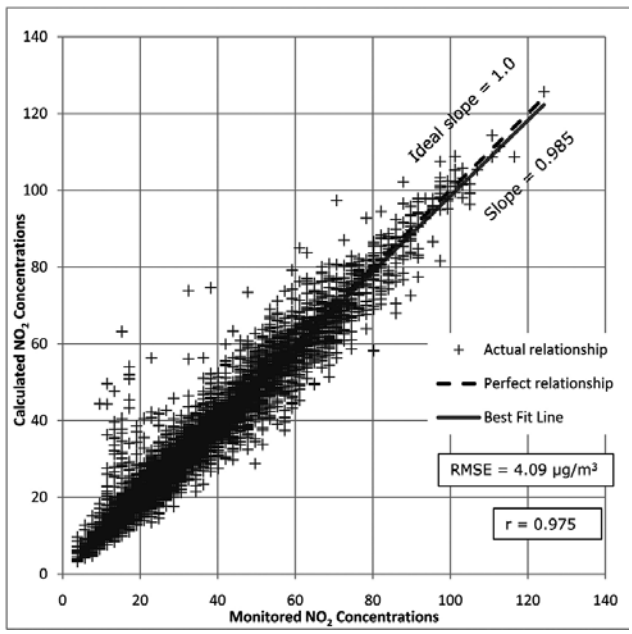

Figure 6. Scatter diagram of hourly NO_2 concentrations at the AQMS after the microcalibration based on run D.

The microcalibration development, from run 23 to run D, increased the error between the calculated and monitored NO_2 concentrations at a few hours, as implied by the comparison between the scatter in the overestimated points on the lower left side of **Figures 5** and **6**. A thorough investigation was undertaken in order to identify the reason for such unexpected behavior of the microcalibration process at these hours. A potential reason was the very high ratio of the monitored NO_X concentration to the monitored NO_2 concentrations, e.g., 7, which was accompanied by a high monitored O_3 concentration at these hours. However, a high calculated NO_X concentration by the air pollution model was accompanied by high calculated NO_2 concentration and low calculated O_3 concentration at these hours. This suggested either imprecise model simulation of the actual atmospheric chemical reactions between NO_X and O_3 due to inaccurate input meteorological data or imprecise monitoring data at these hours.

The high monitored NO_X concentration resulted in a high increase in the NO_X background concentration due to the microcalibration at these hours. Such a high increase in the NO_X background concentration substantially increased the calculated NO_2 concentration, resulting in a big difference between the calculated and low monitored NO_2 concentrations at these hours. At some of these hours, for which the NO_2 concentration was underestimated before any calibration, the microcalibration iterations increased the background NO_2 concentration in order to increase the calculated NO_2 concentration, which changed the NO_2 underestimation into an increasingly greater NO_2 overestimation. At the rest of these hours, for which the NO_2 concentration was overestimated before any calibration, the reduction in calculated NO_2 concentration due to the microcalibration iterations was masked by the increase in calculated NO_2 concentration due to the high NO_X background concentration.

4. Impact of traffic profiles on the macro- and microvalidation

As mentioned in Section 2, the hourly and monthly traffic flow profiles were considered in the set-up of the air pollution model by use of a special text file, a FAC file. The impact of the traffic profiles on the macro and micro levels was investigated by turning off this FAC file in the final microcalibrated version of the Dunkirk AQMA model, corresponding to run D in **Table 2**. The exclusion of the traffic profiles did not have a significant impact on the calculated annual mean NO_2, NO_X, and O_3 concentrations as shown in **Table 2**. Therefore, it was concluded that the consideration of the traffic profiles in the air pollution model was not important for the macrovalidation.

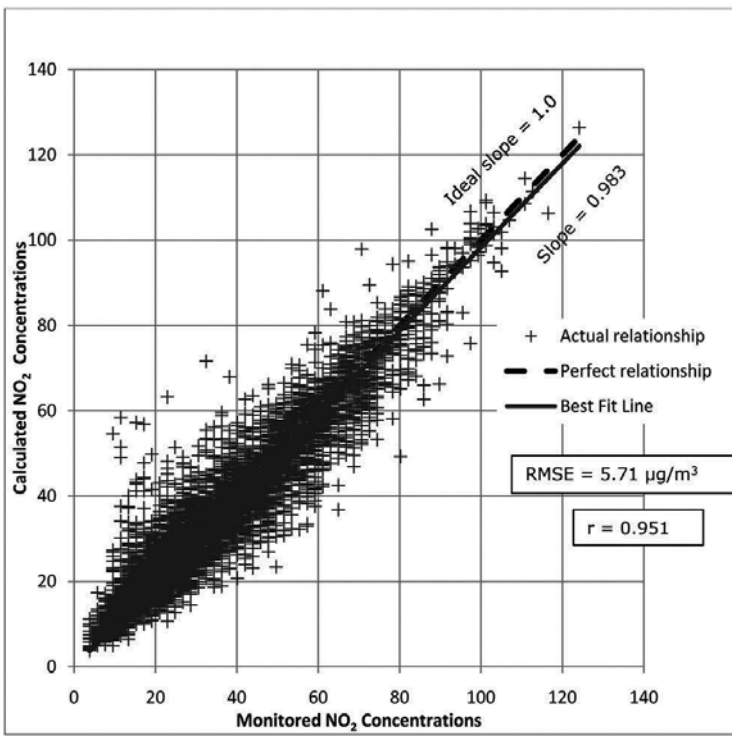

Figure 7. Scatter diagram of hourly NO_2 concentrations at the AQMS after the microcalibration based on run D without a FAC file.

On the other hand, the exclusion of the traffic profiles slightly worsened the hourly calculated NO_2 concentrations as shown in **Figure 7**. This was indicated by the higher RMSE, the lower *r*, and the slightly lower slope of the best fit line through the origin, without a FAC file in **Figure 7** compared to with a FAC file in **Figure 6**. Therefore, it was concluded that the incorporation of the traffic profiles in the air pollution model could further improve the microvalidation by reducing the RMSE between the calculated and monitored hourly NO_2 concentrations by 28.4%.

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5. The calibration of background concentrations versus the use of grid sources

Grid air pollution sources are used in ADMS-Urban to model residual, poorly defined or diffused emissions in urban areas, such as the emissions from domestic heating sources and minor roads [26]. This enables ADMS-Urban to model emissions from sources that are not defined explicitly in the air pollution model. Therefore, Nottingham City Council uses grid sources in ADMS-Urban to compensate for the difference between rural and urban background concentrations [8]. However, the capability to model emissions from such air pollution sources is only available in ADMS-Urban, not in ADMS-Roads. Hence, the Dunkirk AQMA air pollution model was set up in ADMS-Urban, with the Rochester rural background concentrations, and this time with a grid source. The air pollution emissions of the grid source were obtained from the UK National Atmospheric Emissions Inventory (NAEI).

The ADMS-Urban model was run to output the 2006 annual mean concentrations of NO_2, NO_X, and O_3 at the AQMS as shown in **Table 3**. Comparing **Table 2** with **Table 3**, the calculated annual mean NO_2, NO_X, and O_3 concentrations from the ADMS-Roads model, with microcalibrated background concentrations, were closer to the corresponding annual means of monitored concentrations than were the calculated annual means from the ADMS-Urban model, with a grid source and rural background concentrations. This indicated that the ADMS-Roads model, with microcalibrated background concentrations only, was more precise than the ADMS-Urban model, with a grid source and rural background concentrations, on the macro level.

Case description	NO_2 annual mean, μg/m³		NO_X annual mean, μg/m³		O_3 annual mean, μg/m³	
	Calculated	Monitored	Calculated	Monitored	Calculated	Monitored
ADMS-Urban with CRS	37.65	35.29	69.31	67.6	35.18	31.0
ADMS-Urban with CRS with trajectory model	37.77	35.29	69.31	67.6	35.07	31.0

Table 3. Monitored versus calculated annual mean concentrations at the AQMS by ADMS-Urban.

The 2006 hourly NO_2 concentrations calculated by the ADMS-Urban model were compared to the 2006 hourly monitored NO_2 concentrations at the AQMS as shown in **Figure 8**. Hence, comparing **Figure 6** with **Figure 8**, the results of the ADMS-Urban model, with a grid source and rural background concentrations, gave a much higher RMSE than did the results of the ADMS-Roads model, with microcalibrated background concentrations only. In addition, the results of the ADMS-Urban model gave a much lower *r*, and a lower slope of the best fit line through the origin, than did the results of the ADMS-Roads model, with microcalibrated background concentrations only. Therefore, the results of the ADMS-Roads model, with microcalibrated background concentrations only, were much closer to the 2006 hourly NO_2

concentrations monitored by the AQMS than were the results of the ADMS-Urban model, with a grid source and rural background concentrations. This indicated that the ADMS-Roads model, with microcalibrated background concentrations only, was much more precise than the ADMS-Urban model, with a grid source and rural background concentrations, on the micro level.

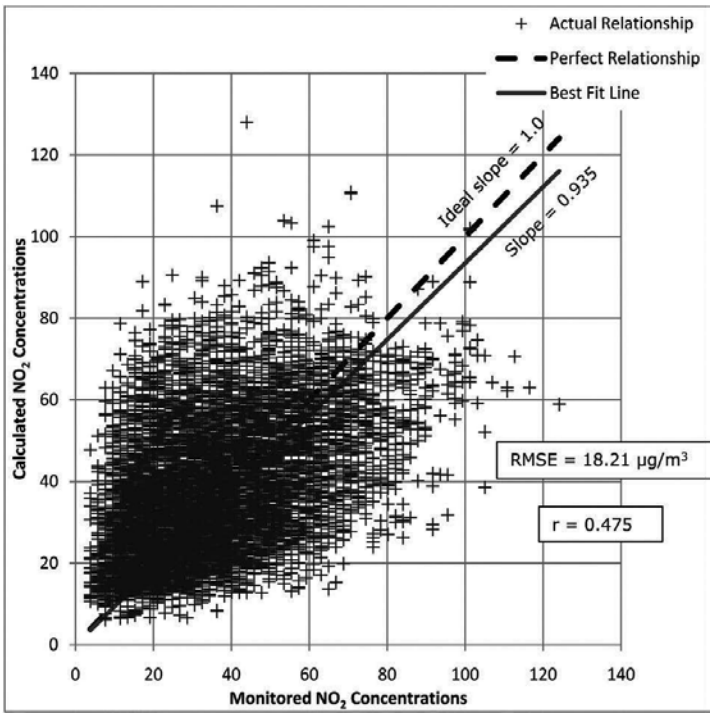

Figure 8. Scatter diagram of monitored versus calculated hourly NO_2 concentrations at the AQMS by ADMS-Urban.

Comparing **Table 1** (run 23) with **Table 3**, the calculated annual mean NO_2, NO_X, and O_3 concentrations from the ADMS-Roads model, with macrocalibrated background concentrations only, were closer to the corresponding annual means of monitored concentrations than were the calculated annual means from the ADMS-Urban model, with a grid source and rural background concentrations. This indicated that the ADMS-Roads model, with macrocalibrated background concentrations only, was more precise than the ADMS-Urban model, with a grid source and rural background concentrations, on the macro level.

In respect of the 2006 hourly NO_2 concentrations, comparing **Figure 8** with **Figure 4**, the results of the ADMS-Urban model, with a grid source and rural background concentrations, gave a slightly higher RMSE than did the results of ADMS-Roads, with macrocalibrated background concentrations only. Both the ADMS-Urban model and the macrocalibrated ADMS-Roads model generally underestimated the 2006 hourly monitored NO_2 concentrations which was indicated by the best fit line through the origin having a slope of less than

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1.0 in both **Figures 8** and **4**. However, the slope of the best fit line in the ADMS-Urban case (in **Figure 8**) was closer to 1.0 than was the slope of the best fit line in the macrocalibrated ADMS-Roads case (in **Figure 4**). Therefore, the tendency of the ADMS-Urban model, with a grid source and rural background concentrations, to underestimate the hourly monitored NO_2 concentrations was less than that of the ADMS-Roads model, with macrocalibrated background concentrations only.

Continuing the comparison of **Figure 8** with **Figure 4**, the results of ADMS-Urban, with a grid source and rural background concentrations, gave a slightly higher *r* than did the results of ADMS-Roads, with macrocalibrated background concentrations only. This implied that the ADMS-Urban model slightly increased the degree of linearity of the actual relationship between the calculated and monitored hourly NO_2 concentrations at the AQMS. Hence, the actual relationship between the calculated and monitored hourly NO_2 concentrations was slightly closer to the perfect straight line relationship in the case of the ADMS-Urban model than it was in the case of the macrocalibrated ADMS-Roads model. The RMSE, *r*, and the slope of the best fit line through the origin indicated that the ADMS-Roads model, with macrocalibrated background concentrations only, was almost as precise as the ADMS-Urban model, with a grid source and rural background concentrations, on the micro level.

The trajectory model of CRS can be used along with a grid air pollution source in ADMS-Urban to adjust the background concentrations in the main model domain, the model application area, on the basis of the grid source emissions [26]. The trajectory model uses the grid source domain, which is usually larger than the main model domain. Then, the trajectory model increases the background concentrations within the nested main model domain, to take account of the emissions in the larger grid source domain. This converts the rural background concentrations within the model application area to urban background concentrations before ADMS-Urban actually starts its calculations of the air pollution concentrations. Therefore, it was decided to investigate the impact of running the ADMS-Urban model with the trajectory model of CRS on the annual mean and hourly calculated air pollution concentrations at the AQMS.

Running the ADMS-Urban model with the trajectory model of CRS did not significantly change the calculated annual mean NO_2, NO_X, and O_3 concentrations at the AQMS from the calculated annual means of these concentrations using the CRS only, as shown in **Table 3**. In addition, comparing **Figure 9** with **Figure 8**, running the ADMS-Urban model with the trajectory model of CRS did not significantly change the RMSE, *r*, or the slope of the best fit line through the origin of the actual relationship between the hourly calculated and monitored NO_2 concentrations at the AQMS. Therefore, it was concluded that using the trajectory model of CRS for running ADMS-Urban did not provide any significant improvement to running ADMS-Urban with the CRS only, on either the macro or the micro level. Therefore, using the trajectory model of CRS did not change the results of comparing the ADMS-Urban model, with rural background concentrations and a grid source, to the ADMS-Roads model, with either macro- or microcalibrated background concentrations.

In terms of the model runtime, running ADMS-Urban with a grid source, rural background concentrations and either the CRS or the trajectory model of CRS required 44 min to calculate

the annual mean and hourly concentrations of NO_2, NO_X, and O_3 at a single output receptor point, the site of the AQMS. On the other hand, running ADMS-Roads with the CRS and either the macrocalibrated or microcalibrated background concentrations required 9 min to calculate the annual mean and hourly concentrations of NO_2, NO_X, and O_3 at the same output receptor point, the site of the AQMS, on the same computer. Therefore, compared to running ADMS-Urban, using ADMS-Roads with the background concentrations calibration technique not only improved the air quality predictions of the air pollution model on the macro and micro levels, but it also saved 35 min of the model runtime for each output receptor point. This saving in the model runtime, when related to an output grid with a large number of receptor points, constitutes a significant reduction in the air pollution model runtime.

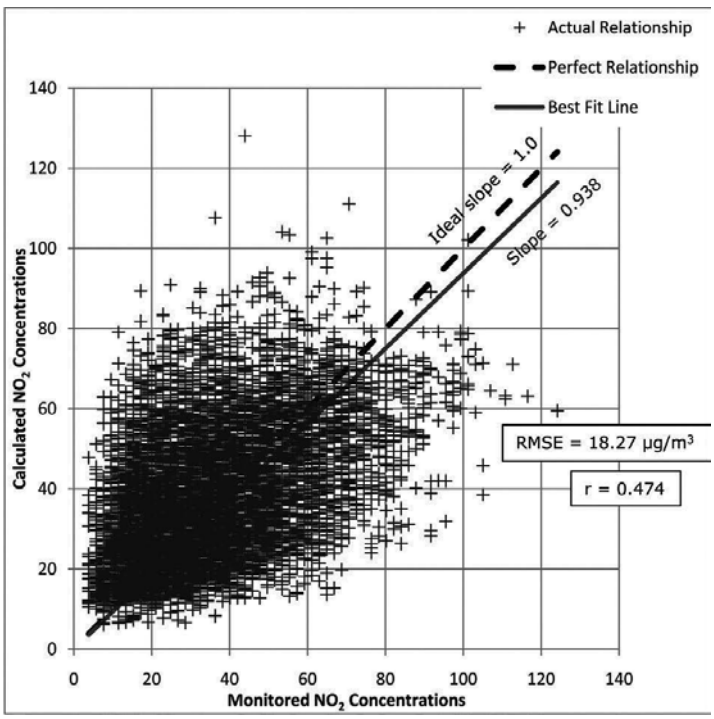

Figure 9. Scatter diagram of monitored versus calculated hourly NO_2 concentrations at the AQMS by ADMS-Urban with the trajectory model of CRS.

6. Conclusions and recommendations

The mathematical algorithm implemented by VBA computer programing in Section 2 was necessary for the processing of large files of primary traffic flow count data that were recorded every 5 min for all of the year 2006.The computer program outputs for each main road in the Dunkirk AQMA were the AADT flow, and the hourly and monthly traffic profiles for the air

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pollution model. The application of this computer program significantly reduced the processing time and effort, which may allow an increase in the number of road links that can be modeled in air pollution dispersion models. This improves the model accuracy, and thus increases the reliability of air quality predictions.

The application of the VBA computer program also helps to avoid the potential human errors that may arise during the manual processing of large files of traffic flow input data, which may further increase the reliability of air pollution dispersion models. The high resolution of the primary traffic flow data for which the program can start the processing makes this computer program suitable for a broad range of other road links with similar or less traffic flow data resolution.

The macrocalibration of background concentrations reduced effectively the error between the calculated and monitored annual means of NO_X, NO_2, and O_3 concentrations. The iterative application of the microcalibration Eqs. (14), (12), and (15) to background concentrations reduced effectively the error between the calculated and monitored annual means of NO_X, NO_2, and O_3 concentrations, and also the error between the hourly calculated and monitored NO_2 concentrations. Further investigation is required into the adaptation of the macrocalibration and microcalibration equations for modeling the air pollution dispersion of inert pollutants, e.g., CO and PM. As chemical reactions will not be considered, the calibration equations may reduce to one equation for the macrocalibration, and one equation for the microcalibration, of the input background concentrations.

For the hours with missing monitored air pollution concentrations, the microcalibration equations were unusable. This was addressed by using the macrocalibrated background concentrations for these hours, as discussed in Section 3.2. As the macrocalibrated background concentrations give less precise calculated concentrations on the hourly level (see Sections 3.2 for details), such a strategy may reduce the reliability of the number of exceedances and percentiles predicted by the air pollution model. Therefore, for the hours with missing monitored air pollution concentrations, further research is needed to investigate the impact of using the macrocalibrated background concentrations on the reliability of the predicted number of exceedances and percentiles by the air pollution model. In case of a significant adverse impact, further research is recommended into the microcalibration of the rural background concentrations of these hours, based on the meteorological data and the microcalibrated background concentrations of other hours with monitored concentrations.

The inclusion of the hourly and monthly traffic profiles in the Dunkirk AQMA air pollution model did not have a significant impact on the error between the annual means of calculated and monitored concentrations. On the other hand, the inclusion of these traffic profiles did reduce the RMSE between the hourly calculated and monitored NO_2 concentrations by 28.4% (see Section 4 for details). As the Dunkirk AQMA air pollution model did not include a large number of road sources, further research is recommended to investigate the impact of including the monthly and hourly traffic profiles on the microvalidation of an air pollution model that has a large number of road sources. This is to correlate between the number of road sources with traffic profiles in the air pollution model and the possible reduction in the RMSE between the hourly calculated and monitored NO_2 concentrations.

In terms of the error between the annual means of calculated and monitored NO_2 concentrations, using ADMS-Roads with only the macro- or microcalibrated background concentrations was more accurate than using ADMS-Urban with a grid source and rural background concentrations. Moreover, in terms of the error between the hourly calculated and monitored NO_2 concentrations, using ADMS-Roads with only the microcalibrated background concentrations was much more accurate, although slightly less accurate with only macrocalibrated background concentrations (see Section 5 for details). Using the trajectory model of CRS in ADMS-Urban did not significantly change the error between the monitored and calculated concentrations otherwise obtained, and so effectively did not change the comparative results between using ADMS-Roads and ADMS-Urban.

Replacing the grid source with either the macro- or microcalibrated background concentrations can save up to 35 min of the model runtime for each output receptor point. This saving in the model runtime, when related to an output grid with a large number of receptor points, constitutes a significant reduction in the air pollution model runtime. The microcalibration mathematical equations did not require any input data to start the iterations, apart from the monitored air pollution concentrations. In comparison, the grid air pollution sources require precise input data for the air pollution emissions which may impede their usage in air pollution modeling of areas without a precise emissions inventory.

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References

[1] Nottingham City Council. Local Transport Plan for Greater Nottingham 2006/7 to 2010/11. 2006.

[2] UK National Air Quality Archive. Air Quality Standards [Internet]. 2010. Available from: http://www.airquality.co.uk/standards.php [Accessed: 2010].

[3] Cai, H., Xie, S. D. A modeling study of air quality impact of odd-even day traffic restriction scheme before, during and after the 2008 Beijing Olympic Games. Atmospheric Chemistry and Physics Discussions. 2010;10:5135–5184. DOI: 10.5194/acpd-10-5135-2010

[4] Ginnebaugh, D. L., Liang, J., Jacobson, M. Z. Examining the temperature dependence of ethanol (E85) versus gasoline emissions on air pollution with a largely-explicit

chemical mechanism. Atmospheric Environment. 2010;44(9):1192–1199. DOI: 10.1016/j.atmosenv.2009.12.024

[5] Majumdar, B.K., Dutta, A., Chakrabarty, S., Ray, S. Assessment of vehicular pollution in Kolkata, India, using CALINE 4 model. Environmental Monitoring and Assessment. 2010;170(1):33–43. DOI: 10.1007/s10661-009-1212-2

[6] Parra, M. A., Santiago, J. L., Martín, F., Martilli, A., Santamaría, J. M. A methodology to urban air quality assessment during large time periods of winter using computational fluid dynamic models. Atmospheric Environment. 2010;44(17):2089–2097. DOI: 10.1016/j.atmosenv.2010.03.009

[7] Jain, S., Khare, M. Adaptive neuro-fuzzy modeling for prediction of ambient CO concentration at urban intersections and roadways. Air Quality, Atmosphere & Health. 2010;3(4):203–212. DOI: 10.1007/s11869-010-0073-8

[8] Nottingham City Council. Detailed Assessment Consultation Document. Pollution Control & Envirocrime Section. 2008.

[9] Namdeo, A., Mitchell, G., Dixon, R. TEMMS: an integrated package for modelling and mapping urban traffic emissions and air quality. Environmental Modelling & Software. 2002;17(2):177–188. DOI: 10.1016/S1364-8152(01)00063-9

[10] CERC. Using ADMS-Urban and ADMS-Roads and the latest government guidance. In: McHugh, C. (ed.) UK tools for modelling NO_X and NO_2. ADMS-Urban and Roads User Group Meeting. 2009.

[11] DEFRA. Technical Guidance on Local Air Quality Management LAQM.TG(09); 2009.

[12] Nottingham City Council. Detailed Assessment 2009. Pollution Control & Envirocrime Section; 2010.

[13] DEFRA. Defra, UK—Environmental Protection—Air Quality—Local Air Quality Management [Internet]. 2010. Available from: http://www.defra.gov.uk/environment/quality/air/airquality/local/support/

[14] Li, M. J., Chen, D. S., Cheng, S. Y., Wang, F., Li, Y., Zhou, Y.,Lang, J. L. Optimizing emission inventory for chemical transport models by using genetic algorithm. Atmospheric Environment. 2010;44(32):3926–3934. DOI: 10.1016/j.atmosenv.2010.07.010

[15] Belalcazar, L. C., Clappier, A., Blond, N., Flassak, T., Eichhorn, J. An evaluation of the estimation of road traffic emission factors from tracer studies. Atmospheric Environment. 2010;44(31):3814–3822. DOI: 10.1016/j.atmosenv.2010.06.038

[16] Barrett, S. R. H., Britter, R. E. Development of algorithms and approximations for rapid operational air quality modelling. Atmospheric Environment. 2008;42(34):8105–8111. DOI: 10.1016/j.atmosenv.2008.06.020

[17] Barrett, S. R. H., Britter, R. E. Algorithms and analytical solutions for rapidly approximating long-term dispersion from line and area sources. Atmospheric Environment. 2009;43(20):3249–3258. DOI: 10.1016/j.atmosenv.2009.03.032

[18] Venegas, L. E., Mazzeo, N. A. Modelling of urban background pollution in Buenos Aires City (Argentina). Environmental Modelling & Software. 2006;21(4):577–586. DOI: 10.1016/j.envsoft.2004.08.013

[19] CERC. ADMS-Roads Version 2.2 User Guide. An Air Quality Management System. Cambridge Environmental Research Consultants Ltd; United Kingdom, 2006.

[20] Nottingham City Council. Second and Third Stage of Air Quality Review and Assessment Report. Pollution Control and Envirocrime Section; 2001.

[21] Met Office: Nottingham Watnall. Available from: http://www.metoffice.gov.uk/weather/uk/em/nottingham_watnall_latest_weather.html [Accessed: 28/10/2010].

[22] DMRB. Environmental Assessment Techniques. Design Manual for Roads and Bridges. 2007;11(3), Annex B:8–31.

[23] Kanji, G. K. 100 Statistical Tests. SAGE Publications; New York, 2006.

[24] Hanna, S. R., Chang, J. C., Strimaitis D. G. Hazardous gas model evaluation with field observations. Atmospheric Environment. Part A. General Topics. 1993;27(15):2265–2285. DOI: 10.1016/0960-1686(93)90397-H.

[25] Hanna, S. R., Strimaitis, D. G., Chang, J. C. Hazard Response Modeling Uncertainty (A Quantitative Method). User's guide for software for evaluating hazardous gas dispersion models. Tech. Rep. prepared for Engineering and Services Laboratory, Air Force Engineering and Services Center, Tyndall Air Force Base, and American Petroleum Institute, by Earth Tech; 1991.

[26] CERC. ADMS-Urban Version 2.2 User Guide. An Air Quality Management System. Cambridge Environmental Research Consultants Ltd; United Kingdom, 2006.

Chapter 2

Particulate Matter Sampling Techniques and Data Modelling Methods

Jacqueline Whalley and Sara Zandi

Additional information is available at the end of the chapter

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Abstract

Particulate matter with 10 μm or less in diameter (PM_{10}) is known to have adverse effects on human health and the environment. For countries committed to reducing PM_{10} emissions, it is essential to have models that accurately estimate and predict PM_{10} concentrations for reporting and monitoring purposes. In this chapter, a broad overview of recent empirical statistical and machine learning techniques for modelling PM_{10} is presented. This includes the instrumentation used to measure particulate matter, data preprocessing, the selection of explanatory variables and modelling methods. Key features of some PM_{10} prediction models developed in the last 10 years are described, and current work modelling and predicting PM_{10} trends in New Zealand—a remote country of islands in the South Pacific Ocean—are examined. In conclusion, the issues and challenges faced when modelling PM_{10} are discussed and suggestions for future avenues of investigation, which could improve the precision of PM_{10} prediction and estimation models are presented.

Keywords: particulate matter, modelling, regression, artificial neural networks, instrumentation and measurement

1. Introduction

Particle pollution—also known as particulate matter or particulates—is a complex but stable gaseous suspension of liquid droplets and solid particles in the earth's atmosphere. Particle pollution is known to have many environmental effects from poor visibility to more serious consequences such as acid rain, which pollutes soil and water. The science of air quality is

complex, and many aspects of the problem are not understood fully. Particles are commonly classified according to their size as either coarse or fine. Fine particles have a diameter of 2.5 μm ($PM_{2.5}$) or less, and coarse particles are 10 μm or less (PM_{10}). Particulate matter that has a diameter over 100 μm tends not to stay airborne long enough to be measured. Fine particles are commonly generated through combustion or by secondary gas to particle reactions. These fine particles are typically rich in carbon, nitrates, sulphates and ammonium ions. Coarse particles are commonly the product of mechanical processes but also include naturally occurring wind-blown particles. A common example of coarse particulate matter is dust containing calcium, iron, silicon and other materials from the earth's crust.

Sources of particulate matter are often classified according to whether they originate from natural or anthropogenic sources. Natural sources include particles suspended in the atmosphere by volcanic eruptions, bush fires and pollen dispersal. Mechanistic processes cause natural particles such as dust and sea-salt particles to be suspended in the atmosphere. Biological sources of particulate matter are also natural sources; these consist largely of fungal spores (≤1 μm) and plant debris (normally < 2 μm) but also include microorganisms, viruses, pollen (≤10 μm) and fragments of living things (e.g. skin cells). Anthropogenic sources of biological particles include sources from farming, horticulture, waste disposal and sewage. Another anthropogenic source is emissions from combustion of fuels, for example, vehicle exhaust. In Europe, anthropogenic sources have been identified as the main contributor to PM_{10} due to urbanisation, high population density and areas of intensive industry. In New Zealand, the main contributors are also anthropogenic but are emissions from winter household heating (i.e. the wide use of wood-burning fires) and industry.

PM_{10} are so minute that they can be inhaled, penetrate the lungs and cause serious health problems. One event which illustrates the effect of particle pollution on human health is the 1952 'Great Smog' in London. Particle pollution from coal burning hung over the city for four days due to cold temperatures and lack of wind. Approximately 4000 deaths were linked to this single event [1]. As a result of events such as the Great Smog and obvious signs of climate change, many countries are now committed to international and national clean air legislation and air quality standards. These agreements require regular reporting of air quality including PM_{10} concentrations.

The economic costs of particulate pollution on a country can be significant. In the European Union in 2015, the cost of air pollution-related deaths was reported to be over US$1.4 trillion. In Israel, it is estimated that 2500 people a year die as a result of exposure to air pollutants [2]. In New Zealand (population ~ 4.4 million), it was reported that, despite relatively low air pollution when compared with other members of the Organisation for Economic Co-operation and Development, during 2012 a total of 1370 deaths, 830 hospital admissions and 2.55 million restricted activity days were linked to PM_{10} pollution [3]. Even low levels of PM_{10} have been found to significantly affect human health.

In order to make informed decisions, as individuals or as policymakers, it is critical that particulate matter is measured and modelled appropriately.

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2. PM_{10} modelling

Models can be designed to estimate, predict or project. Discontinuities in data represent a real obstacle for time series analysis and prediction. Thus, estimating PM_{10} is important in situations where small periods of ground-truth data, acquired from sensors, are missing. Prediction models allow us to determine that something will happen in the future based on past data, generally with some level of probability, and are based on the assumption that future changes will not have a significant influence. In this sense, a prediction is most influenced by the initial conditions—the current situation from which we predict a change. Predicting short-range PM_{10} is important in order to identify days in which PM_{10} levels spike so that people with medical conditions which make them vulnerable to air pollution, such as asthmatics, can avoid exposure. It also allows for initiatives such as free public transport days to reduce commuter traffic volumes and thus reduce PM_{10} concentrations on a predicted high day. Models that allow for long-range projections are also important in order to assess the impact of different air quality management scenarios. A projection determines with a certain probability what could happen if certain assumed conditions prevailed in the future. Most PM_{10} models are designed to predict short range hourly, mean daily or maximum daily PM_{10} concentrations one day ahead.

A wide variety of techniques, ranging from simple to complex, have been used to predict PM_{10} concentrations. Mechanistic models are complex three-dimensional physiochemical models requiring theoretical information to simulate, using mathematical equations, the processes of particulate matter transportation and transformation (e.g. the air pollution model (TAPM) [4]). Such models are complex and time-consuming to implement and often prove inaccurate. Mechanistic models require a wide variety of input variables for which ground-truth data are not available. These missing data are either estimated or the model is simplified and all begin with meteorological forecasting, introducing both errors and uncertainties to a model.

Statistical models aim to discover relationships between PM_{10} concentrations and other explanatory variables. Statistical models work on a number of assumptions. Machine learning algorithms, on the other hand, are largely free of such assumptions and learn from the data they are presented with, finding patterns and relationships that are not necessarily obvious in the data. Machine learning approaches also tend to be good at modelling highly non-linear functions and can be trained to accurately generalise when presented with new, unseen data. As a result, machine learning methods have on the whole proven to be better at predicting PM_{10} concentrations than statistical models. This chapter focuses on statistical and machine learning approaches to PM_{10} modelling and prediction.

The vast majority of models in the last decade have been developed using a data-driven approach and have their origins in statistical modelling and machine learning. These models use ground-level sensor data and make no attempt to model the physical or chemical processes involved in PM_{10} generation, transportation and removal. They are reliant on measurements of pollutants and meteorological variables which are accurate only within a small area around

the monitoring stations. Thus, any model is limited by coverage, reliability and distribution of monitoring stations.

There are several steps in building an empirical PM_{10} model (**Figure 1**). The first is data acquisition from various types of particulate matter sensor. The next step is cleaning and preparing the raw data for analysis, including handling missing data, suspected errors and outliers. The next step, variable selection, is central to the performance of most models [5]. The aim of variable selection is to simplify the model by reducing the dimensions and removing any variables that do not significantly contribute to the model. The model is then built based on this subset of variables. Once a model is established, it is tested, after validation where required, by exposing the model to new data and measuring how well it predicts.

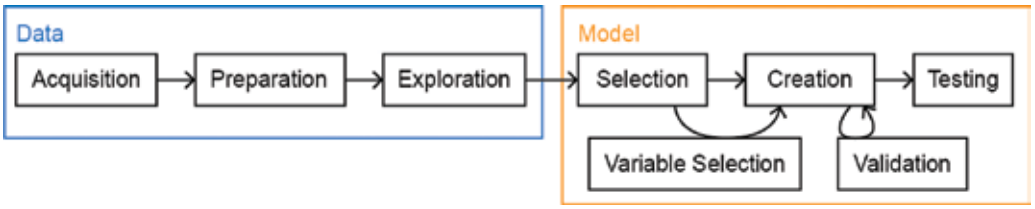

Figure 1. Key steps in the modelling process.

2.1. Particulate matter sampling techniques

The most common instruments for measuring particulate matter measure either its concentration or size distribution. The most accurate measurements are obtained from instruments that use a gravimetric (weighing) method. Air is drawn through a preweighed filter, and particles collect in the filter. The filter is then removed and reweighed. This approach has the added advantage that particles collected in the filter can be analysed chemically [6]. This method involves careful pre- and post-conditioning of the filter. Filter choice is also important as substrates are sensitive to environmental factors such as relative humidity. PTFE-bonded glass fibre has been found to be the most stable type of filter [7]. Accurate weighing is essential, and precise weighing protocols must be followed for results to be comparable [7]. This method is the most widely adopted by regulatory bodies including the EPA and the EU. However, it is not the most pragmatic method for PM_{10} modelling purposes because it is not real time and provides only average data for the period the filter was deployed. A manual process and consequently high operating costs limit the applications of this method. However, gravimetric measurements may be useful to provide a quick snapshot of PM_{10} at a site in order to determine locations for more intensive monitoring [8].

The TEOM™ sensor is the most commonly used instrument based on the microbalance method. TEOM™ uses a filter which is mounted on the end of a hollow tapered tube made of quartz. Particles collect on the filter and cause the oscillation frequency of the quartz tube to vary. PM_{10} measurements can be logged in near real time. A study which examined the

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measurements on PM_{10} in New Zealand using microbalance measurement instruments found that the measurements were not equivalent to those from gravimetric methods [9].

Real-time monitoring of PM_{10} concentrations can be achieved using optical instruments. These instruments measure either light scattering, light absorption or light extinction caused by particulate matter. The most common instrument is an optical particle counter (OPC) which uses a light source, normally a laser diode, to illuminate particles and a photodetector to measure light scattered by those particles. Measurements may be periodically verified and calibrated using data from gravimetric instrumentation. OPC instruments have lower purchase and operating costs than gravimetric meters, but their lower precision and sensitivity mean that they are not considered appropriate for compliance monitoring [8]. However, the low cost of OPC instruments and real-time monitoring capability make OPCs suitable for particulate matter research.

Regardless of the data collection methods used, PM_{10} models are reliant on accurate and complete time series data from geographically localised monitoring stations.

2.2. Explanatory variables

Suspended PM_{10} regardless of location is dependent on many factors such as meteorological properties of the atmosphere, topo-geographical features, emission sources and the physical and chemical properties of the particles (size, shape and hygroscopicity). Many natural environmental factors influence PM_{10} concentrations from the time of year, to the weather, to extreme events such as volcanic eruptions and earthquakes. The effect of extreme events in nature on PM_{10} concentrations is well documented: high PM_{10} levels have been reported during heatwaves in Greece [10], as a result of forest fires [11], and in the aftermath of the Christchurch earthquakes in New Zealand [12]. Relatively low PM_{10} concentrations are observed during the monsoon season in India [13]. Of the myriad complex interrelated potential explanatory variables, only a small number have been used in the modelling of PM_{10} concentrations.

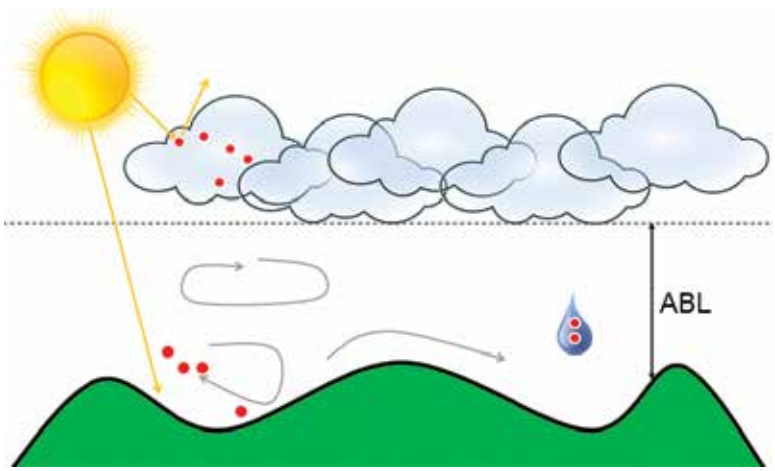

Figure 2. Particulate matter and the atmospheric boundary layer.

One key factor commonly used to explain and evaluate trends in PM_{10} data is the impact of meteorological conditions. The atmospheric boundary layer (ABL) is the lowest part of the earth's atmosphere (**Figure 2**). The thickness of the ABL can vary from 100 to 3000 m and extends from the ground to the point where cumulus clouds form. In the ABL wind, temperature and moisture fluctuate rapidly, and turbulence causes vertical and horizontal mixing. Suspended in the ABL, particles may undergo physical and chemical transformations triggered by factors such as the amount of water vapour, the air temperature, the intensity of solar radiation and the presence or absence of other atmospheric reactants. It is these physical processes, which help to explain why meteorological variables have such an influence on PM_{10} concentrations.

Having accurate and complete input data is critical to the success of any PM_{10} prediction model. As a result, most models make use of data that are readily recorded using weather station sensors. In cases where data are incomplete, the instance is often removed rather than imputed because of errors which may be introduced by estimation processes. The outputs of numerical weather forecast models can also be used as input variables in PM_{10} models. However, this is not common because of the uncertainties such variables introduce to PM_{10} predictions [14, 15].

Wind speed and temperature are the meteorological explanatory variables most frequently used in PM_{10} prediction models (**Table 1**). Wind variables have been found to be useful proxies for physical transportation factors; wind is critical to the horizontal dispersion of PM_{10} in the ABL. Wind direction controls the path that the PM_{10} will follow, while wind speed determines the distance it is carried and the degree to which PM_{10} is diluted due to plume stretching. The effect of wind speed and direction on PM_{10} varies with the geographical characteristics of a location. Low wind speed can be associated with high PM_{10} [16, 17]; this is common in hilly or mountainous regions. Conversely, in coastal or desert regions, high wind speeds result in high PM_{10} concentrations due to salt or dust suspension. In Europe, PM_{10} concentrations are significantly influenced by long-range transport contributions, which are independent of local emissions, so both wind direction and speed have a significant impact [18]. In Invercargill, New Zealand, where there are no close neighbours and thus little long-range transboundary PM_{10}, wind speed explains most of the variability in PM_{10} concentrations [19].

Cold temperatures increase the likelihood of an inversion layer forming in many locations. An inversion exists where a layer of cool air at the earth's surface is covered by a higher layer of warmer air. An inversion prevents the upward movement of air from the layers below and traps PM_{10} near the ground. As a result, cold temperatures tend to coincide with high concentrations of PM_{10}. However, in some locations days with high temperatures, no clouds and stable atmospheric conditions result in high PM_{10} [17]. In other locations when the difference between daily maximum and minimum temperatures is large and the height of the ABL mixing layer is low, high PM_{10} concentrations are observed [20].

PM_{10} levels can be reduced by rain, snow, fog and ice. Rain scavenging, a phenomenon in which below-cloud particles are captured and removed from the atmosphere by raindrops, is considered to be one of the major factors controlling the removal of PM_{10} from the air. The degree to which PM_{10} is removed is dependent on rainfall duration and intensity [21]. While rainfall is a primary factor in PM_{10} concentrations, it has not been used widely in models. This

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is in part due to the fact that in some countries, there is no rain for long periods of time or little rainfall in summer. The lack of rain data means that it is not often included in PM_{10} models [14].

Study reference		**[16]**	**[26]**	**[25]**	**[34]**	**[33]**	**[35]**	**[35]**	**[39]**	**[14]**	**[23]**
Country of study (ISO 3166-1 alpha 3)		GRC	GRC	PRT	CHL	MYS	AUT	CZE	TUR	SAU	MYS
Predicted variable											
PM_{10}	Daily		Y	Y	Y	Y	Y	Y		Y	Y
	Hourly	Y							Y		
Explanatory variables											
	PM_{10} lag	Y		Y	Y	Y	Y	Y		Y	Y
Co-pollutants	CO_2			Y		Y				Y	Y
	SO_2			Y		Y				Y	Y
	NO			Y		Y					
	NO_2			Y		Y				Y	Y
	O_3					Y					Y
Meteorological data	Temperature	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Temperature lag						Y	Y			
	Wind direction	Y						Y	Y	Y	
	Wind direction lag										
	Wind speed	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Wind speed lag										
	Precipitation	Y			Y		Y				
	Solar radiation	Y									Y
	Sunshine hours										
	Air pressure				Y				Y		
	Dew point		Y								
	Humidity (%)	Y		Y	Y	Y			Y	Y	Y
	Cloud cover							Y			
	Date/time	Y	Y				Y	Y			
	Seasonal effects	Y						Y			
	Spatial variables										

Table 1. Explanatory variables used in recent MLR models for predicting PM_{10} concentrations.

Relative humidity has been used more frequently in models than rainfall. The relationship between PM_{10} concentration and relative humidity also depends on other meteorological conditions. For example, if humidity is high and there is also intense rainfall (such as during a monsoon season), then humidity has a negative correlation with PM_{10} due to rain scavenging. If high humidity is not accompanied by rainfall but is accompanied by high temperatures,

humidity has been found to contribute to higher PM_{10} concentrations. It has been suggested that when the relative humidity is over 55%, then PM_{10} concentrations are affected [22].

High solar radiation has also been shown to result in lower PM_{10}. When solar radiation is high, the surface of the earth is warmer; as a consequence the exchange of heat in the air results in turbulent eddies that disperse suspended particles [23].

Autocorrelation is a basic structural feature of the meteorological variables used in PM_{10} models. When a numeric time series correlates with its own past and future values, this is known as autocorrelation or *lagged correlation*. A positive autocorrelation indicates persistence and a tendency for a system to remain in the same state from one observation to the next. For example, if today is rainy, then tomorrow is more likely to be rainy. Most PM_{10} models rely solely on meteorological data from the same day (day t) and do not consider lagged ($t - n$) or lead ($t + n$) variables. However, the use of lagged variables has consistently increased the predictive power of such models. McKendry [24] found that one-day lagged ($t - 1$), two-day lagged ($t - 2$) and lead ($t + 1$) rainfall and lead ($t + 1$) wind direction contributed to models for estimating daily maximum PM_{10}. Lead ($t + 1$) daily mean temperature is also known to have good explanatory power for PM_{10}, whereas lagged daily mean temperature contributes to a lesser degree. PM_{10} is also autocorrelated and persistent, and therefore including lagged PM_{10} in the set of explanatory variables strengthens the predictive power of a model [24–26].

Co-pollutants—gases such as nitrogen monoxide (NO), nitrogen dioxide (NO_2), carbon monoxide (CO) and sulphur dioxide (SO_2)—have been found to be useful explanatory variables when used in conjunction with meteorological variables [24, 25, 27]. In many countries and especially in urban areas, road transportation is considered to be the largest contributor to PM_{10}. Road vehicles not only emit exhaust but also resuspend particulate matter [28]. Where data on traffic are not available, CO and NO_x can be used as a proxy for exhaust emissions [27].

Land usage can also influence PM_{10} concentrations, and therefore, land use type may be a useful explanatory variable. One study discovered spatial variations in PM_{10} with higher concentrations in commercial areas than in residential and industrial areas [29]. However, land use classifications are not common in PM_{10} models.

Another factor affecting PM_{10} concentrations is time. Various temporal variables have been used in models of PM_{10} concentration. Variables that reflect the seasonal cycle, such as sine and cosine of Julian day, are important for mean daily PM_{10} prediction because they reflect the dry, warm conditions typical in summer and therefore the role of photochemical production in increasing particulate matter concentrations [24]. Similarly, binary variables are sometimes used to indicate whether a period is cold or warm. For urban areas variables that reflect diurnal and weekly cycles are important due to high-density commuter and industrial traffic on weekdays contributing significantly to PM_{10} levels [16]. In urban areas of New Zealand PM_{10} has distinct diurnal cycles, with peaks between 10 pm and midnight and 8 am and 10 am, which have been found to be independent of population density [30]. Over the last 10 years, most PM_{10} models have included temporal variables.

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A very recent approach to estimating PM_{10} concentrations is the use of satellite-based remote sensing in addition to ground-level meteorological variables. MODIS (Moderate Resolution Imaging Spectroradiometer) and MISR (Multi-angle Imaging Spectroradiometer) images are analysed using algorithms designed to calculate how much direct sunlight is prevented from reaching the ground by aerosol particles—the aerosol optical depth (AOD). Several studies have used AOD to estimate $PM_{2.5}$ and PM_{10} concentrations [31] and have shown that there is a high linear correlation between particulate matter concentrations and AOD [32]. One study published in 2014 used AOD along with meteorological variables to predict ground-level PM_{10} but did not evaluate the degree to which including AOD influenced the outcome of PM_{10} predictions [32].

2.3. Regression methods

Regression methods have been used as prediction and estimation tools in a wide range of disciplines including environmental pollution and climate studies. These methods are simple to implement and compute and provide models that are easily interpretable, hence their wide adoption. Among regression methods, multivariate linear regression (MLR) is probably the most commonly used statistical method for modelling air pollution and PM_{10}. **Table 1** summarises some recent MLR PM_{10} models reported in the literature, highlighting the explanatory variables which contributed to each model.

MLR is simply a process of finding a line that best fits a multidimensional cloud of data points. The line of best fit is computed to be the line in which the squared deviations of the observed points from that line are minimised. In other words, the constant term β_0 and the coefficients are calculated so that the average error ε is zero. This line of best fit provides a model (Eq. 1), which can be used to explain the relationship between one continuous response variable y, in this case PM_{10}, and two or more explanatory variables x_i:

$$y_i = \beta_0 + \sum_{i=1}^{n} \beta_i x_i + \varepsilon_i \tag{1}$$

In MLR, it is assumed that a linear relationship exists between the response variable and the explanatory variables, all variables are normally distributed, there is little or no multicollinearity in the data (explanatory variables should not be highly correlated), and the residuals ($e_i = y_i - \hat{y}_i$) are homoscedastic (the variance around the predictor line is the same for all values of the response variable).

MLR models are considered to be limited models of PM_{10} concentration due to the inability to extend the response to non-central locations of the explanatory variables and to meet the other assumptions of the model [14, 33]. Despite possible nonconformity with one or more of the assumptions, MLR has been used extensively for predicting PM_{10} and is often used as a benchmark to which other methods are compared. Much of the PM_{10} modelling reported in the literature does not fully provide or explicitly address the data preparation and exploration

steps. Thus, it is often difficult to ascertain whether poor performance of MLR models is due to the fact that the data do not meet the assumptions of MLR, and therefore, MLR is a poor choice of model, or that the researchers chose not to refine the model when used only as a benchmark. In practice the assumption of linearity cannot be confirmed, and linear regression models are considered to be acceptable provided there are only minor deviations from this assumption. PM_{10} and its explanatory variables typically do not meet the assumption of linearity [22]. Often the variables do not have a normal distribution due to the presence of outliers.

Issues of linearity, non-normal distribution and homoscedasticity can be addressed by transforming the variable concerned. Such transformations are undertaken to linearise the relationship between the response and explanatory variables making model fitting simpler. Variable transformations should be handled carefully because in some cases, the transformation can introduce multicollinearity. Hourly PM_{10} concentrations in Athens are reported to have a logarithmic distribution so modellers performed a log transform of the PM_{10} response variable in order to improve the homoscedasticity of the residuals [16]. In Chile, maximum 24-hour moving average PM_{10} was also found to be logarithmically distributed. Again a log transform was used to normalise the data, and extreme outliers were removed [34]. After outlier analysis, all data were then normalised to ensure constant variance for each variable. In the case of Graz, Austria, the PM_{10} was gamma distributed, and a generalised linear modelling (GLM) approach using a log-link function was compared with MLR [35]. GLM is a generalisation of MLR that relates a linear model to its response variable by a link function and therefore allows for response variables that are not normally distributed. Little difference was observed between the two models, suggesting that the simpler MLR method was a better option than GLM in that case. Studies in other locales have reported that PM_{10} was normally distributed [26] or that PM_{10} concentrations were right skewed [14]. Studies have found that the use of curvilinear transformations of input variables (e.g. inverse transformation of wind speed) may result in improved regression models (e.g. see [36]).

Multicollinearity can be identified by examining the correlations among pairs of explanatory variables. However, looking at correlations only among pairs of predictors is not the best approach as even when pairwise correlations are small, it is possible that a linear dependence exists among three or more variables. Some PM_{10} modelling studies report on the linear correlation of PM_{10} with each of the possible explanatory variables but fail to examine correlations between those explanatory variables. A few recent studies have examined the correlations between pairs of explanatory variables but have not used this information to reduce multicollinearity [25]. Some researchers have used variance inflation factors (VIF) to ensure the assumption of no multicollinearity in the data before creating linear regression models [16, 22].

One approach for variable selection, in order to establish a parsimonious model, is stepwise regression: a systematic method for adding and removing terms from a multilinear model based on their statistical significance in a regression. The method begins with an initial model and then compares the explanatory power of incrementally larger (forward) or smaller (backward) models. In [16], a backward stepwise method was adopted to establish linear

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regression models for daily average PM_{10} prediction one day ahead for four different sites in Athens. One problem with stepwise refinement is that a combination of variables to add or remove may be missed where their combined effect or non-effect is hidden by collinearity.

Pires et al. [25] investigated a number of alternative linear regression models including principal component regression (PCR), independent component regression (ICR), quantile regression (QR) and partial least squares regression (PLSR).

PCR uses principal component analysis (PCA) to create new variables, or principal components, that are orthogonal and uncorrelated linear combinations of the original explanatory variables. Linear regression is then used to determine a relationship between PM_{10} and selected principal components. PCR showed no improvement over MLR in terms of model performance [25]. In an earlier study [26], a PCR model for predicting one-day-ahead mean PM_{10} in Thessaloniki, Greece, was found to perform slightly better than MLR (for which no input selection was undertaken); PCR also better predicted high daily mean spikes in concentration.

ICR is another method that extends linear regression, but in this case, the input variables are independent components—linear combinations of latent variables—that are considered to be non-Gaussian and statistically independent.

QR models the relationship between a set of predictor variables and specific percentiles (or quantiles) of the response variable and thus gives a more complete picture of the effect of the predictors on the response variable. QR has been used very little for modelling pollutants. One study that compared models developed using QR with MLR found that QR was better at predicting hourly PM_{10} concentrations [14].

Like PCR, PLSR combines PCA and MLR, but in PLSR latent variables are selected such that they provide the maximum correlation with the response variable. If all the latent variables are used, then the results of PLSR will be very similar to those of PCR. PLSR is the most flexible of the extensions to MLR modelling and can be used in cases where MLR cannot. For example, PLSR is applicable where there are a large number of explanatory variables, and as a result, multicollinearity exists.

A comparison of regression models found the size of the data set is critical to performance [25]. Both ICR and QR were found to perform poorly on a large data set. MLR, PCR and PLSR all gave similar results and performed best on a large data set. On a smaller data set, the models that removed the correlation of the variables—PCA, ICR and PLSR—performed best.

Another recent regression-based modelling approach is cluster-wise linear regression which is founded on the 'mixtures of linear regression' statistical framework [37, 38]. When compared with generalised additive non-linear models, cluster-wise linear models performed extremely well and further exploration of such models has been recommended [37].

Some research suggests that, in general, non-linear regression methods outperform their linear counterparts. One study using AOD from satellite imagery along with meteorological variables—temperature, wind speed, wind direction, relative humidity and planetary boundary layer height—to compare a non-linear regression model with a linear regression model found that the non-linear models outperformed their counterparts [32]. In the non-linear

model, non-linear functions were used to reflect the non-linear influences of the explanatory variables. For example, an exponential function of relative humidity was used to account for the growth of the particle size with increasing humidity (Eq. 2):

$$PM_{10} = \left(e^{\beta_{0+}\beta_T(T)+\beta_{WD}(WD)}\right) \times \left(e^{\beta_{RH}(RH)}\right) \times \left(AOD^{\beta_{AOD}}\right) \times \left(PBL^{\beta_{PBL}}\right) \times WS^{\beta_{WS}} \quad (2)$$

Some researchers have also explored using generalised additive models (GAMs) to model PM_{10} concentrations [14, 18, 27]. GAMs are a nonparametric extension of GLM that are flexible and able to handle non-linear relationships well. Like GLM, the response variable may have an exponential distribution (e.g. gamma, Poisson, exponential, etc.). GAMs assume that the mean of the response variable is dependent on additive predictors through a non-linear link function g and establish the nature of the relationship using smoothing functions which are determined by the data (Eq. 3):

$$g\left(E\left[y \mid x\right]\right) = \beta_0 + f_1\left(x_1\right) + \ldots + f_p\left(x_p\right) \quad (3)$$

In a daily mean PM_{10} GAM that employed co-pollutants, the previous day's mean PM_{10}, and meteorological explanatory variables, significant differences were seen between the mean observed and predicted PM_{10} concentrations [14]. The model was also found to underestimate occurrences of high PM_{10}. GAMs have not been used widely for PM_{10} modelling, possibly as they can be prone to overfitting, hard to interpret and computationally expensive.

2.4. Artificial neural networks

Artificial neural networks (ANNs) are the most common of the machine learning approaches used to model PM_{10} and have seen rapid growth in this field since the year 2000. ANNs are suited to modelling complex and dynamic non-linear systems and are particularly useful because they can be trained to accurately generalise when presented with new information. Air pollution models have been developed using ANNs for modelling and forecasting single air pollutant indicators such as ozone (O_3), NO_2, SO_2 and particulate matter. In almost all cases, ANNs, properly designed and trained, have been found to provide more accurate predictions than traditional linear statistical approaches.

ANNs are a family of computational machine learning algorithms inspired by the way biological nervous systems process and learn from information. ANNs consist of a large number of interconnected nodes, called neurons, that work together to learn and identify patterns. They have three or more layers: one input layer consisting of one or more neurons, one or more hidden layers where the learning occurs and one output layer. The neurons in this interconnected network send signals to each other along weighted connections. The most common variant of ANNs used in PM_{10} prediction is a feedforward ANN in which signals

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travel in one direction from input neurons (explanatory variables) to output (response variable).

A multilayer perceptron (MLP) is a fully connected feedforward neural network and is in essence a logistic regression classifier that facilitates non-linear transformations between inputs and outputs. The weighted values of each input are passed to the hidden layer which generates output weightings for each neuron. Each neuron j receives incoming signals from every neuron i in the previous layer. The effective incoming signal N_j to the node j is the weighted sum of all the incoming signals (Eq. 4) [39]:

$$N_j = \sum_{i=1}^{m} x_i W_{ji} \tag{4}$$

where m is the number of input neurons converging into neuron j, x_i is incoming signal and W_{ji} is a synaptic weight associated with each x_i. An activation function is then applied to the N_j to produce the output signal of the neuron j. The most commonly used activation function in backpropagation (BP) networks is the logistic sigmoid function shown in Eq. 5:

$$\sigma\left(N_j\right) = \frac{1}{\left(1 + e^{-N_j}\right)} \tag{5}$$

In order to train a neural network, the weights of each input have to be adjusted so that the error between the expected output and the predicted output is reduced. Backpropagation is a widely used method for computing the error derivative of the weights. In BP, the input data are repeatedly presented to the neural network. If the desired output is not achieved, the error weighting is propagated backwards through the network and the synaptic weights adjusted. Once the ANN has been trained, the network can be used to perform a forecast using a testing data set.

When using ANNs to model PM_{10}, it is important to ensure that there is sufficient data to adequately train the network. PM_{10} concentration time series data are typically noisy and contain outliers. A suggested 'rule of the thumb' is that a very noisy target variable requires over 30 times as many training cases as weights. The necessity for copious training data is one of the main impediments to using ANNs [40]. A validation data set is used to tune the model by exposing it to new data prior to testing its ability to predict [41]. In the PM_{10} modelling literature, many researchers do not mention the way in which they dealt with the training and testing of their models beyond the fact that they measured the difference between the observed and predicted values. It is important when using machine learning approaches to consider the size of the data sets and the approach used to train, validate and test models.

Spatial and temporal variations of PM_{10} concentration are attributed to complex interactions between high numbers of input variables which mean an even higher number of weights are

needed to train a network with a fully connected topology. The more inputs, the more time that an ANN model takes to develop. As with regression models, eliminating irrelevant or interdependent variables can improve an ANN's performance—thus keeping the model complexity low while achieving the best fit possible. Often other bioinspired optimisation methods are used in order to select the inputs for an ANN. Genetic algorithms (GAs) use a natural selection process, inspired by Darwinian evolution, to try to find the fittest solution. Individual solutions evolve through mechanisms of reproduction and mutation to produce new solutions from which only the fittest or best survive, thus finding the optimal solution—in this case the best subset of explanatory variables for PM_{10} concentration. A GA can be used to set the initial weights of the input variables for an ANN; indeed, there are some instances where GAs have been used to select the best inputs for ANN modelling of PM_{10} concentrations [24, 42].

A common rhetoric throughout work using ANNs to model PM_{10} concentrations is that ANNs are more appropriate for modelling PM_{10} than conventional deterministic approaches such as regression because ANNs are better at dealing with multiple correlated factors, uncertainty and non-linearity [43–45]. It is also generally accepted that ANNs are useful in cases where a full theoretical approach is not available and a large number of different variables are involved [46].

The first reported use of an ANN as a prediction model for ground-level SO_2 pollution was in 1993 [47]. In 1999, an MLP was used to model hourly NO_x and NO_2 pollutant concentrations in central London using basic hourly meteorological data. Their results have shown that the ANN models outperformed their previous attempts to model the same pollutants using regression-based models [48]. Another early study which compared ANNs with MLR for predicting daily mean and maximum PM_{10} and $PM_{2.5}$ reported that neural network models showed little if any improvement over regression models [24]. Despite this finding, considerable interest and work continue in the area of ANNs for modelling air quality. One study reported that multilayer ANN models are able to predict the exceedances of PM_{10} concentration thresholds in about 75% of cases, suggesting that ANNs might be suitable for predicting episodic events in which air pollution is high [20].

Since then, the vast majority of air quality prediction models developed using ANN methods have used the MLP variant. MLP models have been used to successfully predict daily average PM_{10} concentrations one day in advance in urban areas of Belgium [49] and for Santiago in Chile [20]. A BP neural network was used to predict mean hourly air pollutant concentrations ten hours ahead (CO, NO_2, O_3 and PM_{10}) for Guangzhou in China [50]. A summary of some models developed in the last ten years using ANNs to model PM_{10} is presented in **Table 2**, contrasting the input variables. These studies have used various ANN architectures and different input parameters to obtain the most predictive models possible for their locations.

A number of studies have compared the performance of ANNs and MLR [17, 33, 43, 46] for the prediction of PM_{10} for various locations. In all cases ANNs were found to outperform MLR. ANNs were also found to give better predictions for PM_{10} concentrations than a deterministic dispersion model [43]. In [15] MLP, a radial basis function ANN and PCR models for prediction of mean hourly PM_{10} concentrations in Cyprus were compared. A variety of meteorological,

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co-pollutant and temporal variables as well as lag PM_{10} were used as input to the models. Among these models, MLP was found to give the most accurate predictions for all the sites investigated.

Study reference		**[33]**	**[54]**	**[54]**	**[54]**	**[23]**	**[62]**	**[63]**	**[51]**	**[51]**	**[52]**	**[42]**	**[42]**
Country of study (ISO 3166-1 alpha 3)		MYS	CHN	CHN	CHN	MYS	NZL	NZL	MEX	MEX	TUR	IRN	IRN
Predicted variable													
PM_{10}	Daily	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Hourly												
Explanatory variables													
	PM_{10} lag	Y				Y					Y		
Co-pollutants	CO_2	Y				Y							
	SO_2	Y				Y							
	NO												
	NO_2	Y				Y							
	O_3	Y				Y							
Meteorological data	Temperature	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Temperature lag										Y		
	Wind direction					Y	Y	Y	Y	Y	Y		
	Wind direction lag										Y		
	Wind speed	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Wind speed lag										Y		
	Precipitation		Y	Y	Y								
	Solar radiation						Y						
	Sunshine hours		Y	Y	Y			Y					
	Air pressure		Y	Y	Y								
	Dew point												
	Humidity (%)	Y	Y	Y	Y		Y		Y	Y	Y	Y	Y
	Cloud cover												
	Date/time											Y	Y
	Seasonal effects												
	Spatial variables		Y	Y	Y			Y					

Table 2. Explanatory variables used in recent MLP models for predicting PM_{10} concentrations.

In order to improve on MLP model performance, one approach is to use clustering algorithms to find relationships between PM_{10} and meteorological variables using the clusters as input to the model [51]. When compared with MLP, it was found that using an ANN with k-means

clusters gave improved predictions for daily mean PM_{10} for Salamanca in Mexico. Another approach reported that improved daily predictions could be achieved by developing separate MLP models for winter and summer periods. PCA was used to create ANN input vectors which were components of the most significant lagged variables within a seven-day period [52].

A study forecasting PM_{10} using different variants of ANNs found that ANN models are effective tools for two-day-ahead prediction of incidences of high PM_{10} concentration. The models were based on four simple input variables—the daily mean wind intensity, wind speed, temperature and barometric pressure [53]. The data were preprocessed to eliminate errors introduced by instrumentation and the input variables normalised in the range of −1 to 1. The best-performing ANN was reported to be that with an Elman topology. The authors concluded that ANN models are an effective tool for early high-level air pollutant warning systems. In a recent study, three ANN models—MLP, Elman and support vector machines (SVM)—for six stations in Wuhan, China, were compared. Because of the rapid development and intensive construction occurring in the area, a Construction Index (CI) was included to represent the dust arising from building works. The time series data were decomposed into wavelet functions, and wavelet coefficients were predicted. All three models were found to give similar results, and when CI was included, the model gave improved predictions for peaks in PM_{10} concentration [54].

2.5. Alternative approaches

A few recent studies have explored other machine learning methods for modelling PM_{10}. One approach is Classification and Regression Trees (CART). CART models are obtained by recursively splitting the data and fitting a simple prediction model within each partition to create a decision tree. In one study CART was found to give better overall prediction than an ANN, but the ANN was found to better reflect temporal trends in PM_{10} [26]. Random forests (RFs) are an ensemble technique which consists of a number of learned decision trees that are used to determine the PM_{10} prediction. Predicting PM_{10} is usually considered to be a regression problem so the RF's tree responses are averaged to obtain an estimate of the dependent variable. One study showed that RFs gave better daily mean PM_{10} predictions than MLP or SVM approaches [5]. This finding is supported by another study which also found that an RF model outperformed an SVM model [55]. Tzima et al. explored a number of machine learning approaches including decision trees and RFs for predicting daily mean PM_{10} and found that logistic model trees—decision trees with logistic regression models at the leaves—gave the best predictions [56].

3. New Zealand PM_{10} trends and models

New Zealand is a country situated in the Pacific Ocean and comprises two main islands, the North Island and South Island, and numerous smaller islands. New Zealand has a mild and temperate maritime climate, but conditions can change rapidly and vary dramatically across

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regions from very wet on the west coast of the South Island, to semiarid in Central Otago, to subtropical in Northland.

New Zealand has one of the best-reported air qualities in the OECD. Legislations and guidelines govern the level of PM_{10} that is acceptable. The World Health Organisation (WHO) has set guidelines for short- and long-term PM_{10} exposure levels. These guidelines state that there should be no more than three exceedances of the daily mean limit of 50 μg/m^3 in a year and annual concentrations should not exceed 20 μg/m^3 [57]. New Zealand also has its own local standards, the National Environmental Standards (NES), which specifies a daily mean PM_{10} threshold of 50 μg/m^3 measured between the hours of midnight and midnight. One exceedance per year is allowed. In 2012, 87% of New Zealand monitoring sites meet the WHO guidelines, while 50% had levels which exceeded the short-term standard [12]. The sites which exceeded WHO guidelines were mainly in the South Island where the winters are colder and a large proportion of home heating uses wood burners. In 2013, Christchurch, also in the South Island, is reported to have exceeded the NES limit on about 30 occasions [58].

During summer, anthropogenic sources are mainly traffic and industry. Thus, PM_{10} concentrations in New Zealand are linked strongly to season. **Figure 3** (left) shows a heat map of PM_{10} concentrations by month and time of day; the highest PM_{10} concentrations occur during the evening in the winter from May to August. As in other parts of the world, PM_{10} concentrations are autocorrelated (**Figure 3**, (right)) and dependent on meteorological conditions.

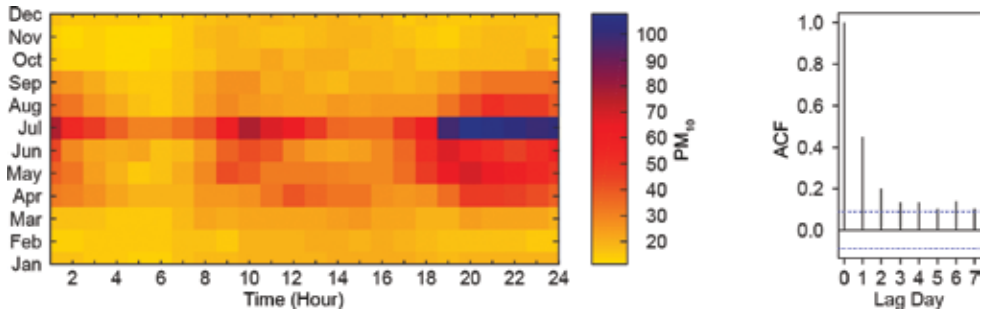

Figure 3. Heat map of PM_{10} concentrations in Timaru for 2012, data source [59] (left) and autocorrelation function (ACF) for a typical week of PM_{10} concentration observations in Auckland (right).

Christchurch is a coastal city with a flat topology surrounded by hills and is therefore subject to complex ABL winds. Wind speed and vertical temperature gradient influence PM_{10} concentrations by dispersing pollutants. A relatively recent study examined a method for studying PM_{10} emission trends by removing meteorological factors [58]. PM_{10} was calibrated and imputed using simple linear regression and transformed using the natural logarithm. Transformed PM_{10} values were then regressed against temperature, the calculated linear dependency was removed and summer observations were scaled so the PM_{10} time series was independent of seasonal fluctuations. This data set was input for multiple linear regression analysis with temperature and wind speed as explanatory variables. The resulting model could only explain 20% of the variation in PM_{10} concentrations. Residuals of the observed and

predicted values were calculated to investigate the variations in PM_{10} that were unexplained by the regression model. These residuals were added to the overall mean of the temperature-corrected PM_{10} data and a simple moving average filter applied to smooth the data. The modelled trend showed peak emissions in 2001 and 2002 with a subsequent steady decline. This trend did not match with those reported by local authorities in their three yearly emission inventories in which a steady decline was reported [58]. However, it is difficult to compare the two. In the inventory, constant emissions are assumed and then modified according to meteorological conditions and are only undertaken every three years. In [58], the method has been modified to allow for emissions that are not constant, and the model is based on hourly observations making it difficult to assess the success of the method.

In 2010, a study was undertaken to identify the influence of weather factors on occurrences of high PM_{10} concentrations—those in which the NES limits were breached—in Blenheim [60]. Blenheim is a small coastal town (population ~ 30,000) in the South Island of New Zealand. The town is on a flat area surrounded by hills on three sides. Blenheim has a dry climate with hot summers and cold winters. A boosted regression tree using a Gaussian link function was used to identify the meteorological variables which best explained the observed variance in PM_{10} concentrations. Mean daily wind speed and average temperature between 8 pm and midnight were found to best explain the variance; these variables were then used as input to a normal regression tree. It was discovered that low wind speed and low temperatures explained the majority of the NES exceedances. A similar result obtained for Invercargill using CART found that low wind speeds and low temperatures in the evening hours also accounted for most of the variation in PM_{10} levels [19]. In both studies, the model was used to account for trends in PM_{10} rather than to predict or estimate PM_{10}.

Much of the PM_{10} modelling undertaken in New Zealand until recently has been for areas in the South Island. This may be due to the fact that there is constant and historic time series data available from a well-maintained network of South Island PM_{10} monitoring stations or that frequent and higher exceedances of PM_{10} limits have been recorded for South Island regions than for regions in the North Island.

An in-depth study of Christchurch's daily mean PM_{10} employing statistical modelling approaches was undertaken using GLM, GAM, generalised additive mixed model with auto-correlated errors (GAMM + AR) and QR [61]. All of the models evaluated used a natural log transform of the PM_{10} response variable as a number of the explanatory meteorological variables impacted PM_{10} concentrations in a negative exponential form. It was concluded that simple linear regression modelling was not a suitable approach as the data violated all of the assumptions. A total of 41 meteorological variables were considered from which a subset of 20 in addition to lag PM_{10} were chosen by forward and backward stepwise selection. Models were built using the response PM_{10} data both without imputation and with missing values imputed by linear interpolation. The GAMM+AR model was found to be the best prediction model and able to explain around 70% of the variability in daily average PM_{10} concentrations [61].

There have been very few models developed using ANNs to estimate or predict PM_{10} concentrations in New Zealand. Gardner and Dorling [48] compared the performance of

different models such as linear regression, feedforward ANNs and CART approaches for modelling mean hourly PM_{10} in Christchurch, New Zealand. As with studies in other parts of the world, ANNs were found to be the best-performing modelling method. In another more recent study, ANNs were combined with a k-means clustering method to group and rank explanatory variables. The data used were from Auckland—New Zealand's most populated city with a population of over 1.4 million. It was found that the inclusion of cluster rankings, derived from k-means cluster analysis, as an input parameter to the ANN model showed a statistically significant improvement in the performance of the ANN model and that the model was also better at predicting high concentrations [62, 63].

Near-ground maximum PM_{10} concentrations for two sites in Timaru, a small rural town, were estimated using a feedforward backpropagation ANN with a hyperbolic tangent sigmoid function [41]. The response and explanatory variables were normalised. Additionally, due to the correlation between the seasonal changes and PM_{10} concentration, the PM_{10} data were divided into high season (winter/autumn) and low season (spring/summer) classes prior to creating the model. The inputs included one-day lagged meteorological variables and one-day lagged PM_{10}, in addition to meteorological variables for the day of estimation. Levenberg-Marquardt optimisation and Bayesian regularisation training were evaluated, and it was found that Bayesian regularisation was the best approach for tuning the weights and bias values for the network. This approach gave good estimations of daily mean PM_{10} concentration for both sites.

Some research has been conducted using TAPM, a deterministic global atmospheric pollutant model, [4] which includes fundamental fluid dynamics and scalar transport equations to predict meteorology and pollutant concentration [64–66]. Localised models of PM_{10} concentrations for two South Island towns, Alexandra (population ~ 5000) and Mosgiel (population ~ 10,000), were developed. Alexandra has a borderline oceanic semiarid climate—the country's coldest, driest and warmest—due to its geographic location as New Zealand's most inland town. Mosgiel is separated from Dunedin city by hills and is situated on a plain. It has a temperate climate with a significant annual average rainfall of 738 mm. TAPM was found to correctly predict daily PM_{10} concentration breaches and non-breaches of the NES 66% of the time in Alexandra and 71% of the time in Mosgiel [65]. Another study has looked at TAPM for simulating PM_{10} dispersion for a single winter in Masterton and also obtained good predictions of PM_{10} [65]. Yearlong PM_{10} was modelled using TAPM for Christchurch city. TAPM was reported to provide an acceptable simulation of ground-level weather and PM_{10} dispersion (with a 4 $\mu g/m^3$ difference in annually averaged concentration of modelled and measured PM_{10}), but the model tended to overestimate wind speed during still nights resulting in low PM_{10} estimates for those periods [66].

4. Summary

Although there are now several models available for predicting PM_{10}, it is difficult to compare them. The complex nature of ambient particulate matter composition and the physical and

chemical transformations that particulate matter can undergo between emission source and sampling location seems to mean that PM_{10} concentrations are largely explained by location-specific variables and events. Meteorological variables used in these localised models tend to be restricted to those which are routinely collected by local authorities.

It is also difficult to compare models because of the variation in PM_{10} instrumentation and measurement approaches used between different studies. In the future, improved sensor technology and lower costs associated with such monitoring could allow for more comprehensive coverage of areas—improving the inputs available for modelling. Ability to sense at different atmospheric levels should also enhance the data and in turn any empirical models. The use of geo-topological features such as elevation and land use could be considered as inputs for modelling as they reflect site-specific conditions and are readily available, but few models utilise these variables. Inclusion of air quality data, such as AOD measurements, from satellite-based remote sensing should also enhance models. Such data have the potential to provide a means of imputing missing values, to verify and enhance the accuracy of sensor-based ground-level observations and to provide additional inputs to models.

While general trends in PM_{10} concentrations can be explained and similarities can be seen between countries and factors contributing to PM_{10}, no empirical comparison can be made between models developed for specific locations. An attempt to develop a single general model for an area found that the general model performed poorly compared with site-specific models [32]. Some of these site-specific issues are removed when a deterministic physiochemical modelling approach is used, but accuracy of such models is currently limited as many of the actual mechanisms involved in pollution generation, dispersal, dilution and removal are not fully understood. However, it is possible that in the future, with better understanding, deterministic models could prove to be the way forwards.

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References

[1] Stone R. Counting the Cost of London's Killer Smog. Science. 2002;298(5601):2107–7.

[2] European Environment Agency. Air quality in Europe-2015 report [Internet]. 2015 . Available from: http://www.eea.europa.eu/publications/air-quality-in-europe-2015 [Accessed: 2016-07-22]

[3] Statistics NZ. Health effects from exposure to PM10 [Internet]. 2012 . Available from: http://www.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/Air/health-effects.aspx [Accessed: 2016-07-22]

[4] CSIRO. The Air Pollution Model (TAPM) [Internet]. 2015 . Available from: http://www.csiro.au/en/Research/OandA/Areas/Assessing-our-climate/Air-pollution/TAPM [Accessed: 2016-07-20]

[5] Siwek K, Osowski S. Data mining methods for prediction of air pollution. International Journal of Applied Mathematics and Computer Science. 2016;26(2):467–78.

[6] Nussbaumer T, Czasch C, Klippel N, Johansson L, Tullin C. Particulate emissions from biomass combustion in IEA countries. In: 16th European biomass conference and exhibition; 2–6 June; Valencia, Spain. 2008.

[7] Brown AS, Yardley RE, Quincey PG, Butterfield DM. Ambient air particulate matter: quantifying errors in gravimetric measurements. NPL Report DQL-AS. 2005 Jan:41.

[8] Ministry for the Environment. Good Practice Guide for Air Quality Monitoring and Data Management 2009 [Internet]. 2009. Available from: https://www.mfe.govt.nz/sites/default/files/good-practice-guide-for-air-quality.pdf [Accessed: 2016-07-22]

[9] Bluett J, Wilton E, Franklin P, Dey K, Aberkane T, Petersen J, et al. PM10 in New Zealand's urban air: a comparison of monitoring methods [Internet]. 2007. Available from: https://www.niwa.co.nz/sites/niwa.co.nz/files/import/attachments/CHC2007_059.pdf [Accessed: 2016-07-22]

[10] Pakalidou N, Katragkou E, Poupkou A, Zanis A, Bloutsos S, Karacostas, T. Analysis of Heat-Wave Events in Thessaloniki and Investigation of Impacts on PM10. In: Helmis G.C., Nastos T.P., editors. Advances in Meteorology, Climatology and Atmospheric Physics. Berlin, Heidelberg: Springer; 2013. p. 663–669. DOI: 10.1007/978-3-642-29172-2_94

[11] Junpen A, Garivait SB, Onnet P, Ongpullponsak A. Spatial and temporal distribution of forest fire PM10 emission estimation by using remote sensing information. International Journal of Environmental Science and Development. 2011;2(2):156–61. DOI: 10.7763/IJESD.2011.V2.115

[12] Ministry for the Environment and Statistics New Zealand. New Zealand's Environmental Reporting Series: 2014 Air domain report [Internet]. 2014 . Available from: www.mfe.govt.nz [Accessed: 2016-07-22]

[13] Tiwari S, Chate DM, Pragya P, Ali K, Bisht DS. Variations in mass of the PM 10, PM 2.5 and PM 1 during the monsoon and the winter at New Delhi. Aerosol and Air Quality Research. 2012;12(1):20–9. DOI: 10.4209/aaqr.2011.06.0075

[14] Sayegh AS, Munir S, Habeebullah TM. Comparing the performance of statistical models for predicting PM10 concentrations. Aerosol and Air Quality Research. 2014;14(3):653–65. DOI: 10.4209/aaqr.2013.07.0259

[15] Paschalidou AK, Karakitsios S, Kleanthous S, Kassomenos PA. Forecasting hourly PM10 concentration in Cyprus through artificial neural networks and multiple regression models: implications to local environmental management. Environmental Science and Pollution Research. 2011;18(2):316–27.

[16] Grivas G, Chaloulakou A. Artificial neural network models for prediction of PM10 hourly concentrations, in the Greater Area of Athens, Greece. Atmospheric Environment. 2006;40(7):1216–29.

[17] Papanastasiou DK, Melas D, Kioutsioukis I. Development and assessment of neural network and multiple regression models in order to predict PM10 levels in a medium-sized Mediterranean city. Water, Air, & Soil Pollution. 2007;182(1–4):325–34.

[18] Barmpadimos I, Hueglin C, Keller J, Henne S, Prévôt ASH. Influence of meteorology on PM 10 trends and variability in Switzerland from 1991 to 2008. Atmospheric Chemistry and Physics. 2011;11(4):1813–35.

[19] Wilton E, Appelhans T, Baynes M, Zawar Reza P. Assessing long- term trends in PM 10 concentrations in Invercargill [Internet]. 2009 . Available from: http://www.environet.co.nz/environet/documents/TrendsPM10InvercargillFinal.pdf [Accessed: 2016-07-22]

[20] Perez P, Reyes J. Prediction of maximum of 24-h average of PM10 concentrations 30 h in advance in Santiago, Chile. Atmospheric Environment. 2002;36(28):4555–61.

[21] Olszowski T. Changes in PM10 concentration due to large-scale rainfall. Arabian Journal of Geosciences. 2016;9(2):1–11.

[22] Ul-Saufie AZ, Yahaya AS, Ramli NA, Rosaida N, Hamid HA. Future daily PM10 concentrations prediction by combining regression models and feedforward backpropagation models with principle component analysis (PCA). Atmospheric Environment. 2013;77:621–30.

[23] Afzali A, Rashid M, Sabariah B, Ramli M. PM10 pollution: its prediction and meteorological influence in PasirGudang, Johor. IOP Conference Series: Earth and Environmental Science. 2014;18: 012100. DOI: 10.1088/1755-1315/18/1/012100

[24] McKendry IG. Evaluation of artificial neural networks for fine particulate pollution (PM10 and PM2.5) forecasting. Journal of the Air & Waste Management Association. 2002;52(9):1096–101.

[25] Pires JCM, Martins FG, Sousa SI V, Alvim-Ferraz MCM, Pereira MC. Prediction of the daily mean PM 10 concentrations using linear models. American Journal of Environmental Sciences. 2008;4(5):445–53.

[26] Slini T, Kaprara A, Karatzas K, Moussiopoulos N. PM10 forecasting for Thessaloniki, Greece. Environmental Modelling & Software. 2006;21(4):559–65.

[27] Munir S, Habeebullah TM, Seroji AR, Morsy EA, Mohammed AMF, Saud WA, et al. Modeling particulate matter concentrations in Makkah, applying a statistical modeling approach. Aerosol and Air Quality Research. 2013;13(3):901–10.

[28] Chen L, Bai Z, Kong S, Han B, You Y, Ding X, et al.. A land use regression for predicting NO2 and PM10 concentrations in different seasons in Tianjin region, China. Journal of Environmental Sciences. 2010;22(9):1364–73.

[29] Yu CH, Fan Z-H, Meng Q, Zhu X, Korn L, Bonanno LJ . Spatial/temporal variations of elemental carbon, organic carbon, and trace elements in PM 10 and the impact of land-use patterns on community air pollution in Paterson, NJ. Journal of the Air & Waste Management Association. 2011;61(6):673–88.

[30] Trompetter WJ, Davy PK, Markwitz A. Influence of environmental conditions on carbonaceous particle concentrations within New Zealand. Journal of Aerosol Science. 2010;41(1):134–42.

[31] Péré JC, Pont V, Mallet M, Bessagnet B. Mapping of PM10 surface concentrations derived from satellite observations of aerosol optical thickness over South-Eastern France. Atmospheric Research. 2009;19(1):1–8.

[32] Sotoudeheian S, Arhami M. Estimating ground-level PM10 using satellite remote sensing and ground-based meteorological measurements over Tehran. Journal of Environmental Health Science & Engineering. 2014;12(1):122. DOI: 10.1186/s40201-014-0122-6

[33] Ul-saufie AZ, Yahaya AS, Ramli NA, Hamid HA. Comparison between Multiple linear regression and feedforward back propagation neural network models for predicting PM 10 concentration level based on gaseous and meteorological parameters. International Journal of Applied Science and Technology. 2011;1(4):42–29.

[34] Díaz-Robles LA, Ortega JC, Fu JS, Reed GD, Chow JC, Watson JG, et al. A hybrid ARIMA and artificial neural networks model to forecast particulate matter in urban areas: The case of Temuco, Chile. Atmospheric Environment. 2008;42(35):8331–40.

[35] Stadlober E, Zuzana H. Forecasting of daily PM10 concentrations in Brno and Graz by different regression approaches. Austrian Journal of Statistics. 2012;41(4):287–310.

[36] Chaloulakou A, Kassomenos P, Spyrellis N, Demokritou P, Koutrakis P. Measurements of PM10 and PM2.5 particle concentrations in Athens, Greece. Atmospheric Environment. 2003;37(5):649–60.

[37] Poggi JM, Portier B. PM10 forecasting using clusterwise regression. Atmospheric Environment. 2011;45(38):7005–14.

[38] Misiti M, Misiti Y, Poggi J, Portier B. Mixture of linear regression models for short term PM10 forecasting in Haute Normandie (France). Case Studies in Business, Industry and Government Statistics. 2015;6(1):47–60.

[39] Ozdemir U, Taner S. Impacts of meteorological factors on PM10: Artificial Neural Networks (ANN) and Multiple Linear Regression (MLR) approaches. Environmental Forensics. 2014;15(4):329–26.

[40] Ordieres JB, Vergara EP, Capuz RS, Salazar RE. Neural network prediction model for fine particulate matter (PM2.5) on the US–Mexico border in El Paso (Texas) and Ciudad Juárez (Chihuahua). Environmental Modelling & Software. 2005;20(5):547–59.

[41] Zandi S, Whalley J, Sallis P, Ghobakhlou A. Estimation of Near Ground PM10 Concentrations using Artificial Neural Networks. In: Weber T, McPhee MJ, Anderssen RS, editors. MODSIM2015, 21st International Congress on Modelling and Simulation; Queensland, Australia. Modelling and Simulation Society of Australia and New Zealand Inc.; 2015. p. 42.

[42] Asghari Esfandani M, Nematzadeh H. Predicting air pollution in Tehran: Genetic algorithm and back propagation neural network. Journal of AI and Data Mining. 2016;4(1):49–54.

[43] Kukkonen J, Partanen L, Karppinen A, Ruuskanen J, Junninen H, Kolehmainen M, et al. Extensive evaluation of neural network models for the prediction of NO2 and PM10 concentrations, compared with a deterministic modelling system and measurements in central Helsinki. Atmospheric Environment. 2003;37(32):4539–50.

[44] Esplin GJ. Approximate explicit solution to the general line source problem. Atmospheric Environment. 1995;29(12):1459–63.

[45] Hewitson B, Crane RG. Neural Nets: Applications in Geography. 1st ed. Netherlands: Springer; 1994. 196 p.

[46] Chaloulakou A, Grivas G, Spyrellis N. Neural network and multiple regression models for PM10 prediction in Athens: a comparative assessment. Journal of the Air & Waste Management Association. 2003;53(10):1183–90.

[47] Boznar M, Lesjak M, Mlakar P. A neural network-based method for short-term predictions of ambient SO_2 concentrations in highly polluted industrial areas of complex terrain. Atmospheric Environment. Part B. Urban Atmosphere 1993;27(2):221–30.

[48] Gardner MW, Dorling SR. Regression modelling of hourly NO(x) and NO_2 concentrations in urban air in London. Atmospheric Environment. 1999;33:709–19.

[49] Hooyberghs J, Mensink C, Dumont G, Fierens F, Brasseur O. A neural network forecast for daily average PM10 concentrations in Belgium. Atmospheric Environment. 2005;39(18):3279–89.

[50] Cai M, Yin Y, Xie M. . Prediction of hourly air pollutant concentrations near urban arterials using artificial neural network approach. Transportation Research Part D: Transport and Environment. 2009;14(1):32–41. DOI: 10.1016/j.trd.2008.10.004

[51] Cortina-Januchs MG, Quintanilla-Dominguez J, Vega-Corona A, Andina D. Development of a model for forecasting of PM10 concentrations in Salamanca, Mexico. Atmospheric Pollution Research. 2015;6(4):626–34. DOI: 10.5094/APR.2015.071

[52] Taşpinar F. Improving artificial neural network model predictions of daily average PM10 concentrations by applying principle component analysis and implementing seasonal models. Journal of the Air & Waste Management Association. 2015;65:800–9. DOI: 10.1080/.

[53] Brunelli U, Piazza V, Pignato L, Sorbello F, Vitabile S. Two-days ahead prediction of daily maximum concentrations of SO_2, O_3, PM10, NO_2, CO in the urban area of Palermo, Italy, Atmospheric Environment. 2007;41(14):2967–95.

[54] Feng Q, Wu S, Du Y, Xue H, Xiao F, Ban X, et al. Improving neural network prediction accuracy for PM10 individual air quality index pollution levels. Environmental Engineering Science. 2013;30(12):725–32. DOI: 10.1089/ees.

[55] Ishak AB, Moslah Z, Trabelsi A. Analysis and prediction of PM10 concentration levels in Tunisia using statistical learning approaches. Environmental and Ecological Statistics. 2016; 23:1-22. DOI: 10.1007/s10651-016-0349-8

[56] Tzima FA, Karatzas KD, Mitkas PA, Karathanasis S. Using data-mining techniques for PM10 forecasting in the metropolitan area of Thessaloniki, Greece. In: IEEE International Conference on Neural Networks; 2007. p. 2752–7.

[57] World Health Organization. WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide: global update 2005: summary of risk assessment [Internet]. 2006 . Available from: http://whqlibdoc.who.int/hq/2006/WHO_SDE_PHE_OEH_06.02_eng.pdf?ua=1 [Accessed: 2016-07-22]

[58] Appelhans T, Sturman A, Zawar-Reza, P. Modelling emission trends from non-constant time series of PM 10 concentrations in Christchurch, New Zealand . International Journal of Environment and Pollution 2010;43(4):354–63.

[59] Environment Canterbury. Data Catalogue [Internet]. 2016. Available from: http://data.ecan.govt.nz/

[60] Wilton E, Rijkenberg M, Bluett J. Assessing long-term trends in PM 10 concentrations in Blenheim [Internet]. 2010 . Available from: http://www.marlborough.govt.nz/Environment/Air-Quality/~/media/Files/MDC/Home/Environment/Air Quality/TrendsInAirQuality9.ashx [Accessed: 2016-07-21]

[61] Scarrott C, Reale M, Newell J. Statistical estimation and testing of trends in PM 10 concentrations: is Christchurch city likely to meet the NES target for PM 10 concentrations in 2013? [Internet]. 2013 . Available from: http://ecan.govt.nz/publications/Reports/PM10TrendsComplete.pdf [Accessed: 2016-07-22]

[62] Elangasinghe MA, Singhal N, Dirks KN, Salmond JA, Samarasinghe S. Complex time series analysis of PM10 and PM2.5 for a coastal site using artificial neural network

modelling and k-means clustering. Atmospheric Environment. 2014;94:106–16. DOI: 10.1016/j.atmosenv.2014.04.051

[63] Elangasinghe MA. Applications of semi-empirical and statistical techniques in urban air pollution modelling [thesis]. University of Auckland; 173 p. Available from: https://researchspace.auckland.ac.nz/handle/2292/23444

[64] Tate A. Wintertime PM10 measurements and modelling in Alexandra and Mosgiel, Otago, New Zealand [thesis]. University of Otago; 2012. 117 p. Available from: http://hdl.handle.net/10523/2255

[65] Xie S, Gimson N, Clarkson T. Modelling wintertime PM10 dispersion in Masterton, New Zealand: a tool for implementing national standards. WIT Transactions on Ecology and the Environment. 2006;86:65–74.

[66] Zawar-Reza P, Kingham S, Pearce J. Evaluation of a year-long dispersion modelling of PM10 using the mesoscale model TAPM for Christchurch, New Zealand. Science of the Total Environment. 2005;349(1–3):249–59.

Chapter 3

Economics and Air Pollution

Fernando Carriazo

Additional information is available at the end of the chapter

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Abstract

This chapter discusses the relationship between economics and air pollution: first, it presents the main characteristics of the economic growth-environmental pressure debate and introduces the concept of environmental Kuznets curve hypothesis (EKC). As an example of the EKC, the estimated relationship between CO_2 emissions and economic growth, using a cross-sectional sample of 152 countries, is reported. Second, the chapter discusses air pollution as a result of a market failure and introduces the main theoretical causes of ambient degradation, acknowledging air pollution externalities as a common problem that leads to overexploitation in the absence of well-defined property rights for the atmosphere. Third, the main instruments for pollution control, including traditional regulation based on standards and the more flexible incentive-based regulation, are presented. Finally, the chapter reviews the main features of cost and benefits related to air pollution emissions.

Keywords: air pollution, air quality, incentive-based regulations, transferable permits, air quality standards, cost-benefit analysis

1. Introduction

Most marketable goods are produced by the manufacturing sector. Industry is, with no doubt, one of the main contributors of economic growth, and its products are the bases of today's lifestyles. The extraction of raw materials from natural resources is part of a transformation process where industries introduce both final products and pollution to society. In 2013, the proportion of value added from manufacturing (VAM) to the gross domestic product varied from 8.6% in low-income countries to 20.8% in middle high-income countries (see **Table A1** in Appendix). Nonetheless, in the information era, traditional manufacturing is no longer the most important sector of the economy. The world average VAM has decreased from 19.2 to 16.3%

during the period 2000–2013. This trend of the VAM is observed in both low-income and high-income countries, possibly explained by a more integrated industry to the services' sector and the emergence of the so-called information economy.

Notwithstanding the trend described above, the negative effects of industrial activity on the environment, which once were perceived exclusively as a local pollution problem, today are widely debated in public policy, aiming not only to preserve the environment, but also to mitigate and adapt to climate change produced by greenhouse gases. Overall, emissions from CO_2 coming from the industrial manufacturing and construction sectors are higher in low-income countries than in high-income countries. Even though VAM decreased worldwide, emissions of CO_2 from the industrial sector do not seem to follow the same path. While CO_2 emissions represented 18% of the total consumption of fossil fuels in 2000, they reached 20% in 2013 (see **Table A2** in Appendix).

Traditionally, analysts of pollution classify this physical phenomenon into three categories: (1) greenhouse gases such as CO_2 or methane, (2) pollution from chemical imbalances, and (3) aerosols.

Regarding the first category, in Ref. [1], NASA estimates that the global levels of carbon dioxide have increased since 1960 from 316 ppm to near 407 ppm in 2016. As documented in Ref. [2], these high levels of CO_2 have caused the world to increase its temperature, and if the current trend persists, by 2035, we would observe a level of 535 ppm, which makes a global average rise of at least 2°C more likely. Today, strong scientific evidence suggests links between concentrations of greenhouse gases and changes in global temperature: a rising of 2°C relative to preindustrial levels could bring as a result risks in food security, significant changes in water availability, collapse of ecosystems, extreme weather events, and irreversible impacts such as the melting of Greenland ice sheet [2].

Chemical imbalances, on the other hand, create pollution mainly in the form of nitrogen oxides and sulfur oxides, which in turn produce acid rain and smog. Chemical imbalances such as the incomplete combustion of fossil fuels produce carbon monoxide which is known for its toxicity and the concomitant risks for the respiratory and circulatory systems.

Aerosols are essentially suspended particles of different sizes that are produced mainly due to the burning of fossil fuels and industrial processes. Particulate matter affects water systems, agriculture, and also the respiratory system of humans. Most local ambient pollution is attributed to chemical imbalances or to aerosols whose main sources are vehicles or combustion processes in industrial facilities. The World Health Organization (WHO) recognizes that air pollution is a major environmental risk to health. In fact, the WHO estimated that ambient air pollution caused 3.7 million premature deaths in both urban and rural areas worldwide in 2012 [3].

The deterioration of air quality from industrial smokestacks' emissions and other sources that are well beyond the control of individuals has triggered public concerns and has inspired heated debates on the trade-off between environmental quality and economic growth. In this chapter, we discuss first the nature of the debate between growth and environmental quality; second, the theoretical causes of environmental degradation from the perspective of econom-

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ics; and, third, the type of economic instruments that economists have proposed to regulate ambient pollution as an alternative to traditional policies based on the so-called command and control approach. Then, we briefly discuss the framework to think about costs and benefits of pollution. In the last section, we discuss some lessons learned.

2. Economic growth and emissions

The relationship between environmental degradation and economic growth has been object of constant debate among environmental economists. During the last two decades, the debate between economic growth and the environment introduced into the discussion the environmental Kuznets curve hypothesis (EKC). Within the framework of the EKC hypothesis, it is expected to observe an inverse U curve relationship between a variable that measures environmental pressure (i.e., air pollution) and economic growth (usually measured as income per capita). The name of environmental Kuznets is related to the bell-shaped relationship between income distribution (inequality) and economic growth that the economist Simon Kuznets suggested in the 1950s. Identifying the patterns of environmental pressure and economic growth allows researchers to understand whether economic growth is part of the solution to environmental problems, or whether, on the contrary, policies aimed to encourage economic growth have detrimental consequences on the environment. As pointed out by De Bruyn and Heintz [4], if we accept the hypothesis of the environmental Kuznets curve, we are recognizing that (1) pollution and environmental degradation is only a temporary phenomenon, and (2) economic growth can be part of the solution to global environmental problems.

Literature on the environmental Kuznets curve is extensive. An overview of some of the most influential empirical studies that have examined the relationship between indicators of environmental pressure and per capita income levels can be found in Ref. [4]. Most studies have used as indicators water and air pollution.

In Ref. [5], the authors examined the relationship between urban air pollution and economic growth. These authors did not find evidence that environmental quality decreases with economic growth. Grossman and Krueger found an inverse U-shaped relationship and calculated the turning points of this relationship.[1] For most pollutants examined, after economies reach US$8000 per capita (in 1985 dollars), environmental pressure starts to decrease.

Ref. [6] overviews the most cited studies during the first half of the 1990s. In particular, it examined the study of Grossman and Krueger mentioned above as well as the estimations from Shafik and Bandyopadhyay [7], Panayotou [8], and Selden and Song [9].

As pointed out by Stern et al. [6], the study in Ref. [7] estimated an environmental Kuznets curve for air pollution measured as total suspended particulate matter, emissions of sulfur oxides, and carbon emissions per capita. In their estimations, these authors found that the EKC hypothesis is accepted for these air pollutants. According to their estimations, the level of

[1] Turning point refers to the level of income at which environmental pressures start to decline.

income at which air pollution starts to decline is between US$3000 and US$4000. In Ref. [9], the authors tested the EKC for the following air pollutants: SO_2, NO_x, SPM, and CO. They found that all pollutants adjust very well to the EKC except carbon monoxide (CO). The following turning points of income were found in this work: for SO_2 US$8709; for NO_x US$11217; for SPM US$10289; and for CO US$5963. However, coefficients were not statistically significant for the latter. In Ref. [6], Stern et al. refer to the study in Ref. [8] in which the author estimated EKC for air pollution (emissions of SO_2, NO_x, and SPM) with a sample of 54 countries. He found that the turning point for air pollution lies between US$3000 and US$5500, depending on the pollutant used in estimations. Most studies surveyed applied reduced forms of the relationship of environmental pressure and economic growth. It should be noted that the economic model corresponds to a descriptive behavior of what it is expected to be observed within the EKC framework.

The basic model of the environmental Kuznets curve hypothesis describes the pattern of the environmental pressure for different growth levels in the economy. Following the explanation of the relationship between income growth and environmental pressure in Ref. [4], environmental pressure increases faster than GDP during the first stage of economic development. This is identified as the first phase of the EKC. The second phase is characterized by an increase in the environmental pressure but at a lower rate than the increase of GDP. In other words, during the second phase, pollution increases at a decreasing rate until the curve reaches a maximum. The third phase starts at the maximum point of environmental pressure. In this phase, the EKC starts to decrease, and if it continues to decrease when income levels tend to infinity, then economic growth is not linked anymore to environmental pressure. In this case, there is an authentic environmental Kuznets curve, in which the pattern of environmental pressure follows an inverted U-shaped curve. If we observe a certain level of income at which environmental pressure starts to increase again, then we are in the presence of the fourth phase, where there is a period of relinking between income and environmental pressure. Some authors have called the environmental pressure-economic growth relationship an N-shaped curve when this phase is observed.

To illustrate the application for estimating the relationship between economic growth and environmental degradation, the following reduced form was considered:

$$CO_2 = \beta_0 + \beta_1 Y_i + \beta_2 Y_i^2 + \beta_3 Y_i^3 + e_i \quad (1)$$

CO_{2i} is per capita emissions of CO_2 of country i, Y_i is GDP per capita of country i. β_0 is a constant that indicates the average level of CO_2 when per capita income does not influence environmental pressure; β_k indicates the importance of GDP per capita on environmental pressure; and the usual assumption of an error term that is normally distributed with mean zero and constant variance $e_i \sim N(0, \sigma^2)$ holds. Results of Eq. (1), estimated by generalized least squares, are reported in **Table 1**. Descriptive statistics and data sources are presented in **Table A3** in Appendix.

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Estimation results suggest that CO_2 emissions follow an N-shaped pattern. As income increases, CO_2 emissions increase until income reaches a turning point. These stages are described by the positive and negative coefficients of the GDP pc and GDP pc squared variables, respectively. This suggests the presence of an environmental Kuznets curve. However, because the coefficient of GDP cubed is statistically different from zero, the estimated relationship between environmental pressure and economic growth suggests the possibility of relinking. In other words, CO_2 emissions increase again at high levels of income. Taking into account the sign of the estimated coefficients and their significance level, we reject H_0: $\beta_1 > 0, \beta_2 < 0 \beta_3 = 0$ and conclude in favor of H_a: $\beta_1 > 0, \beta_2 < 0 \beta_3 > 0$. This suggests that the relationship between CO_2 emissions follows an N-shaped polynomial and not a parabolic (inverted U-shaped) polynomial as the environmental Kuznets curve predicts (see **Figure 1**).

Variable	Parameter estimate	SE	*T* value	*P* value
Constant	0.0220	0.1289	0.17	0.8646
GDP pc	0.0016**	0.000182	9.09	<0.0001
GDP pc squared	−6.59E−8**	1.599E−8	−4.12	<.0001
GDP pc cubed	7.52E−13*	3.24E−13	2.32	0.0219

Dependent variable. CO_2 per capita emissions.
*Significant at 5% level;
**Significant at 1% level.
$R^2 = 0.2295$.

Table 1. GLS estimation for model (1).

The results from model in Eq. (1) are useful to determine the income level for which the relationship between environmental pressure and economic growth starts to decline given that the turning point is a maximum.

To calculate the turning points, we set up the first derivative of the fitted cubic polynomial equal to zero and solve for Y. Therefore, given in Eq. (2):

$$CO_2 = \beta_0 + \hat{\beta}_1 Y_i + \hat{\beta}_2 Y_i^2 + \hat{\beta}_3 Y_i^3 \tag{2}$$

the first-order condition for maximum (minimum) is given in Eq. (3):

$$\frac{\partial CO_2}{\partial Y} = \hat{\beta}_1 + 2\hat{\beta}_2 Y_i + 3\hat{\beta}_3 Y_i^2 = 0 \tag{3}$$

By solving Eq. (3) for Y_i, we find the turning points. The fitted model suggests that CO_2 per capita emissions will follow an N pattern: as income per capita increases, CO_2 emissions increase until we reach an income level per capita near to US$18,000. After this income level,

pollution decreases. However, this decline in environmental pressure is not permanent. When the economy reaches a level of income per capita near US$40,000, CO_2 emissions are likely to increase again. In other words, there is a relinking of the environmental pressure-economic growth positive relationship. This suggests a cyclical behavior of pollution and economic growth.

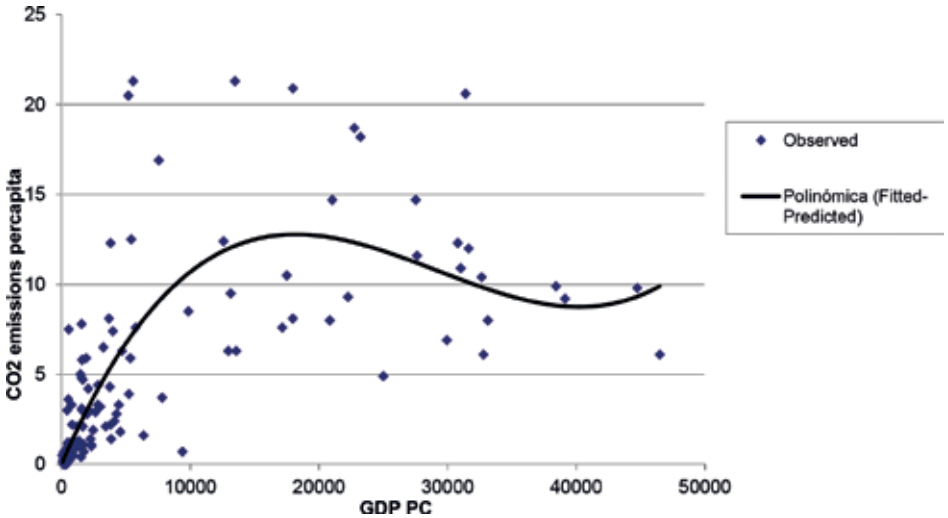

Figure 1. CO_2 emissions and economic growth.

The estimated model is suggestive but not conclusive: the predicted pattern follows an N-shaped curve. The observed behavior of CO_2 per capita emissions suggests that at very low income levels, air pollution occurs at minimum levels. This could be related to low levels of industrial activity. Nevertheless, as per capita income grows, the economy shifts from an agricultural intensive sector to a more industrial intensive sector. This brings as a result more pollution. However, once the economy reaches the turning point of income, air pollution decreases. This decline may be associated with the development of cleaner technologies or a more service-intensive (nonpollutant) sector of the economy. In Ref. [4], the authors discuss other possible explanations for the inverted U-shaped relationship. They argue that at higher income levels, one may observe behavioral changes and changes in preferences that are related to a cleaner environment. For example, individuals are more likely to be willing to pay for environmentally friendly vehicles and ecoproducts. Also, institutional changes may influence the income-pollution relationship; at lower income levels of an economy, institutions that regulate the environment are not very well developed, whereas in rich economies, environmental agencies are more likely to enforce pollution reductions. In fact, the most developed economic instruments designed for air pollution control occur in middle high- or high-income countries.

The next section discusses the theoretical causes that help explain pollution from the point of view of economics. This analysis is highly related to what the first phase of the EKC suggests:

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in the absence of pollution control, either from environmental agencies or from economic incentives, higher levels of pollution are likely to occur. Excessive pollution levels cause damages that affect the well-being of individuals who cannot directly control emissions. The question we intend to answer in the next session is, "How can air pollution be explained from an economics' theoretical perspective?" In the subsequent sections, we will discuss the strategies to curve pollution levels.

3. Air pollution as an externality

External effects or *externality* is one of the most basic concepts evoked by economists when looking at problems of environmental pollution (e.g., see [10–12]). In the economics framework, an externality is an important source of market failure that arises when the production or consumption activities of a person or a firm influence the well-being of a bystander. The side effect of the activity on others is typically unintended and uncompensated. A negative externality occurs when the impact of the bystander is adverse and a positive externality when it is beneficial.

Air pollution is essentially a negative externality: it imposes external costs to people who are external to the transaction of a polluting product. Further, economists typically define air pollution as a negative externality in production.

From an economics perspective, demand law suggests an inverse relationship between price and the quantity consumed of a marketable product. However, when a product does not have a very well-established market, this product will be most likely underpriced. This is the case of natural systems such as air or water. The lack of property rights for these natural inputs and the absence of environmental regulation or legal protection to pollution receptors make a firm to perceive air as an input that can be freely used, like a common resource, thus neglecting all external costs imposed to other agents of the economy. In other words, if there were well-defined property rights for air, firms would have to buy the right to pollute it and emissions could be internalized through a market mechanism. When buyers and sellers do not take into account the external costs of their actions while deciding how much to consume or to produce, the market equilibrium is not efficient and the price of a good does not necessarily reflect its social value.

Figure 2 illustrates the previous point through a simple supply and demand analysis. Let us consider the industry of lead smelter. Factories that produce recovered lead ingots from recycled batteries emit pollution. For each ingot produced, certain amount of smoke enters the atmosphere. Because exposure to lead smoke may cause poisoning that affects the central nervous system, especially in children, this smoke is a negative externality. In other words, because there is no market where the external costs that smoke emissions impose to individuals are reflected in the price of lead ingots, this externality affects the efficiency of the market outcome. How?

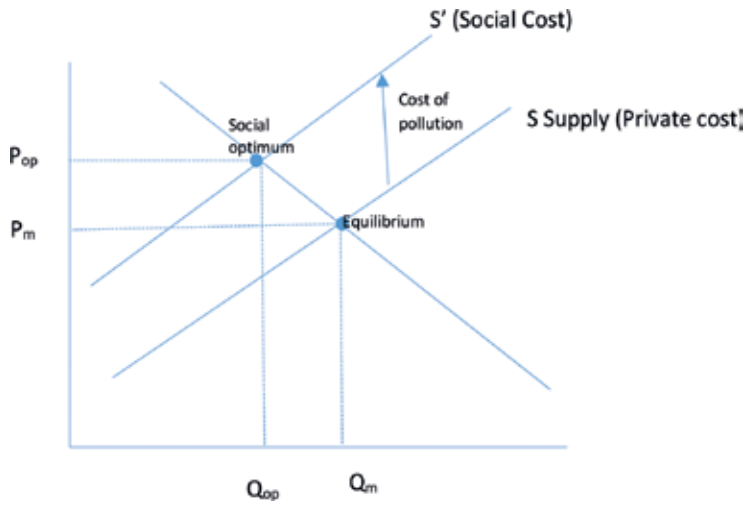

Figure 2. Air pollution externality.

If firms ignore the externality, they decide how much to pollute and will benefit, in the absence of any regulation, from the existence of pollution. With unregulated emissions, firms do not have the incentives to consider the pollution costs to bystanders affected by smokestacks' emissions. Polluters receive the total revenue of pollution; however, the victims of pollution assume all the costs produced by smoke because they cannot control or influence the production decisions of emitters. In this case, the market equilibrium is found where supply equals demand at the equilibrium quantity (Q_m) and the equilibrium price (P_m). In the presence of an externality, however, the cost to society of producing lead ingots is larger than the private costs to the lead producers. The social cost S' includes the private cost plus the cost to the victims of the externality who are affected by pollution. This social cost includes both the private costs of lead producers and the cost to individuals affected by the emissions. If all firms that pollute are forced to pay the full social costs (S') of production that include the external costs of emissions, the competitive supply S (private marginal cost) will shift upward to S' (social cost of pollution) and the externality is internalized. The market price for lead ingots will be higher (P_{op}), and the quantity sold will be lower (Q_{op}). The combination (Q_{op}, P_{op}) is a social optimum, because it denotes the desirable amount of lead ingots produced from the stand point of society as a whole. This graph reveals that, at the social optimum, reductions in pollution are accompanied by reductions in the supply of the product that contributes to emissions.[2]

In Section 4, we discuss some public policies proposed to achieve the optimal outcome where the externality is internalized as well as private solutions for the treatment of externalities. For this purpose, we focus on environmental quality as a problem of the commons.

[2] S, the supply curve, is the marginal cost for the firm that maximizes profits. At the optimum, price = marginal cost for a competitive firm. S' is the marginal social cost of production, which in turn equals the marginal cost of production plus the marginal external cost of emissions.

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4. Environmental policies to correct air pollution externalities

Environmental quality shaped by air pollution entails a critical commons problem. Economists have long discussed that externalities arise when resources are treated as commons that are shared by many users without payment. In the absence of rationing, the presence of free-access common property resources may lead to the overutilization of the resource thus creating market inefficiencies [13]. The atmosphere has the characteristics of a common property resource. Economic theory that focuses on market failures arising from incomplete systems of property rights defines clean air in the atmosphere as a particular form of commons problem: pure public goods that are nonexcludable and nonrival in consumption (see [14]). This type of public goods provides benefits to people or firms at zero marginal costs and does not exclude someone from enjoying them. The presence of such pure public goods has triggered governments to design environmental policies. Some of these policies follow a command and control approach that does not account for economic responses. Often, economists argue that these policies are not cost-effective, sometimes bring unintended consequences on the environment, and sometimes are ineffective. This approach has essentially used legislation to correct pollution externalities through uniform standards. Other environmental policies have emerged taking into account economic incentives, making pollution an expensive activity in the form of pollution taxes and marketable permit system. We will briefly discuss these approaches in the following session.

4.1. Standards

A standard is a legal form of regulation that limits how much pollutants a firm can emit. Under this approach, the government usually implements the so-called command and control policies to regulate polluting activities. In other words, the environmental authorities regulate behavior directly by dictating a maximum level of pollution that a factory may emit (safety standard), or by requiring firms to impose the adoption of abatement technologies for reducing emissions (technology standard).

4.1.1. Safety standard

Early regulation of air pollution was based mainly on the so-called safety or ambient standard. Safety standard can be simply defined as a maximum level for some pollutant in the ambient environment. Safety standards are usually expressed as average concentration levels over some period of time. In the US, the Clean Air Act, last amended in 1990, requires the US Environmental Protection Agency (EPA) to set National Air Quality Standards (NAQS) for the so-called criteria pollutants: particle pollution, photochemical oxidants and ground-level ozone, carbon monoxide, sulfur oxides, nitrogen oxides, and lead. For example, the US standard for particulate matter less than 10 $\mu g/m^3$ (PM10) requires that a 24 h average of 150 $\mu g/m^3$ not be exceeded more than once per year on average over a 3-year period.[3] Safety

[3] A table summarizing NAQS is reported by the US Environmental Protection Agency at https://www.epa.gov/criteria-air-pollutants/naaqs-table#4.

standards are essentially stock levels of pollution that are set to protect the health of people or their well-being. The safety standard approach was conceived in terms of rights and fairness rather than efficiency. The vision of the safety standard requires pollution reductions to minimum levels to eventually eliminate damage to the environment and risks to people's health. This approach is based on legislation that seldom mentions cost-benefit tests. Advocates of this approach usually argue that in the case of environmental protection, costs should not be involved in people's decision-making processes. Safety standards, however, usually are accompanied by high compliance costs and are subject to several criticisms: first, they are considered inefficient in the sense that regulators may have chosen pollution levels that are too high compared to standards based on costs and benefits. Second, safety standards are not necessarily cost-effective. In other words, not always the maximum amount of safety is achieved with the available resources. The final criticism to safety standards is that pollution control may have a regressive impact on income distribution. This means that higher prices of goods consumed triggered by environmental regulation affect a greater proportion of the incomes of poor households than those of richer households [15]. A comprehensive discussion of the potential regressive impacts of pollution control measures is reported in Ref. [16].

4.1.2. Emission standard

An emission standard is a flow variable and may be defined as the maximum level applied to the quantities of emissions coming from a pollution source. Sometimes this standard is known as a performance standard because it is an end result of compliance that the environmental authority requires from final pollution sources. In this case, the authority regulates quantities. In the case of air pollution, a standard may be expressed as a quantity of material per unit of time, per unit of output, or per unit of input. For example, an emission standard may be set as SO_2 emissions per kilowatt-hour of electricity produced, or sulfur content of coal used in power generation. A common emission standard is the mobile source air pollution program where EPA establishes emission standards for new cars by mandating ceilings on emissions per mile of operation.[4] It is worth noting that the compliance of emission standards does not mean that ambient standards are met. Complex chemical processes may change the physical properties of the pollutant, and the environmental authorities usually do not have control over the numbers of cars circulating, making it difficult to control the ambient air quality.

From a theoretical economic point of view, emission standards are set taking into account what economists define as efficient pollution levels. An efficient pollution level may be defined as the one that maximizes the net benefits of reduction. This level occurs theoretically where the marginal costs of pollution reduction equals its marginal benefits. Efficient levels are illustrated in **Figure 3a, b** where total costs and benefits of pollution reduction (**Figure 3a**) and marginal costs and benefits of reduction are shown. Starting from an initial and not efficient level of emissions e_1, the efficient level occurs at e^* where net benefits are maximum. At this point, the slope of the total benefits function (marginal benefit) equals the slope of the total cost function (marginal cost). The point e^* is illustrated in **Figure 3b**. The pollution level where total costs of

[4] Details found in the Clean Air Act Mobile Source Fuels Civil Penalty Policy Title II of the Clean Air Act 40 C.F.R. Part 80 Fuels Standards Requirements.

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reduction equal total benefits of reduction would lead to too much control of pollution from an efficiency point of view. In practice, the environmental authority sets a standard to the theoretical level e^* and enforces the standard by measuring pollution at the source and putting fines to firms in case detecting possible violations.

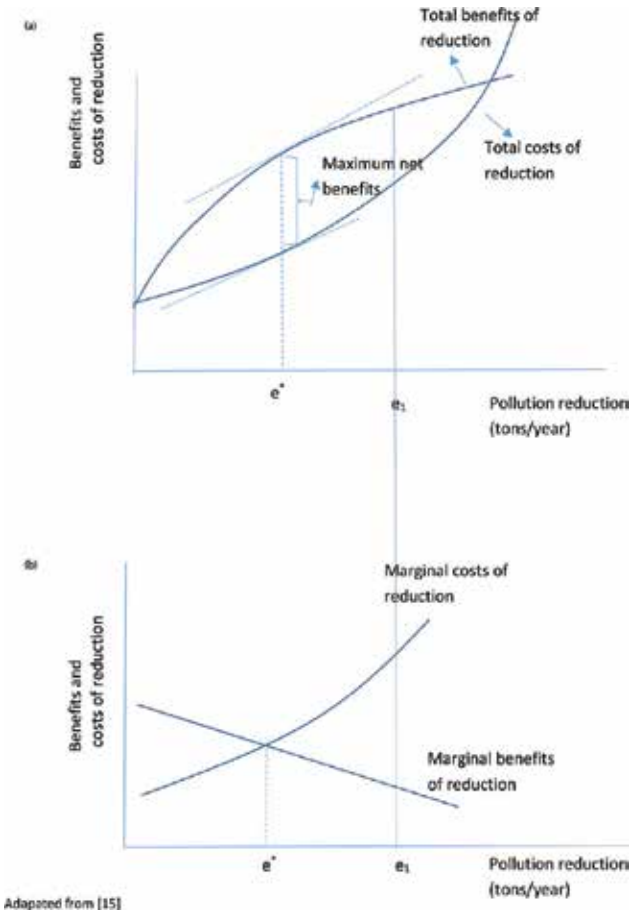

Figure 3. (a, b) Emission standard.

Under the scheme of emission standards, firms have the flexibility to choose the pollution abatement method to meet the standard. Other forms of regulation focus on the practices and technology adoption that emitters must adopt to meet the mandated goal of emission levels. We will briefly discuss technology standards below.

4.1.3. Technology standards

An environmental agency may specify a particular type of technology that a polluter must adopt in the production process. Also it may specify the characteristics of the inputs that must

be used in production or the pollution abatement technologies such as stack gas scrubbers to directly reduce emissions. Overall, this regulatory approach receives the name of *best available technology*. The technology prescribed usually meets two conditions: it must achieve significant reductions in pollution, but at a reasonable cost (see [12]).

The standards described above in this section are labeled as quantity or direct regulation. Under this regulatory view, policy makers control pollution through legal regulations by specifying emissions ceilings or the technology to be used in production or abatement. Under the regime of standards, the regulator (government) is responsible for how much pollution is allowed to each firm, mandating a specific technology, and, most importantly, for the monitoring of emissions to verify compliance and the implementation of monetary penalties to noncomplying sources [17]. Economists have claimed that the standards regime is not cost-effective (it does not achieve the emission target at a minimum cost), and have proposed a regulatory approach based on economic incentives. The idea behind incentive-based regulation was to make polluting an expensive activity and lower the costs of pollution control by letting the pollution abatement decisions to firms and not to the regulator [15]. The following section briefly discusses the major regulatory schemes based on incentives, namely the pollution tax and the marketable permit system.

4.2. Incentive-based regulation for air pollution control

Two historical economic views of environmental problems have shaped the incentive-based regulation for pollution control. The first view follows the tradition of Pigou [18] who thought on environmental pollution as a negative externality problem. In the presence of an externality such as air pollution, Pigou argued that imposing a per unit tax on the emissions firms would internalize the externality. The tax rate would be equal to the marginal social damage caused by the last unit of pollution at the efficient allocation [17]. The second economic view to tackle environmental problems follows the tradition of Ronald Coase [19] who thought of environmental pollution as a problem of public good. Coase demonstrated that in the absence of transaction costs and free riding, it is possible to achieve an efficient pollution level through negotiation regardless of who has the legal right to pollute or prevent pollution [19]. This theorem is usually evoked when analyzing the initial distribution of permits in a marketable permit system, a trading market in pollution rights.

4.2.1. Emission taxes

One of the main criticisms of command and control as a regulatory approach is that standards hinder the cost-effectiveness of pollution control since they allow little flexibility in the means of achieving lower pollution levels. The cost-effectiveness rule says that cost-effectiveness is achieved if all sources that control pollution experience the same marginal cost of reduction [10]. In the standards approaches to regulate the environment, firms are forced to take similar shares of the pollution control burden by imposing uniform standards in both performance and technology. In practice, abatement costs are very heterogeneous among firms placing command and control methods as not cost-effective. The regulator could employ nonuniform standards to each source in order to achieve cost-effectiveness; however, the information

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burden that the regulator would have to carry, in order to know each source's abatement costs, is in practice unfeasible or very expensive [14]. Theoretically, a Pigouvian tax is a way by which the government can achieve the pollution reduction goal by giving factory owners an economic incentive to reduce pollution. Since a Pigouvian tax places a price to the right to pollute, the higher the tax, the larger the reduction. The environmental authority can achieve the desired level of pollution by setting an emission tax at the appropriate level. Theoretically, this tax is fixed at the efficient pollution level where the marginal damage of emissions equals the marginal abatement costs [20]. This tax is shown in **Figure 4**.[5]

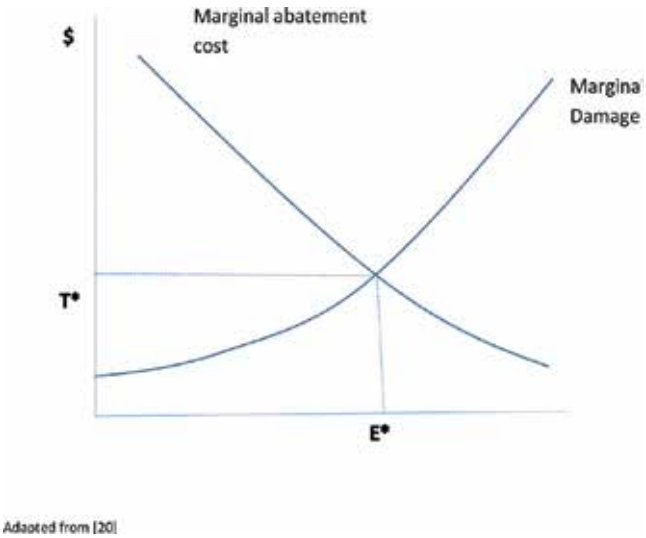

Figure 4. Emission tax.

One of the main advantages of emission taxes is their economic efficiency. When the marginal cost of pollution reduction at one factory is greater than that at another factory, the overall costs can be reduced without changing the ambient pollution level by decreasing pollution at the low-cost site and increasing it at the highest cost site [15]. In other words, the emissions tax leads firms with lower marginal abatement cost to larger reductions compared to those from firms with larger marginal abatement costs. Each firm would reduce its emissions until its marginal reduction costs equal the tax; then, marginal abatement costs will be the same for all sources and the cost-effectiveness rule holds.

The previous point is illustrated in **Figure 5**. Firm A has lower marginal abatement costs than firm B. This difference may be attributed in practice to differences in production technologies. The production technology used by firm A makes reductions less costly than reductions at firm B.

[5] The main difference between graph 3 and 4 is that graph 3 is in terms of pollution reductions and graph 4 is in terms of emissions.

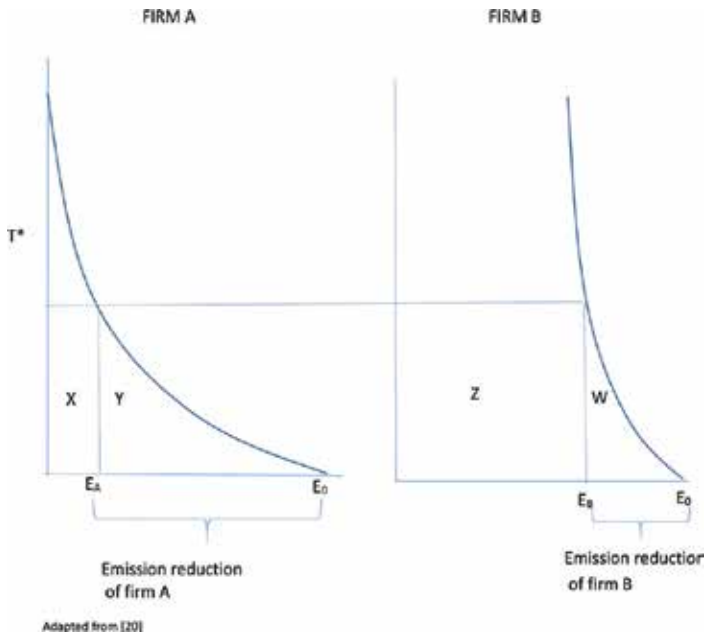

Figure 5. Emissions reductions and abatement costs.

The emission tax leads firm A to reduce from E_0 (initial level of emissions) to E_A, while firm B would reduce a lower quantity, from E_0 to E_B. In other words, because of the tax, the firm with lower marginal abatement costs would have a larger proportion of the emissions reductions. Note that total abatement cost for firm A equals area Y and for firm B it equals area W. The tax bill that would be sent to firm A equals area X, whereas for firm B the tax bill is much larger and equal to area Z. One of the main advantages of emission taxes is that the regulator does not need to know information on abatement costs to reach efficient levels of emissions. Economists, in general, prefer the tax to standards because the tax reduces pollution more efficiently. In our example, firm A, which has lower marginal costs of abatement, responds to the tax by reducing emissions substantially to avoid the tax. Firm B, for which it is more expensive to reduce emissions, responds to the tax just by reducing less and paying the tax. The emission tax approach differs from standards where, notwithstanding the marginal reduction costs for each firm, all firms have to reduce the same amount, thus making uniform standards non-cost-effective.

4.2.2. Tradable emissions rights

Following the tradition of Ronal Coase [19], who proposed to view harmful effects such as smoke or noise as rights, economists used his key insight to propose a price-based remedy for pollution control, based on a system of transferable or tradable emission permits. A detailed review of this regulatory approach is found in Refs. [20–22]. The basic idea of the system of marketable permits for emissions is that each firm must have permits to generate emissions. Each permit specifies the quantity of emissions allowed to the firms. For example, each permit

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represents one unit of emissions. If firms do not comply with the specified emissions, they would be subject to strong sanctions. Emission permits are allocated among firms. The initial allocation is a centralized decision, the total number of permits is less than the current amount of emissions, and some or all sources would receive fewer permits than the current amount of emissions. The total number of permits is chosen to achieve a level of emissions to guarantee the desired ambient quality. The permits are marketable; that is, they can be bought and sold by any firm participating in the market. The idea behind this approach is that firms that can reduce pollution more easily, because of facing lower marginal abatement costs, would be willing to sell permits, and firms that face high costs of reduction would be willing to buy whatever amount of permits they need. The free market for pollution rights is also seen by economists as an efficient way to achieve optimal pollution levels for society. In the design of marketable emissions programs, economists create a market for pollution externalities. This approach combines some of the advantages of standards with the cost-effectiveness of the Pigouvian tax: the agency administering the program determines the total number of permits (like in a standard approach) and the transferability of permits allows cost-effective pollution abatement like in the tax system.

The US has a long tradition of tradable emission programs. These programs have also become more popular around the world. Early attempts by the US EPA to implement tradable emissions rights include the offset policy. Under this program, new sources of emissions may be located in regions where pollution standards were not met ("nonattainment" regions) as long as they offset their new emissions from existing sources by reducing their emissions below their current legal requirements. EPA would certify these reductions as emission reduction credits. These credits created a supply that became transferable to new sources interested in entering the area (demand). One of the advantages of the offset policy is that it helped improve air quality without impeding economic growth. The early offset program expanded to the US Emissions Trading Program which established three trading programs: offset, bubble, and netting policies. The offset policy worked as explained before. The netting policy allowed existing sources to expand, avoiding technological requirements for pollution control as long as any net increase in emissions fell below an established limit. The bubble policy diverges from the offset policy because emission reduction credits could be traded among existing sources in localized air quality regions ("bubbles"), whereas in the offset policy *new* sources are allowed to acquire credits from existing sources.

Tradable emission rights have been applied in the US to control inputs such as lead in gasoline [23], to reduce ozone-depleting chemicals mandated by the Montreal Protocol, to tackle acid rain through the Sulfur Allowance Program, to reduce NO_x with the NO_x Budget Program, and to reduce volatile organic matter (VOM) emissions with the Emissions Reduction Market System (ERMS) in the Chicago area [17]. One of the most important emission trading programs is California's Regional Clean Air Incentives Market (RECLAIM) to control nitrogen oxides and sulfur in a market with more than 400 industrial polluters participating [21]. Non-US experience includes the European Union Emissions Trading Scheme to reduce industrial greenhouse emissions. Emissions trading programs have also been implemented to control particulate matter in Santiago de Chile [24].

5. Cost and benefits of air pollution control

The economic analysis of cost and benefits of air pollution control entails two main complexities: first, the taxonomy of the costs of environmental regulation is extensive and evaluated from different points of view, And second, many of the benefits of air pollution control and better ambient quality are nonmarketable, and, therefore, they cannot be easily monetized. For many policy makers, the costs of environmental regulations are mainly defined by the cost to government of administering environmental laws and regulations. These administrative costs are mainly from monitoring and enforcement. On the other hand, from the point of view of private firms there are costs of compliance given by the capital and operating expenditures to meet emission standards. Part of the compliance costs may be shared also by the regulator, for example, when clean technologies are subsidized. Other "negative costs" or nonenvironmental benefits linked to environmental regulation include productivity impacts of a cleaner environment or innovation stimulation. Other costs encompass the discouragement of investment, for example, when a firm closes because it cannot afford expensive pollution abatement technologies, retarded innovation, and other economic impacts including the loss of jobs. Jaffe et al. report in Ref. [25] that the direct compliance costs for the private sector in the US represent nearly 2.6 of gross national product (close to US$125 billion). The biggest share of these compliance costs is business pollution abatement expenditures (61%). Other components of these costs included personal consumption abatement (11%), government abatement (23%), government regulation and monitoring (2%), and research and development (3%).

Indirect benefits from air pollution control also include those that are nonmarketable. Acid rain and exposure to particulate matter can affect human health and are usually linked to increased risks of lung disorders such as asthma and bronquitis. They also have been linked to premature mortality in adults and children. Acid rain or particulate matter may cause damages in ecosystems, including water bodies and forests, and may also damage human-made infrastructure [26]. Bazhaf et al. estimated 1980s benefits from reducing sulfur dioxide in a 50% in the Adirondacks region in the US. The authors found annual benefits of US$24 million from improvements in recreational fishing, US$700 million to grain crop producers, and US$800 annual value for the commercial timber sector [27]. Other efforts to measure benefits of air quality improvements include the monetary estimation of the impacts of air pollution in human health. The World Bank estimated that mortality and morbidity from urban air pollution in India and China meant annual losses that varied between 2 and 3% of GDP [28]. More recently, Cropper and Khanna evaluate the methods used by the World Bank to calculate economic costs related to the exposure of outdoor air pollution [29]. They recommend the usage of estimates of disability-adjusted life years (DALYs) lost due to outdoor air pollution produced by the Global Burden of Disease model (GBD) to calculate the monetary value of a statistical life. Other economic literature on nonmarket valuation has estimated the impacts of air pollution on property values. These studies use direct methods of environmental valuation, using as a framework Rosen's hedonic pricing model [30]. The basic idea of this analytical framework is to learn about the price of air quality (a nonmarketable good) by observing the housing market. The main assumption is that housing is a marketable good composed by a bundle of heterogeneous characteristics that do not have a market, including air quality of the

neighborhood where the house is located. By observing the housing market, we can indirectly observe trade-offs that individuals are willing to make with respect to a characteristic. A consumer of housing will be willing to pay more for a house located in a clean air environment than for an identical house in a polluted area, *ceteris paribus*. This differential in price is an approximation of the willingness to pay or implicit price of air quality. A long tradition of hedonic models examining the impact of air pollution on property values has been in the nonmarket valuation literature. These estimations are important efforts that economists have done to find a monetary value for air quality. Applications of the hedonic model relating property values with air pollution include [31–36]. More recent applications include but are not limited to Refs. [37–42]. All of these studies estimated the so-called hedonic price function that relates property values to air pollution levels and other characteristics of the structure of dwellings (e.g., area, bathrooms, number of rooms) and other local environmental amenities such as proximity to parks, proximity to main roads, proximity to noxious facilities, among others. Results from the valuation exercises are very contextual and differ from one city/country to another. For example, in Ref. [36] researchers found that in the US a unit ($\mu g/m^3$) reduction in total suspended particles increases mean housing values between 0.2 and 0.4%. In other application of the hedonic model, the authors found that marginal implicit prices for PM10 air pollution in Santiago de Chile ranged from US$3 to US$6 depending on the estimation method [40]. In Bogotá, Colombia, the marginal implicit price for pollution measured as the annual average PM10 levels varied from US$ 0.31 in lower income neighborhoods to US$4.58 in higher income neighborhoods, also depending on the estimation method [42]. All hedonic valuations estimating the impact of pollution on property values suggest that air pollution improvements capitalize into housing values. These results from hedonic model estimations are important for policy makers related to air pollution control because benefits from housing capitalizations could be considered in cost-benefit analysis of pollution abatement.

6. Discussion

The debate on economic growth and the environment is not a closed one among economists [43]. The relationship between economic growth and pollution is suggestive if we accept the environmental Kuznets curve. In this chapter, we illustrated the nature of the EKC and the possible explanations for the inverse U relationship between economic growth and pollution. One of the most echoed explanations for the inverse U relationship is the impact of regulation on pollution abatement [5].

Usually, pollution is perceived as an economic problem associated with production and consumption; however, it is important to recognize the public good properties of the environment that may cause pollution. Pollution could be explained, from an economic point of view, as a by-product of a market failure. The environment is perceived by polluters as a public good that is a nonrival and a nonexclusive good for which property rights are not well defined. This kind of market failure makes firms perceive polluting activities as costless; however, this chapter has shown that there are external costs of air pollution. If the objective of an environmental authority is to reduce pollution, we have discussed the basic nature of instruments that

the regulator may use for pollution control, emphasizing on standards and economic-incentive-based regulation: namely the Pigouvian tax and Emissions Trading. This chapter highlighted the advantages of incentive-based regulation compared to the command and control approach for pollution abatement, as it is frequently discussed within the economics discipline.

However, environmental regulation is not limited to standards or to the incentive-based regulation pointed out in this chapter. Other approaches to environmental regulation include decentralized policies where coordination among polluters is facilitated because of small numbers involved. Within these approaches, there are liability laws, well-defined property rights, and moral suasion (Field, 1994). Other environmental policies initiatives that have been used recently in developing countries rely on information and public disclosure as an initiative for industrial pollution abatement. An emblematic program using this approach is the Program for Pollution Control, Evaluation, and Rating (PROPER), launched in Indonesia [44]. As economists recognize the potential trade-offs between clean air and economic growth, and the impossibility of a zero pollution environment, economics as a discipline has proposed incentive-based regulation as a cost-effective way to control air pollution. The application of incentive-based regulation is compatible with the Intended National Determined Contributions (INDCs) agreed upon in the United Nations' Framework Convention on Climate Change (UNFCCC) Conference of the Parties (COP21) in Paris in December 2015, to tackle global climate change. As this chapter suggests, economics as a discipline complements the efforts of governments and engineers to reduce industrial pollution.

Appendix

Group of countries	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
High income	17.9	17.0	16.6	16.4	16.3	16.1	16.1	15.9	15.4	14.4	15.0	14.9	14.8	14.9
Upper middle income	24.7	24.5	23.3	24.1	24.1	23.9	23.9	23.9	23.7	23.0	23.2	21.6	21.2	20.8
Middle income	22.8	22.6	21.9	22.6	22.6	22.5	22.5	22.4	22.2	21.5	21.6	20.5	20.2	19.8
Lower middle income	17.1	17.2	17.3	17.3	17.5	17.4	17.5	17.4	17.2	16.6	15.9	16.8	16.6	16.3
Low income	10.2	11.2	11.5	11.3	11.0	10.5	10.4	10.4	10.2	9.9	9.0	9.1	8.8	8.6
World	19.2	18.5	18.1	18.2	18.2	18.0	18.0	17.8	17.3	16.4	16.8	16.5	16.3	16.3

Source: World Development Indicators.

Table A1. Manufacturing, value added (% of GDP).

Group of countries	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
High income	14	14	13	13	13	12	13	12	12	12	12	13	12	12
Upper middle income	23	22	22	22	24	25	24	24	25	25	25	26	25	25
Middle income	23	22	22	22	23	24	24	24	24	24	24	26	25	25
Lower middle income	21	21	21	22	21	22	23	23	23	22	23	23	22	22
Low income	48	48	47	48	47	47	48	45	46	45	44	39	38	37
World	18	18	17	17	18	18	18	18	19	19	19	20	20	20

Source: World Development Indicators.

Table A2. CO_2 emissions from manufacturing industries and construction (% of total fuel consumption).

Variable name	Definition	Mean	SD	Min	Max
Y	2000 year GDP at market prices, constant 1995 US$ (WB estimates)	6468.67	10518.07	45.605	46485.31
CO_2	Carbon dioxide emissions (CO_2), metric tons of CO_2 per capita (Year 2000)	4.303521	5.33227	0	21.3

CO_2 per capita emissions used for Eq. (2) comes from United Nations data base. GDP from 2003 World Development Indicators by the World Bank (WB).
One hundred and fifty-two countries are included in the sample. The 2000 per capita GDP variable is measured at market prices in constant 1995 US$ (WB estimates). According to the United Nations Definition, the variable CO_2 measures emissions in metric tons of CO_2 per capita. Emissions of CO_2 refer to the release of greenhouse gases and/or precursors into the atmosphere over a specified area and a period of time. In our case, the time period for emissions is year 2000 and each area corresponds to each country in the 152 observations sample.

Table A3. Descriptive statistics.

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References

[1] NASA. Global Climate Change [Internet]. 2011. Available from: http://climate.nasa.gov/vital-signs/carbon-dioxide/ [Accessed: 2016-07-05]

[2] Stern, N. The economics of climate change. American Economic Review. 2008; 98(2): 1–37. doi:10.1257/aer.98.2.1

[3] WHO. Fact sheet N°313 [Internet]. 2014. Available from: http://www.who.int/media-centre/factsheets/fs313/en/ [Accessed: 2016-07-07]

[4] De Bruyn, S.M., Heintz R.J. The environmental Kuznets curve hypothesis. In: Jeroen, C.J., van den Bergh, M. editors. Handbook of Environmental and Resource Economics. Cheltenham, UK: Edward Elgar Publishing; 1999. 1328 p.

[5] Grossman, G.M., Krueguer A.B. Economic growth and the environment. Quarterly Journal of Economics. 1995; 110, 353–77. doi:10.2307/2118443

[6] Stern, D.I., Common M.S., Barbier E.B. Economic growth and environmental degradation: a critique of the environmental Kuznets curve. World Development. 1996; 24, 1151–60. doi:10.1016/0305-750X(96)00032-0

[7] Shafik, N., Bandyopadhyay, S. Economic growth and environmental quality: time series and cross-country evidence. Policy Research Working Papers. WPS 904.Washington, D.C:1992. Available from: http://documents.worldbank.org/curated/en/833431468739515725/pdf/multi-page.pdf [Accessed:2016-07-01].

[8] Panayotou, T. Empirical tests and policy analysis of environmental degradation at different stages of economic development. Discussion Paper 1, Geneva: International Labor Office; 1993. Cited in De Bruyn, S.M., Heintz, R.J. (1999).

[9] Selden, T.M., Song, D. Environmental quality and development: in there a Kuznets curve for air pollution emissions? Journal of Environmental Economics and Management. 1994; 27, 147–62. doi:10.1006/jeem.1994.1031

[10] Baumol, W.J., Oates, W.E. The Theory of Environmental Policy, 2nd edn. Cambridge, UK: Cambridge University Press; 1988. 299 p.

[11] Verhoef, E.T. Externalities. In: Jeroen C.J., van den Bergh, M., editors. Handbook of Environmental and Resource Economics. Cheltenham, UK: Edward Elgar Publishing; 1999. 1328 p.

[12] Sterner, T. Policy instruments for environmental and natural resource management. Resources for the Future, The World Bank, Swedish International Development Cooperation Agency; RFF Press: Washington, D.C; 2003. 504 p.

[13] Ostrom, E., Dietz, T., Dolšak, N., Stern, P.C., Stovich, S., Weber, E.U., editors. Committee on the Human Dimensions of Global Change. Division of Behavioral and Social Sciences and Education. Washington, DC: National Academy Press; 2002. doi: 10.17226/10287

[14] Stavins, R. The problem of the commons: still unsettled after 100 years. American Economic Review. 2011; 101, 81–108. doi:10.1257/aer.101.1.81

[15] Goodstein, E.S. Economics and the Environment, 2nd edn. New York, NY: John Wiley & Sons, Inc.; 1999. 558 p.

[16] Kosonen, K. Regressivity of environmental taxation: myth or reality? European Commission, Taxation Papers; 2012. Working Paper N. 32. Available from: https://ec.europa.eu/

taxation_customs/sites/taxation/files/docs/body/taxation_paper_32_en.pdf. [Accessed: 2016-07-01]. DOI 10.2778/30289.

[17] Tietemberg, T.H. Emissions Trading, Principles and Practice, 2 nd edn. Washington, DC: Resources for the Future; 2006. 233 p.

[18] Pigou, A.C. The Economics of Welfare. London: Macmillan; 1920. 875 p.

[19] Coase, R. The problem of social cost. Journal of Law and Economics. 1960; 3, 1–44. doi: 10.1086/466560

[20] Field, B. Environmental Economics: An Introduction. McGraw-Hill: New York; 1999. 482 p.

[21] Tietemberg, T.H. Lessons from using transferable permits to control air pollution in the United States. In: Jeroen C.J., van den Bergh, M., editors. Handbook of Environmental and Resource Economics. Cheltenham, UK: Edward Elgar Publishing1999. 275–292 pp.

[22] Koustal, P. Tradable permits in economic theory. In: Jeroen C.J., van den Bergh, M., editors. Handbook of Environmental and Resource Economics. Cheltenham, UK: Edward Elgar Publishing; 1999. 1328 p.

[23] Kerr, S., Maré, D. Transaction Costs and Tradable Permit Markets: The United States Lead Phasedown.[internet]. 1998. Available from: http://papers.ssrn.com/sol3/papers.cfm?abstract_id=1082596 [Accessed: 2016-07-8]

[24] Montero, J.P. et al. A market-based environmental policy experiment in Chile. Journal of Law and Economics. 2002; 45(1): 267–287. doi:10.1086/324657

[25] Jaffe, A.B., Peterson, S.R., Portney, P.R., Stavins, R.N.. Environmental regulation and competitiveness of U.S. manufacturing: what does the evidence tell US?. In: Stavins, R.N., editor. Economics of the Environment: Selected Readings, 5th edn. New York, NY: W.W. Norton & Company, Inc.; 2005. 676 p.

[26] Vallero, D. Fundamentals of Air Pollution, 5th edn. Elsevier, Inc-Academic Press: Cambridge, MA; 2014. 996 p. doi:10.1016/B978-0-12-401733-7.01001-X

[27] Banzhaf, S., Burtraw, S.D., Evans, D., Krupnick, A. Valuation of natural resourceimprovements in the Adirondacks. Washington-RFF, Report. Sept 2004. Available from: http://www.rff.org/blog/2005/valuation-natural-resource-improvements-adirondacks [Accessed: 2016-06-02].

[28] Bolt, K., Hamilton, K., Pandey, K., Wheeler, D. The cost of air pollution in developing countries: new estimates for urban areas. World Bank Development Research Group Working Paper. 2001. Cited in Dasgupta et al. (2002) (Reference [43]).

[29] Cropper, M., Khanna, S. How should the World Bank estimate air pollution damages? Discussion Paper. Sept 2014. RFF DP 14–30. Available from:http://www.rff.org/files/sharepoint/WorkImages/Download/RFF-DP-14-30.pdf. [Accessed: 2016-07-02].

[30] Rosen, S. Hedonic prices and implicit markets: product differentiation in pure competition. Journal of Political Economy. 1974; 82, 34–55. doi:10.1086/260169

[31] Ridker, R.G., Henning, J.A. The determinants of residential property values with special reference to air pollution. Review of Economics and Statistics. 1967; 49, 246–25. doi: 10.2307/1928231

[32] Smith, V.K., Huang, J.C. Hedonic models and air pollution: twenty-five years and counting. Environmental and Resource Economics. 1993;3, 381-394. doi:10.1007/BF00418818

[33] Chattopadhyay, S. Estimating the demand for air quality: new evidence based on the Chicago housing market. Land Economics. 1999; 75, 22–38. doi:10.2307/3146991

[34] Zabel, J.E., Kiel, K.E. Estimating the demand for air quality in four U.S. cities. Land Economics. 2000; 76(2),174–194. doi:10.2307/3147223.

[35] Kim, C.W., Phipps, T., Anselin, L. Measuring the benefits of air quality improvement: a spatial hedonic approach. Journal of Environmental Economics and Management. 2003;45, 24–39. doi: 10.1016/S0095-0696(02)00013-X

[36] Chay, K.Y., Greenstone, M. Does air quality matter? Evidence from the housing market. Journal of Political Economy. 2005; 113(2), 376–424. doi:10.1086/427462

[37] Anselin, L., Le Gallo, J. Interpolation of air quality measures in hedonic house price models: spatial aspects. Spatial Economic Analysis. 2006; 1, 31–52. doi: 10.1080/17421770600661337

[38] Neill, H.R., Hassenzahl, D.M., Assane, D.D. Estimating the effect of air quality: spatial versus traditional hedonic price models. Southern Economic Journal. 2007; 73(4), 1088–1111.

[39] Anselin, L., Lozano-Gracia, N. Errors in variables and spatial effects in hedonic house price models of ambient air quality. Empirical Economics. 2008; 34, 5–34. doi:10.1007/s00181-007-0152-3

[40] Vásquez, L.F., Dresdner, J., Aguilar, R. The value of air quality and crime in Chile: a hedonic wage approach. Environment and Development Economics. 2011; 16, 329–355. doi:10.1017/S1355770X10000483

[41] Mínguez, R., Montero, J.M., Fernández-Avilés, G. Measuring the impact of pollution on property prices in Madrid: objective versus subjective pollution indicators in spatial models. Journal of Geographic Systems.2012; 15, 169–191. doi:10.1007/s10109-012-0168-x

[42] Carriazo, F., Ready, R., Shortle, J. Using stochastic frontier models to mitigate omitted variable bias in hedonic pricing models: a case study for air quality in Bogotá, Colombia. Ecological Economics. 2013; 91, 80–88. doi:10.1016/j.ecolecon.2013.04.005

http://dx.doi.org/10.5772/65256

[43] Dasgupta, S., Laplante, B., Wang, H., Wheeler, D. Confronting the environmental Kuznets Curve. Journal of Economic Perspectives. 2002; 16(1), 147–168. doi: 10.1257/0895330027157

[44] Afsah, S., Blackman, A., García, J.H., Sterner, T. Environmental Regulation and Public Disclosure: The case of PROPER in Indonesia. Washington, DC: RFF, Press. 2013; 152 p.

Case studies

Chapter 4

Atmospheric Pollution and Microecology of Respiratory Tract

Chunling Xiao, Xinming Li, Jia Xu and Mingyue Ma

Additional information is available at the end of the chapter

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Abstract

This chapter elaborates the source and ingredients of atmospheric pollutants, the microecology of respiratory tract in animals and humans and the effect of atmospheric pollution on it and thus clarifies the relationship between air pollution and microecology of the respiratory tract based on the experiments.

Keywords: atmospheric pollution, microecology, respiratory tract, health

1. Introduction

This chapter includes five aspects such as atmospheric pollution, the hazards of atmospheric pollution to the body, microecology of respiratory tract, atmospheric pollution and microecology of respiratory tract in humans and atmospheric pollution and microecology of respiratory tract in rats; this chapter describes the source of atmospheric pollution, the classification of atmospheric pollutants (particulates, sulfur oxides, nitrogen oxides, pollution by polycyclic aromatic hydrocarbons (PAHs)), acute and chronic influence of atmospheric pollution to the body, the composition and physiological function of microecology of respiratory tract and focuses on the effect of air pollution on the microecology of respiratory tract. The results of oropharyngeal flora detection in children and rats have shown that atmospheric pollution can cause increase in the detection rates and flora density of both normal flora and pathogenic bacteriatherefore resulting in microecology imbalance. Exposure to air pollutants for different length of time can lead to corresponding acute or chronic injury of the respiratory system. Compared with total suspended particles (TSP) and PM_{10}, $PM_{2.5}$ can stay for long time in atmosphere and has long conveying distance, so that it has more influence on the quality of atmospheric environment and human health.

2. Atmospheric pollution

2.1. Source of atmospheric pollution

The most principal hazardous substances in atmospheric pollution include sulfur oxides, nitrogen oxides, dust, hydrocarbons and carbon oxides. Sulfur oxides mainly come from the combustion of fossil fuels, such as petroleum, coal and natural gas. Nitrogen oxides mainly come from vehicle exhaust, including multiple substances, such as NO and NO_2. Dust is the particulate whose diameter is generally 0.002–500 μm and mainly comes from the fuel combustion, solid crushing, sand storm and secondary hazardous substances, such as ozone and sulfuric acid mist, belonging to photochemical smog. The poisonous gases in industry, such as toluene and gasoline as well as heavily metals entering atmosphere such as mercury, chromium and zinc, would cause atmospheric pollution [1, 2].

Atmospheric pollution is caused by one or multiple kinds of pollutants in gaseousness, gas dispersoid, or particulate that exist in the atmosphere and it can cause harmful or abnormal effects to mankind and other organisms. The principal hazardous substances in atmospheric pollution include sulfur oxides, nitrogen oxides, dust, hydrocarbons and carbon oxides, etc. and atmospheric pollution has become a worldwide problem because of its wide source and large scope of influence.

The sources of atmospheric pollution mainly include fuel combustion, industrial production and transportation. The pollutants discharged by fuel combustion approximately account for 70% of total pollutants in atmosphere of China, 20% by industrial production and 10% by transportation. Among them, over 95% of pollutants discharged by fuel combustion come from coal burning; therefore, the main source of atmospheric pollution in China is the flue dust discharged from direct coal burning.

2.2. Classification of atmospheric pollution

According to their attributes, atmospheric pollution can be divided into three categories, i.e., chemical pollution, biological pollution and physical pollution. Among them, chemical pollution is the most diversified and it is the key point of atmospheric pollutants because of its wide scope of pollution. Chemical pollutants are discharged into atmosphere mainly in the form of waste gas. According to their physical states in atmosphere, they can be divided into gaseous and particulate states. Common gaseous atmospheric pollutants include SO_2, O_3, CO and NO_2. The particulates closely related to hygiene include total suspended particles (TSP) and inhalable particulate (IP). TSP is the generic term of various particulates dispersed in the atmosphere in the state of gas dispersoid. The particle size of TSP is 0.1–100 μm and the toxic substances contained in them, such as As, Cd, Pb, Ni and polycyclic aromatic hydrocarbon, can change the activity and metabolic status of cell enzyme. IP or PM (10) means the particulates whose aerodynamic equivalent diameter (AD) are less than 10 μm and it is then generally divided into coarse particulates (2.5 μm < AD < 10 μm), fine particulates or PM2.5(AD ≤ 2.5 μm) and ultra-fine particulates (UFPs) (AD ≤ 0.1 μm) by the particle size (**Figure 1**). Coarse particulate is mainly produced by mechanical actions, but the latter two kinds of particulates mainly come from fuel combustion, including direct discharge after combustion and generation from gaseous pollutants through chemical transformation [1–4].

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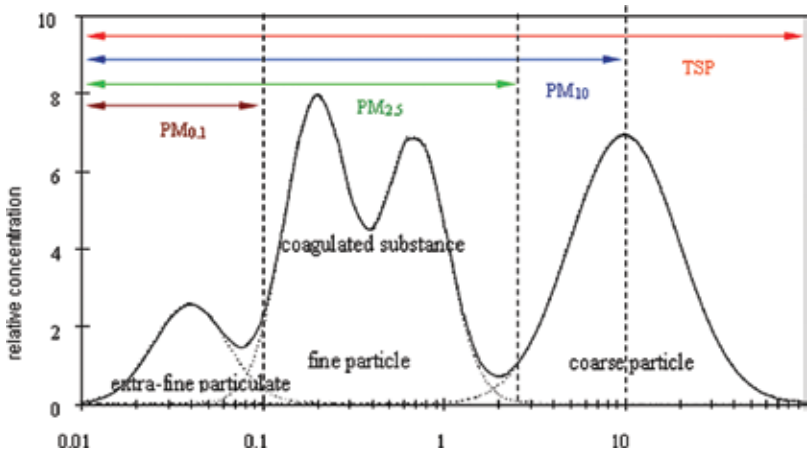

Figure 1. Particle aerodynamic diameter.

Atmospheric pollutants can be classified by different division methods, such as source, chemical component, particle size and indoor or outdoor environment. The pollutants directly discharged into the atmosphere are called primary pollutants and the pollutants produced after chemical reactions with other environmental substances are secondary pollutants. There are obvious correlations between them, but that is not always unchanged, since the lowered level of primary pollutants as precursor would not always automatically make the level of secondary pollutant correspondingly lower. For example, ozone (O_3) is mainly generated by nitrogen oxides (NO_X). When the discharge of NO_X is decreased, the level of O_3 in the atmosphere sometimes conversely rises. The particulates in atmospheric pollutants (PM) are commonly divided into three categories according to the aerodynamic diameter. Coarse particles (2.5–10 μm, PM_{10}) come from the aggregation of road dust, building scrap, or tiny combustion particles; fine particulates (<2.5 μm, $PM_{2.5}$) and extra-fine particulates (<0.1 μm, UFPs) are mainly formed in the combustion process of fossil fuel. US EPA has already limited the environmental control standard for $PM_{2.5}$ and PM_{10}. $PM_{2.5}$ is the subcomponent of PM_{10} and there is a standard specially stipulated for $PM_{2.5}$, so as to guarantee that $PM_{2.5}$ with smaller volume and greater permeability through respiratory tract can be effectively controlled because it is more influential. At present, little is known about the potential risks of UFPs. There is evidence to show its greater quantity, more contents of transition metals and oxidation-reduction chemical components, which can more easily penetrate blood circulation. Therefore, it has greater toxicity than $PM_{2.5}$ and PM_{10}. However, the corresponding control standard has not yet been promulgated [1–4].

2.2.1. Particulates

In the generally acknowledged atmospheric pollutants, the epidemiological links between particulates and population health effect endpoint is the closest. The quantitative hazard assessment of particulates on health has become one of the hot spots that received attention from international organizations in recent years, such as WHO and EU. It is stipulated by America that the mean daily value and mean annual value of IP (PM_{10}) shall

respectively be 0.15 and 0.05 mg/m^3. It is stipulated in GB3095–1996 promulgated by China in 1996 that the secondary standard for the mean daily value of PM_{10} shall be 0.15 mg/m^3 and the mean annual value shall be 0.10 mg/m^3. In 1997, Environmental Protection Agency of the United States (US EPA) was the first to promulgate the $PM_{2.5}$ standard and its mean daily value was strictly stipulated to be 0.065 mg/m^3 and its mean annual value to be 0.015 mg/m^3 [1–4].

The toxicity of particulates, both PM_{10} and $PM_{2.5}$ can increase the risk of lung cancer. It is shown by the American research that sulfate, nitrates, hydrogen ion, carbon element, secondary organic compounds and transition metals are all concentrated on the fine particulates, but Ca, Al, Mg and Fe elements are mainly concentrated on coarse particulates and they have different impacts on the human body. The hazard of $PM_{2.5}$ on the human body is greater than PM_{10}, which has become a new target of policy for environmental air control. With the development of transportation and the increase of motor vehicles as well as the increasing disruption of environment, the pollution of $PM_{2.5}$ gets more and more serious. It is found by researches that the ratio of atmospheric $PM_{2.5}$ in TSP is increased year by year and 96% of the particulates that deposit in the lower respiratory tract are $PM_{2.5}$. The $PM_{2.5}$ in the urban atmosphere mainly comes from the exhaust of traffic waste gas (18–54%) and the secondary pollution of gas dispersoid (30–41%) [1–4].

2.2.2. Sulfur oxides

The toxicity of SO_2: Sulfur dioxide is one of the main atmospheric pollutants. It is stipulated by the second class of national standard for atmosphere that the content of SO_2 in air shall be <0.06 mg/m^3. SO_2 can be adsorbed on the surface of $PM_{2.5}$ to enter deep into the respiratory tract, increasing its toxicity by three to four folds. SO_2 would be oxidized into SO_3 through metallic particle catalysis, whose risk is four times that of individual effects of SO_2. The carcinogenic action of benzo(a)pyrene, i.e., BaP can be increased under joint action of SO_2 and B[a]P.

Researches on the relations between SO_2 and the incidence risk of lung cancer. It is shown by epidemiological researches that SO_2 is closely related to the incidence of lung cancer. It is shown by the analysis of the mortality rate of lung cancer in Qingdao by Hu Yan [5]. that SO_2 in the atmosphere of downtown Qingdao has certain correlations with the increase in the mortality rate of lung cancer. Chen Shijie et al. [6] utilized grey relativity model to calculate the mortality rate data for lung cancer of mass and the mean annual concentration data of SO_2. It is shown by the results that the grey relativity between the mortality rate of lung cancer and the SO_2 8 years ago is the greatest, thus the latent period of SO_2 to cause lung cancer is 7 years. It is shown by the cohort research of 16,209 individuals in Norway that long-term exposure to SO_2 is related to the risk of lung cancer. Through the data collected by American Cancer Society for 16 years and the data of risk factor for the cause of death of 500, 000 Americans, Turner et al. [7]. found that SO_2 in air is related to the total morbidity rate and mortality rate of lung cancer. Through the cohort research of 57,613 workers exposed to SO_2 in pulp and papermaking industry in 12 countries, Moon et al. [8] found that the relative risk of lung cancer of the workers in the nonexposed group compared with those in the exposed group is 1.49 and its 95% *CI* is 1.14–1.96 [5, 6].

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2.2.3. Nitrogen oxides (NO_X)

The toxicity of nitrogen oxides: The nitrogen oxides in atmosphere mainly come from the waste gas exhausted from combustion of automobile, coal and petroleum. It is stipulated by the national atmosphere secondary standard that the content of NO_X in air shall be <0.10 mg/m^3. Photosynthesis shall occur at hydrocarbons under the effect of ultraviolet ray. Photochemical smog shall be produced and it might induce canceration.

Researches on the population epidemiology of nitrogen oxides: Through spatial geographical distribution map and Spearman rank correlation analysis to the death data of lung cancer in Jiangsu Province and to the air data of corresponding period, Lu et al. [9] found the positive correlation between the concentration of NO_X and the standardized mortality rate of lung cancer. It is shown by the cohort research of 16,209 individuals in Norway: long-term exposure to NO_X is related to the risk of lung cancer. When the concentration of NO_X is increased by 10 μg/m^3, the relative risk of lung cancer would be 1.11 and its 95% *CI* is 1.03–1.19. It is shown by the case-control study by Nyberg et al. [10]. in Sweden that with the NO_X pollution caused by exposure to traffic pollution for over 30 years, the risk of lung cancer is increased by one to two folds.

2.2.4. Pollution of polycyclic aromatic hydrocarbon

The toxicity of PAHs: Most PAHs in atmosphere are generated by the incomplete combustion of fuel in life and production activities of mankind. Multiple kinds of PAHs have been assessed to have carcinogenesis, especially among 16 kinds of PAHs under priority control publicized by US EPA, some of which are strongly carcinogenic compounds. PAHs themselves have no toxicity, but they present carcinogenic actions through metabolizing activation after entering the body. PAHs can also react with O_3, NO_X, and HNO_3, etc. and they might be converted into the compounds with stronger carcinogenic actions or mutagenic actions. At present, the PAHs proved to have relatively strong carcinogenesis by animal experiments which include benzo(a) pyrene, benz(a)anthracene, benzo(b)fluoranthene, dibenzo(a,h)pyrene and dibenz(a,h)anthracene, etc., among which the carcinogenic action is the strongest.

3. The hazard of atmospheric pollution to the body

With the increase of population density and the development of modern industry in the city, the influence of atmospheric pollution on mankind gets more and more serious. It is shown by many epidemiological evidence that the diseases of respiratory system are related to the low quality of air. At present, about 600 million tons of atmospheric pollutants are discharged annually into air worldwide, which is the direct cause for the increased number of hospitalized patients and outpatient patients, the decreased attendance rate of students and the increased morbidity rate of allergy. Life and environment shall be a unity and the life existence of living bodies that must adapt to the internal and external environment, since the maladjustment between them would merely cause diseases or death. Life and environment must be a unity of opposites and they are inseparable. All organisms have the limit of adapting to the

environment and anyone beyond the limit would lose life. Mankind is incessantly adapted to the environment and ceaselessly transforms the environment. Through the long-term repeated actions of low concentration of toxic and harmful pollutants in the environment on the body, chronic hazards would be incurred, which is caused by the sedimentation of poison itself in the human body or gradual accumulation of the tiny injuries to the body. At present, the world is confronted with three major environmental problems, i.e., destruction of ozone layer, acid precipitation and greenhouse effects, which are all related to the atmospheric pollution. The reason for the destruction of ozone layer is the impact of certain compounds, mainly including Freon. Under the photochemical action of ultraviolet ray, the harmful compounds in atmosphere might produce the active compounds to destruct the ozone layer, such as NO, NO_2, HO_2, Cl^- and ClO^-. The hole would be formed at the ozone layer after destruction and the function of ozone layer to block and absorb ultraviolet ray would be weakened. Excessive contacts of population with ultraviolet ray might cause corresponding diseases [11–14].

The combustion of large quantities of fuels would produce large quantities of CO, CO_2, SO_2, and NO_X, etc., which are discharged into the atmosphere and shroud in the air above the ground surface. As a result, the pH value of precipitation is lower than 5.6 and acid precipitation is thus formed. CO_2 can absorb long wave radiation, such as infrared ray and greenhouse effects can be formed by warmer weather aggravating the multiplying of pathogenic agents. Since varied kinds of atmospheric pollution are complicated, the categories of pollutants are multitudinous and their influences on human health are complicated (**Figure 2**).

Human and environment are closely related and the atmospheric pollution incurred by the progress of industrial civilization has substantial impacts on mankind. It is shown by researches

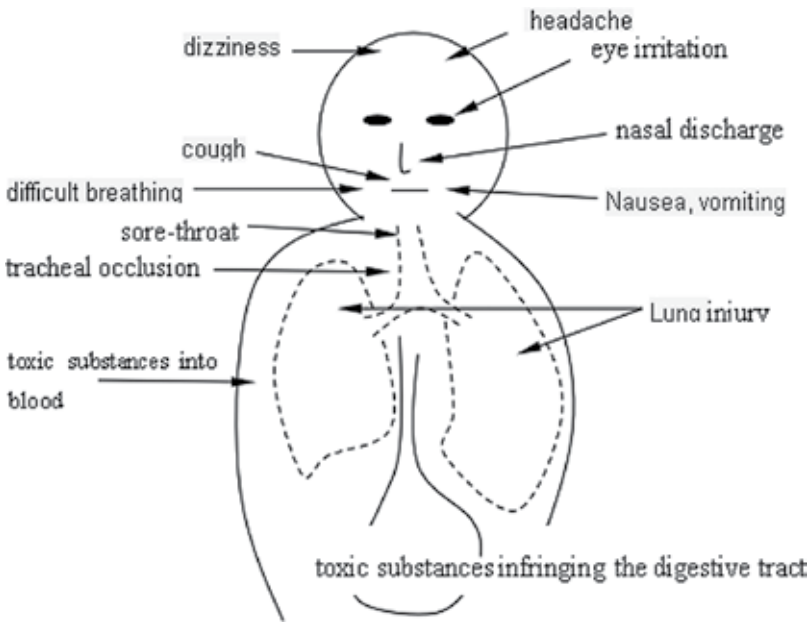

Figure 2. Polluted air is harmful to health.

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of environmental epidemiology that from slight symptom of respiratory system to the increase in outpatient number of cardiopulmonary diseases and mortality rate, all are closely related to atmospheric pollution. As estimated by World Health Organization (WHO), annual death of 800 thousand people and 4.6 million disability adjusted life expectancy (DALY) worldwide are related to urban atmospheric pollution. In 1775, British doctor Pott posed the hypothesis of flue gas pollution to be carcinogenic and in the mid-twentieth century, the occurrence of London smog incidents made people extensively pay attention to the health hazard of atmospheric pollution. However, its influence on health has not yet been fully recognized by mankind. The health hazard caused by atmospheric pollution has become a problem worldwide and it has caught the attention of multitudinous domestic and foreign scholars. Since the health effect is complicated, it is usually the results of multiple factors through comprehensive actions. Thus, it is very difficult to draw the explicit conclusion of a certain factor that causes a certain result. The influence of atmospheric pollution on health is mainly divided into acute influence and chronic influence [12–15].

3.1. Acute influence

The acute influence of atmospheric pollution on humans includes direct actions and indirect actions. For example, acute intoxication of certain poison belongs to direct actions; indirect actions can aggravate the deterioration of respiratory system or heart diseases, etc. and thus accelerate death of patients. There are a few domestic reports on the acute influence of atmospheric pollution on human body and it is pointed by some scholars that when the mean daily concentration of each index of atmospheric pollutants does not exceed the health standard but instantaneous discharge surpasses the maximum allowable concentration for single exposure, the importance of its influence on health might exceed the average day concentration. Therefore, the acute hazard of atmospheric pollution could be more concealed and its severity might be easily neglected by people [16–18].

3.2. Chronic influence

The chronic influence of atmospheric pollution on human body is a kind of composite action, which is manifested in multiple aspects. The early hazard of low concentration atmospheric pollution to human body is usually not fully manifested in the form of disease. Instead, it is mostly manifested as the preliminary effect of diseases. It is shown in the immunologic dysfunction of body and the change in blood and circulation system, which might induce and promote the incidence of allergic diseases, respiration system diseases and other diseases in human body. At the time of performing the epidemiological research, in order to reduce the influence of smoking and occupational factors, etc., children are preferably selected as the investigation objectives [13, 19–21].

From the metabolic process of body, it can be seen that human body and environment are closely related. Human body performs substance exchange with the surrounding environment through metabolism and dynamic equilibrium is normally kept between the substances in environment and human body. In case of abnormal changes in environment, normal physiological functions of human body would be affected with varying degrees and human body is

capable to adjust own physiological functions, so as to adapt to the incessantly changing environment. That normal physiological adjustment function of adapting to the environmental changes is formed by mankind in the long-term development process. If the abnormal changes in environment are within a certain limit, human body can adapt to them. If the abnormal changes of environment surpass the limit of normal physiological adjustment of human body, an anomaly might incur in certain functions and structures of the human body and even in pathological changes. That kind of environmental factor that might incur pathological changes in human body is called environmental pathogenic factor. The diseases of mankind are mostly caused by biological, physical and chemical pathogenic factors. The substances that cause environment pollution include chemical factors, biological factors and physical factors, etc. and they might become pathogenic factors when they reach a certain extent [19–21].

The contents of pollutants in environment are usually very few under many circumstances, after mankind lives in that kind of low concentration polluted environment for several months, several years and even tens of years, chronic hazards to the body might be gradually incurred to cause diseases. Disease is a process of pathological changes in the function, metabolism and morphology of body under the actions of pathogenic factors. Human body has certain compensatory ability against the functional hazard caused by disease factors. Some changes are compensatory and some changes are traumatic in the developing process of diseases and both changes coexist. When the compensatory action is stronger, body can still keep relative stability and the clinical symptoms of diseases would temporarily not emerge. When the pathogenic factors stop acting, body shall develop to restore health. But the compensatory ability is restricted. If the pathogenic factors continue to act, the compensatory functions would gradually be obstructed and the body will manifest unique clinical symptoms and signs of various diseases. The reacting process of human body to environmental pathogenic factors is shown in **Figure 3**.

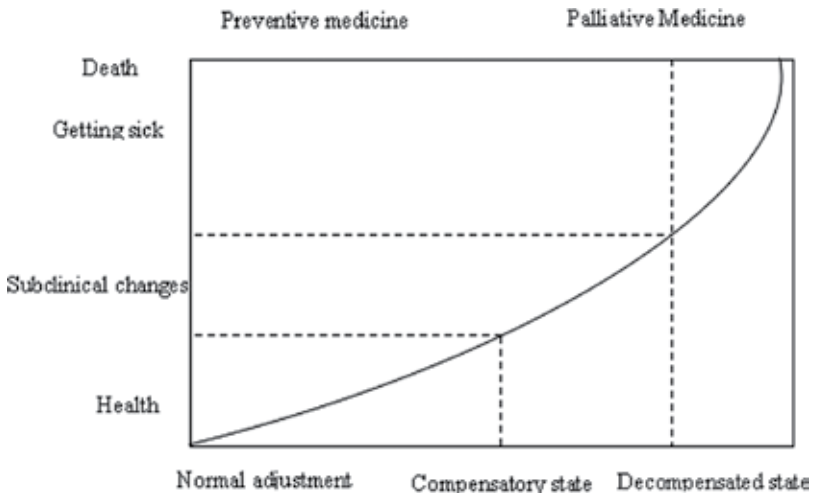

Figure 3. The process that human body react to environmental pathogenic factors.

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The incidence and development of disease is generally divided into the latent period (without clinical manifestation), prodromal period (with slight general discomfort), obvious clinical symptom period (when typical symptoms of the certain disease emerge) and prognostic period (restoring health or deterioration to death). Under the circumstance of acute intoxication, the first two periods of disease might be very short and obvious clinical symptoms and signs might appear very soon. Under long-term micro action of pathogenic factors (such as certain chemical substances), the first two periods of disease can be considerably long. Patients might have no obvious clinical symptoms and signs and instead, they might look healthy. But under continued actions of pathogenic factors, obvious clinical symptoms and signs might emerge. In addition, its resistance to other pathogenic factors (such as bacteria and virus) would decrease and therefore, that kind of person is in the latent period or compensatory state. The stage during which injuries have already occurred by a certain hazard but the clinical symptoms have temporarily not emerged, shall be deemed as the early period of disease (clinical prophase or subclinical state) [19–21].

It is shown by researches that 80–90% of mankind cancer is related to environmental factors, among which 90% are caused by chemical factors and 5% are caused by radioactive factor. But the chemical substances and radioactive substances within the contact scope of mankind mainly come from environmental pollution.

4. Microecology of respiratory tract

The respiratory system of mankind is composed of the following organs: Nose, pharynx, throat, trachea, bronchus and lung. Nose, pharynx and throat are the upper respiratory tract and trachea and bronchus are the lower respiratory tract. Lung is the place where gas exchange is performed. Other organs are the passages of gas and are generically called respiratory tract. Like the permanent planting of flora in other systems or parts of the body, that in the respiratory tract observes a certain regular pattern. For the microorganisms that exist at each part of human respiratory tract, their category and quantity are relatively stable under normal circumstances. Bacteria that are often separable from healthy human body mainly include alpha streptococcus, Neisseria Trevisan, Hemophilia, oral myxococcus, staphylococcus, corynebacteria and a certain anaerobe. Relatively single population of alpha streptococcus mainly lives on the surface of epithelial cells at oral pharynx and oropharynx mucosa. Saliva contains such immunological materials as immune globulin and lysozyme. As a result, as the long-term permanently planted bacteria at oral pharynx, alpha streptococcus has stronger immunity than the crossing bacteria. Alpha streptococcus normally accounts for over 90% of bacteria at oral pharynx [22–24].

The microenvironment of each part of respiratory tract is very complicated. Besides the ecological space of body and the process of its gas exchange with outside environment, which constitute important components of the microscopic environment of respiratory tract, microbiocoenosis (including bacteria, virus, mycoplasma and actinomycete, etc.) inevitably intervenes soon after the birth of individuals, since the passage is open to the exterior. They shall parasitize in different ecological niches of respiratory tract and they shall not be

pathogenic under health. Some can be combined with the receptors on the cell surface, so as to obstruct the invasion of the pathogenic microorganisms that enter later, playing the role of a barrier. The microbiocoenosis at ecological niches of human body, such as the upper respiratory tract and trachea, has relationship of mutual synergia and normal commensal flora can help host to resist the invasion of pathogenic microbes. That protective mechanism includes: (i) aiming at the competition of the same nutrient, disturbing the nutrition of attacking pathogenic bacteria; (ii) aiming at the affinity of the same category of cells, i.e., the commensal bacterium that has relatively big binding force to the receptors on the cells of the host, disturbing the attachment effect of attacking pathogenic bacteria; (iii) through irritating the host to produce immune factor, aiming at the attacking bacteria with "cross" antigen, exerting adverse influences to make the pathogenic bacteria not be permanently planted without difficulty; (iv) through producing bacteriocin or antibiotic, specifically or not disturbing the colonization action of identical or relevant sensitive microorganisms [22–24].

Oral pharynx is the passage to link oral cavity and nasopharynx with the lower respiratory tract and esophagus. Since it is connected with the external environment, it is a physiological part where many bacteria infringe the lower respiratory tract and lung to cause infection. About the general regular patterns for the cognition to microecology of human body, the core population combination that parasitizes at the oral pharynx of a healthy body shall fluctuate within harmonious physiological scope and they shall compose an important microbial barrier for oral pharynx, so as to resist the change in the external environment. The pathogenic bacteria would firstly be permanently planted at the mucosa part represented by oral pharynx, such as mouth and nasopharynx. Then it would reach the lower respiratory tract and lung and would further cause corresponding pathological reaction of the body. Oral pharynx has a high content of organic matter and rich nutrition and is exposed in very complicated microbial environment. Thus it has many opportunities to contact various bacteria. But the categories of bacteria that live at oral pharynx is actually very limited and the total biomass of bacteria is only moderate.

5. Research examples: atmospheric pollution and microecology of respiratory tract in human

In order to explore the early harmful effects of the atmospheric pollution on human health and screen sensible markers, children of school-age who are the susceptible population to the atmospheric pollution are selected in our studies. And the researches were performed in different polluted zones with the different extent of the atmospheric pollution of three cities in the northeast of China, including Benxi, Shenyang and Dalian. Measurement and multiple-analysis were performed to analyze the relationship between the atmospheric pollution and the microecology of the upper respiratory tract.

5.1. The influence of atmospheric pollution on the symptoms and diseases of children's respiratory system

Since the circumstances of air pollution in the three cities are difficult, contrast analysis was performed on the health status of the respiratory system of children in the three cities and the

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children in the contrast regions, so as to have an in-depth understanding of the influence that atmospheric pollution might produce to the body. By comparison of the three cities, children from Benxi and Shenyang are high in the positive rate of the general inflammatory symptoms and diseases of the respiratory system (such as cough, expectoration, sustained cough, sustained cough with expectoration, pneumonia, etc.), but the rate is relatively low in Dalian due to the more serious atmospheric pollution in Benxi and Shenyang (**Table 1**). Significant differences are found in the disease and incidence circumstance of the respiratory system among children in three cities ($P < 0.01$).

Through the investigations to the symptoms and diseases of the respiratory system of 18,000 children in different polluted zones of different cities, it is shown by the analysis of the investigation that the extent of atmospheric pollution is positively related to various symptoms and diseases of respiratory system. In the analysis to each kind of symptoms and diseases, it is all selected into the equation under the condition of $P < 0.05$ for atmospheric pollution and the OR value is all >1, which means atmospheric pollution can increase the risk of symptoms and diseases of children's respiratory system. Thus it is a hazardous factor to cause symptoms and diseases of the respiratory system.

	Benxi region	Shenyang region	Dalian region	Control
Symptom				
Cough	58.6 (3169)	59.5 (3500)	43.6 (1719)	44.1 (947)
Persistent cough	4.0 (216)	4.2 (246)	2.2 (87)	1.9 (40)
Expectoration	36.0 (1947)	37.4 (2198)	28.1 (1108)	26.8 (575)
Persistent cough and expectoration	2.4 (127)	2.6 (151)	1.1 (43)	1.2 (25)
Wheezing	7.3 (396)	1.4 (82)	0	1.4 (29)
Respite	4.5 (241)	2.2 (132)	2.5 (100)	1.4 (29)
Disease				
Bronchitis	10.2 (550)	17.3 (1019)	22. 5 (887)	9.9 (211)
Asthmatic bronchitis inflammation	1.7 (94)	3.3 (195)	5.3 (210)	2.0 (43)
Pneumonia	9.1 (494)	22.1 (1297)	17.5 (692)	11.6 (246)
Eczema	8.9 (479)	12.1 (710)	7.2 (282)	7.5 (159)
Asthma	2.2 (121)	6.1 (359)	10.3 (407)	5.0 (106)
Urticaria	5.8 (314)	7.8 (457)	5.0 (196)	5.5 (116)
Allergic rhinitis	2.0 (106)	3.7 (217)	2.6 (103)	2.2 (46)

**$P < 0.01$.

Table 1. The prevalence rate and the positive rate of children' respiratory system symptoms and diseases in different city.

5.2. The influence of atmospheric pollution on flora of the respiratory tract

According to the atmospheric monitoring data and urban function zoning of Shenyang, heavily polluted regions and light polluted regions are selected (**Table 2**). By means of the

	TSP $\overline{X}$ ± SD	SO_2 $\overline{X}$ ± SD	NO_x $\overline{X}$ ± SD
Heavily polluted region	0.432 ± 0.097	0.169 ± 0.075	0.089 ± 0.021
Light polluted region	0.190 ± 0.041*	0.012 ± 0.041*	0.019 ± 0.058*

*$P < 0.05$.

Table 2. The yearly mean value of the main substance that pollute air in light and heavy polluted regions of Shenyang city (mg/m^3).

cluster sampling, samples of 40 and 50 children (half boys and half girls) aged 6–8 from different regions were selected respectively in 2000, while samples of 40 and 40 children were selected respectively in 2002. They were required not to use antibiotic drugs in the past 3 months. The qualitative location and quantitative analysis was performed to the flora status at the upper respiratory tract of children in the heavily polluted regions and contrast regions, so as to explore the influence of atmospheric pollution on microecology of the respiratory tract.

Normal flora at oral pharynx is the natural barrier for the body. If such barrier is destructed by harmful factors, the flora would be imbalanced in quantity and category. As a result, the ability of the defense barrier would decline, exogenous pathogenic bacteria would intrude, or endogenous conditions would cause massive multiplication of pathogenic bacteria to incur infection. It is shown by our research that the density of the aerobic and anaerobic bacteria at oral pharynx of children in the seriously polluted air regions is higher than that of the light polluted regions and the abnormal flora also emerged, showing status of the excessive growth of flora at pharynx of children in the heavily polluted regions (**Table 3**). It is also shown that the detection rate of the conditional pathogenic bacteria of children in the heavily polluted regions was higher than that of the contrast regions. Permanent colonization of the pathogenic bacteria also emerged at oral pharynx of children in the heavily polluted regions, such as beta streptococcus, Klebsiella pneumonia, Serratia liquefaciens and Pseudomonas maltophilia (**Table 4**). The mutual antagonism and mutual restriction objectively exist in the microbes and this antagonism mainly comes from the microbial community. If the community is destructed, it

	Aerobic		Anaerobic		Total number	Amount (lgX ± SD)
	Detectable number	Amount	Detectable number	Amount		
Heavily polluted region (40)	763	2.883 ± 0.245*	837	2.923 ± 0.255*	1600	3.147 ± 0.260*
Light polluted region (50)	546	2.737 ± 0.926**	321	2.507 ± 0.050**	867	2.899 ± 0.194**

*$P < 0.05$.
**$P < 0.01$.

Table 3. The distribution of bacteria cluster in children' pharynx in different polluted regions (CFU/ml).

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Bacterial species	Heavily polluted region ($N = 40$)		Control ($N = 50$)			
	Detection number	Detection rate (%)	Detection number	Detection rate (%)		
Aerobic						
Neisseria	33	82.5	39	78.0	0.82	>0.05
Streptococcus pneumoniae	21	52.5	17	34.0	3.12	>0.05
Streptococcus	11	27.5	24	48.0	3.93	<0.05
Group D streptococci	6	15.0	5	10.0		
Streptococci	2	5.0	0	0.0		
Liquefied Serratia	1	2.5	0	0.0		
Pseudomonas maltophilia	1	2.5	0	0.0		
Diphtheroids	1	2.5	2	4.0		
Klebsiella pneumoniae	1	2.5	0	0.0		
Staphylococcus aureus	0	0.0	1	2.0		
Micrococcus	0	0.0	1	2.0		
Anaerobes						
Peptostreptococcus	19	47.5	18	36.0	1.21	>0.05
Veillonellaceae	16	40.0	29	58.0	2.88	>0.05
Bacteroides	9	22.5	12	24.0	0.03	>0.05
Excellent bacillus	8	20.0	6	12.0	1.08	>0.05
Fusobacterium nucleatum	4	10.0	9	18.0	1.15	>0.05
Propionic acid bacteria	2	5.0	2	4.0		
Lactobacillus	1	2.5	2	4.0		
Iraq Actinomyces	1	2.5	0	0.0		
Clostridium perfringens	1	2.5	2	4.0		
Bifidobacterium	0	0.0	1	2.0		

Table 4. The comparison of the bacteria examined from children's pharynx in light and heavy polluted regions.

would be favorable for the translocation of alien microbes and the multiplication of the pathogenic bacteria. It is shown by the research studies that the atmospheric pollution can cause the changes in the microbial community at oral pharynx of the body. Therefore, the atmospheric pollutants as exogenous factors have caused the destruction to the microecology balance of the upper respiratory tract. It is shown by the research that the incidence rate of the chronic inflammation at the upper respiratory tract and the incidence rate of chronic bronchitis of pupils in the heavily polluted regions are both higher than that of the light polluted regions (**Tables 5** and **6**), which can be attributed to the microecology imbalance at pharynx of the upper respiratory tract caused by the atmospheric pollution.

	Cough	Persistent cough	Expectoration	Cough and expectoration	Persistent cough and expectoration	Wheezing	Respite
Light polluted region							
First grade	59.7 (597)	3.9(39)	37.5(375)	33.3(333)	2.2(22)	1.8(18)	2.5(25)
Sixth grade	54.8 (544)	4.2(42)	34.1(338)	29.9(297)	1.9(19)	0.6(6)	2.6(26)
Total	57.3 (1141)	4.1(81)	35.8(713)	31.6(630)	2.1(41)	1.2(24)	2.6(51)
Heavily polluted region							
First grade	64.5 (632)	4.2(41)	42.4(416)	38.4(376)	3.1(30)**	2.0(20)	2.2(22)
Sixth grade	61.7 (586)**	5.9(56)**	38.7(367) **	34.5(327)**	3.9(37)*	1.3(12)	2.1(20)
Total	63.1 (1218)**	5.0(97)	40.6(783)**	36.4(7037)**	3.5(67)**	1.7(32)	2.2(42)

*$P < 0.05$.
**$P < 0.01$.

Table 5. The positive rate of children' respiratory system symptom in different polluted regions in Shenyang City (%).

	Bronchitis	Asthmatic bronchitis	Pneumonia	Asthma	Eczema	Urticaria	Allergic rhinitis
Light polluted region							
First grade	20.5(205)	3.4(34)	21.5(215)	6.3(63)	15.3(153)	7.4(74)	2.3(23)
Sixth grade	13.0(129)	3.1(31)	21.1(209)	5.6(56)	12.8(127)	10.1(100)	4.5(45)
Total	16.8(334)	3.3(65)	21.3(424)	6.0(119)	14.1(280)	8.73(174)	3.4(68)
Heavily polluted region							
First grade	23.0(225)	4.1(40)	25.9(254)	8.8(86)**	12.8(125)	5.5(54)*	1.9(19)
Sixth grade	14.2(135)	2.6(25)	22.4(213)	5.8(55)	8.7(83)*	9.8(93)	5.1(48)
Total	18.7(360)	3.4(65)	24.2(467)*	7.3(141)*	10.8(208)**	7.6(147)	3.5(67)

*$P < 0.05$.
**$P < 0.01$.

Table 6. The children' prevalence rate of respiratory disease in different polluted regions of Shenyang City (%).

It is shown by the dynamic analysis for 2 years that the detectable rate of alpha streptococcus at oral pharynx of children in the heavily polluted regions is lower than that of the contrast regions and the difference is significant (**Table 4**). Alpha streptococcus is the normal flora at the pharynx mucosa and it can be antagonistic against the abnormally permanent planting of the certain pathogenic bacteria, playing the role of a barrier. The reduction in the alpha streptococcus of children in the heavily polluted regions would certainly decrease the protective effects of the pharynx barrier and would increase the susceptibility to infection. It is

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considered by some scholars that alpha streptococcus can be taken as the probiotics for the upper respiratory tract. It can play a role in preventing and treating the infection of the upper respiratory tract by means of biological oxygen capture, flora adjustment and biological antagonism, which is consistent with our research results. It is also shown by this research that atmospheric pollution can cause the reduction of alpha streptococcus and further cause the increase of other abnormal flora. As a result, the infection of the respiratory tract might be increased. Therefore, we consider that alpha streptococcus can be used as one of the probiotics for the upper respiratory tract and it needs to be further developed as a microbial preparation, so as to prevent and treat infection of the upper respiratory tract and maintain the microecology balance.

6. Atmospheric pollution and microecology of respiratory tract in rats

To explore the regulation mechanisms of air pollution-related molecules and demonstrate the effects of air pollutants on the microecology of the respiratory tract, Wistar rats were used to expose to simulated real atmospheric pollution. It is beneficial to provide theoretical basis for the influence of air pollutants to the health of human body and the prevention and control of atmospheric pollution [25].

6.1. The influence of total suspended particulate (TSP) to the respiratory tract flora of rats

The relationships between normal microbiota and the host include ecosystems, biomes, species and individuals. Normal microbiota are divided into origin flora and foreign flora. The origin flora has more obvious physiological function if they come in contact with the host tissue cells closely, therefore providing more protection to the host and vice versa. Microecological balance is the ecological balance at the cellular level and molecular level, including microorganisms and host aspects. Normal microbiota of local ecological balance have obvious biological antagonist action to the pathogenic bacteria. Microecological imbalance will occur when this balance is damaged or influenced by external environmental factors.

From the ecological perspective, the normal flora of pharynx is the natural protective barrier. If the bacterial flora imbalance in quantity and types occur when the barrier suffers from the damage of harmful factors, the defensive barrier will break; therefore, the invading exogenous pathogen or some kind of endogenous bacteria will multiply and cause infection.

6.1.1. The influence of TSP to the bacterial species of respiratory tract of rats

TSP was collected continuously at a flow rate of 1000 l/min from areas of coal burning, traffic congestion in Shenyang city (China) during heating season and made into suspension by ultrasonic oscillation. The aerobic and anaerobic bacteria on the oropharynx of Wistar rats were analyzed before or after exposure to TSP. The results are shown in **Table 7**.

Klebsiella pneumoniae and *Staphylococcus aureus* were not detected on the oropharynx of Wistar rats before exposure to TSP, however, *K. pneumoniae* were detected on the oropharynx of six rats ($P < 0.01$) and *S. aureus* were detected on the oropharynx of two rats after they exposure to

Bacterial species	No TSP exposure(n = 78)		TSP exposure(n = 67)	
	Rats number	Detection rate	Rats number	Detection rate
Escherichia coli	20	25.6	16	23.9
Diphtheroid bacillus	13	16.7	9	13.4
Group A *Streptococcus*	31	39.7	33	49.3
Neisseria	20	25.6	11	16.4
Streptococcus oralis	13	16.7	6	9.0
Klebsiella pneumoniae	0	0.0	6	9.0*
Staphylococcus aureus	0	0.0	2	3.0
Streptococcus pneumoniae	7	9.0	6	9.0
Staphylococcus epidermidis	11	14.1	14	20.9
Moraxella lacunata	29	37.2	26	38.8
Micrococcus	6	7.7	5	7.5
Veillonella	17	21.8	13	19.4
Lactobacillus	6	7.7	10	14.9
Peptostreptococcus	38	48.7	24	35.8
Bacteroides	21	26.9	21	31.3
Clostridium perfringens	6	7.7	4	6.0
Eubacterium	17	21.8	16	23.9
Peptococcus	6	7.7	5	7.5

*Compared with no TSP exposure, $P < 0.01$.

Table 7. The comparison of bacterial detection rates before or after TSP exposure (%).

TSP (**Table 7**). The detection rate of *Escherichia coli* on the oropharynx of rats was higher after low, medium and high concentration of TSP exposure, furthermore, the detection rate of peptostreptococcus was proportional to the concentration of TSP (**Table 9**). *E. coli* and peptostreptococcus are the normal flora of intestinal tract; therefore, the detection results show that the bacterial species on the oropharynx of Wistar rats changed after exposure to TSP, resulting in the colonization of pathogenic bacteria and intestinal bacteria.

Normal flora plays an important role for maintaining ecological balance and stable internal environment. So many microaerobic, aerobic and anaerobic bacteria colonize in the upper respiratory tract of healthy people including 21 genera and 200 species of bacteria. The normal flora of respiratory tract are affected by the normal flora on the oropharynx because of the anatomic relationship. Compared with the control group, the detection rates of aerobic and anaerobic bacteria had no significant difference after low, medium and high concentration of TSP exposure because of the acute toxicity experiment with short time but large dose of TSP exposure. The changes of microbiocenosis in respiratory tract affected by air pollution

are gradual, there will be more apparent changes after long-time exposure to pollutants eventually.

6.1.2. The influence of TSP to the density of bacteria in the respiratory tract of rats

The first step of bacterial colonization is adhesion. Similar to other biological cells, the bacterial proteins can be ionized into positively-charged amino ($-NH^+$) and negatively charged carboxyl ($-COO^-$) in solution and the degree of ionization is related to the pH of the environment. Negatively charged bacteria mainly exist in the alkaline solution and vice versa. The isoelectric point of gram-positive bacteria is pH 2–3 and the isoelectric point of gram-negative bacteria is pH 4–5; therefore, all the bacteria are negatively charged in neutral or weak alkaline medium with pH 7.2–7.6 and the gram-positive bacteria contain much more negative charges than gram-negative bacteria. For charge dependence, the bacterial adhesion increases with the pH of the environment. Moreover, the damage of host immune defense (such as malnutrition, basic disease, etc.) also can provide conditions for bacterial adhesion. However, there is mutual constraint relationship among bacterial flora such as space competition and nutrient competition.

The density of bacteria on the oropharynx of rats before or after TSP exposure was determined and the results are shown in **Tables 8** and **9** and **Figure 4**. There was no significant difference in the bacterial density of aerobe flora, however, the density of anaerobic bacteria flora was markedly reduced ($P < 0.05$) on the oropharynx of rats after TSP exposure compared with the results before TSP exposure (**Tables 10** and **11**). The density of bacteroides and eubacterium of anaerobic bacteria were markedly reduced ($P < 0.05$) on the oropharynx of rats after TSP exposure compared with the results before TSP exposure. There was no significant difference

Group	*n*	Aerobic bacteria	Anaerobic bacteria	Average bacterial density
No TSP exposure	78	2.4111 ± 0.5868	2.5983 ± 0.4850	2.5047 ± 0.5447
TSP exposure	67	2.3353 ± 0.6566	2.4039 ± 0.6178*	2.3696 ± 0.6361

*Compared with no TSP exposure, $P < 0.05$.

Table 8. The comparison of bacterial density before or after TSP exposure.

Time	*n*	Aerobic bacteria	Anaerobic bacteria**	Average bacterial density*
No TSP exposure	78	2.4111 ± 0.5868	2.5983 ± 0.4850*	2.5047 ± 0.5447**
1st day after TSP exposure	31	2.4223 ± 0.7594	2.5503 ± 0.6115*	2.4863 ± 0.6868*
8th day after TSP exposure	36	2.2605 ± 0.5534	2.2778 ± 0.6033	2.2691 ± 0.5749

*Compared with 1st day, $P < 0.05$.
**Compared with 8th day, $P < 0.05$.

Table 9. The comparison of bacterial density in different time after TSP exposure.

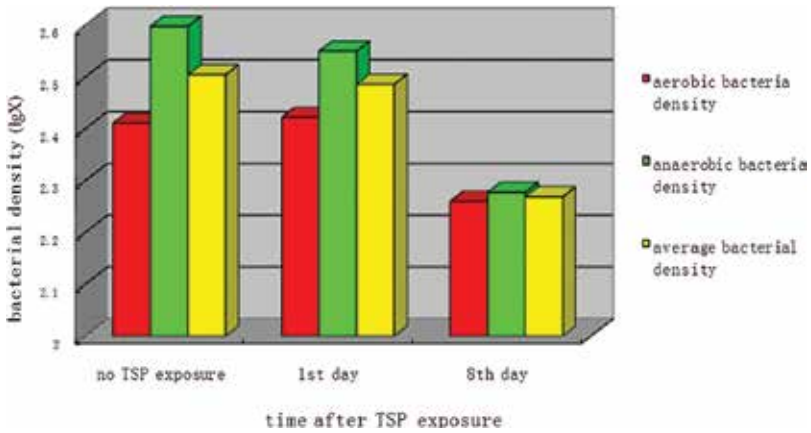

Figure 4. The bacterial density in different time after TSP exposure.

in the bacterial density of aerobe flora before TSP exposure and on the first day after TSP exposure, however, the density of aerobic bacteria was downturn on the 8th day after TSP exposure and the density of anaerobic bacteria flora was obvious decline with the passage of time ($P < 0.05$) on the oropharynx of rats after TSP exposure compared with the results before TSP exposure (**Tables 7** and **9**). The average density of bacteria were significantly decreased

Group	*n*	Aerobic bacteria	Anaerobic bacteria	Average bacterial density
Control	18	2.2531 ± 0.7952	2.2922 ± 0.6948	2.2531 ± 0.7837
Low concentration	16	2.4139 ± 0.4964	2.4936 ± 0.4958	2.4537 ± 0.4706
Middle concentration	18	2.3190 ± 0.7240	2.5295 ± 0.7411	2.4243 ± 0.7299
High concentration	15	2.3698 ± 0.5890	2.2915 ± 0.5143	2.3306 ± 0.5448

Table 10. The comparison of bacterial density in different TSP concentrations.

Aerobic bacterial species	No TSP exposure		TSP exposure
	N	$\overline{X}_{lgX}$–S_{lgX}	$\overline{X}_{lgX}$–S_{lgX}
Escherichia coli	8	1.1899 ± 0.8867	0.8677 ± 0.8393
Diphtheroid bacillus	2	2.6276 ± 0.2129	1.8891 ± 0.5826
Group A *Streptococcus*	15	2.2335 ± 0.3932	2.3379 ± 0.4941
Neisseria	2	2.2516 ± 0.6799	1.6901 ± 0.3012
Streptococcus pneumoniae	2	2.8037 ± 0.2129	1.6901 ± 0.5502
Moraxella lacunata	9	1.9036 ± 0.7001	2.0308 ± 0.2370

Table 11. The comparison of aerobic bacterial density before or after TSP exposure.

($P < 0.05$). There was no apparent change in the bacterial density of aerobe flora because of the short interval time of TSP exposure. The density of peptostreptococcus, bacteroides and eubacterium of anaerobic bacteria were markedly reduced ($P < 0.01$ or $P < 0.05$) on the oropharynx of rats over time after TSP exposure. The results showed that the flora on the oropharynx of rats changed and there were apparent changes in the normal flora especially the obvious reduction of anaerobic bacteria as the extension of the TSP exposure time. The normal flora in upper respiratory tract plays an important role in maintaining immune defense of respiratory tract. They can serve as antigens to stimulate the host to produce antibodies, at the same time, stimulate macrophages, enhanced the activity of interferon, release toxic terminal metabolites such as bacteriocin, bacteriocidin and fatty acid to constitute a biological barrier to prevent the invasion of invading bacteria. Studies have shown that anaerobic bacteria, especially the obligate anaerobic bacteria are the vast majority in number in the respiratory tract and they are indigenous microflora and physiological microbes. Obligate anaerobic bacteria have a strong biological antagonism effect, they integrate with mucosal epithelial cells closely forming a layer of biofilm to provide the space-occupying protective effects on the host cells. If the barrier is destroyed, the obligate anaerobic bacteria cannot survive because of the destruction of the anaerobic environment, foreign bacteria will invade. In summary, the normal flora on the oropharynx can be damaged by air pollution, thus it is easy for pathogenic bacteria to colonize.

6.1.3. The influence of TSP concentration to the flora of respiratory tract of rats

Normal flora is influenced by various aspects especially the physiological functions and the pathological features of the host, so that, different groups of rats are used to explore the relationship between the density of aerobic and anaerobic bacterial flora and the concentration exposed to TSP and the results are shown in **Table 10** and **Figure 5**.

Compared with control group, there was no significant difference in the bacterial quantity (**Table 16**), however, the quantity of aerobic bacteria fluctuated up and down with different dose of TSP exposure, the quantity of anaerobic bacteria was increased related to the dose of

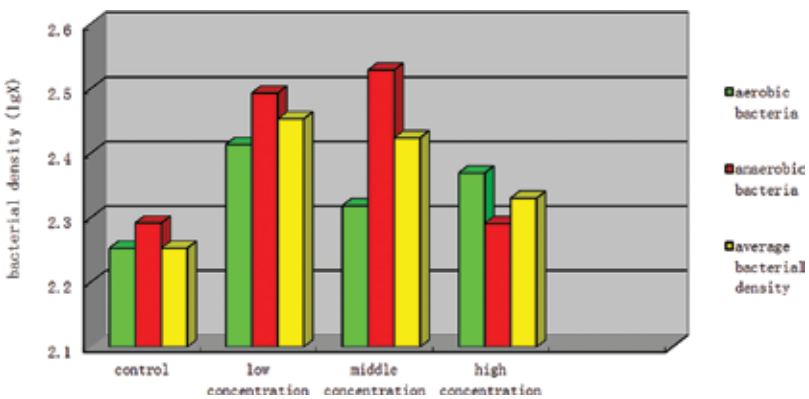

Figure 5. The bacterial density in different TSP concentration.

TSP exposure, but decreased in high concentration group ($P > 0.05$). The average density of bacteria had no significant difference too. In the aerobic bacteria, the density of *E. coli* increased obviously after TSP exposure, the density of Neisseria of high concentration group was obviously higher than that of control group and low concentration group ($P < 0.05$) on the oropharynx of rats. In the anaerobic bacteria, the density of Veillonella in the high concentration group was significantly decreased ($P < 0.05$) on the oropharynx of rats.

The colonization of bacteria is influenced by the host factors and the normal microbiota. Tension of the mucous membrane of the host, cilia movement and various discharge are important factors affecting bacteria to colonize. Normal microbiota members mainly come from anaerobic bacteria, especially the obligate anaerobe. Normal flora is an important factor in protecting human health. The results confirmed that air pollution can cause changes in the microecological condition of respiratory tract. Anaerobic bacteria on the oropharynx are the important biological barrier to resist external environmental change. Therefore, when the concentration of air pollutants is not very high, the quantity of anaerobic bacteria increases to maintain the ecological balance and prevent invading bacteria from colonizing. However, when the concentration of air pollutants is very high, the quantity of anaerobic bacteria decreases to result to the microecological imbalance in respiratory tract. Thus, it is easy for pathogenic bacteria and intestinal bacteria to colonize.

6.1.4. The changes of flora in the upper respiratory tract of rats before or after TSP exposure

The results of aerobic bacteria and anaerobic bacteria on the oropharynx of rats before or after TSP exposure are shown in **Tables 11** and **12**.

Compared with no TSP exposure, there was no significant difference in the bacterial density of aerobe flora after TSP exposure (**Table 11**). In the aerobic bacteria, the detection rate of group A Streptococcus increased after TSP exposure ($P > 0.05$), which is different from the results of our research on crowd. Maybe the resident flora of rats and humans are different and the results only come from the acute toxicity experiment with short time but large dose of TSP exposure. In the anaerobic bacteria, the density of bacteroides and eubacterium were obviously reduced ($P < 0.05$) on the oropharynx of rats after TSP exposure (**Table 12**).

	No TSP exposure		**TSP exposure**
Anaerobic bacterial species	***N***	$\overline{X}_{lgX}$–S_{lgX}	$\overline{X}_{lgX}$–S_{lgX}
Peptostreptococcus	14	2.2253 ± 0.5418	2.1307 ± 0.6799
Veillonella	5	1.6403 ± 0.3261	1.4897 ± 0.2718
Bacteroides	8	2.3978 ± 0.4495	1.7658 ± 0.6459*
Eubacterium	5	2.6384 ± 0.4277	2.1574 ± 0.4548**

*Compared with no TSP exposure, $P < 0.05$.
**Compared with no TSP exposure $P < 0.01$.

Table 12. The comparison of anaerobic bacterial density before or after TSP exposure.

6.2. The influence of PM_{10} to the respiratory tract flora of rats

Particulate matter less than 10 μm in diameter (PM_{10}) was collected using low flow of PM10 air sampling instrument and glass fiber filter film with 55 mm diameter from areas of coal burning in winter heating season, traffic and life pollution area in Shenyang city (China). Wistar rats were used as a model to analyze the changes of flora on the oropharynx before or after exposure to PM_{10}. The results are shown in **Tables 13–15**.

Bacterial species	No PM10 exposure		PM10 exposure %		$\chi 2$	*p*
	n	%	*n*	%		
Bacillus subtilis	8	17.0	3	6.4	1.647	0.198
Staphylococcus aureus	6	12.8	16	34.0*	4.807	0.027
Staphylococcus epidermidis	22	46.8	20	42.6	0.043	0.836
Acinetobacter baumannii	8	17.0	9	19.1	0.000	1.000
Neisseria	6	12.8	2	4.2	1.230	0.267
Group A *Streptococcus*	18	38.3	15	31.9	0.187	0.666
Micrococcus luteus	9	19.1	2	4.2*	3.706	0.050
Staphylococcus saccharolylicus	11	23.4	3	6.4*	4.113	0.040
Peptostreptococcus	13	27.6	12	25.5	0.000	1.000
Eubacterium aerofaciens	4	8.5	5	10.6	0.000	1.000
Veillonella parvula	20	42.6	9	19.1*	4.987	0.025
Clostridium harmless spore	3	6.4	6	12.8	0.492	0.486
Actinomyces israelii	2	4.2	4	8.5	0.178	0.677
Clostridium difficile	1	2.1	5	10.6	—	—.
Streptococcus anginosus	0	0.0	3	6.4	—	—
Clostridium death	0	0.0	4	8.5	—	—
Fusobacterium nucleatum	0	0.0	6	12.8	—	—

*Compared with no PM10 exposure, $P < 0.05$.

Table 13. The comparison of bacterial detection rates before or after PM_{10} exposure ($n = 47$).

Group	Aerobic bacteria ($n = 44$)	Anaerobic bacteria ($n = 47$)	Total bacteria ($n = 47$)
No TSP exposure	331.2 ± 3.5	177.3 ± 2.1	601.9 ± 1.9
TSP exposure	452.3 ± 2.9	251.8 ± 2.2*	796.5 ± 1.8
t(paired)	−1.769	−2.407	−2.620
P	0.084	0.020	0.012

*Compared with no PM10 exposure, $P < 0.05$.

Table 14. The comparison of the bacterial density before or after PM_{10} exposure (CFU/ml, G ± S).

Group	*n*	Aerobic bacteria	Anaerobic bacteria	Total bacteria
Control	12	241.3 ± 8.1	177.0 ± 2.9	630.2 ± 2.6
1st day	12	854.9 ± 1.3*	288.9 ± 1.9	1011.8 ± 1.5
7th day	11	164.6 ± 4.0	334.0 ± 2.3	627.3 ± 1.8
30th day	12	581.7 ± 1.8*	222.1 ± 1.9	865.2 ± 1.5
F		3.700	1.414	1.622
P		0.019	0.252	0.198

*Compared with control group, $P < 0.05$.

Table 15. The comparison of bacterial density in different time after PM_{10} exposure (CFU/ml, G ± S).

6.2.1. The influence of PM_{10} to the detection rate of respiratory tract flora of rats

Compared with no PM_{10} exposure, the detection rate of *S. aureus* increased significantly ($P < 0.05$), the detection rate of *Micrococcus luteus, Staphylococcus saccharolylicus* and *Veillonella parvula* decreased obviously ($P < 0.05$). It was interesting that three strains of *Streptococcus anginosus*, four strains of Clostridium death and six strains of *Fusobacterium nucleatum* were detected on 30th day after exposure to PM_{10} (**Table 14**).

6.2.2. The influence of PM_{10} to the density of respiratory tract flora of rats

After PM_{10} exposure, the total bacterial content and the density of anaerobic flora on the oropharynx of rats were significantly higher than that before PM_{10} exposure ($P < 0.05$) (**Table 14**, **Figure 6**). Compared with no PM_{10} exposure, the detection rate of *V. parvula*

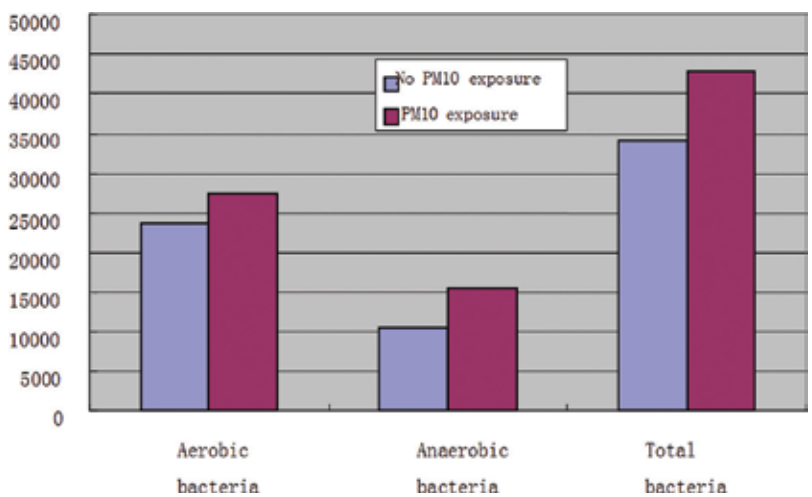

Figure 6. The comparison of the bacterial number before or after PM_{10} exposure.

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decreased obviously, however, the density of it increased significantly after PM_{10} exposure ($P < 0.05$). Moreover, the density of *Actinomyces israelii* increased significantly after PM_{10} exposure ($P < 0.05$).

6.2.3. The changes of flora density in the upper respiratory tract of rats before or after PM_{10} exposure

Compared with no PM_{10} exposure, the density of peptostreptococcus and *V. parvula* increased significantly in the same rats ($P < 0.05$).

6.2.4. The changes of flora density in the upper respiratory tract of rats at different time after PM_{10} exposure

The density of aerobic flora on the oropharynx of rats on the first day after PM_{10} exposure was significantly higher than that in control group ($P < 0.05$), while it decreased on 7th day but increased obviously on 30th day after PM_{10} exposure($P < 0.05$). The total bacterial content and the density of anaerobic flora on the oropharynx of rats were increased volatility after PM_{10} exposure, but no significant difference (**Table 15**, **Figure 7**).

The results of present study showed that if the microecological balance was affected by some abnormal interference and destruction, there would be quantitative, qualitative or positioning changes in microbial population; therefore, the function of microbial biological barrier was weakened, so that some physiologic microbes became pathological bacteria. The results of this study showed that the detection rates of *S. aureus, Clostridium difficile, S. anginosus, Clostridium death* and *F. nucleatum* were much higher after PM_{10} exposure than that before PM10 exposure. The density of aerobic and anaerobic flora on the oropharynx of rats were significantly higher, furthermore, the density of *V. parvula* and *A. israelii* increased significantly than that before PM_{10} exposure ($P < 0.05$) (**Table 14**).

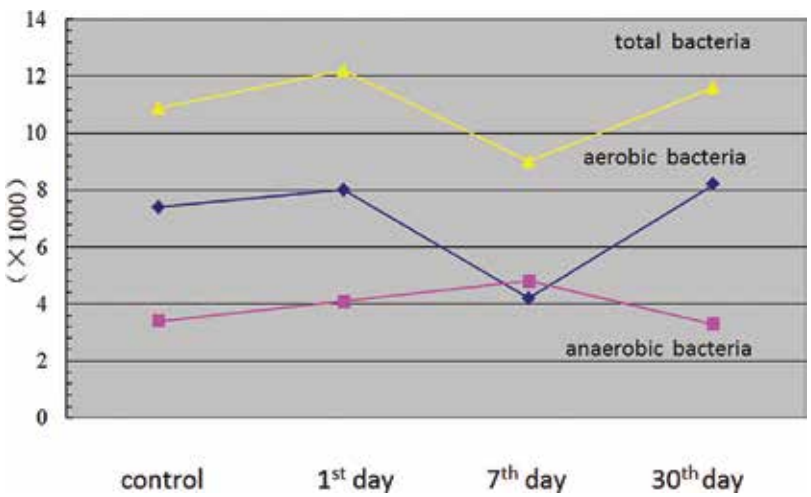

Figure 7. The changes of bacterial number in different time after PM_{10} exposure.

The results of Wistar rat animal model of air pollution showed that after PM_{10} exposure, the density of anaerobic flora was significantly higher than that before PM_{10} exposure in the same rats ($P < 0.05$) and the total bacterial content had increased significantly ($P < 0.05$). In the anaerobic flora, the density of peptostreptococcus and *V. parvula* increased significantly ($P < 0.05$). All of these suggest that the air pollutants can lead to the increase of the bacterial intensity including both normal flora and pathogenic bacteria, therefore resulting in microecological imbalance.

The results of **Figure 8** showed that the density of aerobic flora on the oropharynx of rats on the first day after PM_{10} exposure was significantly higher than that in control group, while it decreased on 7th day but increased obviously on 30th day after PM10 exposure ($P < 0.05$). Which suggested that large amount of bacteria bred and resulted to microecological imbalance after 1 day PM_{10} exposure. With the extension of time to PM_{10} exposure, there was a compensatory state after 7 days PM_{10} exposure, however, the microecological balance was broken again after 30 days PM_{10} exposure and led to a significant increase in bacterial number and density ($P < 0.05$).

C. difficile is the normal flora in human intestine, which can lead to false membranous enteritis. It can be detected on the oropharynx of rats on 30th day after PM_{10} exposure, which suggests that ectopic metastases occur in the flora at that time. In recent years, the infection of the lungs and pleura caused by anaerobic bacteria especially peptostreptococcus and *F. nucleatum* is increasing and the proportion is as high as 50–80%. The present results showed that the density of anaerobic flora especially *F. nucleatum, V. parvula* and peptostreptococcus was significantly higher than that before PM_{10} exposure which provide further proof that the exposure to the air pollutants for long time can lead to respiratory infections.

Infection is a process in which microbe, host and environment interact with each other. Under the conditions of microecological balance, normal flora is harmless and beneficial to the host, however, the quantitative, qualitative, or positioning changes of normal flora will lead to infection under the condition of microecological imbalance. The results of our present experiments indicated that the density change of bacterial flora could act as a sensitive indicator of

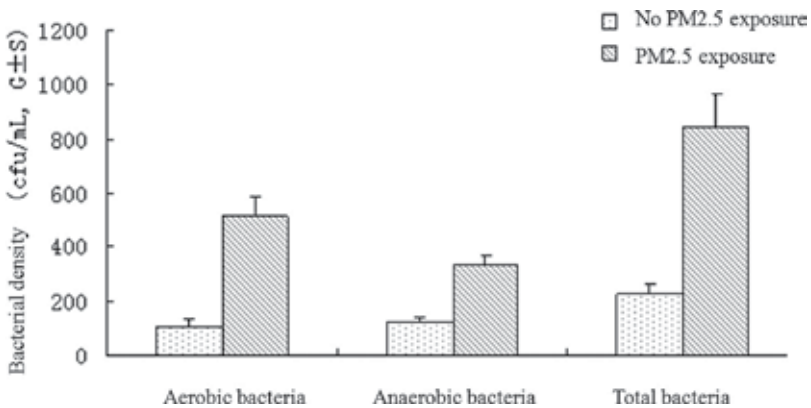

Figure 8. The comparison of bacterial density before or after $PM_{2.5}$ exposure ($^{\Delta}P < 0.01$).

air pollution effecting on human body. Air pollution could cause microecological imbalance in the respiratory tract of human body, furthermore, lead to the infection of respiratory tract and other respiratory diseases.

6.3. The influence of $PM_{2.5}$ to the respiratory tract flora of rats

Particulate matter less than 2.5 μm in diameter ($PM_{2.5}$) was collected using portable air sampling instrument and glass fiber filter film with 47 mm diameter from areas of coal burning in winter heating season and the traffic downtown area in Shenyang city (China). Wistar rats were used as model to analyze the changes of the respiratory tract flora on the oropharynx before or after exposure to $PM_{2.5}$. The results are shown in **Figures 8** and **9**.

6.3.1. The influence of $PM_{2.5}$ to the detection rate of respiratory tract flora on the oropharynx of rats

Compared with no $PM_{2.5}$ exposure, the detection rates of Cellulomonas/Mycobacterium, *Actinomyces naeslundii, Bacteroides ureolyticus* and *Gemella morbillorum* increased significantly ($P < 0.05$). The detection rates of *E. coli* and *Staphylococcus sciuri* increased significantly too ($P < 0.01$). The detection rates of *K. pneumoniae, Staphylococcus gallinarum* and *Staphylococcus lentus* increased but had no statistical significance ($P > 0.05$). *Streptococcus pneumoniae, S. anginosus, S. aureus, Mannheimia haemolytica, Plesimonas shigelloides* and *Pasteurella pneumotropica* were detected after $PM_{2.5}$ exposure which did not exist on the oropharynx of rats before $PM_{2.5}$ exposure (**Table 16**).

6.3.2. The influence of $PM_{2.5}$ to the density of respiratory tract flora on the oropharynx of rats

Compared with the results before $PM_{2.5}$ exposure, the density of aerobic and anaerobic bacteria as well as the total population of bacteria significantly increased ($P < 0.01$) (**Table 17**). In the aerobic bacteria, the density of *S. sciuri, Staphylococcus xylosus* and so on increased obviously. The density of some bacteria such as *S. anginosus, S. aureus* and *S. pneumoniae* which were

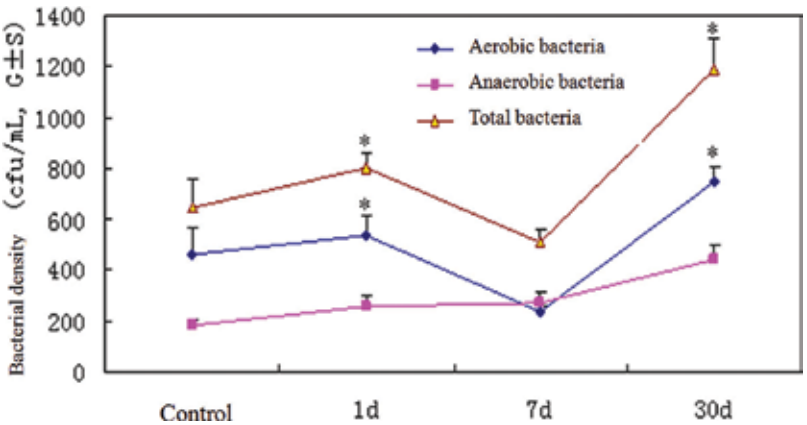

Figure 9. The changes of bacterial density in different time after $PM_{2.5}$ exposure(* $P < 0.05$).

Bacterial species	No PM2.5 exposure		PM2.5 exposure		
	Number of rats	Detection rate	Number of rats	Detection rate	*p*
Escherichia coli	3	3.9	20	26.0	0.000$^{\Delta}$
Cellulomonas/Microbacterium	5	6.5	14	18.2	0.027*
Staphylococcus sciuri	2	2.6	17	1.3	0.000$^{\Delta}$
Klebsiella pneumoniae subsp pneumoniae	1	1.3	4	5.2	0.363
Streptococcus pneumoniae	0	0	1	1.3	0.156
Streptococcus anginosus	0	0	6	7.8	0.066
Staphylococcus aureus	0	0	1	1.3	0.156
Actinomyces naeslundii	2	2.6	10	12.9	0.016*
Bacteroides ureolyticus	5	6.5	17	22.1	0.027*
Gemella morbillorum	1	1.3	8	10.4	0.039*
Mannheimia haemolytica (Algae Pasteur bacteria)	0	0	3	3.9	0.061
Plesimonas shigelloides	0	0	1	1.3	0.156
Pasteurella pneumotropica	0	0	1	1.3	0.156
Staphylococcus gallinarum	3	3.9	9	11.68	0.071
Serratia marcescens	6	7.8	8	10.4	0.575
Enterobacter gergoviae	5	6.5	7	9.1	0.548
Staphylococcus lentus	5	6.5	11	14.3	0.113
Aerococcus viridans	3	3.9	6	7.8	0.303

*Compared with no $PM_{2.5}$ exposure, $P < 0.05$.
$^{\Delta}$Compared with no $PM_{2.5}$ exposure, $P < 0.01$.

Table 16. The comparison of bacterial detection rates before or after $PM_{2.5}$ exposure (%).

Group	Aerobic bacteria	Anaerobic bacteria	Total bacteria
No PM2.5 exposure	107.13 ± 27.69	122.54 ± 15.40	229.67 ± 430.83
PM2.5 exposure	512.71 ± 78.18$^{\Delta}$	330.95 ± 35.64$^{\Delta}$	843.65 ± 120.61$^{\Delta}$
P	0.000	0.008	0.000

$^{\Delta}$Compared with no PM2.5 exposure, $P < 0.01$.

Table 17. The comparison of bacterial density before or after $PM_{2.5}$ exposure (CFU/ml, G ± S).

detected only after $PM_{2.5}$ exposure also increased. In the anaerobic bacteria, the density of some bacteria such as *Clostridium tertium* and *G. morbillorum* which were detected only after $PM_{2.5}$ exposure increased too.

6.3.3. The changes of flora density on the oropharynx of rats at different time after $PM_{2.5}$ exposure

Compared with the results before $PM_{2.5}$ exposure, the density of aerobic bacteria and the total population of bacteria increased significantly on 1st day ($P < 0.05$) but reduced significantly on 7th day while it increased obviously on 30th day ($P < 0.05$), however, the density of anaerobic bacteria had no significant difference in different time after $PM_{2.5}$ exposure (**Table 18**).

Group	*n*	Aerobic bacteria	Anaerobic bacteria	Total bacteria
Control	13	461.65 ± 101.23	182.58 ± 23.59	644.23 ± 116.37
1st day	12	539.25 ± 72.91*	260.08 ± 36.54	799.33 ± 63.54*
7th day	13	235.92 ± 24.91	274.81 ± 35.95	510.73 ± 44.15
30th day	14	746.96 ± 64.85*	443.82 ± 49.69	1190.79 ± 123.00*
P		0.038	0.145	0.029

*Compared with no $PM_{2.5}$ exposure, $P < 0.05$.

Table 18. The comparison of bacterial density in different time after $PM_{2.5}$ exposure (CFU/ml, G ± S).

On the basis of TSP and PM_{10}, the impact of $PM_{2.5}$ on respiratory tract micro ecology of rats was mainly focused on. In the research, the density of bacteria and the change of bacterial population were analyzed and the results showed that the detection rate of bacteria and bacterial amount especially aerobic bacteria increased significantly after $PM_{2.5}$ exposure. The density of aerobic bacteria on the oropharynx of rats fluctuated over time on compensated and decompensated states, which increased significantly on 1st day but reduced significantly on 7th day while it increased obviously on 30th day after $PM_{2.5}$ exposure. The total bacterial amount also showed the same trend, the density of most aerobic and anaerobic bacteria was higher than that before $PM_{2.5}$ exposure and some genera were detected after $PM_{2.5}$ exposure. All the results showed that $PM_{2.5}$ could change the microecology on the oropharynx of rats to microecological imbalance. Compared to previous experimental results of PM_{10}, the pathogenic bacteria of group B *Streptococcus* and *Staphylococcus intermedius* were not detected, only 1 case of *S. aureus* was detected after $PM_{2.5}$ exposure. The density of bacteria was higher than no $PM_{2.5}$ exposure, but the bacterial species were less than the previous PM_{10} experimental results, maybe this phenomenon was caused by the less sample size. In addition, some scholars thought that fine particulate matters were easy to deposit in the region of the bronchi and alveoli and possibly entered into blood circulation to damage the microecology of respiratory tract. Present studies have shown that the atmospheric pollutant particles with diameter less than PM_{10} can enter into the respiratory tract directly and keep in the deep lungs which are not easy to be discharged out of the body due to their small diameter without nasal block. $PM_{2.5}$ is also known as inhaled lung particulate matter and it has important effect on atmospheric visibility and air quality, which has attracted worldwide attention. Compared with the coarse atmospheric particulates, the particle size of $PM_{2.5}$ is small and there are a lot of poisonous and harmful substances in rich, moreover, $PM_{2.5}$ can stay for a long time in the

atmosphere and has a long conveying distance, so that it has more influence on the quality of atmospheric environment and human health.

7. Conclusions

Air pollution caused destruction of microecological balance in upper respiratory tract. The amounts of oropharyngeal aerobic and anaerobic bacteria were higher in children from heavily polluted area than those lightly polluted area. The detection rate of conditional pathogenic bacteria in children from heavily polluted area is higher than that in the control area. After 2 years of dynamic analysis, the detection rate of alpha streptococcus in children from heavily polluted area was lower than that in the control area and the difference was significant. Reduction of alpha streptococcus is bound to decrease the protective effect of the pharynx barrier, but increase susceptibility to infection.

Animal model of atmospheric pollutants was used to analyze the flora on the oropharynx of Wistar rats. The results showed that *K. pneumoniae* and *S. aureus* were not detected before air pollutant exposure, however, they were detected after air pollutant exposure and the detection rate of *K. pneumoniae* increased significantly. Thus Streptococcus pneumoniae, *S. aureus* and *K. pneumoniae* were thought as the common pathogens of bacterial pneumonia. The microecological environment on the oropharynx of rats changed after air pollutants exposure, it was easy for pathogens to colonize and cause respiratory infections.

After air pollutants exposure, the density of aerobic bacteria on the oropharynx of rats showed no significant change, however, the quantity of anaerobic bacteria decreased significantly and had significant decrease in trend over time. The average density of bacteria reduced with significant difference over time. The density of bacteroides and eubacterium of anaerobic bacteria were significantly reduced on the oropharynx of rats after air pollutants exposure compared with the results before air pollutants exposure. These results showed that air pollutants led to a decrease in respiratory total biomass and anaerobic bacteria and therefore declined the resistance of anaerobic bacteria, causing dysbacteriosis and colonization of pathogenic bacteria, which strongly proved that the anaerobic bacteria in normal flora played an important role in terms of colonization resistance.

The quantity of aerobic and anaerobic bacteria on the oropharynx of rats had no significant change with air pollutants but fluctuated with the concentration of air pollutants. The quantity of anaerobic bacteria on the oropharynx of rats had a downward trend in a high dose group (**Figure 8**). The density of Neisseria in aerobic bacteria increased significantly in high concentration group, however, the density of Veillonella in anaerobic bacteria reduced significantly. These results showed that the pathogenic and potential pathogenic bacteria were easier to colonize with the increasing concentration of air pollutants, but the quantity of total bacteria had no obvious change, which confirmed that the bacteria on the oropharynx of rats fluctuated within a certain range under certain conditions.

Air pollution could cause ecological imbalance and damage to the micro communities on the respiratory tract which led to the content of normal flora especially anaerobic bacteria decrease; therefore, the pathogenic bacteria colonized easily and caused respiratory diseases.

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References

[1] Gautam S, Yadav A, Tsai CJ, Kumar PA. A review on recent progress in observations, sources, classification and regulations of $PM_{2.5}$ in Asian environments. Environ Sci Pollut Res Int. 2016; 23(21):21165–21175.

[2] Liu Z, Wang Y, Hu B, Ji D, Zhang J, Wu F, Wan X, Wang Y. Source appointment of fine particle number and volume concentration during severe haze pollution in Beijing in January 2013. Environ Sci Pollut Res Int. 2016; 23(7):6845–6860.

[3] Park SU, Lee IH, Joo SJ. Spatial and temporal distributions of aerosol concentrations and depositions in Asia during the year 2010. Sci Total Environ. 2016; 542 (Pt A):210–222.

[4] Paraskevopoulou D, Liakakou E, Gerasopoulos E, Mihalopoulos N. Sources of atmospheric aerosol from long-term measurements (5 years) of chemical composition in Athens, Greece. Sci Total Environ. 2015; 527–528:165–178.

[5] Hu Y. Evaluation on effects of Qingdao air pollution on economic loss of human health. Chin J Public Health. 2003; 19(8):940–941.

[6] Chen S, Li X, Zhou L. Quantitative study by grey system on the latent period of lung cancer induced by air pollutants. Chin J Epidemiol. 2003; 24(3):233–235.

[7] Turner MC, Krewski D, Chen Y, Pope CA 3rd, Gapstur SM, Thun MJ. Radon and nonrespiratory mortality in the American Cancer Society cohort. Am J Epidemiol. 2012; 176(9):808–814.

[8] Moon EK, Son M, Jin YW, Park S, Lee WJ. Variations of lung cancer risk from asbestos exposure: impact on estimation of population attributable fraction. Ind Health. 2013; 1 (1):128–33.

[9] Lu Y, Hu X, Zhao J, Wang P, Qin Y, Wu M. The geographic information system of air pollution and the mortality of lung cancer in Jiangsu Province. China Cancer. 2003; 12 (7):374–377.

[10] Nyberg F, Gustavsson P, Järup L, Bellander T, Berglind N, Jakobsson R, Pershagen. Urban air pollution and lung cancer in Stockholm. Epidemiology. 2000;11(5):487–495.

[11] Ma M, Li S, Jin H, Zhang Y, Xu J, Chen D, Kuimin C, Yuan Z, Xiao C. Characteristics and oxidative stress on rats and traffic policemen of ambient fine particulate matter from Shenyang. Sci Total Environ. 2015; 526:110–115.

[12] Bandowe BA, Meusel H, Huang RJ, Ho K, Cao J, Hoffmann T, Wilcke W. PM2.5-bound oxygenated PAHs, nitro-PAHs and parent-PAHs from the atmosphere of a Chinese megacity: seasonal variation, sources and cancer risk assessment. Sci Total Environ. 2014; 473–474:77–87.

[13] West JJ, Cohen A, Dentener F, Brunekreef B, Zhu T, Armstrong B, Bell ML, Brauer M, Carmichael G, Costa DL, Dockery DW, Kleeman M, Krzyzanowski M, Künzli N, Liousse C, Lung SC, Martin RV, Pöschl U, Pope CA 3rd, Roberts JM, Russell AG, Wiedinmyer C. What we breathe impacts our health: improving understanding of the link between air pollution and health. Environ Sci Technol. 2016; 50(10):4895–4904.

[14] ZEpton MJ, Dawson RD, Brooks WM, Kingham S, Aberkane T, Cavanagh JA, Frampton CM, Hewitt T, Cook JM, McLeod S, McCartin F, Trought K, Brown L. The effect of ambient air pollution on respiratory health of school children: a panel study. Environ Health. 2008; 7:16.

[15] Nidhi, Jayaraman G. Air quality and respiratory health in Delhi. Environ Monit Assess. 2007; 135:313–325.

[16] Maitre A, Bonneterre V, Huillard L, Sabatier P, de Gaudemaris. Impact of urban atmospheric pollution on coronary disease. Eur Heart J. 2006; 27:2275–84.

[17] Basagaña X, Jacquemin B, Karanasiou A, Ostro B, Querol X, Agis D, Alessandrini E, Alguacil J, Artiñano B, Catrambone M, de la Rosa JD, Díaz J, Faustini A, Ferrari S, Forastiere F, Katsouyanni K, Linares C, Perrino C, Ranzi A, Ricciardelli I, Samoli E, Zauli-Sajani S, Sunyer J, Stafoggia M; MED-PARTICLES Study group. Short-term effects of particulate matter constituents on daily hospitalizations and mortality in five South-European cities: results from the MED-PARTICLES project. Environ Int. 2015; 75: 151–158.

[18] Zhang JJ, Cui MM, Fan D, Zhang DS, Lian HX, Yin ZY, Li J. Relationship between haze and acute cardiovascular, cerebrovascular, and respiratory diseases in Beijing. Environ Sci Pollut Res Int. 2015; 22(5):3920–3925.

[19] Curtis L, Rea W, Smith-Willis P, Fenyves E, Pan Y. Adverse health effects of outdoor air pollutants. Environ Int. 2006; 32:815–830.

[20] Villeneuve PJ, Weichenthal SA, Crouse D, Miller AB, To T, Martin RV, van Donkelaar A, Wall C, Burnett RT. Long-term exposure to fine particulate matter air pollution and mortality among Canadian women. Epidemiology. 2015; 26(4):536–45.

[21] Ballester F, Tenias JM, Perez-Hoyos S. Air pollution and emergency hospital admissions for cardiovascular diseases in Valencia, Spain. J Epidemiol Community Health. 2001; 55:57–65.

http://dx.doi.org/10.5772/66039

[22] De Steenhuijsen Piters WA, Sanders EA, Bogaert D. The role of the local microbial ecosystem in respiratory health and disease. Philos Trans R Soc Lond B Biol Sci. 2015; 370(1675):1–13.

[23] Martin C, Burgel PR, Lepage P, Andréjak C, de Blic J, Bourdin A, Brouard J, Chanez P, Dalphin JC, Deslée G, Deschildre A, Gosset P, Touqui L, Dusser D. Host-microbe interactions in distal airways: relevance to chronic airway diseases. Eur Respir Rev. 2015; 24 (135):78–91.

[24] Brugman S, Perdijk O, van Neerven RJ, Savelkoul HF. Arch Immunol Ther Exp (Warsz). 2015; 63(4):251–268.

[25] Xiao C, Li S, Zhou W, Shang D, Zhao S, Zhu X, Chen K, Wang R. The effect of air pollutants on the microecology of the respiratory tract of rats. Environ Toxicol Pharmacol. 2013; 36:588–594.

Chapter 5

The Air Quality Influences of Vehicular Traffic Emissions

Sailesh N. Behera and Rajasekhar Balasubramanian

Additional information is available at the end of the chapter

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Abstract

The number of automobiles has been steadily increasing in cities as a consequence of rapid urbanization and economic growth. It has been widely reported that vehicular emissions are strongly correlated with the level of urban air pollution. The major primary air pollutants that are linked to direct emissions from on-road vehicles include soot (black carbon), carbon monoxide (CO), and nitric oxide (NO). Human exposure to these air pollutants is of health concern. Therefore, it is important to investigate air pollutants of traffic origin (e.g., BC, CO, and NO) in ambient air at different locations of cities and to assess the effects of vehicles on the urban air quality. With this goal in mind, we carried a systematic study in Singapore (the fourth most densely populated country in the world) with concurrent measurements of BC, NO, and CO in ambient air at four different locations having variations in traffic flows and meteorology. We then assessed the relationship between traffic flows and prevailing levels of the three air pollutants, and studied the association of these air pollutants among each other and with diverse meteorological conditions. The major outcomes of the study are discussed.

Keywords: urban air quality, traffic emissions, aerosol black carbon, carbon monoxide, nitric oxide, diurnal variations

1. Introduction

Aerosol black carbon (BC) is a widespread environmental pollutant, which is generated from combustion processes of carbonaceous materials at high temperature. Being the principal light absorbing aerosol species with specific absorption coefficients ranging from 11 to 12 $m^2 g^{-1}$ at 650 nm, BC absorbs radiation that lowers the single scattering albedo [1, 2]. As a result, the amount of reflected radiation is reduced, and the radiation absorbed by the atmosphere is increased. Due to the nature of higher porosity, BC adsorbs other species from the vapor phase,

especially organics that are potentially mutagens or carcinogens [3, 4]. For example, BC adsorbs polycyclic aromatic hydrocarbons (PAHs) with four rings or more, those are carcinogenic in nature [5, 6]. In the environment, BC is inhaled frequently by humans that can be deposited in the lungs or other airways causing severe health effects on a long- or short-term basis [7, 8]. In addition, BC provides surfaces that may catalytically promote certain other reactions in the atmosphere [9]. This can be seen in the environment with higher levels of BC that affect ozone (O_3) and nitrogen oxide (NO_x) concentrations due to heterogeneous destruction of O_3 molecules on particles [10]. For example, Dasch and Cadle [11] observed that oxidation of sulfur dioxide (SO_2) is also catalyzed by BC aerosol.

During the past two decades, research on BC has been increasing rapidly due to its role on local air quality causing visibility problems and adverse health effects, regional air quality affecting cloud microphysics, and global climate change due to its positive radiative effects [12–18]. Previous studies have also reported that BC of local origin can be transported over a long distance due to its long lifetime (order of several days to several weeks depending on meteorology) in the atmosphere [19]. Major anthropogenic sources of BC include on-road vehicles, domestic heating, industrial activities, and refuse burning, among which vehicular emissions are particularly dominant [20–22].

Human exposure to ambient BC, carbon monoxide (CO), and nitric oxide (NO) is of concern in urban environments due to a high population of vehicles [23, 24]. Such higher exposure to BC, CO, and NO causes several health effects, including myocardial infarction and pneumonia, elevated inflammatory markers of cardiovascular disease, diminished heart rate variability, and ventricular tachyarrhythmias. [23–27].

In general, two types of studies are performed to characterize the vehicular emissions, i.e., direct tests on engines and experiments in the ambient air. In the past, several studies focused on engine particle emission through engine test sites or chassis dynamotors [28–30]. However, to get the real atmospheric particulate characterization, it is essential to perform field measurements. Although some studies [31, 32] were performed under stationary measurements in recent years, these studies were, however, confined to a particular measurement site. Therefore, ambient characterization with on-road measurements under real moving traffic conditions are the best experiments that can provide better scientific knowledge on traffic emissions.

In the past, several detailed studies were undertaken in Singapore to assess the status of air quality and particulate matter characterization [33, 34]. However, these studies did not address the effects of vehicles on aerosol BC and association of NO and CO with BC. The latest study by Kalaiarasan et al. [35] investigated traffic-generated airborne particles in naturally ventilated multistorey residential buildings of Singapore and the potential health risk at different vertical heights. To get a clear picture on the role of vehicles on ambient air quality, it is essential to get the status of air quality at different urban locations with varying traffic loads. The status of ambient air quality at human breathing levels (2 m above ground) can be augmented to health risk models to assess the risk due to location of a particular area.

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Some limited studies have been carried out in the urban areas of developed countries recently to assess the effects of vehicles on ambient air quality [36–40]. These studies monitored the air quality status (BC, CO, and NO) near locations of heavy and low traffic flows for comparative assessment of air quality. However, no such studies were undertaken in tropical areas within Southeast Asia. We undertook a preliminary study in Singapore for the first time. Being a heavily motorized country, the traffic system in Singapore (the fourth most densely populated country in the world) is of particular interest in the region of Asia Pacific from air pollution point of view [35, 41, 42]. In connection to health problems, bronchial asthma is one of the common respiratory disorders in Singapore and is commonly observed that about 1 of 5 children is asthmatics [35]. Therefore, there is a strong need to investigate traffic-generated pollutants (BC, CO, and NO) in ambient air at different locations and to assess the effects of vehicles on the ambient air quality.

The present study was designed with the following objectives: (1) measurements of levels of BC, NO, and CO in ambient air at four different locations having variations in traffic flows and meteorology, (2) assessment of the relationship between traffic flows and prevailing levels of the three air pollutants, and (3) examination of the association of these air pollutants among each other and with the meteorology.

2. Materials and methods

2.1. Characteristics of study area

The study was conducted in Singapore, which is located at the tip of the Malayan Peninsula (1°09'N to 1°29'N and 103°36'E to 104°25'E) with areas of about 699 km^2 and population of about 4.7 million [35]. Singapore's geographical location and maritime exposure, uniform temperature, pressure, and high humidity characterize its climate. The temperature ranges from 32°C for a high and 24°C for a low with a daily mean humidity of 84.4%. Singapore is the fourth most densely populated country in the world with a population density of 6369 persons/km^2 as at 2008 and can be considered as a land scarce country [35]. Early to 1995 and after independence, Singapore's public transport systems were managed separately, but in 1995 most land transport functions were brought together by the Land Transport Authority (LTA) [43]. With the plans and policies for implementation to provide a world class transport system, since 1998 onwards Singapore is known to be a heavily motorized country with the most developed road system in the Asia-Pacific region [44].

In Singapore, the number of automobiles has been steadily increasing as a consequence of rapid urbanization and economic growth. As an example, based on LTA data, the total number of vehicles in Singapore were 688,811 in 1999 and 956,704 in 2011 (LTA, 1999; LTA, 2011) [45, 46]. In the present study, we have studied the effects of vehicles on local air quality at the human breathing level in 2010. We give the vehicle details in the break-up of different categories for 2009, as described below. As per the data of LTA (1999), cars accounted for 61% of the total vehicle population. These cars are all gasoline driven and a majority of them have catalytic converters. Goods vehicles, running on gasoline and diesel, accounted for

18%. The gasoline driven two wheelers, i.e., motorcycles and scooters accounted for approximately 16% of the vehicle population. Buses and taxis accounted for 2% and 3%, respectively, and these vehicles run with diesel fuel. Although the percentage composition of the buses and taxis in the total population is low, the overall vehicle kilometers traveled by them on road are quite high.

Singapore's strategy for reducing air pollution from vehicles focuses on two main aspects: improving the fuel quality to reduce emissions and management of traffic to control the increase in the number of vehicles [47]. The high expense of owning and operating a vehicle in Singapore has effectively controlled the growth of vehicles. In addition, car owners pay annual road taxes based on the engine capacity of their vehicles. "Singapore has also introduced a full-fledged use of the Electronic Road Pricing System, commonly known as ERP, for vehicles on major expressways. According to this system, vehicles traveling on certain expressways at certain time periods have to pay a toll" (LTA, 2011). The Singapore government introduced a quota system known as Certificate of Entitlement (COE), in January 1990. Under this system, a vehicle entitlement is valid for 10 years from the date of registration of the vehicle. If the owner wishes to continue using the vehicle on expiration of the vehicle entitlement, he has to pay a revalidation fee.

2.2. Details of the measurement program

To achieve the objectives of this study, field measurements were carried out at four different sites: (i) road side in the National University of Singapore campus (NUS), (ii) road side in the vicinity of expressway (EXW), (iii) central express tunnel (CTE), and (iv) a remote site near coastal region (RME). These measurement sites were selected on the basis of intensity of traffic flow. The parameters of measurement included were air quality (BC, CO, and NO), meteorology (wind speed and wind direction), and traffic volumes (car, bus, taxi, motorbike, and others). **Table 1** presents the details of the field measurement program with information of the monitoring sites.

Measurement site	Site code	Characteristics of sites	Details of field measurement			
			Air quality	Meteorology	Traffic survey	Duration of study
Road side in National University of Singapore campus	NUS	Medium traffic volume	BC, CO, NO	Wind speed, wind direction	Car, bus, taxi, bike, etc.	13–16 May, 2010
Road side of expressway	EXW	Higher traffic volume	BC, CO, NO	Wind speed, wind direction	Car, bus, taxi, bike, etc.	22–23 July, 2010
Central express (CTE) tunnel	CTE	Higher traffic volume	BC, CO, NO	Wind speed, wind direction	Car, bus, taxi, bike, etc.	02–04 June, 2010
Remote site near coastal region	RME	Negligible traffic influence	BC, CO, NO	Wind speed, wind direction	Car, bus, taxi, bike, etc.	05 May, 2010

Table 1. Monitoring sites and details of the measurement program.

To get insights into the behavior of air pollution patterns during weekdays and weekends, measurements at the NUS site were conducted from Thursday to Sunday. Automated equip-

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ment was maintained in an air conditioned mobile enclosure with the sampling tubes protruding out from the window of the van. A video camera was used to capture the traffic flows and to characterize the composition of the vehicles traveling at each measurement site. In the tunnel site, CTE, the monitoring point was at the center of the tunnel, so that the field measurements could be carried out under smooth flowing traffic conditions. The wind speed and wind direction were measured with digital wind vane and anemometer, respectively.

2.3. Measurements of BC, CO, and NO

In this study, field equipment was placed at a distance of 5.0 m from the road at all measurement sites except the tunnel site, CTE. Air pollutants were measured at approximately 2.0 m height above the ground except the NUS site. Measurements at NUS were done on the rooftop of the Atmospheric Research Station, which is about 15 m above the ground level.

For the measurement of BC in ambient air, the optical and thermal techniques are widely used. In the past, various research groups used these techniques to estimate the BC concentration [48]. However, aethalometer is the only equipment that is used for the real-time measurement of BC [49]. To get measurement information with short-term peaks in quasi-real time, aethalometer is the best option. The model AE–20UV Aethalometer (Magee Scientific Company, Berkeley, CA) was used to measure BC in real time for this study. The sample flow rate was 5.0 L/min, and sampling time base was 5 min. The principle of measurement method is based on the optical attenuation of light by particles collected on the quartz fiber filter, which is summarized as follows: (i) light from a stabilized lamp is split and passes the sampling portion and a reference blank portion of the filter, (ii) the intensity of the transmitted light is determined by the light sensors placed after the filter, (iii) thus, changes of light transmitted through the sampling portion of the filter due to collected absorbing aerosols are detected and recorded as changes in optical attenuation, and (iv) finally, assuming a constant specific attenuation cross-section of BC, the concentration of BC is calculated. We used the original laboratory calibration factor of 17 m^2/g, as recommended by the manufacturer.

CO was measured by a real-time CO analyzer, Thermo Environmental Instruments (TEI), Model 48C. The instrument is based on the principle that CO absorbs infrared radiation at the wavelength of 4.6 μm. The model 42C, NO–NO_2–NO_X analyzer from TEI, was used to measure NO based on the principle of chemiluminescence. It is based on the principle that NO and O_3 react to produce a characteristic luminescence with intensity linearly proportional to the NO concentration.

3. Results and discussions

3.1. Overall results

Table 2 presents the overall results of the ambient levels of BC, CO, and NO at four measurement sites. On the basis of 1 h observations, the mean BC varied as the lowest at the RME site (1.7 $\mu g/m^3$) to the highest at the CTE site (45.6 $\mu g/m^3$). Similarly, CO and NO varied from 336.3

ppbv (parts per billion by volume) at the RME site to 8322.4 ppbv at the CTE site and 60.0 ppbv at the NUS site to 100.2 ppbv at the EXE site, respectively. Concurrent measurements of NO at CTE and RME were not possible due to instrumental problems. From the results obtained at the four sites, it can be clearly seen that the tunnel, CTE, experienced the highest air pollution followed by the expressway site, EXW, and the site with the least influence of vehicles (RME) experienced the lowest air pollution. Therefore, the contribution of vehicles to the ambient levels of these air pollutants is significant. The highest levels of BC and CO measured at the CTE site can be attributed to the high traffic flow of vehicles of different types with reduced ventilation as well as little homogeneous mixing of air pollutants inside the tunnel. The relation between various types of the volume of vehicles and the resulting levels of air pollutants during the measurement period can provide better insights into the role of vehicles on the profiles of the measured concentrations of air pollutants. This aspect of the study is discussed in a subsequent section.

Measurement site*	**BC (μg/m³)**			**CO (ppbv)**			**NO (ppbv)**		
	Mean	**SD**	***N***	**Mean**	**SD**	***N***	**Mean**	**SD**	***N***
NUS	6.2	3.8	94	582.9	404.3	95	60.0	58.3	87
EXW	7.2	1.9	45	1384.8	325.4	46	100.2	37.1	47
CTE	45.6	9.6	14	8322.4	886.6	15	NA	NA	NA
RME	1.7	0.5	23	336 .3	162.5	24	NA	NA	NA

*Experimental results are based on average of 1 h observations.
SD: standard deviation; *N*: number of observations; NA: not applicable.

Table 2. Overall results of BC, CO, and NO during the measurement period.

To compare the results obtained from this study with those from other appropriate studies reported in the literature, **Table 3** presents the relevant data on BC, CO, and NO. In general, the ambient concentrations of BC, CO, and NO in Singapore are comparable to those reported for other urban areas of the world.

Location	**BC (μg/m³)**	**CO (ppbv)**	**NO (ppbv)**	**Particular of site**	**Reference**
Essen East, Germany	NA	1 921.4	60.0	Urban	[50]
Düsseldorf–Mörsenbroich, Germany	NA	2707.4	111.2	Urban	[50]
Helsinki, Finland	1.5	NA	NA	Sub Urban	[21]
Los Angeles, USA	4.4	230	NA	Urban	[38]
Toronto, Canada	NA	NA	70.5	Urban	[37]
Aachen, Germany	9.4	NA	NA	Urban	[36]
Aachen, Germany	1.5	NA	NA	Rural	[36]
Barcelona, Spain	3.6	NA	14.2	Urban	[40]

NA: not available.

Table 3. Ambient concentrations of BC, CO, and NO reported for different parts of the world.

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3.2. Diurnal variations of BC, CO, and NO

To get a better insight into the diurnal variations of BC, CO, and NO, the measurement data obtained for every 5-min interval observations were averaged during all hours of the day. In this section, we have considered two measurement sites (NUS and EXW) for examining diurnal variations, because of the similar pollution status of BC at these two sites. However, the tunnel site, EXW, and the remote site, RME, experienced different pollution status and traffic flows (**Table 2**). Therefore, in this paper, we discuss the pollution and traffic patterns at the CTE site in a separate section. The site, RME, did not show any specific trends of air pollution. **Figure 1** shows the diurnal variations of BC, CO, and NO with hourly average observations at NUS. BC showed pronounced peaks in the morning traffic hours (7:00–11:00) with a maximum concentration of 8.9 μg/m^3 at 9:00, indicating that vehicular emission is an important source of BC (**Figure 1**). Similarly, CO and NO showed pronounced peaks in the morning traffic hours (7:00–

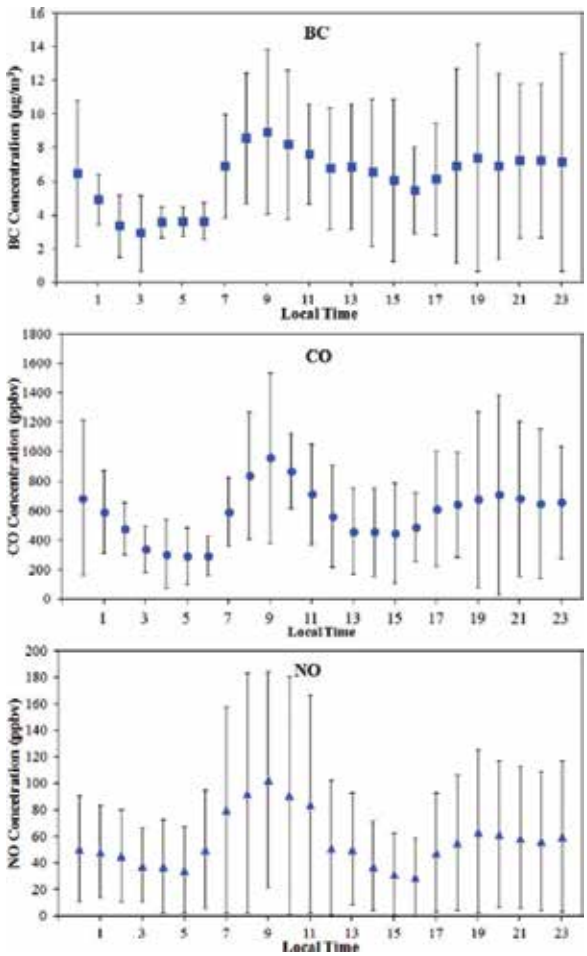

Figure 1. Diurnal variations of concentrations BC, CO, and NO at the NUS site. The error bars represent the corresponding standard deviations.

11:00) with maximum concentrations of 959.7 and 102.8 ppbv, respectively, at 9:00, indicating that vehicular emission is an important source of CO and NO (**Figure 1**).

The concentrations of BC, CO, and NO remained low in the afternoon. There were no significant evening rush hour peaks, which could be probably due to unstable atmospheric conditions induced by long hours of sunshine and hence improved dispersion of traffic emissions during the afternoon. Moreover, being a tropical country, the solar radiation in Singapore remains high even during the evening rush hour, thereby resulting in the enhanced vertical mixing of air pollutants. This leads to dilution of BC, CO, and NO concentrations. Relatively higher concentrations in the early morning could be attributed to low mixing heights and reduced dispersive conditions. Being a coastal city, Singapore's air quality is also influenced by land breeze and sea breeze. Land breeze blowing during early morning brings in contaminated air, whereas the sea breeze that blows during afternoon brings in relatively clean air. In addition, the traffic flow of heavy-duty/utility vehicles is relatively higher in the morning than during other periods of the day.

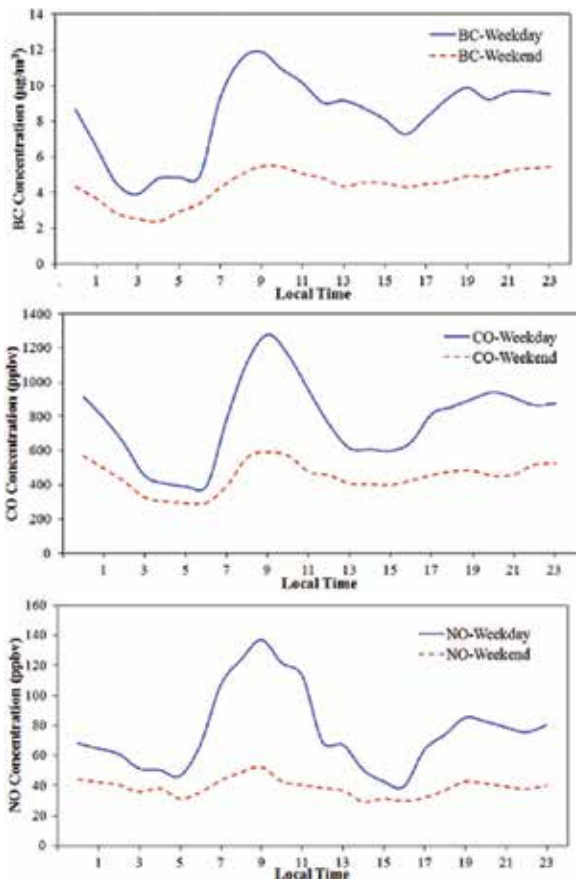

Figure 2. Diurnal variations of concentrations of BC, CO, and NO at the NUS site during weekday and weekend.

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In the case of the expressway site, EXW, it was observed that the traffic volume remained high on the expressway even in the noon; there was no decrease in the concentration in the afternoon in contrast to the diurnal profiles of BC, CO, and NO concentrations in the ambient air at NUS. Furthermore, the effect of dispersion on the concentrations of air pollutants does not play a significant role at this site because the measurements were performed at the ground level.

To examine the effect of rush hour traffic, the mean diurnal variations of BC, CO, and NO were divided into weekdays (Thursday and Friday) and weekends (Saturday and Sunday). This is to be noted that the measurement at the NUS site was done for 4 days with two weekdays and two weekends. **Figure 2** shows the diurnal trends of these pollutants during weekdays and weekends. The morning peak, which is usually seen between 7:00 and 11:00 on weekdays, does not exist on weekends. The peak concentrations on weekdays were 11.9 μg/m^3, 1279.6 ppbv, and 137.1 ppbv for BC, CO, and NO, respectively. However, during weekends, the peak values seemed to be less than half of respective peaks of weekdays (the peaks on weekend were 5.5 μg/m^3, 590.8 ppbv, and 52.1 ppbv for BC, CO, and NO, respectively). From these observations, it can be confirmed that vehicles play a major role in making significant contributions to the prevailing air pollution levels in Singapore.

In the eastern United States, at Uniontown, PA, the diurnal variation of BC concentrations was measured in 1990 [49]. A clear peak was observed between 6:00 and 10:00. There was no significant evening rush hour peak, but instead slightly elevated concentrations were observed from 20:00 to 23:00. Pakkanen et al. [21] analyzed in Helinski reported that the peak in hourly average BC concentrations was observed during the morning and evening rush hours on weekdays. Weekends showed relatively stable hourly average concentrations. Therefore, based on these comparisons, it appeared that our observations on BC due to traffic emissions were consistent with those from the other studies reported in the literature.

3.3. Measurements in the tunnel site

The field measurements at CTE were conducted under varying traffic composition to examine the effect of traffic volume and the composition on the levels of BC and CO measured in the traffic tunnel. Air sampling and traffic surveys were performed on 2 days from 13:40 to 19:25 on day 1 and from 9:45 to 16:10 on day 2. The average traffic count on the first day was 3900 vehicles per hour, and the vehicle fleet comprised 8% motorbikes, 3% heavy-duty vehicles, 24% pickups and vans, 49% cars, 1% buses, and 15% taxis. On the second day, the traffic count was around 4400 vehicles per hour, almost 10% higher as compared to the first day. The traffic composition was similar to the first day consisting of 6% motorbikes, 3% heavy-duty vehicles, 24% pickups and vans, 48% cars, 1% buses, and 18% taxis. Heavy-duty vehicles, pickups and vans, buses, and taxis are driven by diesel fuel, whereas the fuel used in motorbikes and cars are with gasoline. Based on the classification by fuel types, the fraction of diesel-driven vehicles on the first day was 0.43 and on the second day was 0.46.

The concentration of BC was 40.9 μg/m^3 on the first day and 49.9 μg/m^3 on the second day. The concentration of CO was 8394.5 ppbv on the first day and 8247.3 ppbv on the second day. Average concentrations of the air pollutants at different sampling locations are also shown in **Table 2**. As can be seen from **Table 2**, the concentration of air pollutants in the CTE was

observed to be much higher than those at other sampling sites. Such higher concentrations at CTE site could be due to a combination of three factors: (1) measurements in the tunnel were performed very close to the emission source (vehicles), (2) there was a substantially high volume of traffic flow in the tunnel, and (3) there was a limited vertical dispersion of BC and CO in the confined environment in the tunnel.

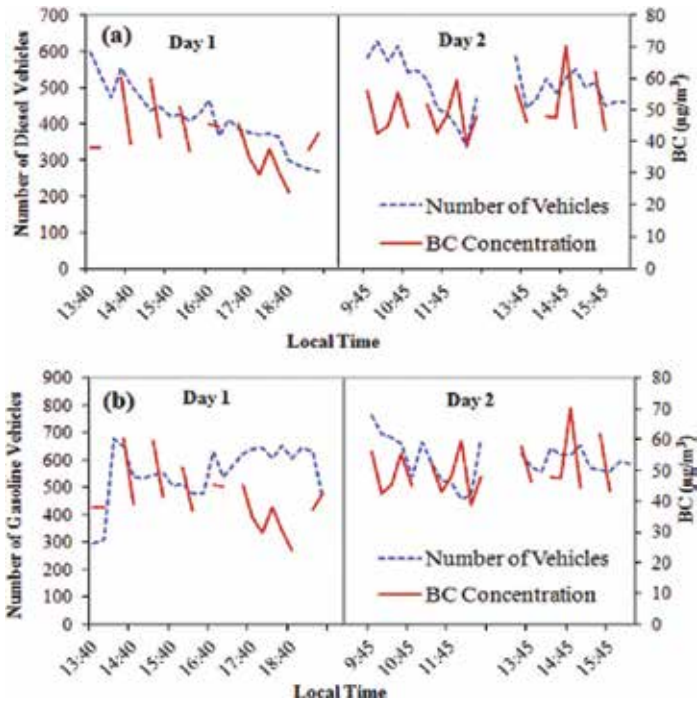

Figure 3. Variations of BC concentration with the number vehicles: (a) diesel-driven vehicles, and (b) gasoline-driven vehicles.

Figures 3(a) and **(b)** show the variations of BC with the total number of diesel-driven vehicles and the total number of gasoline-driven vehicles, respectively, for 2 days. It should be noted that the concentration plots were made based on the observations of 5-min interval. The concentration of BC could not be continuously obtained in the tunnel during these 2 days as the instrument was automatically set in the calibration mode for every hour. Hence, the variations of BC in **Figures 3(a)** and **(b)** do not show their continuous profiles. The increase in the frequency of calibration is especially important as the instrument tends to be saturated with such a high concentration of BC in ambient air of the tunnel. The variations of CO concentration with traffic are shown in **Figures 4(a)** and **(b)**.

The variation in the concentration of BC followed closely the trend in percentage of diesel-driven vehicles. However, such a similarity in the trends of concentration of BC and gasoline vehicles was not seen. For example, on the first day of the air sampling in the tunnel, there was a gradual decrease in the number of diesel-driven vehicle between 16:40 and 19:40, which was

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accompanied by a similar decrease in the concentration of BC. On the other hand, the number of gasoline vehicles remained almost the same. The link between the number of diesel-driven vehicles and the corresponding change in the concentration of BC is further strengthened by the variability in the concentration of CO. In other words, the concentration of CO follows a trend very similar to the number of gasoline vehicles, which is in contrast to the pattern of BC.

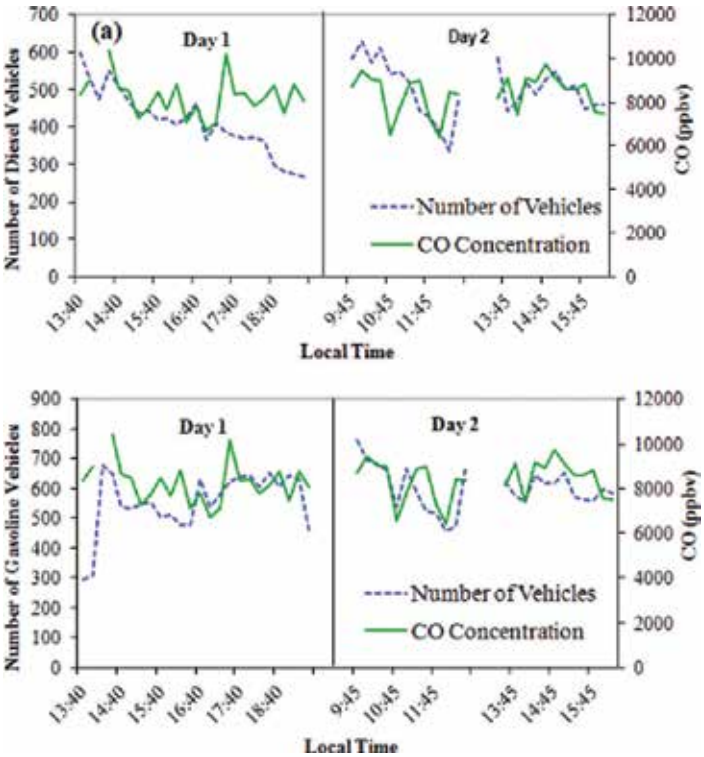

Figure 4. Variations of CO concentration with the number of vehicles: (a) diesel-driven vehicles, and (b) gasoline-driven vehicles.

This finding strongly suggests that diesel-driven vehicles emit much higher levels of BC than gasoline-driven vehicles. This observation is consistent with the previous studies, which have indicated that BC is more abundant in heavy-duty diesel-fuelled exhaust than in light-duty gasoline-fuelled vehicle exhaust [51, 52]. The amount of BC emitted is highly variable, and depends on the vehicle condition, its age, quality of the fuel used, maintenance, the speed of vehicles, and the operating modes of drivers. Miguel et al. [52] did the measurement of CO and BC in two bores of the Caldecott tunnel in California: one bore was influenced by heavy-duty diesel truck emissions; a second bore was reserved for light-duty vehicles. Miguel et al. [52] found that concentration of BC in truck-influenced bore was higher by a factor of 5 in spite of higher traffic volume in other bore. The concentration of CO was higher in the bore dominated by light duty vehicles. According to Miguel et al. [52] light-duty gasoline-driven vehicles and heavy-duty diesel trucks, emitted, respectively, 30 ± 2 mg and 1440 ± 160 mg of

fine BC particles per kg of fuel burned. Gray and Cass [53] estimated that diesel-driven vehicles were responsible for 60% of total BC emissions. Steiner et al. [54] showed that the contribution of BC to total suspended particulate matter (TSP) emitted from a spark ignition engine is 11%. On the other hand, the contribution of BC to TSP in the case of particles emitted from diesel engine ranges from 50% to 80%.

3.4. Correlation of BC with CO and NO

From the time-series plots of BC and CO with vehicles (in Section 3.2), it has been observed that the traffic flow makes a major contribution to the existing levels of these air pollutants. To further confirm this trend, we reviewed the relevant literature on studies done in various parts of the world. From the literature review, it could be concluded that vehicular emissions are strongly correlated with the levels of CO and NO under normal conditions and these pollutants are indicators of traffic emissions in the urban environment [55, 56]. From the estimation of USEPA [57], it is clear that transportation sources were responsible for nearly 72% of total CO emissions, 40% of NO_x, and 31.5% of hydrocarbons in the United States in 1991. In the UK west Midlands conurbation, of which Birmingham is the major city, 98% of CO and 85% of NO_x emissions arise from road traffic. In Australia, the local transport was responsible for up to 63% of NO_x emissions and 95% of CO emissions. In Mexico City, traffic accounts for 99% of the CO and 70% of NO_x [58].

Therefore, to provide further evidence that BC is influenced by traffic flows and to estimate the contribution of traffic flows toward the total ambient levels of BC, the statistical correlation of BC with CO and NO were examined. In this interpretation, we considered the data from the sites of NUS and EXW as we made observations of air pollutants in the open ambient air at

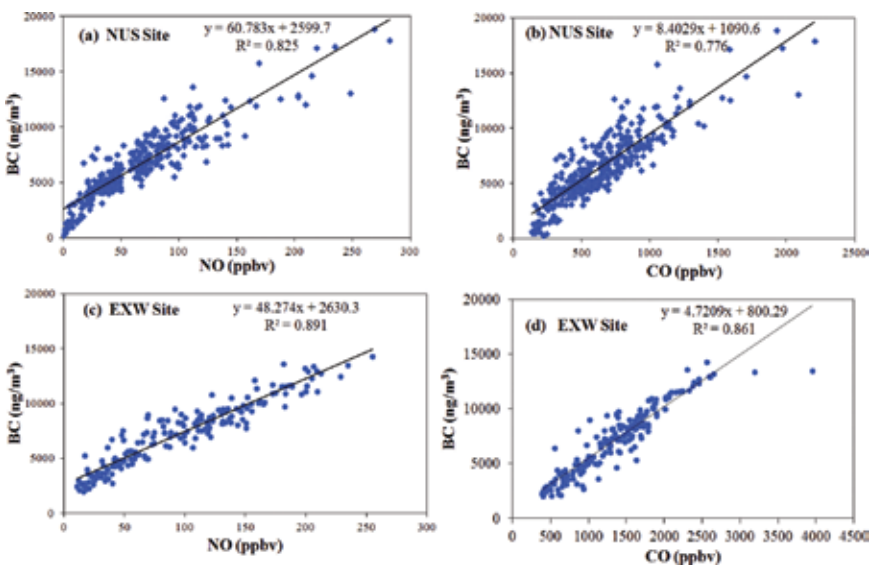

Figure 5. Correlation plots of BC with NO and CO at the NUS and EXW sites: (a) BC vs. NO at NUS, (b) BC vs. CO at NUS, (c) BC vs. NO at EXW, and (d) BC vs. CO at EXW.

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these measurement sites with substantial traffic volumes. Linear regressions of 15 min observations of BC versus NO and BC versus CO were performed from the data of these two sites individually. High correlations of BC with NO ($R^2 = 0.825$, $n = 343$, $p < 0.01$) and with CO ($R^2 = 0.776$, $n = 352$, $p < 0.001$) were observed at NUS (**Figures 5a** and **b**). Similarly, the EXW site also experienced high correlations of BC with NO ($R^2 = 0.891$, $n = 183$, $p < 0.001$) and with CO ($R^2 = 0.861$, $n = 181$, $p < 0.001$) as shown in **Figures 5(c)** and **(d)**, respectively. These high correlations suggest a strong association between vehicular emissions and levels of BC. With these highly significant correlations, we have attempted to assess the contribution of vehicles to the BC concentration in the ambient air using the linear regression equations obtained from the plots of BC and CO in **Figures 5(b)** and **(d)** at the sites, NUS and EXW.

A linear regression of BC with CO yielded the following equations at NUS (Eq. 1) and EXW (Eq. 2):

$$BC = 8.4 \times CO + 1090.6 \tag{1}$$

$$BC = 4.7 \times CO + 800.2 \tag{2}$$

where BC is in ng/m^3 and CO is in ppbv. To estimate the contribution of BC from vehicular sources, we assumed that 90% of the total emission of CO is traffic generated. Therefore, the background CO in the absence of traffic would be 58.3 and 138.5 ppbv at NUS and EXW, respectively (i.e., 10% of 1 h average CO concentration; see **Table 2**). For these values of CO, Eqs. (1) and (2) gave an estimate of BC concentration of 1.6 and 1.4 μg/m^3 at NUS and EXW, respectively, in the absence of traffic. The average ambient concentrations of BC at NUS and EXW were 6.2 and 7.2 μg/m^3 (**Table 2**). Thus, the average concentrations of BC due to traffic were 4.6 and 5.8 μg/m^3 at NUS and EXW. Finally, the contributions of vehicular traffic to the total BC concentration were estimated to be 74% and 80% at NUS and EXW, respectively. While EXW is close to the road, NUS is relatively far from the road. Consequently, the roadside environment involving human exposure to higher BC emissions could cause more health effects. It should be noted that the approach to estimate the traffic contribution toward air pollutants through the regression analysis has been used by other investigators. For example, Lim et al. [59] estimated the contribution of traffic to PAHs concentration using a regression equation.

3.5. Correlation of vehicles with BC, CO, and NO

As described in Section 3.3, the traffic flow showed a significant influence on the levels of BC and CO. It is also evident from the past studies [55, 56] that vehicles have a dominant influence on the ambient levels of NO concentration. Therefore, to examine the role of vehicles on the levels of BC, CO, and NO, correlation coefficients were estimated between various species using Minitab 15 English. In this approach, we classified the entire observations into two parts with respect to the site characteristics (results are presented in **Tables 4a** and **b**). It should be noted that the observations were made in the ambient air at the sites, NUS and EXW. However,

in the case of CTE, the observations were made inside the tunnel. In the correlation analysis, all pollutant concentrations and their corresponding traffic numbers (diesel-driven and gasoline-driven) were included at both NUS and EXW sites. The correlation analysis at CTE was made separately to ensure that the assessment of correlation was based on traffic flows only.

	BC	**CO**	**NO**	N_d	N_p
BC	1.00				
CO	**0.85**^{III}	1.00			
NO	**0.92**^{III}	0.21	1.00		
N_d	**0.84**^{III}	**0.29**^{II}	**0.85**^{III}	1.00	
N_p	**0.27**^{II}	**0.82**^{III}	**0.88**^{III}	**0.49**^{III}	1.00

N_d: number of diesel-driven vehicles; N_p: number of gasoline-driven vehicles.
Bold marks are statistically significant. Superscripts II and III denote that correlation is significant at $P < 0.01$ and $P < 0.001$.

Table 4(a). Correlation matrix for NUS and EXW sites.

	BC	**CO**	N_d	N_p
BC	1.00			
CO	**0.91**^{III}	1.00		
N_d	**0.92**^{III}	**0.56**^{II}	1.00	
N_p	**0.53**^{II}	**0.94**^{III}	**72**^{III}	1.00

N_d: number of diesel-driven vehicles; N_p: number of gasoline-driven vehicles.
Bold marks are statistically significant. Superscripts II and III denote that correlation is significant at $P < 0.01$ and $P < 0.001$.

Table 4(b). Correlation matrix for CTE site.

Table 4(a) presents the correlation matrix of the sites of NUS and EXW. The observations from this correlation analysis are summarized as follows: (1) BC concentration showed significant correlations with CO and NO, (2) BC concentration showed strong and significant correlations with the number of diesel-driven vehicles monitored during that period, (3) CO concentrations showed a strong and significant correlation with the number of gasoline-driven vehicles observed during that period, and (4) NO showed strong and significant correlations with both gasoline-driven and diesel-driven vehicles. Hence, it could be concluded that the BC concentrations were mainly influenced by the diesel-driven vehicles. However, the CO concentrations were predominantly influenced by the gasoline-driven vehicles.

Table 4(b) presents the correlation matrix of the site, CTE. It should be noted that the parameter, NO concentration, was not included due to unavailability of observational data. Overall, it can be seen that the concentrations of BC and CO were influenced by diesel-driven and gasoline-driven vehicles, respectively.

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3.6. Effect of wind speed and wind direction on pollutants

In general, the variability of pollutant concentration levels strongly depends on the origin of the air masses arriving at the sampling site and the concentration of pollutants in the ambient air is influenced by the direction from which wind blows. In this study, we did not find any specific trends between wind directions and the concentrations of air pollutants at the measurement sites. For example, at the NUS site, we observed that winds mainly blew from the Northeast during the morning rush hour. However, on the following day, there was a change in the wind direction. To assess the role of meteorology in the variation of ambient levels of BC, CO, and NO, we estimated correlation coefficients between pollutant levels, wind speed, and wind direction using Minitab 15 English. As explained in Section 3.5, we considered the NUS and EXW sites in a single platform for correlation analysis and further interpretation. **Table 5** presents the correlation matrix of BC, CO, NO, wind speed, and wind direction. It was observed that the wind direction had lower correlations with the air pollutant concentrations during the measurement period, indicating that there was little change of air pollution levels with the change in wind direction. The reason for such observations could be due to the fact that winds blowing from the South bring in clean marine air resulting in relatively low concentrations of air pollutants; however, the northerly winds originated from land air mass. Being influenced by human activities, the land air mass led to the enhancement in the level of air pollutants.

	BC	CO	NO	WS	WD
BC	1.00				
CO	**0.85**^{III}	1.00			
NO	**0.92**^{III}	0.21	1.00		
WS	**-0.68**^{III}	**-0.72**^{III}	**-0.67**^{III}	1.00	
WD	**0.14**^{I}	**0.16**^{I}	**0.15**^{I}	**0.13**^{I}	1.00

WS: wind speed; WD: wind direction.
Bold marks are statistically significant. Superscripts I and III denote that correlation is significant at $P < 0.05$ and $P < 0.001$.

Table 5. Correlation matrix meant for meteorology at NUS and EXW sites.

In general, wind speed is considered as one of the important parameters affecting the concentration of air pollutants. It determines the time taken to travel from a source to a given receptor and the total area over which the air pollutant would be dispersed. The wind speed showed strong and significant negative correlations with BC, CO, and NO. Higher wind speeds result in better mixing of air pollutants, causing their dilution. As the wind speed gets lower, air pollutants tend to get accumulated due to poor dispersion of air. Therefore, the concentration of BC < CO, and NO increased at low wind speeds. Harrison et al. [7] compared the daily mean elemental carbon concentration and wind speeds at Birmingham, and found quite a similar relationship as the one observed in this study.

4. Conclusions

The present study investigated the on-road emissions of BC, CO, and NO under real moving traffic conditions at four different measurement sites (NUS, EXW, CTE, and RME) of Singapore, an urban environment in the Asia-pacific region. On the basis of 1 h observations, the mean BC varied from the lowest value at the RME site (1.7 μg/m^3) to the highest one at the CTE site (45.6 μg/m^3). Similarly, CO and NO varied from 336.3 ppbv at the RME site to 8322.4 ppbv at the CTE site and 60.0 ppbv at the NUS site to 100.2 ppbv at the EXE site, respectively. At the NUS site, BC showed pronounced peaks in the morning traffic hours (7:00–11:00) with a maximum of 8.9 μg/m^3 at 9:00, and CO and NO showed pronounced peaks in the morning traffic hours (7:00–11:00) with maximum values of 959.7 and 102.8 ppbv, respectively, at 9:00, indicating that vehicular emission is an important source of BC, CO, and NO. The concentration of air pollutants in the tunnel site, CTE, was observed to be much higher than those at other measurement sites. The study revealed that diesel-driven vehicles had a major influence on the ambient BC concentration. However, gasoline-driven vehicles had more influence on ambient CO concentrations. The contribution of on-road vehicles to the total BC concentration was estimated to be 74% and 80% at NUS and EXW, respectively. The statistical analysis of data obtained in this study showed significant correlations between BC, CO, and NO, confirming that the on-road vehicles were the dominant source of these air pollutants. A significant negative correlation between wind speeds and concentrations of BC, CO, and NO was observed, confirming that the lower wind speed was mainly responsible for the accumulation of air pollutants in the sampling location due to poor dispersion of air.

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References

[1] Bond TC, Bergstrom RW. Light absorption by carbonaceous particles: An investigative review. Aerosol Sci. Tech. 2006; 40: 27–67.

[2] Ban-Weiss GA, Cao L, Bala G, Caldeira K. Dependence of climate forcing and response on the altitude of black carbon aerosols. Clim. Dynam. 2012; 38: 897–911.

[3] Pitts Jr JN. Formation and fate of gaseous and particulate mutagens and carcinogens in real and simulated atmospheres. Environ. Health. Persp. 1983; 47:115–140.

[4] Viidanoja J, Kerminen VM, Hillamo R. Measuring the size distribution of atmospheric organic and black carbon using impactor sampling coupled with thermal carbon analysis: Method development and uncertainties. Aerosol Sci. Technol. 2002; 36: 607–616.

[5] Li H, Chen J, Wu W, Piao X. Distribution of polycyclic aromatic hydrocarbons in different size fractions of soil from a coke oven plant and its relationship to organic carbon content. J. Hazard. Mater. 2010; 176: 729–734.

[6] Liu JJ, Wang XC, Fan B. Characteristics of PAHs adsorption on inorganic particles and activated sludge in domestic wastewater treatment. Bioresour. Technol. 2011; 102: 5305–5311.

[7] Harrison RM, Deacon AR, Jones MR, Appleby RS. Sources and processes affecting concentrations of PM_{10} and $PM_{2.5}$ particulate matter in Birmingham (UK). Atmos. Environ. 1997; 31: 4103–4117.

[8] Donaldson K, Tran L, Jimenez LA, Duffin R, Newby DE, Mills N, MacNee W, Stone V. Combustion-derived nanoparticles: a review of their toxicology following inhalation exposure. Part. Fibre Toxicol. 2005; 2: 10. DOI: 10.1186/1743-8977-2-10

[9] Rey deCastro B, Wang L, Mihalic JN, Breysse PN, Geyh AS, Buckley TJ. The longitudinal dependence of black carbon concentration on traffic volume in an urban environment. J. Air Waste Manage. Assoc. 2008; 58: 928–939.

[10] Bizjak M, Tursic J, Lesnjak M, Cegnar T. Aerosol black carbon and ozone measurements at Mt. Krvavec EMEP/GAW station. Slovenia. Atmos. Environ. 1999; 33: 2783–2787.

[11] Dasch JM, Cadle SH. Atmospheric carbon particles in the Detroit urban area: Winter-time sources and sinks. Aerosol Sci. Technol. 1989; 10: 236–248.

[12] Gardiner K, Trethowan NW, Harrington JM, Rossiter CE, Calvert IA. Respiratory health effects of carbon black: A survey of European carbon black workers. Br. J. Ind. Med. 1993; 50: 1082–1096.

[13] Conant WC, Nenes A, Seinfeld JH. Black carbon radiative heating effects on cloud microphysics and implications for aerosol indirect forcing, 1, Extended Ko¨hler theory. J. Geophys. Res. 2002; 107: 4604. DOI: 10.1029/2002JD002094

[14] Menon S, Hansen J, Nazarenko L, Luo Y. Climate effects of black carbon aerosols in China and India. Science 2002; 297: 2250–2253.

[15] Sharma S, Brook JR, Cachier H, Chow J. Gaudenzi A, Lu G. Light absorption and thermal measurements of black carbon in different regions of Canada. J. Geophys. Res. 2002; 107: 4771. DOI: 10.1029/2002JD002496

[16] Venkataraman C, Habib G, Eiguren-Fernandez A, Miguel AH, Friedlander SK. Residential biofuels in South Asia: Carbonaceous aerosol emissions and climate impacts. Science 2005; 307: 1454–1456.

[17] Highwood EJ, Kinnersley RP. When smoke gets in our eyes: The multiple impacts of atmospheric black carbon on climate, air quality and health. Environ. Int. 2006; 32: 560–566.

[18] Gautam R, Hsu NC, Lau KM. Premonsoon aerosol characterization and radiative effects over the Indo-Gangetic Plains: Implications for regional climate warming. J. Geophys. Res. 2010; 115: D17208. DOI: 10.1029/2010JD013819

[19] Dumka UC, Moorthy KK, Kumar R, Hegde P, Sagar R, Pant P, Singh N, Babu SS. Characteristics of aerosol black carbon mass concentration over a high altitude location in the Central Himalayas from multi-year measurements. Atmos. Res. 2010; 96: 510–521.

[20] Hamilton RS, Mansfield TA. Airborne particulate elemental carbon: its sources, transport and contribution to dark smoke and soiling. Atmos. Environ. 1991; 25: 715–723.

[21] Pakkanen TA, Kerminen V, Ojanen CH, Hillamo RE, Aarnio P, Koskentalo T. Atmospheric black carbon in Helinski. Atmos. Environ. 2000; 34: 1497–1506.

[22] Behera SN, Sharma M. Reconstructing primary and secondary components of $PM_{2.5}$ composition for an urban atmosphere. Aerosol Sci. Technol. 2010; 44: 983–992.

[23] Schwartz J, Litonjua A, Suh H, Verrier M, Zanobetti A, Syring M, Nearing B, Verrier R, Stone P, MacCallum G, Speizer FE, Gold DR. Traffic related pollution and heart rate variability in a panel of elderly subjects. Thorax. 2005; 60: 455–461.

[24] Zanobetti A, Schwartz J. Air pollution and emergency admissions in Boston, MA. J. Epidemiol. Community Health. 2006; 60: 890–895.

[25] Maisonet M, Correa A, Misra D, Jaakkola JJ. A review of the literature on the effects of ambient air pollution on fetal growth. Environ. Res. 2004; 95: 106–115.

[26] Dockery DW, Luttmann-Gibson H, Rich DQ, Link MS, Mittleman MA, Gold DR, Koutrakis P, Schwartz JD, Verrier RL. Association of air pollution with increased incidence of ventricular tachyarrhythmias recorded by implanted cardioverter defibrillators. Environ. Health. Persp. 2005; 113: 670–674.

[27] Zeka A, Sullivan JR, Vokonas PS, Sparrow D, Schwartz J. Inflammatory markers and particulate air pollution: characterizing the pathway to disease. Int. J. Epidemiol. 2006; 35: 1347–1354.

[28] Kleeman MJ, Schauer JJ, Cass GR. Size and composition distribution of fine particulate matter emitted from motor vehicles. Environ. Sci. Technol. 2000; 34: 1132–1142.

[29] Sakurai H, Tobias HJ, Park K, Zarling D, Docherty S, Kittelson DB, McMurry PH, Ziemann PJ. On-line measurements of diesel nanoparticle composition and volatility. Atmos. Environ. 2003; 37: 1199–1210.

[30] Schneider J, Hock N, Weimer S, Borrmann S, Kirchner U, Vogt R, Scheer V. Nucleation particles in diesel exhaust: composition inferred from in situ mass spectrometric analysis. Environ. Sci. Technol. 2005; 39: 6153–6161.

[31] Charron A, Harrison RM. Fine ($PM_{2.5}$) and coarse ($PM_{2.5–10}$) particulate matter on a heavily trafficked London highway: Sources and processes. Environ. Sci. Technol. 2005; 39: 7768–7776.

[32] Virtanen A, Ronkko T, Kannosto J, Ristimaki J, Makela JM, Keskinen J, Pakkanen T, Hillamo R, Pirjola L, Hameri K. Winter and summer time size distributions and densities of traffic-related aerosol particles at a busy highway in Helsinki. Atmos. Chem. Phys. 2006; 6: 2411–2421.

[33] Balasubramanian R, Qian WB, Decesari S, Facchini MC, Fuzzi S. Comprehensive characterization of $PM_{2.5}$ aerosols in Singapore. J. Geophys. Res. 2003; 108: 4523. DOI: 10.1029/2002JD002517

[34] Balasubramanian R, Qian WB. Characterization and source identification of airborne trace metals in Singapore. J. Environ. Monitor. 2004; 6: 813–818.

[35] Kalaiarasan M, Balasubramanian R, Cheong KWD, Tham KW. Traffic-generated airborne particles in naturally ventilated multi-storey residential buildings of Singapore: Vertical distribution and potential health risks. Build. Environ. 2009; 44: 1493–1500.

[36] Schneider J, Kirchner U, Borrmann S, Vogt R, Scheer V. In situ measurements of particle number concentration, chemically resolved size distributions and black carbon content of traffic-related emissions on German motorways, rural roads and in city traffic. Atmos. Environ. 2008; 42: 4257–4268.

[37] Beckerman B, Jerrett M, Brook JR, Verma DK, Arain MA, Finkelstein MM. Correlation of nitrogen dioxide with other traffic pollutants near a major expressway. Atmos. Environ. 2008; 42: 275–290.

[38] Ntziachristos L, Ning Z, Geller MD, Sheesley RJ, Schauer JJ, Sioutas C. Fine, ultrafine and nanoparticle trace element compositions near a major freeway with a high heavy-duty diesel fraction. Atmos. Environ. 2007a; 41: 5684–5696.

[39] Ntziachristos L, Ning Z, Geller MD, Sioutas C. Particle concentration and characteristics near a major freeway with heavy-duty diesel traffic. Environ. Sci. Technol. 2007b; 41: 2223–2230.

[40] Pérez N, Pey J, Cusack M, Reche C, Querol X, Alastuey A, Viana M. Variability of particle number, black carbon, and PM_{10}, $PM_{2.5}$, and PM_1 levels and speciation: influence of road traffic emissions on urban air quality. Aerosol Sci. Technol. 2010; 44: 487–499.

[41] May AD. Singapore: The development of a world class transport system. Transport Rev. 2004; 24: 79–101.

[42] See SW, Balasubramanian R, Wang W. A study of the physical, chemical, and optical properties of ambient aerosol particles in Southeast Asia during hazy and non-hazy days. J. Geophys. Res. 2006; 111: D10S08. DOI: 10.1029/2005JD006180

[43] LTA. Land Transport Authority Singapore. 1996; 3-16. (https://www.lta.gov.sg/content/dam/ltaweb/corp/PublicationsResearch/files/ReportNewsletter/White-Paper.pdf). Last access on July 2012.

[44] Jessie WS, Yuan W. The efficacy of safety policies on traffic fatalities in Singapore. Accid. Anal. Prev. 1998; 30: 745–754.

[45] LTA, 1999. Land Transport Authority Singapore, Statistics of 1999.

[46] LTA, 2011. Land Transport Authority Singapore, Statistics of 2011.

[47] Miyamoto K. Transport-Environment Issues and Countermeasures in Various Metropolises, in World Conference on Transport Research Society, Institute for Transport Policy Studies, Urban Transport and the Environment, an International Perspective, Tokyo, Editors: Nakamura H, Hayashi Y, May AD. 2004. pp 253–402.

[48] Chow CJ, John GW, Watson GJ, Parichett CL, Pierson RW, Frazier AC, Purcell GR, The DRI thermal/optical reflectance carbon analysis system: description, evaluation and applications in U.S. air quality studies. Atmos. Environ. 1993; 27A: 1185–1201.

[49] Allen GA, Lawrence J, Koutrakis P. Field validation of a semi-continuous method for aerosol black carbon (aethalometer) and temporal patterns of summertime hourly black carbon measurements in southwestern PA. Atmos. Environ. 1999; 33: 817–823.

[50] Pfeffer HU. Ambient air concentrations of pollutants at traffic-related sites in urban areas of North Rhine-Westphalia, Germany. Sci. Total Environ. 1994; 146: 263–273.

[51] Watson JG, Chow JC, Lowenthal DH, Pritchett LC, Frazier CA, Neuroth GR, Robbins R. Differences in the carbon composition of source profiles for diesel and gasoline powered vehicles. Atmos. Environ. 1994; 28: 2493–2505.

[52] Miguel AH, Kirchstetter TW, Harley RA. Onroad emissions of particulate polycyclic aromatic hydrocarbons and black carbon from gasoline and diesel vehicles. Environ. Sci. Technol. 1998; 32: 450–455.

[53] Gray HA, Cass GR. Source contributions to atmospheric fine carbon particle concentrations. Atmos. Environ. 1998; 32: 3805–3825.

[54] Steiner D, Burtscher H, Gross H. Structure and disposition of particles from a spark-ignition engine. Atmos. Environ. 1992; 26: 997–1003.

[55] Fenger J. Urban air quality. Atmos. Environ. 1999; 31: 4877–4900.

[56] Sawyer RF, Harley RA, Cadle SH, Norbec JM, Slott R, Bravo HA. Mobile sources critical review. Atmos. Environ. 2000; 34: 2161–2181.

[57] USEPA. National air quality and emissions trends report. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, NC 27711. 1992; EPA454/R-93-031.

[58] Onursal B, Gautam SP. Vehicular air pollution: Experiences from seven Latin American urban centres. World Bank Technical Paper No. 373, Washington D.C., USA. 1997; 132–140.

[59] Lim LH, Harrison RM, Harrad S. The contribution of traffic to atmospheric concentrations of polycyclic aromatic hydrocarbons. Environ. Sci. Technol. 1999; 33: 3538–3542.

Chapter 6

Air Pollution Monitoring: A Case Study from Romania

Gabriela Iorga

Additional information is available at the end of the chapter

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Abstract

The first section of this chapter provides an up-to-date general view of air pollution/air quality topic. It indicates main pollutants and their sources and impacts and presents and discusses current air quality standards and air quality indexes worldwide; how datasets are acquired, gathered and analyzed and how the measurements are then interpreted are also presented. Recent works containing updated and detailed technical discussions for each issue addressed and additional web resources are mentioned. The great importance of air pollution monitoring is emphasized. Second, in the international context of incomplete information on air pollution in East Europe, the chapter includes a section presenting an assessment of air pollution at some sites in Romania together with its evolution from the beginning of the monitoring up to present. Availability of PM_{10}, $PM_{2.5}$, NO_x, SO_2 and CO concentrations is site and pollutant dependent and varies from 3 to 9 years. Investigation of temporal and spatial variation of pollutant levels, as well as of PM_{10} and $PM_{2.5}$ relationships with the measured gaseous air pollutants and with meteorological variables, includes correlation and linear regression analysis and temporal-trend analysis; coefficient of divergence was calculated to check up on the air pollution inter-sites' differences and pollutant seasonal variation intra-site.

Keywords: air pollution, air quality standards, air quality index, particulate matter, gaseous pollutants, temporal trends, East Europe

1. An introduction to air pollution monitoring

The challenge of modern society to take air pollution abatement measures based on scientific knowledge has encouraged the scientists to study the atmospheric composition changes, the short- and long-term pollutant effects and impacts and to simulate air pollution scenarios all over the world. The advances achieved in the field of air pollution during the past decades are

due to numerous detailed investigations, the application of a large number of techniques and the acquisition of abundant monitoring data.

First, the aim of this chapter is to provide an up-to-date general view of air pollution/air quality topic. Second, in the international context of incomplete information on air pollution in East Europe, the chapter includes a section presenting an updated image of air pollution at some sites in Romania together with its evolution from the beginning of the monitoring up to present.

The substances that accumulate in atmosphere in such a concentration and for enough long time that they may harm the living organisms or produce damage to building materials are called ***pollutants***. World Health Organization gives us the following definition of ***air pollution***: "Air pollution is contamination of the indoor or outdoor environment by any chemical, physical or biological agent that modifies the natural characteristics of the atmosphere." [1] Air pollution can be also defined as "when gases or aerosol particles emitted anthropogenically build up in concentrations sufficiently high to cause direct or indirect damage to plants, animals, other life forms, ecosystems, structures or works of art" [2]. Although both definitions refer to accumulation of a pollutant in atmosphere, the second one is a restrictive definition to anthropogenic influence on air composition. In this respect, the ***air quality*** (AQ) collocation, which is often used to express the status of air pollution, can be viewed as a measure of the anthropogenic perturbation of the natural atmospheric state. The quality of the air depends on the amount of pollutants, the rate at which they are released from various sources and how quickly the pollutants are deposited or disperse. Good air quality refers to clean, unpolluted air. The meteorological conditions influence significantly the amount of pollutants in a region: low winds, temperature inversions and topography with mountains can trap the pollutants close to the ground, leading to an increased amount of pollutants over the region. Conversely, the presence of a strong and persistent wind over an area with significant pollutant emissions but located in a plain can disperse very quickly the air pollutants.

Air pollution comes from many different ***sources*** such as factories, electrical power and chemical plants, chimneys, landfills, oil refineries, smelters, solid waste disposal farming, home and business activities, etc. In addition, all transportation activities using cars, buses, trucks, trains, boats and airplanes contribute to air pollution. Pollution can also result from wildfires, volcanic eruptions, dust storms or windblown dust. As a result, air pollutants can have natural or anthropogenic sources, could come from mobile (e.g. automobiles) or stationary sources (e.g. industrial facilities), could be emitted by local sources and may travel or be formed over long distances affecting therefore large areas. Pollutants in atmosphere can be primary pollutants (emitted directly to the atmosphere) or secondary pollutants (formed by chemical reactions involving primary pollutants and other constituents within the atmosphere). In highly populated metropolitan areas where air pollutants result from a combination of stationary sources and mobile sources, we encounter the so-called air pollution hotspots.

However, the air pollution refers not only to ambient, ***outdoor pollution***, but also to ***indoor pollution***. Pollution within enclosed spaces, such as schools, homes, building offices and various workplaces, can come from tobacco smoke, mould, chemicals released from household products or synthetic fabrics, different paintings or dyes. This chapter is focused on outdoor pollution. However, I must note a very detailed and recently published report that summarizes

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the main standards and guidelines related to the key indoor air pollutants developed by various international agencies worldwide in reference [3].

Below is an introduction to the most widespread air pollutants together with their main sources, and impacts they can have, pollutants that are frequently monitored in most of the networks (**Figure 1**).

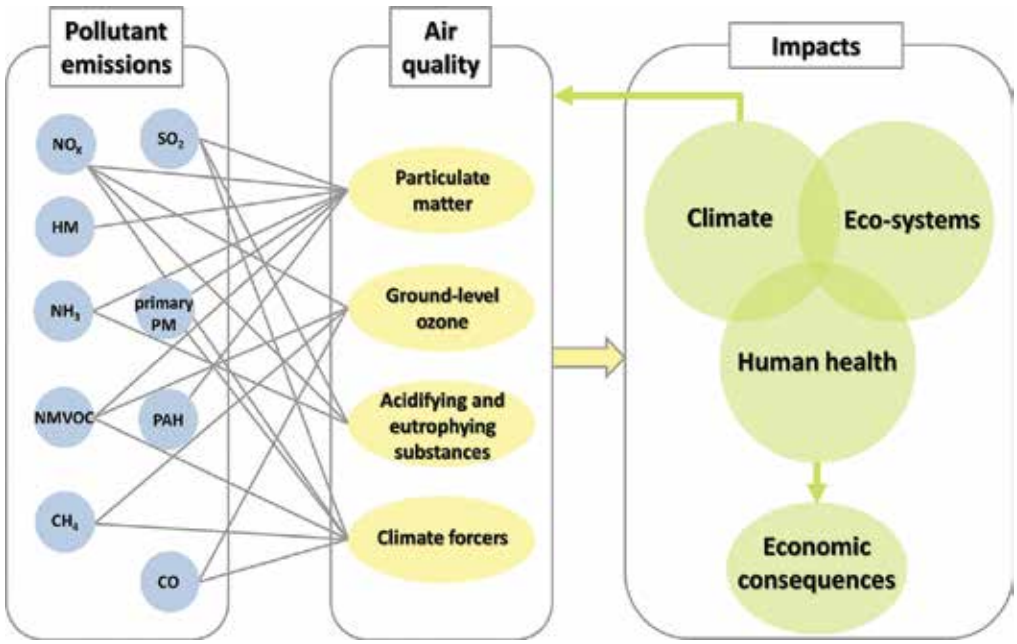

Figure 1. Diagram showing connections between most widespread air pollutants, air quality and impacts. Adapted from brochure [10] and modified.

Particulate matter (PM) in atmosphere is a mixture of particles (solid and liquid) covering a wide range of sizes and chemical compositions. PM_{10} ($PM_{2.5}$) refers to all particles with a diameter less than 10 (2.5) μm. In cities, PM originates predominantly anthropogenically, from several source categories, such as local industrial emissions, vehicular traffic and long-range transport, and can be enhanced by natural sources of coarse particles not easily controllable (e.g. re-suspended windblown dust, sea salt, etc.). Apart from meteorology, even street configuration and urban morphological characteristics may influence the pollutant accumulation or dispersion via the airflow pattern [4, 5]. The main precursor gases for secondary PM are sulfur dioxide (SO_2), nitrogen oxides (NOx), ammonia (NH_3) and volatile organic compounds (VOCs). PM particles are of major concern because of their potential health impact. Although the specific biological mechanisms are not completely understood, many epidemiological studies [6] show the associations between short- and long-term exposure to PM

and hospital admissions, medication use, respiratory symptoms and reduced pulmonary function, or even increased mortality. Heart rate alterations associated with exposure to mixtures of ambient concentration of particulate matter, carbon monoxide (CO) and nitrogen dioxide (NO_2) were observed in epidemiological studies and animal experiments [7]. Health effects are more strongly associated with exposure to fine fraction $PM_{2.5}$ than to the coarse fraction of PM_{10} [8], and the risk of their occurrence is especially high in urban areas.

Sulfur dioxide (SO_2) is an acidic gas formed by oxidation of sulfur (S), mainly through combustion of fossil fuels containing S. The electricity generation sector is the most important source of SO_2. Areas where coal is widely used for domestic heating are important sources of SO_2, as well. Volcanoes are the biggest natural source of sulfur oxides. SO_2 aggravates heart diseases and asthma and can reduce lung function and irritate the respiratory tract. It contributes to the formation of particulate matter, and of acid rain, which damages forests, crops, buildings and ecosystems in rivers and lakes.

Nitrogen oxides (NOx) is the name of a group of highly reactive gases containing nitrogen and oxygen in different amounts. The principal source of NOx is road traffic, but fuel combustion from industrial facilities is another source. Nitrogen monoxide (NO) makes up the majority of NOx emissions, although newer diesel vehicles may emit as much as 55% of their NOx as NO_2 [9]. Oxidation of NO emissions also leads to NO_2. In the presence of sunlight, NO_2 reacts with hydrocarbons and produces photochemical pollutants as ground-level O_3. NO_2 is associated with adverse effects on health of liver and blood. It can aggravate lung diseases leading to respiratory symptoms and increased susceptibility to respiratory infection. NOx contributes to the formation of particulate matter, to acid deposition and to eutrophication of soil and water.

Carbon monoxide (CO) results from incomplete combustion of fuels but can also be formed by oxidation of hydrocarbons and other organic compounds. Sources of CO include road traffic (high levels of CO are registered in heavy traffic congestion with old cars and trucks), industrial processes, residential wood burning and forest fires. CO can react with other pollutants producing ground-level ozone or with O_2 creating CO_2, which is an important warming agent. In a warmer atmosphere, higher levels of NO_3 could appear and thus PM level may increase. CO can lead to significant reduction of oxygen to heart and central nervous system, and, therefore, headaches, dizziness and fatigue appear.

Non-methane volatile organic compounds (NMVOCs) include a variety of chemicals, coming from both anthropogenic (paints, road transport, dry-cleaning and other solvent uses) and biogenic (vegetation) sources, with the emitted amounts dependent on species and on temperature. Certain NMVOC species, such as benzene (C_6H_6) and 1,3-butadiene, are directly hazardous to human health. NMVOCs are also precursors of ground-level ozone.

Ground-level ozone (O_3) is not directly emitted into the atmosphere, but it forms in the atmosphere from a chain of chemical reactions from certain precursor gases: NOx, CO, NMVOCs and methane (CH_4). It irritates the airways of the lungs, may decrease the lung function and aggravates the asthma even at very low levels. O_3 damages plants and ecosystems,

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and it can lead to premature mortality. Ozone is also a greenhouse gas contributing to warming of the atmosphere.

Other pollutants of interest are ammonia (NH_3) and methane (CH_4), coming mainly from agriculture, waste management and energy production; benzo[a]pyrene (BaP), resulting from incomplete combustion of various fuels for domestic home-heating, in particular wood and coal burning, waste burning, coke and steel production and road traffic; toxic metals: arsenic (As), cadmium (Cd), lead (Pb) and nickel (Ni), emitted mainly from the combustion of fossil fuels, metal production and waste incineration; and black carbon (BC), which is a product of incomplete combustion of fossil fuels; BC results mostly from traffic and industry.

Air qusformed in ***networks of air quality monitoring stations*** owned by national governments, but regional, provincial administrations or some non-profit organizations might collaborate, too. Nowadays, automatic and/or manual AQ networks operate in numerous countries. The various air pollutants are monitored depending on national or regional interests, network capabilities and/or personnel and funding available. Among the usual ***determination techniques,*** the chemiluminescence (NO_x), UV fluorescence (SO_2), non-dispersive infrared spectroscopy (CO), UV photometry (O_3) and gas chromatography with photo ionization detector (C_6H_6) find themselves out. Measurements of PM_{10} and $PM_{2.5}$ are obtained by gravimetric analysis and those of heavy metals (Pb, Cd, Ni, As) by atomic absorption spectrometry. A presentation of advances in instrumentations and methodologies for measuring atmospheric composition from space, aircraft and the surface can be found in the reference [11], laboratory techniques being also included. Datasets are acquired through measurements made on an hourly or daily average basis, and concentration values are evaluated and reported. The registered volume or mass concentrations, expressed as ppb, ppm or $\mu g\ m^{-3}$ for example, are used to assess and inter-compare the air quality levels at different scales: local, regional and global. When long-term data are available, a long-term trend analysis can be performed. When an assessment of the AQ is desired, the ambient air measurements must be evaluated in conjunction to data on anthropogenic emissions and their trends for all available pollutants, or at least the main pollutants.

Air pollution is mostly regulated by ***emission standards*** and taxes and by ***air quality standards***. The air quality standards have scientific basis on epidemiological, toxicological investigations on humans and animals and intensive researches on pollution impact on ecosystems. Because the time pattern of air pollution is important in relation to pollution impacts, the objective of a standard is to establish the limit values and alert thresholds for pollutant concentrations in ambient air with the general aim to avoid, prevent and/or reduce harmful effects on human health and on the environment. Air quality standards are expressed as guidelines (WHO) or standards (US EPA, EU). A detailed review of air quality policy in the USA and the EU, including the main legislation acts and emission standards, is presented in [12], and reference [13] provides a review of air quality management actions. A comparative table on limit values for the main pollutants SO_2, NO_2, CO, PM_{10} and $PM_{2.5}$ including some other countries worldwide is provided in **Table 1**.

Pollutant	Time period	European Union		US EPA NAAQS	WHO	Australia	British Columbia	South Africa	Mexico	China*	India*
		Value	Observations	Air quality limit value/guideline							
$_{10}$	1 year	40, 20	For protection of human health 35/year, since 2010	50	20			50	50	40	60
	24 h	50	For protection of human health 7/year	150 1/year	50	50	50	120 4/year	120	50	100
$_{2.5}$	1 year	25		15	10	8	8		15	100	40
	24 h			35	25						
	1 year	20	For protection of ecosystems		50	20	50	50	0.03 ppm	20	20
	24 h	125	3/year	365 1/year	20	80	260	120 4/year	0.13 ppm 1/year	50	80
$_{2}$	1 h	350	24/year	75 ppb		200	900	350 88/year		150	
	3 h			0.5 ppm 1/ year							
	10min				500						
	1 year	30	For protection of ecosystems								
NO_2	1 year	40	For protection of human health	53 ppb	40	30	60	40		40	40
	24 h						200			80	80
	1 h	200	For protection of human health 18/year	100 ppb	200	120	400	200 88/year	0.21 ppm 1/year	120	
	1 year	120	Long-term goal for protection of human health: AOT40 from 1 h values within period May–July								
	1 h	6000	Long-term goal for protection of ecosystems: AOT40 from 1 h	120 ppb							

Pollutant	Time period	European Union		US EPA NAAQS	WHO	Australia	British Columbia	South Africa	Mexico	China*	India*
		Value	Observations	Air quality limit value/guideline							
			values within period May–July								
	8 h			0.07 ppm	100						
	24 h									4	
	8 h	10°		9 ppm 1/year	10°	9°	11°	10° 11/year	11 ppm 1/year		2°
	1 h			35 ppm 1/year	30°		28°	30° 88/year		10°	4°

Data compiled from references [13–16].
*China sets standards for three levels of air pollution, here is shown the most restrictive one, for residential areas; +for protection of ecosystems; "x/year" represents not to be exceeded × times per year; units of measure are μg m^{-3}, unless where ° is indicated, when mg m^{-3} must be considered; ppm—parts per million; ppb—parts per billion.

Table 1. Comparison of current worldwide air quality limit or target values/guidelines.

The data from monitoring stations are also used to calculate ***air quality index (AQI)***. This is a common way to present to the people the air quality status by the government agencies, in both developed and developing countries. The higher the AQI value, the higher the percentage of the population that is likely to experience severe adverse health effects. AQI can be calculated for both short (hourly, daily) and for long-term (annual) periods. AQI is constructed in order to match the air quality standards of the country where it is used. A general formula to compute an AQI is the following:

$$AQI_{pollutant} = \frac{\text{pollutant concentration reading}}{\text{standard concentration}} x100 \quad (1)$$

The AQI is generally based on a number of subindices for individual pollutants. The classification of air quality is based on the subindex with the highest value. Currently, there are numerous AQIs, but we do not have a methodology internationally accepted to construct these indexes. Most of them are defined using the main common gaseous pollutants: CO, NO_2, O_3, SO_2 and particulate matter (PM_{10} and $PM_{2.5}$). Sometimes, other pollutants, such as C_6H_6, NH_3 or Pb, are added. **Table 2** presents a compilation of some current existing AQI, the health risk category and implications for the population. At the end of **Table 2**, the AQ classification, the color code and how the AQI is computed, as provided by Rhenish Institute for Environmental Research at the University of Cologne (EURAD), are shown [17]. For the rest of the regions included in **Table 2**, the appropriate references for AQI calculation are provided.

Most state or local agencies report the AQI on their public web sites. Real-time monitoring data and forecasts of air quality that are color coded in terms of the air quality index are available from US Environmental Protection Agency's AirNow web site www.airnow.gov. Real-time AQI visual map for more than 60 countries over the world is available at https://waqi.info. To convert an air pollutant concentration to an AQI or conversely, EPA has also developed a calculator [23]. As one observes, the AQI is country or city specific, and even the interpretation of an AQI varies considerably from one region to other; this makes the comparison of calculated values in various regions difficult. To minimize these difficulties within its boundaries and to facilitate the international comparison of near real time of AQ, European Union introduced in 2006 the Common AQI in the framework of CITEAIR Project [22]. Moreover, the AQIs do not take into account the coexistence of all the air pollutants. Reference [24] shows how a multi-pollutant and multi-site AQI could be designed in order to get an aggregate measure of air pollution. However, the AQI has the advantage to concentrate multiple and multi-scale measurements in a unique indicator and allows to follow the evolution of air quality in a given region or city providing timely and understandable information for population and supporting local authorities governments in decisions to prevent and avoid adverse health effects. Critical and comparative reviews of the existing AQIs and proposal of alternatives are provided by references [25–27].

Canada [18]

Health risk category	**AQHI**	**Health messages**	
		At risk population	**General population**
Low	1–3	Enjoy your usual outdoor activities	Ideal air quality for outdoor activities
Moderate	4–6	Consider reducing or reschedu -ling strenuous activities outdoors if you are experiencing symptoms	No need to modify your usual outdoor activities unless you experience symptoms such as coughing and throat irritation
High	7–10	Reduce or reschedule strenuous activities outdoors. Children and the elderly should also take it easy	Consider reducing or rescheduling strenuous activities outdoors if you experience symptoms such as coughing and throat irritation
Very high	Above 10	Avoid strenuous activities outdoors. Children and the elderly should also avoid outdoor physical exertion	Reduce or reschedule strenuous activities outdoors, especially if you experience symptoms such as coughing and throat irritation

China [19]

Air pollution level (color code)	**AQI**	**Health implications**	**Cautionary statement (for $PM_{2.5}$)**
Good (green)	0–50	Air quality is considered satisfactory, and air pollution poses little or no risk	None

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China [19]

Air pollution level (color code)	AQI	Health implications	Cautionary statement (for $PM_{2.5}$)
Moderate (yellow)	51–100	Air quality is acceptable; however, for some pollutants there may be a moderate health concern for a very small number of people who are unusually sensitive to air pollution	Active children and adults, and people with respiratory disease, such as asthma, should limit prolonged outdoor exertion
Unhealthy for sensitive groups (orange)	101–150	Members of sensitive groups may experience health effects. The general public is not likely to be affected	Active children and adults, and people with respiratory disease, such as asthma, should limit prolonged outdoor exertion
Unhealthy (red)	151–200	Everyone may begin to experience health effects; members of sensitive groups may experience more serious health effects	Active children and adults, and people with respiratory disease, such as asthma, should avoid prolonged outdoor exertion; everyone else, especially children, should limit prolonged outdoor exertion
Very unhealthy (purple)	201–300	Health warnings of emergency conditions. The entire population is more likely to be affected	Active children and adults, and people with respiratory disease, such as asthma, should avoid all outdoor exertion; everyone else, especially children, should limit outdoor exertion
Hazardous (maroon)	Above 300	Health alert: everyone may experience more serious health effects	Everyone should avoid all outdoor exertion

India [20]

Air pollution level	AQI	Associated health impacts
Good	0–50	Minimal impact
Satisfactory	51–100	May cause minor breathing discomfort to sensitive people
Moderately polluted	101–200	May cause breathing discomfort to people with lung disease such as asthma, and discomfort to people with heart disease, children and older adults
Poor	201–300	May cause breathing discomfort to people on prolonged exposure, and discomfort to people with heart disease
Very poor	301–400	May cause respiratory illness to the people on prolonged exposure. Effect may be more pronounced in people with lung and heart diseases
Severe	401–500	May cause respiratory impact even on healthy people, and serious health impacts on people with lung/heart disease. The health impacts may be experienced even during light physical activity

US EPA [21]

Level of health concern (color code)	AQI	Sensitive groups	General population
Good (green)	0–50	None	None
Moderate (yellow)	51–100	Unusually sensitive individuals may experience respiratory symptoms	None
Unhealthy for sensitive groups (orange)	101–150	Increasing likelihood of respiratory symptoms and breathing discomfort in active children and adults and people with lung disease, such as asthma	None
Unhealthy (red)	151–200	Greater likelihood of respiratory symptoms and breathing difficulty in active children and adults, people should reduce prolonged or heavy outdoor exertion	Possible respiratory effects
Very unhealthy (purple)	201–300	Increasing severity of cardiovascular symptoms and impaired breathing likely in active children and adults and people with lung diseases	Significant increase in respiratory symptoms
Hazardous (maroon)	301–500	Serious aggravation of heart or lung disease and premature mortality in people with cardiopulmonary disease and older adults	Serious risk of respiratory effects in general population

Europe [22]

Pollution (color code)	CAQI	Observations
Very low (green)	0–25	CAQI do not replace the pre-existent AQI. CAQI are designed to give a dynamic picture of the air quality situation in each European city and to allow an AQ comparison of all EU cities in an easily understandable way. Three indices exist: • An hourly index, which describes the air quality today, it is based on hourly values and updated every hour • A daily index, which stands for the general air quality situation of yesterday, it is based on daily values and updated once a day • An annual index, which represents the city's general air quality conditions throughout the year and compare to European air quality standards. This index is based on the pollutants year average compared to annual limit values, and it is updated once a year A general background AQI for outdoor air quality experienced by the average citizens, and a roadside AQI for AQ on busy streets for people living, working, walking, people in cars and busses are also calculated
Low (lime)	25–50	
Medium (light orange)	50–75	
High (gold)	75–100	
Very high (dark red)	Above 100	

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AQ category, AQI and code color as provided by EURAD [17]					
$AQI = Max\left(\frac{O_3(24h)}{100}, \frac{NO_2(24h)}{90}, \frac{PM_{10}(24h)}{50}, \frac{SO_2(24h)}{125}, \frac{CO(24h)}{10000}\right)x50$					
very good	good	satisfactory	sufficient	poor	very poor
below 10	10-20	20-30	30-50	50-80	above 80

Data compiled from references indicated in table for each location.

Table 2. Examples of current worldwide Air Quality Index by location (country, region or city).

Apart from evaluation of air quality at various spatial scales, air pollution monitoring provides essential information to validate the predictive methods and dispersion models, which represent an important set of tools for simulating air pollution scenarios.

One concern that must be mentioned here is the future changes in air quality that will result from climate changes. Many studies indicated a warmer and a more humid climate, with a higher frequency of occurrence of heat waves, of stronger local storms and a higher probability of decrease in frequency of mid-latitude cyclones. Shortly, a warmer and more humid climate will increase the CO_2 and VOC levels, will determine (region-specific) increases or decreases of O_3, a greater conversion of SO_2 to sulfate will take place, and patterns of NOx will be affected. Due to an increased presence of reactive gaseous species even $PM_{2.5}$ speciation might be changed, and this will, in turn, affect the Earth's radiative balance. Simulations of future changes in air quality that will result from changes in both meteorological forcing and air pollutant emissions are presented by Glotfelty et al. [28] up to 2050 following the IPCC AR4 SRES A1B scenario. It shows that global air quality is projected to degrade by the mid-21st century on global average, but the changes are regional in nature: for example, $PM_{2.5}$ level will reduce in Europe and Africa, whereas it will increase in South and Southeast Asia, Indonesia, Australia and South America.

Moreover, thinking about the future long-term air pollution, we must also consider that changes in future air quality will have economic consequences whose projections must be also analyzed. With respect to this, the very recent report "The Economic Consequences of Outdoor Air Pollution" [29] supplies us with a comprehensive assessment of the regional and global economic consequences of outdoor air pollution for the period 2015–2060. Linking the pollutant emissions to labor productivity, healthcare expenditures and changes in crop yields (market costs) and to mortality and morbidity/illness (non-market costs), the projections are indeed of great concern, even if they are subject to uncertainties. The results indicate, among other consequences, that "by 2060, a large number of deaths are projected to take place in densely populated regions with high concentrations of $PM_{2.5}$ and O_3 (especially China and India) and in regions with aging populations, such as China and Eastern Europe. The projected mortality effects of $PM_{2.5}$ exposure are much larger than those of O_3. The market costs of air pollution, flowing from reduced labor productivity, additional health expenditures and crop yield losses, are projected to lead to global annual economic costs of 1% of global gross domestic product (GDP) by 2060. The projected GDP losses are especially large in China (–2.6%), the

Caspian region (–3.1%) and Eastern Europe (Non-OECD EU –2.7% and Other Europe –2.0%), where air pollution impacts lead to a reduction in capital accumulation and a slowdown in economic growth. In per capita terms, the average global welfare costs from mortality and morbidity are projected to increase from less than USD 500 per person in 2015 to around USD 2 100-2 800 in 2060" [29].

One can, therefore, have an idea about the severe global economic consequences of air pollution and the need of stronger policies to improve the air quality results to be of huge importance for all of us. Within this context, to monitor air pollutants is of great necessity of two-fold importance: in order to take informed decisions, to develop and strengthen the political strategies when the societal and economic challenges are addressed and also to respond to the scientific questions of atmospheric sciences.

2. Case study: assessment of air pollution over Northern Romania

2.1. The air pollution monitoring in Romania

A systematic air pollution monitoring in Romania started in early 2000s, beginning with Bucharest, the capital of Romania, and has been gradually developed to the rest of the country. Before 2000s, air pollution was investigated in some fixed points of interest (next to industrial sources, traffic hot spots, parks...) only by manual sampling. The number of fixed sampling points was city dependent and variable in time (for example, Bucharest had between 14 and 5 sampling points); decreasing trend was due to technical issues; 30 min and sometimes 24 h were used as sampling periods for total suspended particles (TSP) and gases NO_2, SO_2, CH_2O, NH_3 and O_3; and TSP were sometimes selectively analyzed for their content of Pb, Cd, Zn and Cu, experimental methods used not being reported. Measured data indicated frequent exceedances of the maximum admitted concentrations (CMA) at that time. For example, between 1996 and 2000 in Bucharest, TSP levels ranged from 150 to 350 $\mu g\ m^{-3}$ (annual average), CMA being of 500 $\mu g\ m^{-3}$. I do not analyze the air pollution before 2000, as measurements were done following local protocols, and the imposed thresholds varied in time, were country specific and were not correlated with the regulations worldwide. All these make the comparison of registered pollutant concentrations in those times with data from other cities very difficult and of very limited usefulness.

Nowadays, a number of 143 monitoring stations of all types, traffic, industrial, urban background, rural and remote background, operate at the country scale. Within the context of air quality monitoring in Europe, reports of the National Environmental Protection Agency (owner of the National Air Quality automatic Monitoring Network) are focused only on compliances with the European Union regulations counting exceedances of the limit values. The very few addressed topics regarding air pollution using some monitoring data in few cities are presented in references [30–35]. Most extensive review image of the air pollution problem in Bucharest metropolitan area was published in 2015 by Iorga et al. [16, 36].

The following part of the chapter focuses on the assessment and analysis of daily concentrations of major pollutants using the longest monitoring datasets available at present.

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2.2. Description of selected stations, data and methods

I selected two urban sites in cities (medium-size) of national importance (Iasi, Cluj-Napoca), with regional role and potential influence at European scale, a regional background site in mountains (Miercurea Ciuc) and the single remote rural background site (Poiana Stampei) for which Romania reports data within EMEP, the **E**uropean **M**onitoring and **E**valuation **P**rogramme for Transboundary Long-Range Transported Air Pollutants (**Figure 2** and **Table 3**). The selected sites are located in different climatic regions of the country have different topography and are expected to be impacted by different pollution sources.

Iasi (IS) is the largest city in North-East Romania; it is located between northernmost hill and a plain, surrounded by uplands, woods and a valley. Iasi has a humid, continental-type climate with summers wetter than winters, with four distinct seasons. Pollution comes from vehicular traffic with old vehicle fleet, construction works, two thermo-electrical power stations and a lack of green spaces. An international airport is located 8 km east of the city center.

Cluj-Napoca (CN) represents the second most populated city in North-West Romania, with a metropolitan area exceeding 420,000 inhabitants. Located in a river valley, surrounded by forests and grasslands, it has a continental climate characterized by warm dry summers and cold winters. Some West-Atlantic influences are present during winter and autumn. The city is an important knob of the European network roads, connecting the country with Western Europe. It has the second main airport in Romania, after Bucharest, at 9 km in its eastern part. Cluj-Napoca has a large industrial park with modern facilities and is an important regional commercial centre, and tourism is well developed in the area. The heating system in Cluj is modern and based on natural gas.

Miercurea Ciuc (MC) is a small city located in a basin surrounded by high mountains with rural settlements. The lowest temperature in the country is frequently registered here. The AQ monitoring station installed here is categorized as rural regional. No major industry exists here; tourism provides the main activities.

Figure 2. Map showing Romania and the air quality monitoring stations included in present research.

Poiana Stampei (PS) is a very small village in North Romania, at the border between the two historical provinces Moldavia in East and Transylvania in West. Over 73% of the village area is represented by forests, and agricultural activity is made with rudimentary field craft.

The air pollution in above sites is compared here with the urban background air pollution in Bucharest, the capital of Romania. Bucharest (approx. 44° 26′ N, 26° 06′ E) represents the most developed city of the country and is located at a relatively equal distance from the Danube River and Carpathian Mountains. The Air Quality Network of Bucharest consists of eight stations that are distributed at different spatial levels (inner core city, larger urban zone and sub-city area) covering the main types of anthropogenic activities. Detailed information about Bucharest can be found in reference [16].

Station name, site designation (population)	Station type	Latitude	Longitude	Altitude (m a.s.l.)	Pollutants included in analysis and the start year of monitoring
Poiana Stampei, PS (837)	Remote rural background	47°19′30″ N	25°08′04″ E	908	PM_{10} (2010), NOx (2010), O_3 (2010), SO_2 (2010), CO (2010)
Miercurea Ciuc, MC (37 176)	Regional rural	46°21′34″ N	25°48′06″ E	710	PM_{10} (2009), NO_x (2009), O_3 (2008), SO_2 (2009), CO (2009)
Cluj-Napoca, CN (324 576)	Urban background	46°46′26″ N	23°35′49″ E	333	PM_{10} (2007), $PM_{2.5}$ (2009), NO_x (2006), O_3 (2006), SO_2 (2006), CO (2006)
Iasi, IS (290 422)	Urban background	47°09′25″ N	27°35′25″ E	44	PM_{10} (2006), $PM_{2.5}$ (2009), NO_x (2006), O_3 (2006), SO_2 (2006), CO (2006)
Bucharest Greater Area (2 272 163), Lacul Morii, LM	Urban background	44°26′33″ N	26°03′36″ E	90	PM_{10}, $PM_{2.5}$, NO_x, SO_2, CO, O_3

Last monitoring year is 2013 for all sites and pollutants. The sampling periods and detailed analysis of pollution corresponding to the selected station in Bucharest used here for comparison are presented in references [36, 39].

Table 3. Stations, monitored pollutants and beginning year of monitoring.

Data used in the present study are extracted from AirBase v.8 database [37] of European Environment Agency (EEA) for background stations in above locations. However, in order to have completeness of data series for Iasi, some $PM_{2.5}$ data were added from a traffic station. Availability of the concentrations is site and pollutant dependent and varies from 3 to 9 years. Most data cover the period from January 1, 2006 to December 31, 2013. I focus here on PM_{10} and $PM_{2.5}$, and NO_x, SO_2 and CO (**Table 3**), as primary gaseous pollutants that accumulate in urban atmosphere and significantly contribute to the photochemical formation of ozone and

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other oxidants and to a fraction of the particulate matter [38]. O_3 daily averages were added in order to seek if they could help to better understand the correlations between particulates and primary gaseous pollutants.

A synthetic database of daily averaged datasets of pollutants from AirBase and local meteorology series (air temperatures, relative humidity, atmospheric pressure, wind speed and direction) was prepared in order to have completeness for all sites for common time periods per site, as **Table 3** specifies. When it was necessary, conversion of hourly gaseous pollutants and local meteorology data to daily averages was done by averaging over 24 h periods from midnight to midnight.

Last monitoring year is 2013 for all sites and pollutants. The sampling periods and detailed analysis of pollution corresponding to the selected station in Bucharest used here for comparison are presented in references [36, 39].

Statistical examination of temporal and spatial variation of PM_{10} and $PM_{2.5}$ concentrations, as well as their relationships with the measured gaseous air pollutants and meteorological variables, includes:

- Correlation analysis, expressed by Pearson coefficients (COR), statistically significant at 95% confidence interval.
- Single and multiple linear regression analysis, between daily PM as dependent variable and meteorological factors and gaseous pollutants as independent variables, respectively.
- Temporal trend analysis for detecting and estimating a monotonic annual and seasonal trend of ambient pollutant concentrations was performed using the non-parametric Mann-Kendall's test and Sen's method using MAKESENS software [40].
- Coefficient of divergence (COD), a self-normalizing parameter, was applied to evaluate the differences in the average concentrations of pollutants at each site for paired seasons and to compare monitoring sites. COD provides information on the degree of uniformity between monitoring stations and seasons. For example, a low COD and a high COR are expected for sites impacted by similar pollution sources. A COD value between 0 and 0.2 will indicate uniformity, and a COD between 0.4 and 1 will indicate heterogeneity. The coefficient of divergence is calculated as:

$$COD_{jk} = \sqrt{\frac{1}{p}\sum_{1}^{p}\left(\frac{x_{ij} - x_{ik}}{x_{ij} + x_{ik}}\right)^2} \tag{2}$$

where j and k stand for the two seasons being compared, p is the number of components investigated and xij and xik represent the average mass concentrations of pollutant i during seasons j and k; j and k stand for different sites when COD definition was applied to inter-site comparisons.

2.3. Emissions

Inventories of emitted air pollutants have been substantially improved during the past few years, in particular for main pollutants, including fine particulates and ozone. WebDab contains all emission data officially submitted to the secretariat of the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) by Parties to the Convention [41]. Romania updated its reports to the emission database WebDab of EMEP in 2015. Pollutant emission trends per site (**Figure 3**) were evaluated using the gridded data from WebDab for the corresponding time periods of ambient mass concentrations of pollutants, considering the national total economic sectors.

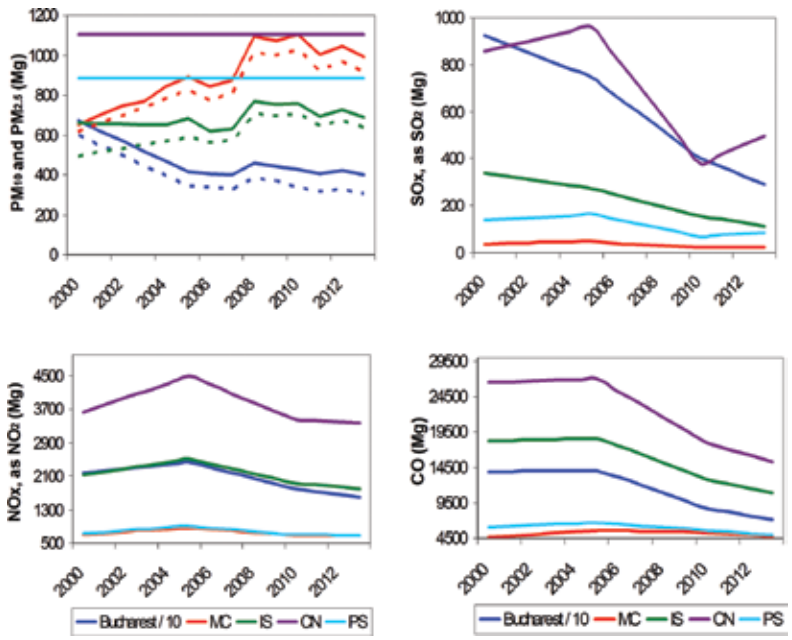

Figure 3. Pollutant emission trends per site as resulted from WebDab emission database, for national total economic sectors. Dotted lines represent $PM_{2.5}$ emissions. Bucharest emissions are included for comparison.

As shown in **Figure 3**, the total emissions of gaseous pollutants decreased for all sites, especially starting with 2006, whereas the PM_{10} and $PM_{2.5}$ emissions show a different pattern: positive trends for IS and MC and stable emissions for CN and PS sites. Even if the particulate emissions in Bucharest are 10 times higher than in all other sites, due to implementation of the environmental development plan, Bucharest has decreased its particulate emissions from about 5970 Mg in 2000 to 3060 Mg in 2013. Emissions of PM seem to be of major concern among the pollutants in Romania. The decreasing trend of gaseous emissions follows the general decreasing trend of emissions (SO_2 decreased by 58%, NOx and CO by about 25%) at EU scale [42], the strongest decrease being for SO_2 (range: 34% for MC–66% for IS), followed by CO (range: 1% for MC–42% for CN).

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2.4. Ambient pollutant concentrations

2.4.1. Levels of PM_{10}, $PM_{2.5}$, NOx, CO, SO_2, O_3

Particulate matter and gaseous pollutant variability are presented in detail in **Figure 4**. **Figure 4a** and **4b** provides a box-plot comparison of the annual levels of daily averages of PM_{10} and $PM_{2.5}$ mass concentrations by site for the corresponding monitoring periods, including EU limit values [43]. Measured ambient annual (mean, median and 95th percentile) PM_{10} concentrations have the highest values at IS urban site and the lowest at PS remote site; at all sites, observations situate below the EU limits with the exception of IS city, where in 2013 a value of 44.64 ± 20.78 μg m^{-3} has been reached. This value is comparable with the value of 45.10 μg m^{-3} representing the mean PM_{10} concentration during 2005–2010 in Bucharest, when a decrease from about 46 to 35 μg m^{-3} was registered. Concentrations higher than 100 μg m^{-3} appear very often at IS (and more frequent than in Bucharest in 2010 [16, 36]) and even in the alpine basin of MC, although here 95th percentile data are below 100 μg m^{-3}. The cleanest air appears to be in PS (mean concentration of 15 μg m^{-3} in 2013), and the urban city CN is the second in rank. $PM_{2.5}$ levels exceed frequently the EU target of 25 μg m^{-3} at both urban sites. Our observations fit very well within the range of European concentrations (from about 20 μg m^{-3} (Finland) to about 75 μg m^{-3} (Bulgaria)), data extracted from Ref. [42] based on 90.4 percentile of daily mean concentration values corresponding to the 36th highest daily mean in 2013.

The average of $PM_{2.5}/PM_{10}$ mass concentration ratios situates between 0.38 (IS) and 0.71 (MC), indicating a higher contribution to PM_{10} samples of coarse particles for IS and of fine fraction for MC. Together with results for CN site (0.63) and Bucharest (from 0.7 for industrial sites to 0.8 for a traffic site in the very centre of the city), our observations are consistent with the $PM_{2.5}/PM_{10}$ mass ratios from 0.5 to 0.9 at most sites across the Europe.

As shown in **Figure 4c**, the annual average SO_2 concentration in IS was 6.92 μg m^{-3} in 2006 and has gradually decreased to 3.45 μg m^{-3} in 2013, and in CN decreased from 6.83 to 5.69 μg m^{-3}. Lower values were observed for regional MC and remote PS sites. The decrease in ambient concentrations of SO_2 and CO in IS was related to lower local emissions of SO_2 and CO based on the positive correlation ambient-emitted SO_2 and CO, respectively (COR_{SO2}=0.94; COR_{CO}=0.96). The same conclusion stands for CO in Cluj-Napoca, but in a lower extent for SO_2 (COR_{SO2}=0.41). For PS site, the ambient SO_2 concentrations increased slightly from 4.67 to 6.91 μg m^{-3}, especially due to intensive use of coal for residential household activities. Multi-annual average temperature at PS is 4.3°C.

The annual averages of NOx and O_3 concentrations show a lower variability at each site, their average values 2006–2013 varying between 40 (42) μg m^{-3} and 55 (34) μg m^{-3} in mid-sized cities IS and CN, respectively.

All the gaseous pollutant concentrations at studied sites here are at least one order of magnitude lower than the values observed in Bucharest. [16, 39].

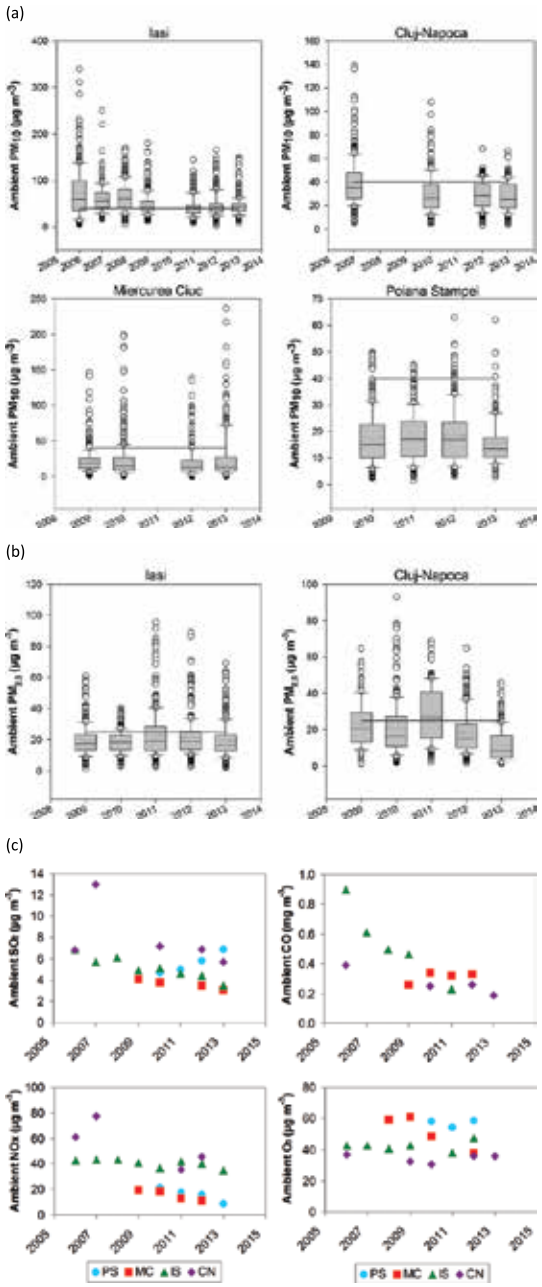

Figure 4. a) Levels of daily PM_{10} mass concentrations for the specified monitoring periods by each site, including median, the 5th, 25th, 75th and the 95th percentiles of their 24 h concentrations. Dotted line within boxes represents the annual average. The EU annual limit value of 40 μg m^{-3} (long continuous line) is included. Black circles represent first and last 5% of observed PM_{10}. b) As in Figure 4a, but for $PM_{2.5}$ concentrations. Long continuous line represents the EU-2015 limit value of 25 μg m^{-3}. c) Annual variation of gaseous pollutants by site..

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2.4.2. Seasonal variability and site inter-comparison

The seasonal variability (**Table 4**) and inter-site comparison (**Table 5**) were investigated using the coefficient of divergence (COD) and coefficient of correlation (COR). As an example, **Figure 5** shows the extreme differences between seasons in Iasi.

		IS				CN	
Season	**Spring**	**Summer**	**Autumn**	**Season**	**Spring**	**Summer**	**Autumn**
Winter	0.22	0.33	0.14	Winter	0.30	0.30	0.17
Spring		0.15	0.17	Spring		0.19	0.34
Summer			0.25	Summer			0.34

		MC				PS	
Season	**Spring**	**Summer**	**Autumn**	**Season**	**Spring**	**Summer**	**Autumn**
Winter	0.39	0.39	0.38	Winter	0.11	0.11	0.07
Spring		0.26	0.29	Spring		0.07	0.12
Summer			0.29	Summer			0.12

Table 4. Seasonal variability per site using COD calculated from multi-seasonal average pollutant concentrations.

COD values for the pairs of seasons ranged from 0.07 to 0.12 at PS site, and this indicates almost no seasonality here. Seasonal changes in pollutant concentrations were modest for Spring-Summer and Winter-Autumn for IS and CN, and surprisingly, some season-to-season variability appears at MC site.

	Coefficients of divergence (COD)					**Inter-sites' correlation coefficients (COR)**			
Site	**MC**	**IS**	**CN**	**Bucharest**	**Site**	**MC**	**IS**	**CN**	**Bucharest**
PS	0.51	0.61	0.56	0.46	PS	0.99	0.6	0.48	0.66
MC		0.28	0.29	0.37	MC		0.71	0.56	0.75
IS			0.21	0.22	IS			0.83	0.95
CN				0.20	CN				0.91

Table 5. Comparisons between sites using COD and COR calculated from multi-annual average pollutant concentrations.

Overall, the inter-site calculated COR indicates a positive correlation among all sites suggesting that they all suffer from the same pollution source categories. A very similar situation was found to characterize Greater Bucharest Area (COR varies from 0.55 to 0.88) and the Greater Athens Area, where COR varies from 0.55 to 0.84 [44]. However, COD values differentiate the sites, showing: air pollution at the remote PS site is very different from that of all the other

sites; cities Iasi, Cluj-Napoca and Bucharest are relatively similar, and air pollution at regional rural MC site is relatively different from the others. The highest contributor to COD value of the paired PS-IS sites is PM_{10}, and highest contributors to COD for the pair MC-Bucharest are NOx, PM_{10} and $PM_{2.5}$.

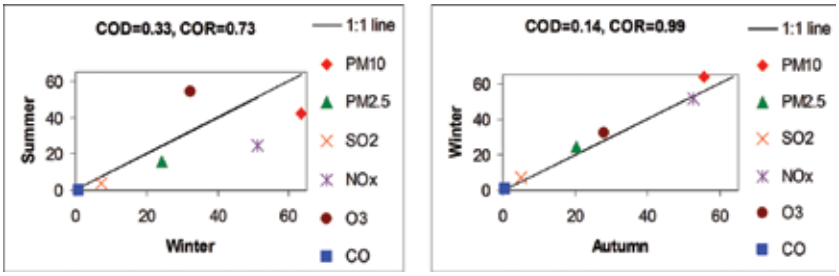

Figure 5. Coefficients of divergence between two seasons for the multi-seasonal average concentrations of pollutants for Iasi site.

2.4.3. Associations between particulate matter levels and gaseous pollutants—meteorology influence

Table 6 synthesizes the relationships between daily PM_{10} and $PM_{2.5}$ and daily averaged gaseous pollutant concentrations over the entire sampling periods up to 2013 per site. It shows good correlations between both PM_{10} ($PM_{2.5}$)-NO_x and PM_{10} ($PM_{2.5}$)-CO, and a less-defined correlation with SO_2. However, the strength of these correlations varies among sites: probably a common road traffic origin in cities IS and CN but with differences in contribution percentages of NOx versus CO (IS has a higher percentage of old vehicles than CN), a lower capability of the area to disperse the pollutants at MC site, low traffic and higher coal and wood combustion at the remote site PS.

Similar correlation coefficients (0.4–0.8 for PM_{10}-NO_x relationship, about 0.4–0.7 for PM_{10}-CO) were reported at different sites in UK and Greece [45]. Bucharest data indicate correlation coefficients of 0.4–0.7 for PM_{10}-NO_x relationship, 0.2–0.5 for PM_{10}-CO relationship and 0.1–0.4 for PM_{10}-SO_2 relationship. The daily mean O_3 concentrations negatively correlated with both PM_{10} and $PM_{2.5}$ could be explained by the reaction of O_3 with NO, which is a major sink for O_3. At the site MC, a positive correlation PM_{10}-O_3 appears. As in some situations in the UK atmosphere [46], short periods with positive correlation PM-O_3 during photochemical episodes were reported in Bucharest Greater Area during 2005–2007 [39]. Our positive correlation might indicate such situations when both PM and O_3 are generated by photochemical activity for MC in warm season, but the calculated coefficient is very low, and probably these episodes are swamped by the 4-year analysis.

The associations between PM_{10} and primary gaseous pollutant levels were investigated further by multiple linear regressions performed using daily mean PM_{10} values and daily averaged gaseous pollutants NO_x, SO_2 and CO for the same periods. For each pollutant, the multiple regressions were performed only for NOx, SO_2 and CO for which single correlation coefficients with PM_{10} were higher than 0.30 (**Table 6**). The multivariate linear regression model is widely

recognized as a useful tool to show associations between primary pollutants [36, 46], to calculate combustion/non-combustion fraction of PM [45] or to predict daily concentrations of PM [47]. For present sites, the model was applied assuming NOx, SO_2 and CO as tracers for anthropogenic activities. In this model, slopes will represent the association of anthropogenic activities with PM_{10} (contribution of anthropogenic activities to PM_{10}), and intercepts are assumed to represent the non-anthropogenic contribution (the natural contribution) to PM_{10}. The natural contributions to PM_{10} thus re-constructed are shown in **Figure 6** for each site.

	IS							**CN**					
	PM_{10}	**$PM_{2.5}$**	**NOx**	**O_3**	**SO_2**	**CO**		**PM_{10}**	**$PM_{2.5}$**	**NOx**	**O_3**	**SO_2**	**CO**
PM_{10}	1.00	0.70	0.56	-0.13	0.34	0.64	PM_{10}	1.00	0.73	0.64	-0.21	0.32	0.53
$PM_{2.5}$		1.00	0.56	-0.19	0.26	0.59	$PM_{2.5}$		1.00	0.46	-0.27	0.32	0.59
NOx			1.00	-0.49	0.36	0.79	NOx			1.00	-0.63	0.25	0.63
O_3				1.00	-0.12	-0.44	O_3				1.00	-0.15	-0.46
SO_2					1.00	0.46	SO_2					1.00	0.31
CO						1.00	CO						1.00
	MC							**PS**					
	PM_{10}	**$PM_{2.5}$**	**NOx**	**O_3**	**SO_2**	**CO**		**PM_{10}**	**$PM_{2.5}$**	**NOx**	**O_3**	**SO_2**	**CO**
PM_{10}	1.00	0.95	0.81	0.05	0.24	0.91	PM_{10}	1.00	–	0.37	-0.13	0.13	0.64
$PM_{2.5}$		1.00	–	–	0.57	–	$PM_{2.5}$		1.00	–	–	–	–
NOx			1.00	-0.18	0.09	0.83	NOx			1.00	-0.45	-0.19	0.54
O_3				1.00	0.33	-0.12	O_3				1.00	-0.07	-0.49
SO_2					1.00	0.07	SO_2					1.00	0.05
CO						1.00	CO						1.00

Table 6. Correlation coefficients between daily PM_{10} and $PM_{2.5}$ and daily averaged gaseous pollutant concentrations.

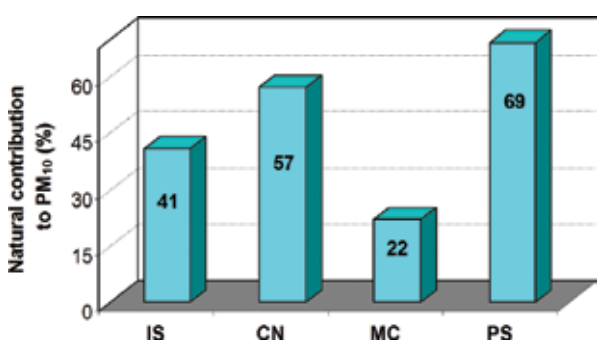

Figure 6. Chart showing re-constructed natural contributions to PM_{10} by site.

Site	Temperature	Atmospheric pressure	Relative humidity	Wind speed	Wind direction
			PM_{10}		
IS (n = 2068)	-0.18	0.15	-0.13	-0.10	-0.12
CN (n = 1327)	-0.22	0.16	-0.12	-0.19	-0.34
MC (n = 1339)	-0.55	0.08	0.22	-0.37	-0.09
PS (n = 1199)	-0.16	0.20	-0.20	-0.40	-0.02
			$PM_{2.5}$		
IS (n = 1733)	-0.32	0.27	0.05	-0.14	-0.13
CN (n = 1327)	-0.48	0.17	0.16	-0.26	-0.30
MC (n = 326)	-0.51	0.05	0.29	-0.42	-0.14

n = number of samples used in analysis.

Table 7. Correlation coefficients between daily PM_{10} and $PM_{2.5}$ and daily averaged local meteorological variables.

Correlation analysis of PM and daily averaged local meteorological variables (**Table 7**) revealed a similar behavior for PM_{10} and $PM_{2.5}$ with all parameters with the exception of PM relationship with the relative humidity.

The negative correlations of PM_{10} and $PM_{2.5}$ with temperature, relative humidity and wind speed indicate dilution of ambient concentrations of PM due to an increased atmospheric boundary layer, scavenging by fog or cloud droplets and deposition onto ground surfaces (precipitation data were not available) and dispersion of particles, especially of fine fraction, by winds. The negative correlation with temperature could be due also to increased emissions (**Figure 3**) or a reduced dispersion (highest coefficients were obtained at MC site) and stable atmospheric conditions (atmospheric pressure) during cold seasons. In cold seasons, low speed wind conditions and lower temperature could result in a lower boundary layer that traps pollution to the ground. In warm seasons, more intense winds, higher temperature (that could reflect positive correlations with solar radiation) and higher boundary layer could result in pollution transport. The multi-annual averages of relative humidity for the corresponding monitored periods have high values for all sites: 72% (IS), 77% (CN), 81% (MC) and 82% (PS). Relative humidity values in the range 70–90% for MC and PS sites appeared frequently, and they were found to be associated with low winds; temperature inversion episodes in MC and PS areas are frequently mentioned in climatology, as well. These combined factors might explain the positive correlation PM_{10}-relative humidity.

The PM_{10} and $PM_{2.5}$ dependence of wind direction (**Figure 7** indicates this dependence for PM_{10}, but $PM_{2.5}$ presents the same distribution) gives certain insights into the distribution of emission sources around the selected monitoring sites. Particulate matter concentrations are associated with southwesterly winds for MC, while in larger cities IS and CN the PM_{10} and $PM_{2.5}$ are distributed relatively equal in all sectors with the exception of NW-NE sector. Highest PM_{10} concentrations (range: 60–80 $\mu g\ m^{-3}$) appear to come from S-SE directions in Cluj-Napoca, and highest PM_{10} (from 100 to 180 $\mu g\ m^{-3}$) come from all directions between NE and NNW in Iasi. At the remote site PS, the highest PM_{10} levels (of about 60 $\mu g\ m^{-3}$) appeared on

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the direction NE-SW, whereas intermediate and low values are associated to all directions from NE to NW.

If one compares meteorological factors that influence the concentration of particulates for the above sites, it results that the most important are temperature, wind speed, humidity and, on the last position, the atmospheric pressure. For Bucharest, the order is changed: wind speed, temperature, atmospheric pressure and humidity. A literature survey revealed that wind speed, relative humidity and temperature seem to compete for the first position, but also their squared terms and interactions between them play some role. In any case, the order of importance of meteorological variable influences on ambient PM levels is regional in nature, and no general conclusion might be drawn.

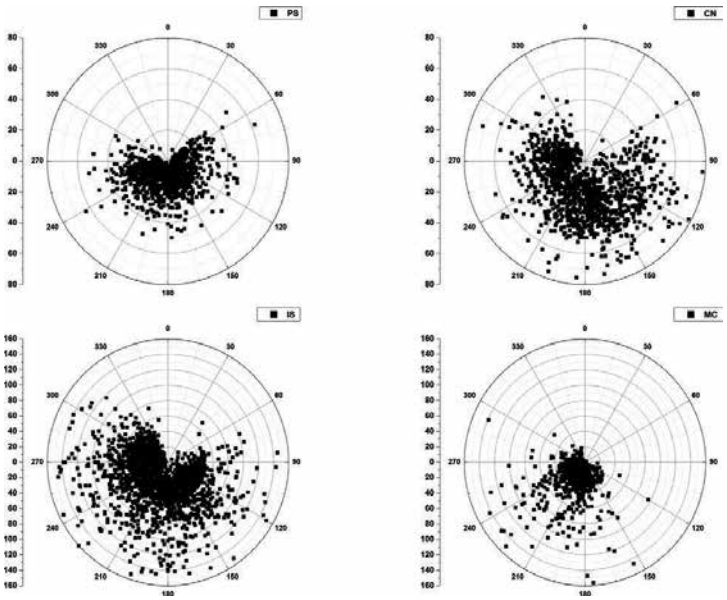

Figure 7. Mass concentrations of PM_{10} with respect to wind direction by each site. $PM_{2.5}$ shows the same pattern.

2.5. Pollutant trends: annual and seasonal

Pollutant annual and seasonal average concentrations at all sites were further investigated in order to determine if ***temporal trends*** could be revealed. Calculated annual trends (**Table 8**) indicate the most pronounced decreases for PM_{10} at IS site (-3.6% yr^{-1}), for NOx at MC (-3.1% yr^{-1}) and stability in CN area. A slight increasing trend for SO_2 was detected at PS remote site. The annual pollutant trends follow only partial trends of emissions, depending on the site, as it results from a comparison of data in **Table 8** with graphs in **Figure 3**. Seasonal trends at PS, MC and CN sites showed the same behaviors as annual pollutant trends for all seasons, whereas in Iasi some seasonality could be observed. Here, PM_{10}, CO and SO_2 decrease mainly during winter, whereas the maximum decrease of NOx appeared in autumn (associated with

maximum increase of O_3). All estimated trends are lower than those calculated for Bucharest area [36, 39], where for example, SO_2 annual trends varied between -1.28% yr^{-1} and -3.73% yr^{-1}. More pronounced reductions in SO_2 (from -6.6 to -14.9% yr^{-1}) were recently reported for UK [48], whereas for various stations across Europe, percentage reductions of $PM_{2.5}$ varied from 7 to 49% during the 2002–2010 period [49].

Site/Pollutant	PS			MC			IS			CN		
	S	Q	Trend	S	Q	Trend	S	Q	Trend	S	Q	Trend
PM_{10}	0	-0.1	Stable	4	1.3	Stable	-15	-3.6	Decreasing	-4	-2.0	Stable
$PM_{2.5}$	nd	nd	nd	nd	nd	nd	0	0.1	Stable	-6	-2.3	Stable
NO_x	-6	-3.8	Decreasing	-6	-3.1	Decreasing	-18	-0.8	Decreasing	-2	-3.8	Stable
O_3	nd	nd	nd	-4	-5.3	Stable	-2	-0.1	Stable	0	0.01*	Stable
SO_2	6	0.8	Increasing	-6	-0.3	Decreasing	-24	-0.4	Decreasing	-4	-0.3	Stable
CO	nd	nd	nd	2	0.02*	No trend	-10	-0.1	Decreasing	-4	-0.03*	Stable

All trends are statistically significant at level of significance at least 0.1 (corresponding to 10% chance there is no trend) unless otherwise indicated.
*Trend non-significant; nd—not determined; S—Mann-Kendall statistics; Q—Sen's slope estimate.

Table 8. Pollutant annual trends, calculated as percent change per unit time.

In southeastern United States, decreasing trends from -5.1 to -9.7% yr^{-1} for SO_2 and decreases of annual mean CO and NOx concentrations at rates ranging from -1.2 to -7.2% yr^{-1} (-6.0 to -9.0% yr^{-1}) were reported [50], which are also higher than the corresponding decreasing rates determined for all selected locations in Romania. However, calculated temporal trends of main pollutants during 1997–2012 in Makkah, Saudi Arabia, indicate both increases (3.4% yr^{-1} for PM_{10}, 6.1% yr^{-1} for SO_2, 4.7% yr^{-1} for O_3) and decreases (-2.6% yr^{-1} for CO, -3.5% yr^{-1} for NO) [51]. Among potential factors responsible for the observed trends all over the world are emissions for traffic, changing weather patterns, construction activities, windblown re-suspensions, emissions of O_3 anthropogenic precursors, whose predominance is of regional nature, but large-scale meteorological phenomena (North Atlantic Oscillation for example), implementation of pollution abatement strategies or the economic crisis influences are also important [52].

3. Conclusion

This study contributes to the knowledge on air pollution in East Europe, presenting an updated assessment of the ground-level concentrations of major air pollutants in different environments, from highest to background values, and using data covering the longest available time period. Ambient air pollution levels, their variability and trends are discussed in the context of air quality status and trends in Bucharest, Europe and worldwide. Specific-air pollutant trends are analyzed in order to show if they follow the trend of pollutant emissions.

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References

[1] World Health Organization. http://www.who.int/topics/air_pollution/en/ [Accessed: 2016-06-23]

[2] Jacobson MZ. Atmospheric Pollution: History, Science and Regulation. Cambridge University Press, Cambridge, UK; 2002. 412 p.

[3] Abdul–Wahab SA, En SCF, Elkamel A, Ahmadi L, Yetilmezsoy K. A review of standards and guidelines set by international bodies for the parameters of indoor air quality. Atmos. Poll. Res. 2015; 6: 751–767.

[4] Hang J, Sandberg M, Claesson L. Pollutant dispersion in idealized city models with different urban morphologies. Atmos. Environ. 2009; 43: 6011–6025.

[5] Gromke C, Jamarkattel N, Ruck B. Influence of roadside hedgerows on air quality in urban street canyons. Atmos. Environ. 2016; 139: 75–86.

[6] Pope CA, Dockery DW. Health effects of fine particulate air pollution: lines that connect. J Air Waste Manag. Assoc. 2006; 56:709–742.

[7] Ramos-Bonilla JP, Breysse PN, Dominici F, Geyh A, Tankersley CG. Ambient air pollution alters heart rate regulation in aged mice. Inhal Toxicol. 2010; 22: 330–339.

[8] Wilson WE, Suh HH. Fine particles and coarse particles: ion concentration relationships relevant to epidemiological studies. J. Air and Waste Manag. Assoc. 1997; 47: 1238–1249.

[9] Grice S, Stedman J, Kent A, Hobson M, Norris J, Abbott J, Cooke S. Recent trends and projections of primary NO_2 emissions in Europe. Atmos. Environ. 2009; 43: 2 154–2 167.

[10] EEA. Air Pollution fact sheet 2014—Romania. Available from: www.eea.europa.eu [Accessed: 2014-12-23].

[11] Laj P, Klausen J, Bilde M, Plaß-Duelmer C, Papalardo G, et al. Measuring atmospheric composition change. Atmos. Environ. 2009; 43: 5351–5414.

[12] Kuklinska K, Wolska L, Namiesnik J. Air quality policy in the U.S. and the EU—a review. Atmos. Poll. Res. 2015; 6: 129–137.

[13] Gulia S, Nagendra SMS, Khare M, Khanna I. Urban air quality management—a review. Atmos. Poll. Res. 2015; 6: 286–304.

[14] World Health Organization. WHO air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Global update 2005. 2006. Available from http://www.who.int/mediacentre/factsheets/fs313/en/ [Accessed: 2016-06-23]

[15] US EPA. Environmental Protection Agency National Ambient Air Quality Standards. Updated in 2015. Available from https://www.epa.gov/criteria-air-pollutants/naaqs-table [Accessed: 2016-06-23]

[16] Iorga G, Balaceanu-Raicu C, Stefan S. Supporting material of annual air pollution level of major primary pollutants in Greater Area of Bucharest. Atmos. Poll. Res. 2015; 6: S1-S18.

[17] EURAD, AQI. Available from: http://www.eurad.uni-koeln.de/index_e.html

[18] (AQHI Canada) Environment and climate change Canada. Air Quality Health Index. http://www.ec.gc.ca/cas-aqhi/default.asp?lang=En&n=065BE995-1

[19] China REF AQI China. http://aqicn.org/city/nanjing/ [Accessed 2016-07-07]

[20] INDIA, Central Pollution Control Board. http://www.cpcb.nic.in/FINAL-REPORT_AQI_.pdf [Accessed 2016-07-07]

[21] US EPA. Technical Assistance Document for the Reporting of Daily Air Quality—the Air Quality Index (AQI). EPA-454/B-09-001. 2009; 31p.

[22] EUCITEAIR http://www.airqualitynow.eu/download/CITEAIR-Comparing_Urban_Air_Quality_across_Borders.pdf [Accessed 2016-07-07]

[23] EPA, AQI calculator_Concentration to AQI https://www.airnow.gov/index.cfm?action=resources.conc_aqi_calc [Accessed 2016-07-07]

[24] Plaia A, di Salvo F, Ruggieri M, Agro G. A multisite-multipollutant air quality index. Atmos. Environ. 2013; 70: 387–391.

[25] Plaia A, Ruggieri M. Air quality indices: a review. Rev Environ Sci Biotechnol. 2011; 10: 165–179.

[26] Cheng WL, Chen Y, Zhang J, Lyons TJ, Pai JL, Chang SH. Comparison of the Revised Air Quality Index with the PSI and AQI indices. Sci. Total Environ. 2007; 382: 191–198.

[27] Elshout S van den, Léger K, Nussio F. Comparing urban air quality in Europe in real time: a review of existing air quality indices and the proposal of a common alternative. Environ. Int. 2008; 34: 720–726.

[28] Glotfelty T, Zhang Y, Karamchandani P, Streets DG. Changes in future air quality, deposition, and aerosol-cloud interactions under future climate and emission scenarios. Atmos. Environ. 2016; 139: 176–191.

[29] OECD. The Economic Consequences of Outdoor Air Pollution: OECD Publishing, Paris; 2016. http://dx.doi.org/10.1787/9789264257474-en

[30] Leitte AM, Petrescu C, Franck U, Richter M, Suciu O, Ionovici R, Herbarth O, Schlink U. Respiratory health, effects of ambient air pollution and its modification by air humidity in Drobeta-Turnu Severin, Romania. Sci. of Total Environ. 2009; 407: 4004–4011.

[31] Haiduc I, Beldean–Galea, MS. Variation of greenhouse gases in urban areas—case study: CO_2, CO and CH_4 in three Romanian cities, in Air Quality–Models and Applications, edited by Popovic D, INTech, UK. 2011; pp. 289–318.

[32] Popescu F, Ionel I, Lontis N, Calin L, Dungan IL. Air quality monitoring in an urban agglomeration. Rom. J. Phys. 2011; 56: 495–506.

[33] Pehoiu G. and Murărescu O. (2012). Pollution and Air Quality in Târgoviste Municipality and Its Surroundings (Romania), Air Pollution—Monitoring, Modelling, Health and Control, Dr. Mukesh Khare (Ed.), ISBN: 978-953-51-0381-3, InTech.

[34] Grigoras G, Mocioaca G. Air quality assessment in Craiova urban area. Rom. Rep. Phys. 2012; 64: 768–787.

[35] Soporan VF, Nascutiu L, Soporan B, Pavai C. Case studies of methane dispersion patterns and odor strength in vicinity of municipal solid waste landfill of Cluj–Napoca, Romania, using numerical modeling. Atmos. Poll. Res. 2015; 6: 312–321.

[36] Iorga G, Balaceanu-Raicu C, Stefan S. Annual air pollution level of major primary pollutants in Greater Area of Bucharest. Atmos. Poll. Res. 2015; 6: 824–834.

[37] EEA, 2015. European Environment Agency. European air quality database, www.Eea.Europa.Eu/data-and-maps/data/airbase-the-european-air-quality-database-1 [Accessed 2015-07-24]

[38] Monks PS. A review of the observations and origins of the spring ozone maximum. Atmos. Environ. 2000; 34: 3545–3561.

[39] Stefan S, Zagar L, Necula C, Barladeanu R, Rada C. Assessment of surface-ozone in Bucharest, Romania focused on trends for three years. Environ. Eng. Manag. J. 2013; 13: 241–250.

[40] Salmi T, Määttä A, Anttila P, Ruoho-Airola T, Amnell T. 2002. Detecting trends of annual values of atmospheric pollutants by the Mann-Kendall test and Sen's slope estimates—

the Excel template application MAKESENS, Helsinki, Finnish Meteorological Institute. Report Code fmi-aq-31.

[41] EMEP/CEIP 2015 Present state of emissions as used in EMEP models; http://www.ceip.at/webdab_emepdatabase/emissions_emepmodels/. [Accessed 2016-05-03]

[42] EEA. Air Quality in Europe—2015 Report. European Environment Agency. 5/2015 www.eea.europa.eu [Accessed 2016-05-03].

[43] Ambient Air Quality Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe, Official Journal of the European Union L 152/11.6.2008, p. 1–44.

[44] Grivas G, Chaloulakou A, Kassomenos P. An overview of the PM_{10} pollution problem, in the Metropolitan Area of Athens, Greece: assessment of controlling factors and potential impact of long-range transport. Sci. Total Environ. 2008; 389:165–177.

[45] Vardoulakis S, Kassomenos P. Sources and factors affecting PM10 levels in two European cities: implications for local air quality management. Atmos. Environ. 2008; 42: 3949–3963.

[46] Harrison RM, Laxen D, Moorcroft S, Laxen K. Processes affecting concentrations of fine particulate matter (PM2.5) in the UK atmosphere. Atmos. Environ. 2012; 46: 115–124.

[47] Dimitriou K, Kassomenos P. A study on the reconstitution of daily PM_{10} and $PM_{2.5}$ levels in Paris with a multivariate linear regression model. Atmos. Environ. 2014; 98: 648–654.

[48] Jones AM, Harrison RM. Temporal trends in sulphate concentrations at European sites and relationships to sulphur dioxide. Atmos. Environ. 2011; 45: 873–882.

[49] Cusack M, Alastuey A, Perez N, Pey J, Querol X. Trends of particulate matter (PM2.5) and chemical composition at a regional background site in the Western Mediterranean over the last nine years (2002–2010). Atmos. Chem. Phys. 2012; 12: 8341–8357.

[50] Blanchard CL, Hidy GM, Tanenbaum S, Edgerton ES, Hartsell BE. The southeastern aerosol research and characterization (SEARCH) study: temporal trends in gas and PM concentrations and composition, 1999–2010. J. Air & Waste Manag. Assoc. 2013; 63: 247–259.

[51] Munir S, Habeebullah TM, Seroji AR, Gabr SS, Mohammed AMF, Morsy EA. Quantifying temporal trends of atmospheric pollutants in Makkah (1997–2012). Atmos. Environ. 2013; 77: 647–655.

[52] Cape JN. Surface ozone concentrations and ecosystem health: past trends and a guide to future projections. Sci. Total Environ. 2008; 400: 257–269.

Chapter 7

Air Pollution Mapping with Bio-Indicators in Urban Areas

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Additional information is available at the end of the chapter

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Abstract

Our study consists of the assessment and the mapping of the atmospheric pollution, which concerns the region of Tiaret city, by using the lichens as bioindicators. In our zone, we did a survey on 25 domains by using Kirschbaum and Wirth 1997 method. This method has enabled to define five classes according to the calculated air quality index (AQI). The dominant class was the one that contained a high pollution degree, which is reflected through the spatial distribution of the lichen species. The achieved map allows us to better visualize the pollution state.

Keywords: air pollution, mapping, lichens, bioindicators, Tiaret, Algeria

1. Introduction

The atmospheric pollution still constitutes a major problem worldwide, especially in the densely populated areas and near industrial sites [1]. More than 2 million people die annually around the world as a result of diseases caused by the air pollution and up to 1.3 million of this figure are inhabitants of the urban areas in developed countries [2]. According to the same source, 1.1 million of these deaths could be avoided if the regulatory limits and norms for air quality were complied with. Facing such a worrisome situation, detection and assessment of atmospheric pollution constitute an important environmental management goal in order to maintain acceptable air quality levels. Several methods exist to monitor compliance with air quality standards and one of them is biomonitoring where responses of living organisms to air quality [3] and environmental pollution in general are recorded [4]. This method can provide qualitative

and/or quantitative information about the levels of atmospheric pollution [3]. Biomonitoring demonstrates the concept of risk of pollution and provides a complementary tool for the assessment of the pollution's environmental impact [5, 6]. The use of biomonitoring can also help raise awareness among the public about the importance of the issue of environmental pollution [5, 6]. From the biomonitoring standpoint, four concepts at different levels of biological organization can be defined as follows [7]: biomarker, biointegrator, bioaccumulator and bioindicator. Lichens are the well-known bioindicators and constitute an ideal means for the air quality assessment. Among the lichen bioindicator methods, the approach of Kirschbaum and Wirth [8] is based on the differences in the lichen's sensitivities toward pollution.

Algeria like other countries in the world is facing a serious air pollution problem. In the case of the City of Tiaret, statistics given by the Algerian National Department of Health indicates that 32 new asthma cases are recorded there every month [9]. Many of the asthma cases are likely due to air quality problems and pollution. Impact of the air pollution on public health in Algeria is often overlooked. Hence health officials have to raise public awareness to cope with the problem. At the same time, assessment and monitoring of atmospheric pollution must be improved. It is against this background, that the purpose of the current study is to assess the atmospheric pollution in the region of Tiaret by the examination and census of the lichen flora. Very little work has been done on the subject of biomonitoring using lichens in Algeria. Therefore, this study is aimed at mapping the classes of lichen species in the context of their sensitivity to different levels of pollution. Specific objectives of the study include:

- to identify the lichen species present in the region of Tiaret, Algeria at the agglomeration level;
- to investigate the use of the chosen bioindicator species in the assessment of air pollution in the Tiaret geographical area;
- to estimate the atmospheric pollution according to the Kirschbaum-Wirth method using the collected lichen data; and
- to develop an air quality map of the region of Tiaret using the collected lichen data and estimates.

2. Materials and methods

2.1. The study zone

The study was conducted in the city of Tiaret, which is located in the northwest of Algeria between the mountainous Tell chain in the north and the mountainous Atlas chain in the south at an altitude of 980 m on average. The climate is Mediterranean and semi-arid with a mean annual rainfall of 400 mm/year. The prevailing winds are from the west and northwest, their average speeds range from 3 to 4 m/s. The population of Tiaret is quadrupled during the period from 1966 to 1998, with the actual numbers increasing from 37,990 to 167,000 inhabitants. As a result, the City of Tiaret has a population density of 136.10 inhabitant/km^2. The rate of

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population growth reached 3.66% per annum between 1977 and 1987, and this was accelerated to 4.11% in 1998 (NOS 1998). The predicted 2015 population of Tiaret is estimated to reach 213,551 inhabitants, and the population growth rate is likely to level off at 2% per annum [10].

Urbanization of the Tiaret city took place in two very different periods, which has resulted in two major and distinct areas in the city. Overall the structure of the city consists of concentric blocks of housing and commercial infrastructure that ultimately converge towards the old urban network of Tiaret City centre. The municipality of Tiaret administers a road network of 200 km, which carries high volume of permanent traffic. Parts of this road infrastructure have deteriorated in recent years. The city has established three types of housing environment zones, which in turn define the type of public roads. This is an indirect outcome of the fact that the network of Tiaret is not organized according to a spatial and functional hierarchy, i.e. it cannot be divided into main boulevards, secondary boulevards, primary public road, secondary public road, etc.

2.1.1. The automobile fleet of Tiaret city

Automobile traffic will have a strong influence on air quality. In December 2008, the automobile fleet of Tiaret city consisted mainly of old vehicle models that do not contain catalysts for removal of exhaust pollutants [11]. More recently, the automobile fleet of the municipality of Tiaret consisted of 31,178 vehicles and up to 75% of them were aged at 20 years or older. Out of this number, there are 1925 cabs, 19,756 gasoline-powered vehicles and 11,422 diesel vehicles. At present, the number of vehicles is almost four times that of 2006 when the total vehicle count stood at 8015 (registration vehicles cards branch of Tiaret, 2010: personal communication). These factors increase the possibilities of the emission of pollutants [12]. This conclusion is further supported by the departments of environment and public health. The road transport has been shown to contribute up to 30% towards the particulate pollution which is of public health significance by causing respiratory disorders in the City of Tiaret. Our work targeted the part of the City of Tiaret to assess the atmospheric pollution. **Figure 1** indicates various sites of research used in our study.

2.1.2. Typology of the monitoring stations of the air quality

Information for the classification and selection of sites for air quality monitoring stations were extracted from reference [13]. The urban monitoring stations were selected so that the data collected at these sites would allow the project team to evaluate the average population exposure level to atmospheric pollution in urban areas of Tiaret (**Figure 1**). Pollution levels detected by these stations should be representative of the average levels of the urban agglomerative pollution. The most important geographical locations for the monitoring stations should be in areas with the highest population densities (e.g. Badr city, Louz city, Rousseau city). Pollution in these urban areas will originate from surface and the mixed or combined sources include the following:

- residential, tertiary, commercial and institutional sources,
- road traffic sources,

- agriculture, forestry and
- others (including the natural sectors)

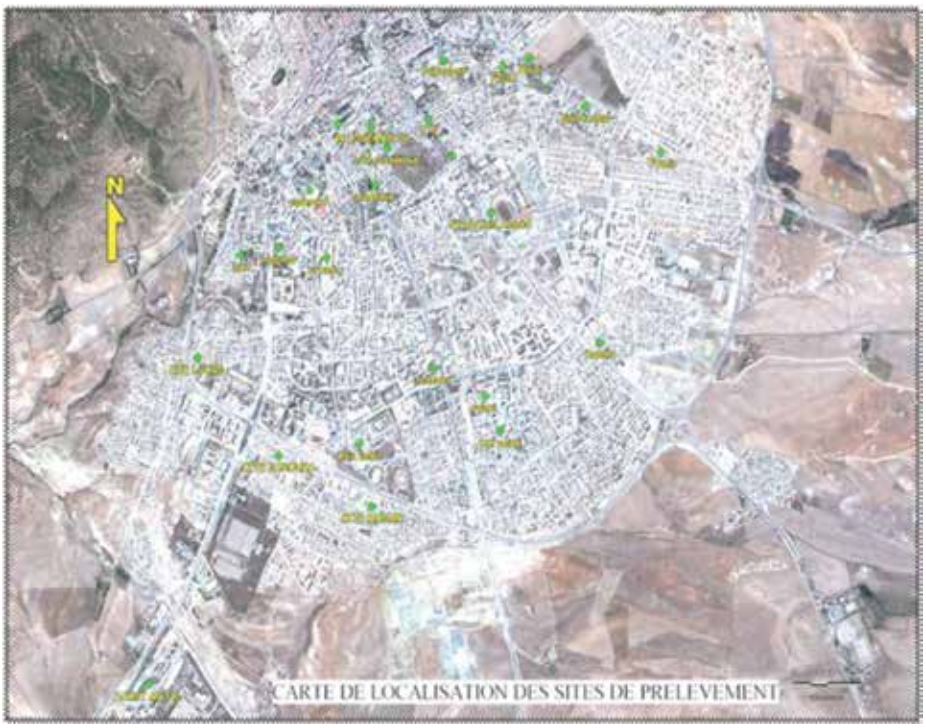

Figure 1. Localization of the research sites in the area of study (region of the city of Tiaret).

Traffic monitoring stations: The objective of these stations is to supply information on the concentration of pollution measured in representative zones of a road infrastructure. These stations can be installed at any type of the studied zone whether it is in a space of a rural or urban domination (Rahma, Volani, La Gare). The category of the issuer which the influence must be dominant on the station is the road transport.

Industrial monitoring stations: The objective of these stations is to supply information on the concentrations of moderate pollution that originate from a combination of several pollution sources and where the pollution accumulation tends to be highest in the vicinity of the industrial source of pollution (SN metal). Dominant sources of pollution will include the following: industry and waste treatment, production, transmission and distribution of energy and others.

National and rural stations: Exposure supervision of the ecosystems and the population in rural areas to the atmospheric pollution can be connected to the said cross-border pollutions (Titanic) using these types of sampling sites.

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2.1.3. Distribution of the population

The study of population distribution is very important for the interpretation of the examined phenomena and the outcomes of these phenomena. At the same time, it is significant in determining the population properties and its spatial distribution, as well as the use describing the said population. For the Tiaret Municipality, the spatial units where the population figures were collected were divided up into three spatial frames:

- County town agglomeration
- Secondary agglomeration (Karman, Ain Mesbah, Senia)
- The scattered zone

As stated above, the population of the municipality of Tiaret has experienced a high population growth rate with the majority being located in the municipality's urban areas. This puts pressure on the life system, especially the housing and the socio-educational equipment, the economy and also the employment opportunities.

2.2. Study design and experimental approach

This work was done over 5 months (from January to May, 2012); the big sections were achieved on different sites during this study. Field work consisted of the sampling site selection, selection of the tree species which carry the lichens, sample collection of said lichens and the exact GPS coordinates of all sampling sites. Once the sample collection was complete, samples were returned to the plant ecology laboratory at the Faculty of Natural and Life Sciences at Ibn Khaldoun University in Tiaret. The samples lichen species were then identified in the same facility using relevant catalogues, the key determination and selected chemical indicators such as K—reactivity with potassium hydroxide and CI—reactivity with bleach. The atmospheric pollution map was constructed using the resources of the URBATIA in Tiaret, Algeria.

To estimate the overall air quality in the urban areas of Tiaret, we adopted the method of Kirschbaum and Wirth [8]. Unless otherwise stated, the Kirschbaum and Wirth method and experimental approach are designated as the method in further text and all the information are from reference [8]. The method is relatively simple and cost-effective to use, and it is ideal for experimental air quality data collection in the municipality of Tiaret. Several stages must be carried out in the application of the method for the measurement and mapping of air pollution/air quality. First, determination of different study zones must be carried out. Second, the tree species, which have lichen symbionts, must be selected. Third, sample collection of lichen and their quantitative and qualitative analyses must be performed, while the final stage involves the mapping of the lichens distribution and the relation to air quality.

It is recommended that the study zones are defined using a 1 km domain, but this spacing can be increased or decreased depending on the given study area. Each domain should contain six judiciously selected trees, and these should be evenly distributed throughout in each domain. Selection of trees carrying lichens should adhere to the following criteria: all sampled trees must be isolated and subjected to the same environmental conditions such as luminosity, humidity and exposure to the wind. The bark characteristics and the development of the

lichens/lichen cover vary among various trees species. Therefore, it is advisable to perform lichen sampling on only trees belonging to a single species, if possible. Alternatively, lichens from one kind of bark in terms of its natural acidity and structure should be sampled. The tree(s) age should be recorded as the tree's average diameter or as the range between the minimum and maximum tree diameters, if more than one tree is joined together and sampled. For our study, the minimum circumference of the sampled tree was equal to 70 cm, and the lichen cover is allowed time to be developed as rate of lichen cover development is slow.

The trees that should be considered first as the best lichen carriers are the ash tree, the poplar and the lime tree. Oak, the sycamore maple and fruit trees, namely the apple tree, the walnut, the cherry tree, are also good lichen carriers. Willows and birches or plane trees are not good candidates as their bark is too acidic and detaches easily and the potential for lichen cover development is very limited, and these trees do not allow for monitoring air quality and pollution. For the current study, the sampled trees had the highest lichen cover and were the most representative of the vegetation in the particular domain. The sampled trees also had comparable characteristics among the different sampling sites. Sampling of oblique trees and those where surface wounds or the friction of the cattle were detected were also excluded from the sampling.

Two types of lichen analyses were performed. The qualitative sampling consisted of establishment of the identity of the lichen species. The qualitative lichen analysis was conducted by examining the species distribution over the most colonized trunk surface area equal to 20 × 10 cm. Quantitative evaluation is done through the measurement of the frequency of the tree colonization by a given lichen species. For this, a 10-compartment grid of 10 × 10 cm per grid was used to evaluate the colonization frequency of each lichen species identified. The samples were examined between 100 and 150 cm above the ground level to avoid the influence of animal excrement and artificial fertilizers on the results of the lichen analysis. Frequency of a given lichen species was recorded manually as the number of compartments that a given lichen species was detected in. The maximal frequency of a particular lichen species on a given tree was 10. This procedure is repeated for all the species present inside the grid and the given domain. The results of the quantitative analyses were recorded using indices. Every sample is the object of an index and the lichen frequencies measured were the object of such indices for a given domain.

On-site collection of lichen was done in one of two ways. The detached lichens (many of the terricolous lichens) were collected manually. The non-detached lichen species, i.e. all the fruticose lichens and most of the foliose lichens, were removed using a knife of a hammer, together with a small piece of the substratum the lichens were attached to. Care was taken not to damage the lichens attached to bark of trees. After sample collection, the lichen samples were placed in plastic bags, which are appropriate for the short-term storage and transport of lichen samples. The sampling location, the substrate and date were recorded on-site. In the laboratory, the fresh material was spread on a bench and allowed air-dry. Then the lichens can be without other precaution placed in the herbarium in a special envelope. Fruticose lichens take up a lot of space and they break easily. Thus, they must be placed in the herbarium when they are still wet and elastic, and only moderate pressure should be applied. In the samples of

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herbarium, the labels contained the following information: the place, the field if necessary the substrate (facilitate the determination), the date of collection and the name of the sample collector [8].

Lichen/sample coding was performed to avoid data confounding. For example, *Xanthoria parietina* E1, the domain LOUZ S16. Samples taken in the City of Tiaret were coded, in order to facilitate the data interpretation and avoid any confusion in it. Fifty species were identified and collected at the city of Tiaret and were coded accordingly to prevent confusion and problem in data interpretation. The actual sample coding of the lichen species is indicated with two columns, one for the species name, and the other for their codes. Samples made in the city of Tiaret were grouped under 25 domains coded with numbers (S1, S2, S3,…) (**Table 1**).

Station	Code	Station	Code
Fida	S1	Asia Kebir	S14
Habitat	S2	Rahma	S15
Academie	S3	Louz	S16
Cite Rousseaux	S4	URBATIA	S17
Ite	S5	PMI Volani	S18
Rue De Frigo	S6	40 Logments	S19
Boulice Amare	S7	SN metal	S20
Volani	S8	Voie d'evitement	S21
Terrain Boumediene	S9	Cite Zaarora	S22
Stade Kaid Ahmed	S10	Cite Manare	S23
Titanic	S11	La Gare	S24
Maidi	S12	Polyvalent	S25
Cite Badr	S13		

Table 1. The coding of the studied domain.

2.2.1. Identification of the lichens

Cuts of the thallus from all collected lichens were identified in the laboratory after examination with a binocular magnifying glass and a microscope. The following determinants were used: the form and the colour of the thallus and fructification, the presence of verrucosis whorl scar (isidium), floury mass (solarium) and other structures. In order to determine the crustose lichens, the use of the microscope is essential [8]. In certain cases, it is enough to bring a good magnifying glass with a magnification of 10× and test for reactivity with potassium hydroxide and little bleach. We note K+ when the potassium hydroxide (KOH at 20%) reacts (otherwise: K-) and C+ (C is for C1, the chlorine), when the bleach reacts (otherwise: C-).We put a quite small drop. At the same time, if the colouring seems fuzzy or not clear, it is absorbed onto a white tissue, making it easier to distinguish the obtained colour. During sampling, a small

knife is useful to lift or remove fragments but it is necessary to avoid damaging the trees and the rare or unique lichen species were not sampled in line with previously reported studies [14]. Taking these limitations into accounts, all the lichen species that were observed in the field were counted and recorded. Thus, the method guidelines were considered limiting and were modified to the conditions in the Tiaret geographical area.

Assessment scale of the air quality classes and map construction: The value of the air quality indices (referred to as AQI in further text) was represented by colours. And for this purpose, the values were grouped into classes. Each class has its own colour. **Figure 2** shows the pollution degree according to the assessment scale of the air quality as outlined in the method [8]. To construct the pollution map, once the identified species and the AQI were calculated for every domain, we have established a pollution degree scale for the different studied domains according to different colours.

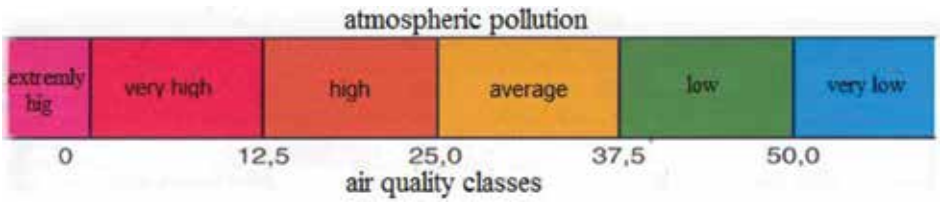

Figure 2. Assessment scale of the air quality classes (by Kirschbaum and Wirth method, 1997).

At URBATIA, all the collected data were exploited to construct a map of atmospheric pollution in the southern region of the city of Tiaret. Using the MapInfo®7.8 software, a map was constructed using the Kriging method interpolation and the vertical mapper™ between the concentration points in elements of the software and for the domain location. Setting of the map was done using AUTOCAD software. By applying the GPS data collected, the geographic coordinates (x, y) were obtained, and the z coordinates are represented by the AQI.

3. Results and discussion

The focus of our work was the assessment and construction of an atmospheric pollution map of the Tiaret city by using lichen bioindicators. We now use the AQI values to show different results that were calculated in each studied domain by the method [8], to allow us to classify the zones according to the atmospheric pollution degree. Then we will clearly indicate the space distribution of the listed lichen species. Then we will introduce the atmospheric pollution map achieved for this region. Finally, a comparison will be made with other previous research.

3.1. Results of the indices of the air quality (AQI)

In order to calculate the indices we have followed this way: Calculating the average frequency of each lichen species existing on the six studied trees and then the average frequency of each species will be added up and the total amount represents the index of the air quality.

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3.1.1. Example of calculation of the AQI in a domain

For illustrative purposes only, we show an example of the AQI for one of the studied domain (see **Table 2** for details), but it should be noted that the index is calculated as a whole for each of the 25 studied domains.

Station	**Academy Latitude: 35° 21'54.08" N Longitude: 1° 19' 09.81" E**						
	Altitude: 1006 m						
	Trees						
Species of lichens	1	2	3	4	5	6	Average frequency of the species
E1	0	10	10	8	0	10	6.33
E5	0	0	0	10	0	0	1.66
E11	10	1	0	9	10	0	5.00
E12	0	9	10	0	0	0	3.16
E16	0	0	0	0	7	0	1.16
E21	10	0	0	0	0	0	1.66
E23	0	0	0	0	10	0	1.66
E43	0	0	0	0	10	0	1.66
E61	0	0	0	1	0	0	0.16
Index of air quality: AQI (sum of frequencies)							22.5

Table 2. An example of calculation of the AQI in the domain ACADEMIE.

3.1.2. AQI results at all studied domains

For all the 25 domains, the total of the frequencies give the value of the AQI by which the pollution degree was assessed and this is done according to different classes (see **Table 3** for details). Results in **Table 3** demonstrate in a global way the air quality of each studied domain. If the value of the AQI is low, then the pollution is high and the status of air quality is critical or worst from the environmental management point of view, if AQI reaches the extreme value of 0. On the other hand, a high value of the AQI indicates a good air quality. According to the **Table 3**, the lowest pollution degree was recorded at the domain RAHMA (AQI = 0), where the pollution is extremely high. In this domain, an illegal rubbish container was identified and could be the source of the air quality problems. Some domains showed low air quality indices, impacted by a limited existing number of lichen species with a low covering rate, e.g. Cite Badr, La Gare, Cite Manare and SN metal. Domains of Badr and La Gare are overcrowded urban areas and busy road traffic junctions. On the other hand, domains of Cite Manare and SN metal contain high levels of industrial activity. These facts provide an explanation for the calculated AQI

An average AQI was recorded at the domain Kaid Ahmed stadium and Titanic, which are opened domains with low degree of building and population coverage. On the other hand,

the domain LOUZ indicates a low pollution degree (AQI = 38.5). This domain is situated close to the north region of the city and distant from the sources of emissions and the industrial units and road traffic, the fact that it is situated in a secluded area towards the city centre. From 25 studied domains, five pollution classes were highlighted. From **Figure 3**, it is clear that 68% of the total number (17 domains) of the studied domains exhibited a high pollution degree and 16% or four domains were very highly polluted. Only 8% (two domains) of the total domains sampled show an average pollution degree. While for the two pollution classes: the extremely high class (4%), the lower class (4%), they represent the lowest number of domains, and it is one domain for each.

Station	Code	AQI	Pollution degree	Type of station
Fida	S1	22.3	High	Urban station
Habitat	S2	20.67	High	Traffic station
Académie	S3	22.5	High	Traffic station
Cite Rousseau	S4	17.67	High	Urban station
Ite	S5	13.83	High	Traffic station
Rue De Frigo	S6	24.83	High	Traffic station
Boulice Amar	S7	23.83	High	Traffic station
Volani	S8	25	High	Traffic station
Terrain Boumedienne	S9	23	High	Urban station
Stade Kaid Ahmed	S10	30	Average	Traffic station
Titanic	S11	33.83	Average	National rural station
Maidi	S12	24.33	High	Urban station
Cite Badr	S13	12.16	High	Urban station
Assia Kebire	S14	18	High	Traffic station
Rahma	S15	0	Extremely high	Traffic station
Louz	S16	38.5	Low	Urban station
URBATIA	S17	18.1	High	Urban station
PMI Volani	S18	19.8	High	Urban station
40 Logements	S19	24.6	High	Traffic station
SN metal	S20	9	Very high	Industrial station
Rue Lourd	S21	23.4	High	National rural station
Cite Zaarora	S22	24.7	High	Industrial station
Cité Manare	S23	8.2	Very high	National rural station
La Gare	S24	11.6	Very high	Traffic station
Polyvalent	S25	19.5	High	Urban station

Table 3. The values of the AQI of the studied zone.

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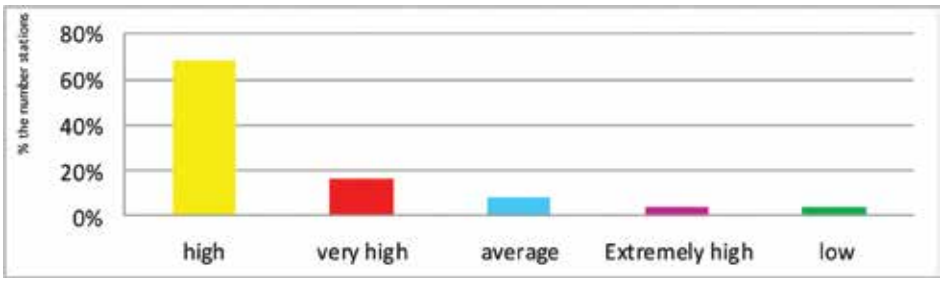

Figure 3. Evolution of the domain numbers according to the pollution classes.

Most of studied domains had an AQI that varies between 12.5 and 25, which corresponds closely to a high pollution degree [8]. In this area, we find different types of domains especially urban and traffic, it reflects an important population density and a significant road infrastructures.

3.2. Results of the pollution classes obtained on each type of domain

From **Figure 4**, we notice that, regardless of the domain's typology, the high and very high two classes are present. In an urban and traffic domain, the high class is mostly present (7/10 and 8/10, respectively). It is also presented in the national industrial and rural domain in a rate of one class per domain. About the high class, we notice it with a low rate (only one domain of each type). A class with an average pollution degree is indicated of each national traffic and rural domain. Though at the domain of urban type, slightly polluted class appears simultaneously, and a class of an extremely high pollution degree.

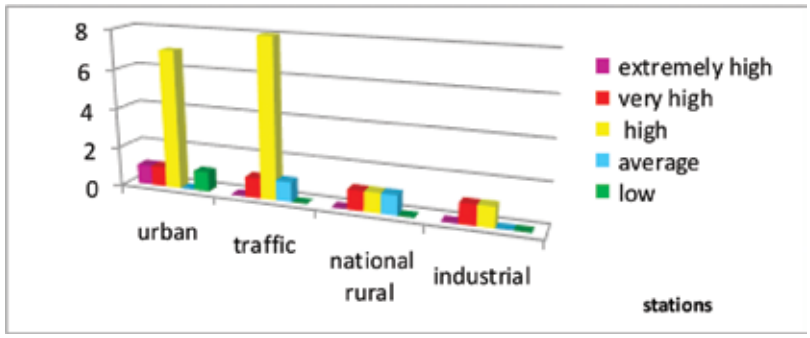

Figure 4. Pollution classes according to the typology of the studied domains.

3.2.1. Average results of AQI according to the studied stations

An average AQI is calculated by averaging the calculated AQI at the different domains of the same typology. From **Figure 4**, we note that the average AQI of all types of domains varies from 21.51 to 19.03, which corresponds to a high pollution degree. Even if these different

domains are in the same class (high), we notice that the urban type shows the highest AQI average (21.51) then the industrial type shows the lowest AQI of 20.33 and 19.59, respectively.

3.3. Spatial distribution of the lichen species

In our study zone, we have observed and identified 50 lichen species, which permitted us to conduct an overall evaluation of the pollution degree in the studied domains. In **Table 4**, the families of lichen species in all domains and the species number in all the families are summarized. Different lichen species are also identified according to the Thallus type, name foliaceous, crustacean, and basidiolichen, and the results are presented in **Figure 5**.

Station in common	AQI 2010 (pollution class)	AQI2013 (pollution class)	Notes
Volani	27.6 (average)	25 (high)	Degradation of air quality
PMI Volani	16.8 (high)	19.8 (high)	Stable air quality
L'Acadimie	20.6 (high)	22.5 (high)	Stable air quality
Cite Rousseaux	37.3 (average)	17.67 (high)	Degradation of air quality
Ite	20.2 (high)	13.83 (high)	Stable air quality
40 Logements	14.5 (high)	24.6 (high)	Stable air quality

Table 4. Comparison of the different AQI at the common domains.

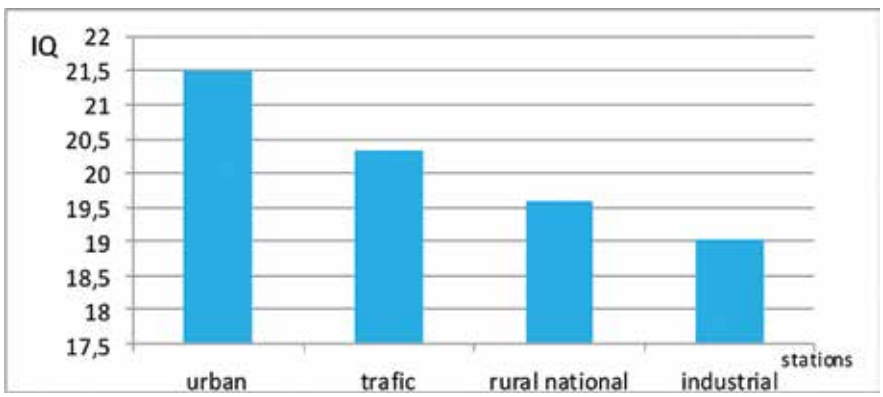

Figure 5. The species number of each thallus type.

Noting that the existence of *Xanthoria polycarpa* indicates a very high pollution, such levels can be inferred for the following domains: SN metal and La Gare. The *Physconia grisea* spp. found at the domains of Academie, Cite Badr and Assia Kebir was indicated as a very resistant species by Fadel et al. [15], which is the case in our study. *Physcia tenella* was found in Titanic domain, and it can be classified as moderately resistant, which reflects the pollution degree found at that station. Some domains like Cite Badr, La Gare, SN metal, Volani, Ite, contain very few lichen species, and thus, their recovery rate was weak. Those domains highly urbanized and

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experience busy road traffic, i.e. the lichen species will represent a high occurrence of diffuse sources of the atmospheric pollution. The overall extent of pollution is increasing. In domains with moderate and high average levels of pollution, a high number of lichen species was observed, i.e. indicating that the ecological conditions there are favourable to the development of the species. This conclusion is demonstrated by the recovery of two different lichen species from the domain of Louz, including the recovery of lichen species that have been shown to be susceptible to atmospheric pollution, namely *Physconia distorta* [16].

3.4. Presentation of the pollution map of the studied zone

After calculating the AQI of each domain, the data were used to produce the pollution map of the study zone, shown in **Figure 6**. Different pollution zones are distinguished by colour coding, which is explained below.

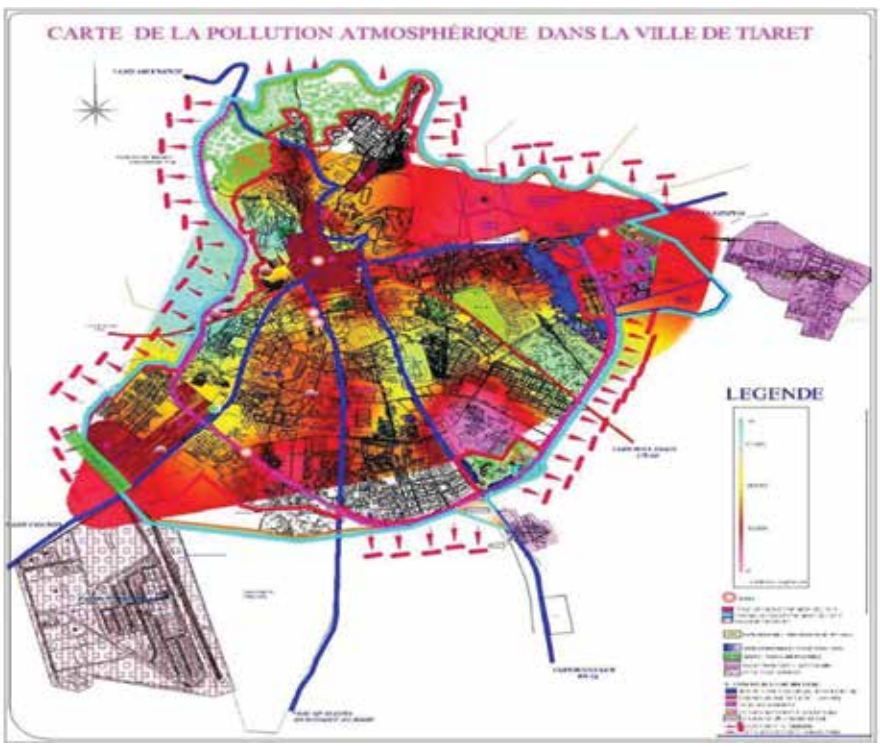

Figure 6. Atmospheric pollution mapping with lichens in Tiaret city.

3.5. Reading of the pollution map

Figure 6 provides a complete picture and evaluation of the atmospheric pollution level in the studied zones. An in-depth analysis of this map results in five pollution classes represented by five gradual colours.

Class 01 refers to an extremely high pollution, indicated on the map by the mauve colour, represented by only one domain Rahma with a pollution degree, AQI = 0 leads us to suggest that the air quality is low in this domain. This is similar to the data on lead pollution from road traffic in the Rahma domain [17]. The apparition of this class can be due to the hazardous waste and to the emanations of exhaust gases issued from the road traffic. Furthermore, this domain is characterized by a relatively steep gradient, which obliges the engine to develop more power and to consume more fuel, thus releasing more pollutants, bringing about a considerable increase in emissions, and a very important area of road traffic. The area also has a high concentration of gas stations with underground fuel storage tanks, which can result in leakage of gasoline and diesel and thus contribute to the atmospheric pollution from filling of reservoirs in gas stations [18].

Class 02 indicates a very high pollution in the domain Badr and La Gare, because of the high population density in these domains, domestic discharges (solvent evaporation), discharges from internal heating (especially in the winter), the road traffic emissions, as well as the important industrial emissions in the domain SN metal and Cite Manare. These findings are similar to those of reference [15], which have demonstrated that the peripheral domains of the urban network are highly polluted and indicated by the results of this study for the SN metal domain. Fadel et al. [15] also reported atmospheric transport of the pollutants from various industrial in the prevailing wind direction. This mechanism could provide an explanation for the pollution levels recorded in this study for the industrial domains, such as SN metal. Therefore, the class 02 sources of atmospheric pollution include those of domestic origin (Cite Badr), those from road traffic (La Gare) and finally those of industrial origin (SN metal). This is in line with findings from previous studies [1, 19, 20].

Class 03: the results of the third class indicate that this zone is affected by a high pollution degree. The affected areas belong to traffic domains (Academie, Volani, Assia Kebir, Habitat, Ite, Rue De Frigo, Boulice Amar, 40 Logements), Urban domains (Polyvalent, Fida, Cite Rousseaux, Terrain Boumediene, URBATIA, PMI Volani, Maidi), a national rural domain (Rue Lourd) and an industrial domain (Cite Zaarora).

These domains are the more representative on the map, and they appear by a yellow colour, which dominate almost the entire map (68%) of the studied zone. They are the busiest and more frequented of the entire city, which favour the pollution to reach a worrying peak. They are also subjected to a strong urbanization and an increase in the number of vehicles, entailing harmful effects on the environment. The last point mainly applies to traffic from old vehicles with a diesel engine and those who use fuels that does not correspond to the regulations of the environmental protection [11]. According to reference [21], the real conditions of the traffic are connected to the urban and rural circle and to the category of the road (highway, express way, medium-sized road, local network), to the function of the road (transit, distribution, residential access), to the mandatory speed limits, to the road's characteristic and to the traffic level (fluid, busy, saturated).

The number of vehicles in the national roads is densely; it is the case of the domain Rue Lourd in our study zone, the fact that the municipality of Tiaret which is conceived as a metropolis of the highland region is often frequented by all kinds of vehicles.

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The industrial zone Cite Zaarora is represented at this class (separately from the industrial zone SN metal, which is present in the class 02, corresponding to a very high pollution degree), this is probably due to the fact that Cite Zaarora, even if it is an industrial zone, still being less active than the SN metal zone, which explains its presence in this class.

One of the reasons that explains the high pollution level in some studied domains is the organizational form of the roads; it is applied with a lot of ignorance, the fact that the radial links are not ensured and the main network was not designed to support the current traffic. It brings about bottleneck between the south and the north (the case of the domain Academie, 40 Logements and Ite in our study zone), which means that the city centre, as an obligatory crossing point, is suffocated by the massive number of vehicles. It causes a network disorganization of the traffic, considered as an essential element of the urban planning [22].

At some domains of this class, for example, Cite Rousseau, PMI and Volani, wind is abated by hindrances formed by a high density of infrastructures, which fosters local pollutant accumulation. This is explained by the absence of the wind that contributes to the accumulation of pollutants close to its sources. In fact the buildings typography can disrupt the normal functioning of the wind and its trajectory and modifying the average characteristic and turbulence of the wind blowing [23].

We note at this class level the existence of the equipment (Maid), services (Volani), agricultural activities (Assia Kabir), and a large amount of schools (Polyvalent), where the pollution directly affects the human health.

Class 04 indicates that the pollution is moderate, it is represented by the blue colour at the domains: Stade Kaid Ahmed, Titanic and La Rue Des Freres Kaidi (W11), these sites are open areas, boosting the dispersion of the atmospheric fallout, which are transported by the wind. There is no topographic obstacle to stop them. It contributes to avoid the localized accumulation phenomena of the pollution. In fact, Loubet et al. [24] explained that the most adverse conditions of the atmospheric pollution dispersion meet when the wind speed is low or nil. Antipolis [22] also confirmed that the wind is an essential factor, which explains the dispersion of the pollutant emitted. It intervenes as long by its direction to orientate the pollution plumes than with its speed to dilute and to bring about the pollutant emissions.

Class 05 represented on the map by a green colour, which is localized at the domain Louz with a maximal AQI equals to 38.5. In fact, this domain is characterized by a low traffic road, its location is opened and wide, in which case the air is considered as slightly polluted. As we move away from the dense centre of the agglomeration (Louz) as the pollution level decreases. Maatoug et al. [17] have effectively confirmed that the opened sites are less polluted, favouring the dispersion of the atmospheric fallout which is carried by the wind.

Our less polluted domain (Louz) is situated far from the emission sources of the industrial units and the sources of urban emanations; this concept is confirmed by Fadel et al. [15], during their research on the bio assessment of the atmospheric pollution in the city of Skikda.

In this regard, and to support our discussion, it may also be considered necessary that we make a comparison between our research and that of Snouci [25] achieved on some common domains

with our study zone (14 common domains) and by using the same methods of the atmospheric pollution assessment.

In order to better illustrate the comparison between the two researches, **Table 4** shows the common domains with their AQI.

According to **Table 4**, we notice that, among the 14 common domains, six of them remained at the some pollution level, five have undergone air quality deterioration, while only three domains have undergone a slight improvement.

Then, about the domains which have shown a similitude of the AQI (Ite, PMI, Volani, Academie, Habitat and Rue Frigo), we note that they have kept the same pollution level in the two researches (high pollution).

The AQI of the domains Volani, Cite Rousseaux, Polyvalent, Assia Kebir and SN metal in our research has been decreased, compared to the calculated pollution level on 2010. We note that the domain Cite Rousseaux has been a subject of an important deterioration of the air quality (AQI decreased from 37.3 in 2010 to 17.7 in 2013), it is due to a heavy urbanization and to the increased rate of the number of vehicles in the city of Tiaret which is increased considerably since 2010. In fact, those domains are situated in the commercial districts of the city of Tiaret. This leads to strong road traffic in those districts by all types of vehicles that continue to increase. The other domain which has been an important cause for of deterioration of the air quality, is the domain SN metal (AQI decreased from 23.4 in 2010 to 9 in 2013) where the industrial emissions have been accumulated during this time, which represent the most important cause of the air quality deterioration.

Our study showed that the AQI of the domains; 40 Logements, Voie D'evitement, Terrain Boumedienne and Stade Kaid Ahmed have increased compared to the study of the year 2010; this is probably due to the elimination of some commercial activities, and to the transfer of certain administrative departments and the closure of some streets in these domains, which lead to a decrease in the road traffic.

In general, we can say that the atmospheric pollution in the region of the city of Tiaret had been in a sharp increase during the last 3 years (2010–2013). In our study, the apparition of a new class that corresponds to an extremely high pollution degree at the domain Rahma can justify such deterioration; due to the household hazardous waste, that are accumulated during a period of time at that domain, and to the population density, also to commercial activities (sale of building materials) and to the road traffic, which is multiplied along this period.

4. Conclusion

Our research focused on the assessment and the mapping of the overall atmospheric pollution in the region of Tiaret by using as bioindicators, lichen species and the total lichen flora. The calculated AQI. values in the 25 domain study zone led to breakdown of the sampling sites into five classes of pollution **(**Kirschbaum and Wirth method, 1997):

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- The first class is represented by the traffic domain Rahma with an API = 0 and extremely high pollution degree.
- The second class corresponds to a very high pollution degree shared between automobiles, industrial and urban domain, where the most representative domains are Cite Badr (AQI= 12.16) and SN metal (AQI = 9).
- The third class is the dominant class, and it corresponds to a high pollution degree with an AQI, which varies between 12.5 and 25. This class groups 17 domains of different typologies (urban, traffics, industrial, rural, and national) located in a severely dense agglomeration and we register the existence of two sites that belong to the industrial zone Zaarora and SN metal.
- The fourth class is called the moderate class, and it is represented by two domains: Stade Kaid Ahmed (AQI = 30) and Titanic (AQI = 33.83); scattered in urban agglomerations with a low population density and in the road sector, relatively low.
- The fifth class is the last class, which comprises only one domain (Louz) with a low urbanization, far from all types of emission sources, with an AQI = 38.5, it reflects a low pollution degree.

Afterwards, we have listed and identified 50 lichen species in the 25 domains in our study zone. The census results and the identification have obviously showed that their number and coverage rates are strictly linked to the pollution degree. The lichen distribution and speciation directly correspond to the pollution degree, based on the observation of the ground. We have classified the collected species in the study zone according to their crustose Thallus types (29 species), foliose (20 species) and basidiolichen (one species). Finally, a pollution atmospheric mapping of the part of the city of Tiaret is achieved according to different classes given by the AQI for the localization of the domains on the map.

In fact, the method permits mapping of pollution for vast geographical areas in a relatively short time, because of the epiphyte vegetation that we have noticed. The used approach also provides an indication about the average pollution in the Tiaret area over several years. Moreover, lichens give us an overall pollution picture in the atmosphere of the Tiaret municipality. Among the measures to improve air quality in the study area, we can cite the renewal of the car fleet which permits without doubt, a decrease in the pollutant emissions of the road traffic; and to improve the adjustment of the combustion used in engines, and to use less pollutant fuels. In a perspective of continuity of this study, it would be interesting to achieve the some work in an extended period of time to assess the pollution evolution during the time. It will also be necessary to achieve this study in partnership with other Algerian cities to estimate the average degree of the pollution in Algeria. It allows us to catalogue the Algerian lichen species. In conclusion, we can say that the bio indicators have provided us with very interesting information, which allowed the detection of the air quality degradation before this one affects severely biotopes or human.

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References

[1] Ramade F. 2005. Elements of Ecology. 6th ed. Dunod. Paris. pp. 83–218.

[2] World Health Organisation (WHO). 2013. Available from: http://www.who.int/mediacentre/factsheets/fs313/fr/ [consultation: 05/2013].

[3] Canha N, Almeida SM, Freitas MC, Wolterbeek HT. 2014. Indoor and outdoor biomonitoring using lichens at urban and rural primary schools. Journal of Toxicology and Environmental Health – Part A: Current Issues 77(14–16): 900–915.

[4] Tandlich R. 2011. 11th International Multidisciplinary Scientific GeoConference SGEM2011, Conference Proceedings. ISSN 1314-2704, June 20–25, 2011, Vol. 2, pp. 947–954. Available from: www.sgem.org.

[5] Garrec J.P. 2007. Vegetable Biomonitoring Pollution of Air and Water. Document Database. Technical Engineering. 62 p.

[6] Association for Supervision and Air Pollution Study of Alsace (ASPA). 2005. Biomonitoring. Report of Alsace Air. No. 8. Available from: http://www.atmo-alsace.net/medias/produits/Reportair_No8_La_biosur.pdf.

[7] Garrec J.P., et Van Haluwyn C. 2002. Plant Biomonitoring of Air Quality: Concepts, Methods and Applications. Ed. Lavoisier. Paris. 117 p.

[8] Kirschbaum U, Wirth V. 1997. The Bio-indicators Lichens Recognize and Evaluate the Quality of the Air. Ed. Les Editions Eugen Ulmer. p. 128.

[9] HDD, Department of Health (Tiaret). 2011.

[10] The Service Department of Tiaret (Algeria). 2012.

[11] Rahal F., Benharat N., Rahal DD., Baba Hamed FZ. 2009. The Influence of Traffic on Air Pollution in the City of Oran. International Symposium Proceedings Environment and Transport in different contexts Ghardaïa, Algeria, February 16–18, 2009.

[12] Maatoug M., Hellal B., Dellal A., Ayad N., Bourbatach M. 2007. Detection of air pollutants from road traffic by using the bioaccumulative effect of flora species

regarding some heavy metals (Pb, Zn, Cu). Pollution Atmosphérique. 196:. pp : 385–394.

[13] Environment Agency and the Energy Management (ADEME). 2010. The air quality in French cities : 2010 Report of the atmospheric index. pp. 7–9. Available from: http://www.ademe.fr/sites/default/files/assets/documents/77296_7219bilan_atmo2010.pdf

[14] Dorléans P. 2006. How to Measure the Urban Air Pollution by Observing Tree Trunks. Lycée Jacques Cœur. 6 p.

[15] Van Haluwyn. C., Lerond M. 1993. The Lichens Guide. Ed. Lechevalier. Paris. 344 p.

[16] Maatoug M., Medkour K., Ait Hammou M., Hellal B., Taibi. 2010. Cartography of atmospheric pollution by the lead from road traffic using transplantation of a lichen bioaccumulator *Xanthoria parietina* in Tiaret city (Algeria). Pollution Atmosphérique. 93–102.

[17] Madany I.M., Ali S.M., Akhter M.S. 1990. Assessment of lead in roadside vegetation in Bahrain. Environment International 16: 123–126.

[18] Fadel D., Dellal A., Djamai R., Laifa A. 2012. Biological estimation of the overall air pollution of a city northeast Algeria by the method of the index of atmospheric purity. Review Ecology Environnement 8: 59–75.

[19] Thibault J. 2003. The Air in Everyday Life: Theoretical and Experimental Approach. Ed. Odile Jacob. Paris. 234 p.

[20] El Yamani Mounia. 2006. Urban Air Pollution. Afsset. France. 6 p. Available from: http://www.cancer-environnement.fr/Portals/0/Documents%20PDF/Rapport/Anses/Afsset/2005_pollution_atmo_urbaine.PDF

[21] Sétra., CETE Lyon., CETE Normandie-Centre. 2009. Road Emissions of Air Pollutants. Information Note. pp. 4–14. Available from: http://www.infra-transports-materiaux.cerema.fr/IMG/pdf/0958w_NI_EEC_92_Emissions.pdf

[22] Gilles M. 2006. Modelling the Dispersion of Pollutants to Scale Intra Urban, Implementation of Morphological Indicators. 16 p. Available from: http://halshs.archives-ouvertes.fr/hal-00130986/document.

[23] C.E.T.E. 2010. The Dispersion of Pollutants to the Edges of Roads. The Air, Health and GES in Public Discussions of Road Projects. 2: 4. Available from: http://www.bv.transports.gouv.qc.ca/mono/1029473/02_Fiche_2.pdf

[24] Petit C., Loubet B., Rémy E., Aubry C., Duguay F., Missonnier J., Cellier P., Ali Feiz A., Blondeau C., Mauclair C., et Durand B. 2013. Local pollution, transport and agriculture. VertigO, Special Issue No. 15. Available from: http://vertigo.revues.org/12774. DOI: 10.4000/vertigo.12774.

[25] Snouci H. 2010. Mapping of Air Pollution in the City of Tiaret Using a Lichen Survey. Engineer Memory. Tiaret University. 68 p.

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