This project is co-financed by the European Union and the Republic of Turkey

Technical Assistance for Improving Emissions Control

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The Role of Emissions Dispersion Modelling in Cost Benefit Analysis Applied to Urban Air Quality Management: Part 1-the Approach

(Version 2: 18 May 2012)

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GLOSSARY OF ACRONYMS

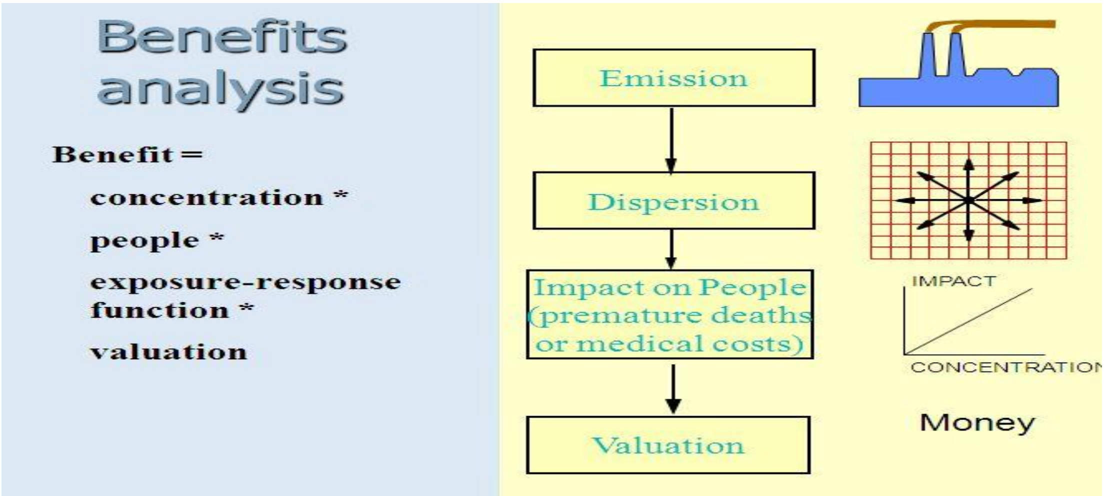
AQMA	Air quality management area
B/C	Benefit-to-cost ratio
BENMAP	Environmental benefits mapping and analysis program
BMBPH	Beijing Municipal Bureau of Public Health
CBA	Cost-benefit analysis
CMH	Chinese Ministry of Health
CO	Carbon monoxide
CV	Contingent valuation
DCF	Discounted cash flow
EEA	European Environment Agency
EIONET	European Topic Centre on Air Pollution and Climate Change Mitigation
EU	European Union
GDP	Gross domestic product
GIS	Geographical Information System
LAQM	Local air quality management
NE	North east
NECD	National Emissions Ceilings Directive
NO_2	Nitrogen dioxide
NOx	Oxides of nitrogen
NPV	Net present value
O_3	Ozone
PC	Personal computer
PM_{10}	Particulate material with an aerodynamic diameter of 10μm or less
PV	Present value
SO_2	Sulphur dioxide
TL	Turkish Lira
UK	United Kingdom
USEPA	United States Environmental Protection Agency
VOCs	(Non-methane) Volatile organic compounds
VOLY	Value of a life-year
VOSL	Value of statistical life
WTP	Willingness to pay

EXECUTIVE SUMMARY

The Project's Terms of Reference (ToR) regarding Activity 1.9 indicated that dispersion modelling should be undertaken as part of the process of developing national emission projections and identifying national emission ceilings. The TA Project Team and staff of MoEU engaged in inventory work discussed this requirement in the Inception Phase. Based on those discussions it was agreed that the described task was not relevant to Activity 1.11 regarding national emissions projections. It was also agreed that the results of the NECD emissions inventory would not be mapped as this would require emissions to be reported as a disaggregated data set, and it was not expected that this would be feasible – since confirmed. It was recognised that the use of dispersion modelling to help identify emission ceilings is more appropriately undertaken at a larger, regional scale. Indeed, the results of European Environment Agency (EEA) studies that investigate marginal damage costs arising from the (modelled) dispersion of emissions at the regional scale - EU member states plus countries to the east of the EU including Turkey – are utilised in the Technical Report on cost-benefit analysis (CBA) which is reported separately.

However, it was agreed that the preparation of a Report introducing the role of dispersion modelling in local and urban air quality planning linked to CBA would be appropriate. This output could be useful to the Ministry of Environment and Urbanisation (and the Ministry of Health) in the near-mid-term future. The present Technical Report, therefore, has been prepared to meet the requirements of the Project's Terms of Reference (ToR) regarding Activity 1.9 as interpreted in the Project's Inception Report and subsequent Progress Reports.

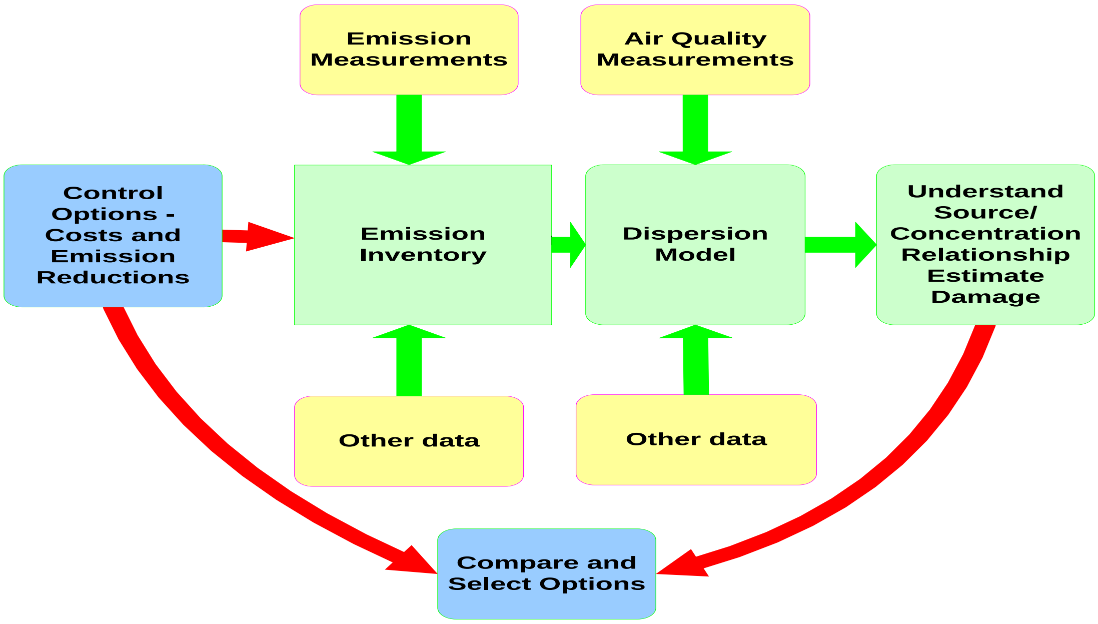
The present Report is the first part of a two-part Report that introduces the 'Impact Pathway Approach' – see chart below - and its application to local/urban scale air quality planning. An important consideration is the geographical scale of the air quality management plan to be prepared. This Report focuses on local scale and urban scale (model) applications as defined by the European Environment Agency (EEA)[1], i.e. local models/plans covering spatial areas of between 1-1000 m and urban scale models/plans covering areas of 1-300 km.



[1] European Environment Agency, 2011. The application of models under the European Air Quality Directive: A technical reference guide

The impact pathway approach involves tracing emissions from their sources through to their ambient concentrations in the atmosphere and assessing their impact on human health in physical and monetary terms. In this approach the significance of a pollutant abatement measure is evaluated, not according to the absolute reduction in emission mass load but in terms of its effects on ambient air pollutant concentrations and likely improvements in human health.

This approach may be applied through a process of air quality management as illustrated in the second chart, below, which links together certain key planning tools – emissions inventory, air quality monitoring data, emissions dispersion modelling, quantifying the effects of air pollutants and the cost-benefit analysis of those identified effects.



The present Part 1 Report introduces all steps involved in this process: it is comprehensive and may be read as a stand-alone document. Part 1 and Part 2 Reports are complementary - there is no duplication of material. Readers should start with Part 1 and, where interested in more in-depth treatment, secondly consult Part 2. Part 1 is organised into eleven (11) separate Sections.

Section 1 provides an overview of the whole impact pathway approach and its use in (i) basing air quality management planning decisions on strategic considerations – maximising the benefits to society relative to the costs to society and (ii) identifying the measures whose implementation will most likely achieve that outcome. The remaining Sections amplify the steps in this process.

Section 2 comments on the scope and requirements of the emissions inventory that is needed as an input to the emissions dispersion modelling step. Two approaches are introduced: (i) a bottom-up approach in which local emission source data are used and (ii) a top-down approach in which aggregated national emissions inventory data are disaggregated into defined areas.

Section 3 introduces the need for ambient air quality monitoring data as essential to (i) defining the air quality 'problem' to be addressed and (ii) providing the mechanism for calibrating the performance of the selected emission dispersion model.

Section 4 explains the practical steps involved in preparing emissions data as an (essential) input to dispersion modelling, focusing on mapping the emissions and their time resolution. Local-scale applications are first considered, followed by larger scale urban applications.

Section 5 summarises the dispersion modelling step and identifies the types of input data needed and factors that have to be taken into consideration when selecting and using an emission dispersion model. Application at local-scale and larger-scale urban areas forms a focus. Many topics are introduced, including:

- The dispersion modelling process
- The different types of dispersion model – Gaussian vs more advanced
- The need for emission source characterisation
- Meteorological considerations in emissions dispersion modelling
- Modelling of local-scale and larger-scale urban areas
- Descriptions of a number of selected emission dispersion models with a summary of their characteristics and suitable applications
- Factors involved in minimising modelling errors and uncertainties
- Model calibration – comparing measured with modelled data.

Section 6 presents a number of case studies in which emissions dispersion modelling has been applied to local and urban scale air quality management issues. They include local-scale studies in the United Kingdom (UK) and larger-scale urban studies in Beijing and Istanbul. The case studies illustrate a number of the issues identified in Section 5.

Section 7 provides a brief introduction to some of the techniques and tools that may be used to map and present geographically distributed ambient air quality data (monitored and modelled). This information is used in the assessment of population exposure to air pollution and in the CBA of abatement measures described in Sections 8 to 11 inclusive.

Air pollution has a range of negative impacts – on human health, on the natural and built environments and on agricultural productivity – but health impacts tend to predominate when impacts are expressed in economic terms. Section 8, therefore, introduces the range of physical impacts of air pollution on human health – illness (morbidity) and premature death (mortality); the concept of population exposure to pollution and its measurement; and exposure-response functions.

Section 9 introduces briefly the concepts involved in valuing the benefits of emissions reduction or other air pollution mitigation measures. Valuing the benefits of reduced illness and reduced premature death are illustrated using as case study an analysis undertaken in Fushun, China. The relative significance of different health impacts is also indicated.

Section 10 summarises the approach to measure cost-effectiveness and the range of cost curves that may be derived in assessing it. Section 11 then introduces the purpose of undertaking CBA and the elements involved. It introduces the concept of discounted cash flow (DCF) analysis applied to CBA and some commonly used terminology. It concludes with comments on the potential application of CBA to local and larger-scale urban air quality planning.

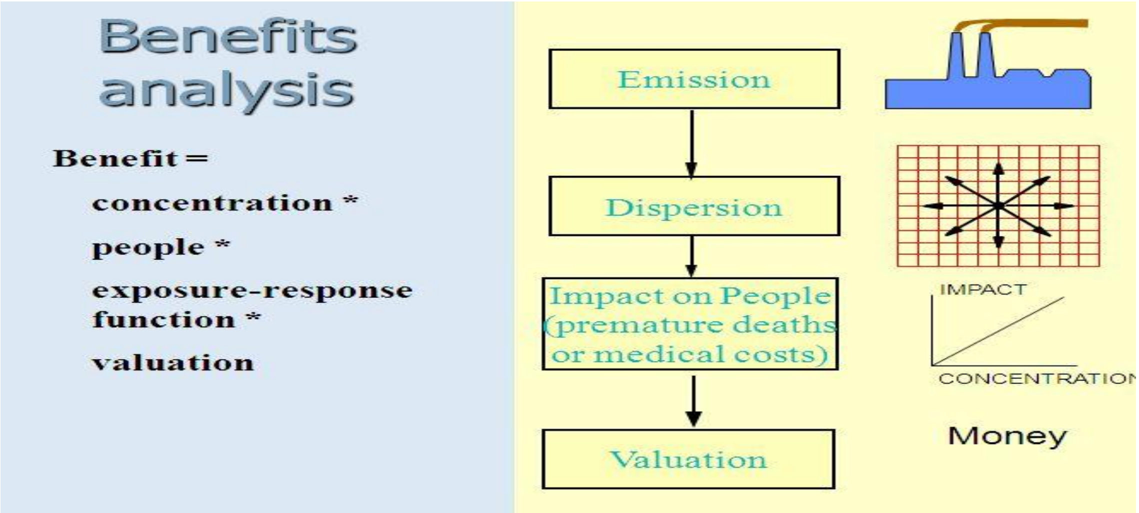
1 Overview of the Impact Pathway Approach

1.1 Introductory Comments

This report describes an approach used widely to help determine an optimal strategy for local air quality management for cities and regions subject to budgetary constraints. In particular, the Impact Pathway approach is used to relate the financial costs of air pollution control measures to their monetary benefits measured in terms of the value of the damage to human health avoided. Dispersion modelling plays an indispensable role in applying this approach.

The approach involves tracing emissions from their sources through to their ambient concentrations in the atmosphere and assessing their impact in physical and monetary terms on human health – the Impact Pathway Approach, see Figure 1-1. The significance of a pollutant abatement measure is thus evaluated in terms of its effects on ambient air pollutant concentrations and its consequences for improvements in human health.

Figure 1-1 The Impact Pathway Approach



Each step of the Impact Pathway Approach is summarised in the present Part 1 Report, which includes a number of illustrative examples and case studies. The Part 2 Report contains more detailed descriptions of a number of the steps and provides further theoretical background.

1.2 Impact Pathway Approach

The Impact Pathway Approach relates:

1. The pollutant emissions from individual and combined sources, to
2. Ambient air quality and pollution concentrations, to
3. The populations exposed to those concentrations, to
4. The physical damage caused to populations by this exposure, to
5. The monetary value of the physical damage caused.

The approach is used to assess the effects of emissions abatement measures on ambient air quality and the impact this has on the amount of physical damage caused by, and the monetary costs of, air pollution. It enables the costs of an abatement measure (e.g. switching to low sulphur fuels) to be compared with the benefits (e.g. avoided damage to human health). In this way, abatement options can be analysed, compared and ranked according to the net benefit they bring to society. The findings of such analyses are used to:

- Inform decisions on air pollution control policy, objectives and strategy.
- Identify priority actions for achieving the human health and welfare objectives of air quality policy at least cost.
- Define a priority action implementation plan that will maximise the benefits to society progressively over time subject to financial constraints.

The focus of the report is confined to the impacts of air pollution on human health. Effects can include increases in levels of premature mortality and in morbidity (e.g. lower respiratory infections). Similar approaches can readily be used to extend it to impacts on other receptors, including natural ecosystems; agriculture, fisheries and forestry; and man-made structures (e.g. bridges and buildings). Impacts on human health are determined by establishing:

(i) The relationships between pollutant emissions and ambient pollutant concentrations

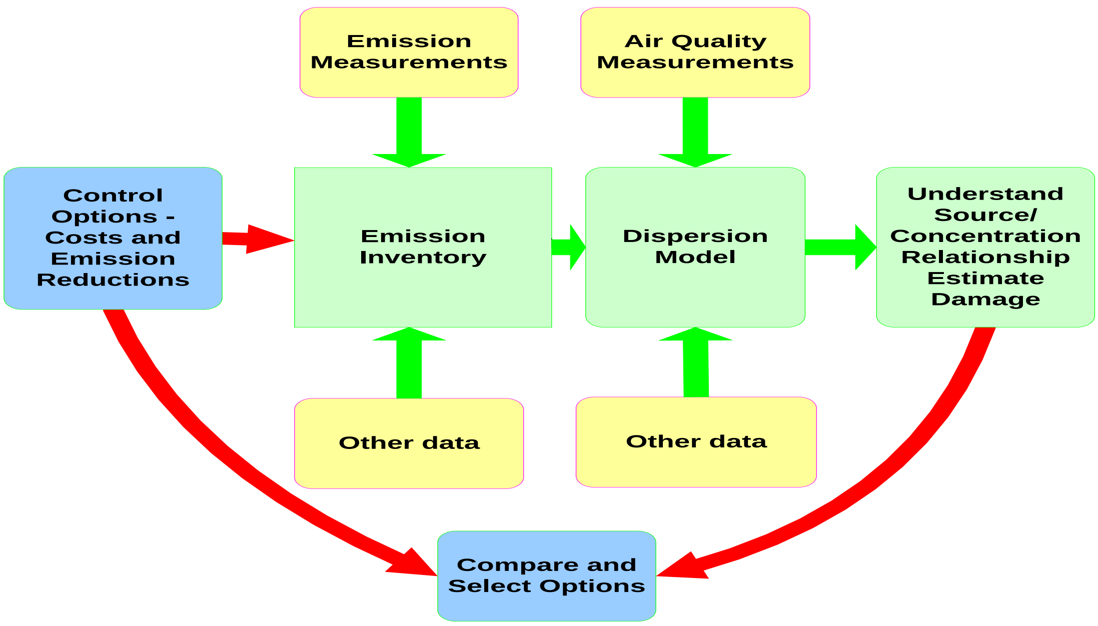
(ii) The relationships between ambient pollutant concentrations and damage to human health. Effects are determined by the exposure of the population to the pollutant and the exposure-response functions calculated for the pollutant

(iii) The economic costs of the damage air pollution causes to human health and reductions in these costs brought by pollution abatement measures

1.3 Strategic Air Quality Management Planning

Strategic air quality planning follows the steps summarised in Figure 1-2, which identifies a number of clearly defined stages introduced below:

Figure 1-2 The Air Quality Management Process



***Stage 1 - Emissions Inventory*:** A compilation of data on emission sources is an essential air quality management planning tool. It enables links to be made between pollutant sources, emissions and concentrations so that the effect on pollutant concentrations of controlling a pollution source can be predicted.

Stage 2 -Air Quality Monitoring Data: they are needed to establish the base-case ambient air quality conditions and pollutant concentrations and for calibrating the dispersion model.

Stage 3 - Dispersion Model: this is used to predict the spatial and temporal distribution of pollution based on an understanding of the meteorology and dispersion characteristics of pollutants. When compared and validated against the monitoring data the model can determine the contribution of individual emissions sources to the overall pollution concentration levels across the study area.

Stage 4 - Economic Model: this takes the outputs from the dispersion model (pollutant concentrations by geographical location) and calculates the exposure of affected populations to specific pollutants (typically PM_{10} or SO_2). Exposure-response functions convert levels of pollutant exposure into impacts on human health. The cost of the damage caused to human health is then calculated and given a monetary value.

Stage 5 - Cost-Benefit Analysis: By using the emissions and dispersion models, the effects on pollutant concentrations of applying abatement measures to specific pollutant sources are estimated and the consequential changes in health impacts quantified and measured in monetary terms. By comparing the costs of abatement measures with the value of the benefits achieved, the net value to society of different measures can be determined. Measures can then be ranked according to the value they add to society.

Stage 6 - Action Planning: Results of the impact-pathway analysis are used to help formulate a practical air quality action plan taking into account the relative benefits to society of alternative measures, as well as any social, institutional, technical, political or financial constraints on their use.

1.4 Local and Larger-Scale Urban Air Quality Management

An air quality management action plan aims to improve air quality in a local or larger scale urban area. This can have the objective of achieving specified air quality standards or objectives, or to improve air quality as much as is ‘economically efficient’. Although fine in theory, establishing the ‘economically efficient’ level in practical terms is very difficult and uncertain. Normally, therefore, the purpose is to comply with predetermined standards or objectives set through government regulation.

Preparing an air quality strategy / action plan involves three phases: (i) assessment (ii) options analysis and (iii) strategy definition and implementation.

1.4.1 Assessment Phase

In order to comply with air quality standards or to meet specific objectives it is necessary to establish what the existing level of pollution is and to form a strategy for reducing it.

An emissions inventory is set up to characterise pollution sources and to generate the emissions data needed within an air quality dispersion model to predict air pollutant concentrations across a geographical area.

At the same time, an air quality monitoring programme is initiated. This measures ambient pollutant concentrations from which the results from the dispersion model can be validated and ambient air quality conditions monitored on implementation of the strategy.

The assessment phase is used to:

- Identify and characterise pollution sources
- Quantify the emissions from those sources
- Determine ambient air quality conditions (monitoring)
- Identify pollutant source-ambient air quality relationships (modelling)
- Establish the relative importance of sources
- Assess the exposure of people, ecosystems, crops and structures to pollutant concentrations
- Quantify impacts using exposure-response relationships
- Evaluate the impacts in monetary terms.

Once the above information is available, the effectiveness of measures to improve air quality can be evaluated. For many studies today, this includes an economic assessment of the costs and benefits of the measures.

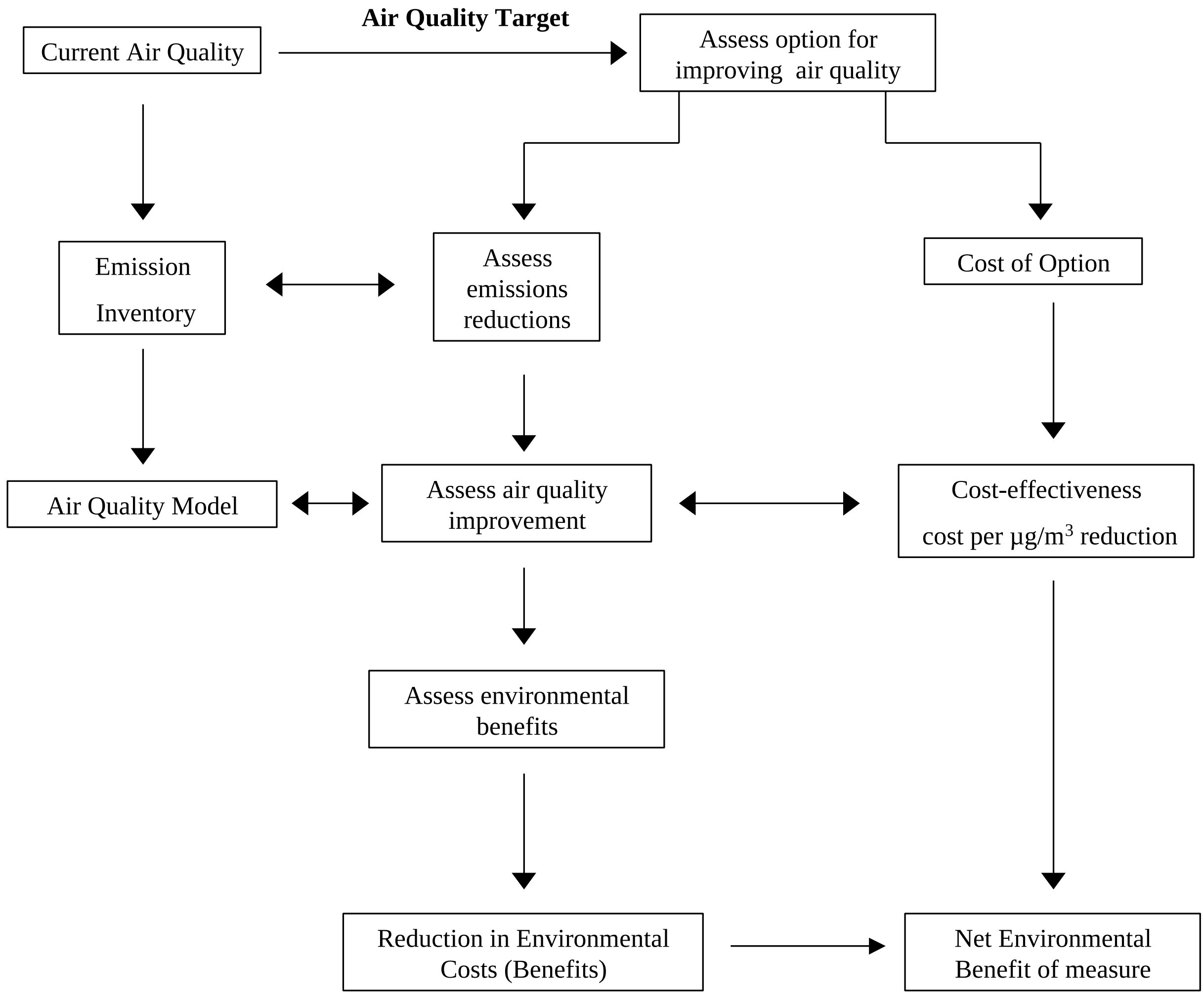
1.4.2 The Options Analysis Phase

At a more detailed level, an air quality strategy also needs to be able to assess and choose between different abatement options. A potentially large number of such measures could be implemented to help improve air quality. A key element is to compare these options and to select the most appropriate combination of measures to achieve the air quality targets or standards. The Analysis of Options Phase can include the following elements:

- Identify short and long-term abatement options, e.g.:
    - energy efficiency and conservation
    - transport planning
    - fuel switching
    - process change/cleaner technologies
    - end-of-pipe solutions
- Estimate changes in population exposure, health effects and damage costs for each abatement option
- Establish the costs of each abatement option and undertake cost-effectiveness and cost-benefit analysis as required
- Rank the abatement options according to their cost-effectiveness (least-cost criterion) or net benefit (cost-benefit analysis)

The aim is to achieve the air quality targets or standards in the most effective way, taking into account both the costs and the benefits of alternative pollution control measures. Figure 1-3 illustrates how an air quality dispersion model is used to assess the impacts on air quality of alternative abatement measures.

Figure 1-3 Using an Air Quality Dispersion Model to Assess the Impacts of Abatement Measures



1.4.3 *The Strategy Development and Implementation Phase*

The final phase draws on the assessment and options phases to construct a feasible strategy for achieving air pollution control options progressively over time in a way that maximises benefits to public health subject to budgetary constraints.

Abatement measures should ideally be ranked according to their net benefits to society. If a cost-benefit analysis is not undertaken, the next best approach is to rank measures according to one of the cost-effectiveness techniques. These aspects are explained in further detail in the Report.

2 THE EMISSIONS INVENTORY

2.1 The role of emission inventories in Local Air Quality Management

Emission inventories are not only important for demonstrating compliance with national emission ceilings, but also for providing input into air quality modelling. This section describes the role of the emissions inventory in the development of a local air quality management action plan.

Emission inventories represent a quantified assessment of where and why pollutants are released into the air. Following an emission, pollutants are mixed in the air, which gives rise to concentrations, exposure, to pollutants and ultimately to health impacts. So the first step in understanding the overall process is to be able to quantify the emissions, and use this as input into dispersion modelling.

However, this usually requires the emissions to be mapped, and with a better time resolution than the typical annual emissions data. So there are fundamental differences between the emission inventory that is needed at the national level, and that which is used for local air quality management, both in terms of application and the creation of mapped emissions.

To created mapped emissions, the emissions from each source are combined with geographical information to allocate emissions to particular areas or locations. For example, emissions from the domestic sector might be combined with a map of population, or housing density, to determine how the emission total should be distributed across the map. The geographical coverage, in this case population, provides a way of “weighting” the emissions that are allocated to each cell in a grid. Different sources are mapped in different ways. Emissions from a power station are different to those from the domestic sector, and are mapped as arising from a single point. The ways in which different sources are mapped is explained in more detail in Section 4.

It is possible to estimate the future impacts of introducing policies and measures by estimating the change to emissions from the relevant sources. The projected emissions can be used as input into the dispersion modelling studies to show how effective the policy is in addressing air quality problems at a specific location.

Emission inventories are therefore an important part of policy formation at the local scale. Furthermore, the inventory of emissions can be used to monitor the progress or effectiveness of policies and measures which have been introduced. For example, if a local measure is introduced to reduce the traffic flow along a section of road (because pollutant concentrations are considered to be unacceptably high), then it is the inventory which provides a quantification of how the emissions have changed across the historic time series.

2.2 The scale of the study area

2.2.1 Introduction

Air quality modelling is usually undertaken at the local scale, but the extent of the term “local” requires some further clarification. For some studies, the extent of the modelling may be a short section of road or even a single industrial installation. At the larger end of the local scale, modelling may be conducted over a larger-scale urban area. These differences are considered in some detail in Section 5.

These different scales usually require different approaches when compiling emission inventories. For assessing very small areas or a “hotspot”, e.g. at a traffic junction, it is best to use a “bottom-up” approach. Where the study area extends to several square kms, perhaps a city centre, it may be more appropriate to use a “top-down” approach. These terms are explained below.

When assessing air quality at the city level it is common to use both approaches. Initially emission maps at the city level can be compiled to identify the general spatial pattern of emissions, and identify areas where there are likely to be the most significant air quality issues. Further work can then be conducted at these specific locations at a much more detailed level.

2.2.2 Street scale emissions (bottom-up approach)

Where the study area is very small, it is sensible to consider each of the sources individually, and sum them to give emission totals. So, as an example, the emissions along a section of road can be determined from traffic count data, information on traffic speeds, and an assessment of the characteristics of the vehicle fleet.

The advantage of using a bottom-up approach is that it is more detailed, and therefore can be considered to provide a higher level of accuracy.

A disadvantage is that compiling information on each of the individual emission sources can be resource and data intensive. So it is generally only sensible to use this approach for areas of a limited size. Furthermore, the bottom-up approach might not necessarily be consistent with other emission estimates compiled at a larger scale.

2.2.3 City scale emissions (top-down approach)

A “top-down” approach can be used where national emission inventories are spatially disaggregated to provide either emission maps or regional/city level emission totals. These totals (either given directly, or summed from the relevant area from the emissions map) can then be used as the emissions inventory for the study area.

An advantage is that the top-down approach is generally a time-efficient way of creating an emissions inventory at the city scale, rather than considering each of the sources of emission, and summing them. A further advantage is that it does ensure consistency with data that have been compiled at larger spatial scales.

A disadvantage of the top-down approach is the limited extent to which data can be spatially resolved from the larger scale. The accuracy of emissions estimated for e.g. one section of road from national or regional emission totals should be considered carefully.

2.3 Local Emission Inventories: Spatial and Time Resolution

At the national level, emission inventories are generally compiled with an annual time resolution, and only provide totals at the national scale. When assessing air quality at the local level, it is necessary to conduct the work at finer spatial and temporal resolutions.

At the city level, it is common to use a 1km x 1km grid to present emission maps. This allows spatial variations to be clearly presented, without requiring an excessive amount of data processing. However it should be remembered that the accuracy or reliability of the emission map is not determined by the resolution of the grid which is used to present the data, but the spatial detail of the data that is used to map the emissions.

Some sources of emission vary considerably across short timescales. For example, emissions from road transport vary across the seasons, across the days of the week, and even on a minute by minute basis. By comparison, emissions from some industrial installations can be almost constant with time, reflecting constant levels of production or output.

The time resolution of the emission inventory that is required very much depends on the application for which it is being used. Annual emission estimates may be sufficient for obtaining a general overview at the city level, but for considering air quality at a single traffic junction, hourly estimates are probably needed

The time resolution of an emissions inventory can be constructed in much the same way as the bottom-up or top-down approaches. Either an annual estimate can be divided into finer resolution time periods e.g. annual to monthly, then to weekly, daily and hourly. Alternatively, it may be possible to use information on emissions at very detailed time resolutions. For example, traffic flow data or emissions from a specific industrial installation may be available on a minute by minute or hourly basis.

Information is included in Section 4 on the practical steps that are needed to handle data to generate the most appropriate emissions datasets. This requires a consideration of the different concepts that have been presented above.

2.4 The use of different assessment metrics: concentration limit values versus exposure or emission reductions

Historically, concentrations have been used as a metric for air quality targets at the local scale. This is because concentrations can be directly linked to exposure and the resulting health impacts.

However, in determining the success of improving air quality at the local level, increased attention is now given to the extent of emission load reductions, not just the achievement of specific concentration values. Primarily, this is because there isn't any specific concentration value of PM that is considered absolutely "safe" – merely that exposure to a lower level of PM brings lower risks to health. So, the extent to which exposure to PM occurs is increasingly considered as a more suitable metric.

3 AIR QUALITY MONITORING TO SUPPORT MODELLING

3.1 Scope and Purpose

The following components or steps are essential for establishing a monitoring programme that could be used to support dispersion modelling undertaken as part of a CBA:

- Select appropriate pollutants
- Determine the averaging times of interest
- Fixed or mobile monitoring site
- Select the monitoring technique
- Design the network and site the stations.

3.1.1 Selection of pollutants

Urban areas can have hundreds of different air pollutants in a typical situation involving contributions from road traffic, industry, fuel combustion and natural sources. Most monitoring campaigns in urban environments in the EU focus on NO_2, SO_2, and PM_{10}, so as to compare the results against prescribed standards.

3.1.2 Averaging times

A major criterion for selecting the monitoring method is the averaging time which can range from 1 hr or less (requires monitors with continuous sensors), through a few hours to a day or so (typically 24 hrs, active sampling followed by laboratory analysis of the collected sample), through to several days or weeks (active or passive samplers with subsequent analysis). Averaging times vary according to the selected compounds but must also be compatible with the objectives of the study: see Part 2 Report for further detail.

3.1.3 Fixed versus mobile monitoring sites

Fixed and mobile stations serve very different purposes.

Fixed stations provide a 24hr, 365 days per year measurement profile at one location. Diurnal and seasonal influences on concentrations are easily calculated using the quite time-resolved data. This information is very useful for determining compliance with air quality standards and identifying long term trends in ambient air quality.

Mobile stations typically consist of vans or caravans that are used for days or weeks at a fixed site, but can also be used to gather samples while in motion, assessing pollution from traffic or other sources. The data generated from such sites is more useful for investigating short term issues or problems.

3.1.4 Monitoring techniques

A range of continuous, semi-continuous and sampling/analytical techniques are available and most air quality assessment programmes will likely require a combination of these. The choice of equipment at a site level is normally influenced by the objectives of the programme, cost, manpower, lab/computer facilities and power/shelter for the monitoring equipment.

3.1.5 Network design and station locations

Network design and the siting of stations depends on the aims of the monitoring programme. If the aim is to evaluate risk to human health through population exposure estimates (perhaps through dispersion modelling), the monitoring should accomplish the following objectives:

- Provide data for exposure to pollutants in residential and hotspot locations
- Provide data for calibrating and modifying dispersion models
- Provide data that can monitor changes in air quality resulting from abatement measures.

When monitoring data are used for calibration of dispersion models the sampling sites should be located selectively at sites that are representative of human exposure or specific source activities. Examples of such sites may include residential areas, industrial or traffic sites, or urban background sites.

This approach allows measured concentrations to be used as the basis for dispersion model calibration. With proper selection and location of the monitoring sites it is possible to use the comparison between measured and modelled concentrations to assess the contribution of local sources. This is an important input to the development of abatement measures or policies.

Note that where a monitoring regime has been put in place to comply with the requirement of a European Directive, this does not make it automatically suitable for dispersion model calibration across wide areas. Dispersion modelling studies are often concerned with quite small areas and may require bespoke monitoring campaigns to be undertaken prior to commencing any modelling.

3.2 Using monitoring data for dispersion model calibration

Air quality models are rarely used without any reference to measurement data. Many modelling applications will use monitoring data for validation purposes, i.e. will compare modelled and measured results (often reverting to statistical methods). Modelling results are particularly useful in assessments to provide supplementary information for the geographical area (on a certain spatial scale) not covered by measurement data.

The approach to model calibration using monitoring data varies according to the pollutant and the averaging time of interest. For instance calibrating a localised dispersion model which is mainly concerned with road traffic may simply involve comparing the monitored annual mean pollutant value (say for PM_{10} or NO_2) with the measured annual mean value. If the air quality standard we are interested in is an annual mean, the model is simply refined in an iterative manner so that agreement is achieved with the measured annual mean. Once model agreement has been achieved, the dispersion model outputs can feed forward into exposure calculations and CBA.

In some instances it is desirable to assess the dispersion model performance at a shorter time-scale e.g. checking performance on an hourly or daily basis in relation to a large industrial source. However, this represents significant challenges as, in most instances, emissions data are not well defined temporally.

In the UK, for example, it is common practice to calibrate models based on comparison with annual mean values where road traffic is the most important source. For industrial sources it is less common to use monitoring data for model calibration though it is possible.

4 Emissions Mapping and Time Resolution

This Section explains the practical steps in preparing emissions data that can be used in dispersion models, focusing on mapping the emissions and time resolution. Local-scale applications are first considered, followed by larger scale urban applications.

4.1 Local-level emission inventories

4.1.1 introduction

The way in which street level emission inventories are compiled is very different to inventories at the city scale. Rather than taking a top-down approach (starting with a total emission load, and then dividing it to assign emissions to particular grid cells or time periods – see Section 4.2), a bottom-up approach us used, in which emissions from individual sources are estimated and then summed to give totals.

The focus below is on emissions from road transport because this is almost always the most significant source in urban areas. However domestic combustion and other sources can also be significant.

4.1.2 Defining the extent, the spatial resolution and the time resolution

The extent of the street level study area is considered in Section 5 below. It may be a length of road, or simply a road junction. It is important to compile the emissions information at spatial and temporal resolutions that are sufficiently detailed to meet the needs of the study.

So, for example, roads are typically included in the emission map as a line. This requires traffic flow data to be available to be able to determine the emissions along the length of the road that is being considered. For particularly detailed studies it may be that individual carriageways of the road are considered, and this then requires the traffic flows for each carriageway to be determined.

The time resolution of the emission estimates will need to match the time resolution of the dispersion modelling. For emissions from road transport, the main variable that changes with time is the traffic flow. Measurement data is usually averaged to hourly values, and this is sufficient for most modelling applications.

4.1.3 Compiling emission totals from localised measurement data

An overview of how emissions are typically estimated for a street level study is given below, starting with traffic flow data. Traffic flow measurements give the number of vehicles passing along the section of road, typically on an hourly basis. It is common for such data to also include information on the different types of vehicles that pass each hour. However, this identifies only some of the information that is needed for emissions calculations. For example, a traffic count point will distinguish between a car, a small van or a lorry. But it will not provide information as to which vehicles are using petrol or diesel, or which Euro standards should be applied to the vehicle fleet.[2].

[2] More sophisticated traffic count systems, such as Automatic Number Plate Recognition (ANPR) do provide this information by cross referencing number plates to registration databases. These systems are relatively expensive to operate, but do provide considerably more detailed information, allowing more reliable emission estimates to be made.

So it is necessary to obtain information on the vehicle fleet. More specifically information on the petrol/diesel mix of the fleet, and the age profiles will be needed so that the appropriate emission factors can be used when making the emission calculation. These data can usually be obtained from national level data, but local variations should be considered. For example, the road being studied may be in a location where local residents are not particularly wealthy, and therefore the average age of the cars using the road is considerably higher than the national average. These kinds of localised factors can make a substantial difference to the emissions that are calculated.

To calculate emissions, traffic flow data is combined with the length of the road to give the total number of kms (by vehicle type) in an hour. This km data is then combined with information on the vehicle fleet (age profiles and petrol/diesel split), and the relevant emission factors to give pollutant emissions from the road vehicles for the corresponding hour. These data are then used as input into the dispersion model.

It is likely that the dispersion modelling study will include the assessment of different scenarios. So it is important to be able to retain the detail of the emissions calculations. This would then allow changes to be made to specific elements of the calculation, and revised emissions to be determined. Some example scenarios that might be investigated to control the levels of emission are:

- **Limiting the traffic flow**: The impact on emissions can be determined by simply reducing vehicle kms from all or different vehicle types.
- **Smoothing the traffic flow**: The impacts of this scenario can only be investigated if the emissions are calculated in a very detailed way, taking into account variations in traffic speeds.
- **Banning HGVs from the road during daytime hours**: This could be achieved by simply setting the HGVs kms to zero for specific hours of the day. However it may be sensible to increase the HGV kms during the night-time hours to represent a certain amount of “displacement”.
- **Restricting cars to Euro 4 or better**: This scenario can be investigated by setting vehicle kms for all cars earlier than Euro 4 to zero. However it is likely that the kms for Euro 4 and better cars would increase above current levels, and consideration should be given to how much the kms might increase.

4.1.4 Height of emission releases

For localised studies, the height of the emission release is important, as it impacts on the way that the emission is mixed in the air. So it is important to include information about emission release heights with each of the sources in the emission inventory.

The most important sources in urban areas are almost always road transport, for which there is no significant variation in the height of the emissions release. Domestic combustion sources will release emissions to air at a higher level, but often also at a uniform height above ground. However, point sources in the area are likely to be large emitters and will need to be considered individually. So, as a general rule, all point sources included in the emissions inventory should be accompanied by information on the stack height with, preferably, more detailed information on the emissions characteristics such as temperature, velocity etc).

4.1.5 Generating mapped emissions at the required spatial and time resolution

Emissions from a section of road are calculated as explained above whilst emissions from individual sources are determined and summed to give a total emission.

For some studies it may be necessary to divide the section of road into different elements. This allows, for example, different speeds to be assigned to different road sections. Lengths of road are almost always divided into individual elements where there is a road junction. This is because the traffic flow either side of the junction can be different. Each section of road is called a road "link". Using this approach, it is possible to calculate emissions from each road link, and if required for input into the modelling, construct emissions from a network of road links.

It was explained above how information from hourly traffic count data are used to generate emission estimates. Exactly the same principles are used for shorter averaging periods, if required, assuming that traffic count data are available. If fine resolution traffic count data is not available, then assumptions must be made about how the traffic flow varies within the averaging period of the available data. However, this introduces substantial levels of uncertainty.

4.2 City level emission inventories

4.2.1 Generating emission totals from national level data

The starting point for a larger-scale urban emissions inventory is to estimate the emission totals for relevant sources (in the city). For a large urban area, this can be achieved by abstracting a relevant portion of the national level emissions inventory. For example, population information might indicate the fraction of the national total that is included in the regional emissions inventory. Consideration would need to be given to the uses of different fuel types in determining the emissions. Alternatively there may be fuel consumption totals for the defined urban area, which allows emission estimates to be calculated.

In the sub-sections which follow it is assumed that an emissions inventory is available with good source sector detail and reliable estimates of pollutant emissions.

4.2.2 Defining the spatial resolution and extent

It is important to clearly define the geographical boundary of the emissions inventory. Typically this is defined by the area that has been included in the city level emissions inventory. It is important that the emissions map to be created matches the area of the emissions inventory.

For a large-scale urban area, the boundary of an emissions inventory may be defined by organisational information – such as the area that is in the control of local Government, or by geographical features – such as a ring road around a city centre. The area of the inventory is called the "**extent**".

The next step is to define the resolution to which the emissions will be mapped within this extent i.e. the size of each grid cell that will make up the extent. For regional area, or very large cities (with a population of ~1 million) it is common to use cells that are 1km x 1km in size. For smaller cities, it may be more appropriate to use smaller grid cells, perhaps as small as 100m x 100m.

Obviously using smaller grid cells allows the emissions to be mapped in more detail, but generates much more data. The resolution of the spatial information that is available to map the emission total should also be considered when deciding the spatial resolution of the inventory map. For example, if population information is available at a 1km x 1km resolution, then there is not much sense in creating an emissions map at 100m x 100m or at an even finer grid of since the "native resolution", i.e. the real accuracy of the emissions map, would still be 1km x 1km – the resolution of the population data.

4.2.3 Point, line and area sources

Different sources have different spatial patterns, and need to be included in emission maps in different ways:

- Emissions from **large point sources**, such as power stations, are usually determined individually. This means that the corresponding emissions can be assigned to a specific point within the emissions map.
- Emissions that arise from numerous smaller sources, which cannot be individually identified, are called **area sources**. These need to be assigned to an area on the emission map. Each of the area sources is paired with a spatial dataset that is considered to represent the distribution of the emission. The spatial dataset is then used to weight the emission total in the corresponding grid cells.

  For example, emissions of NOx from the domestic sector can be paired with a map of population. If grid cell X includes 1% of the population total, then 1% of the NOx emission from the domestic sector is assigned to this cell. In this way, the emissions are distributed across the emissions map with the same pattern as the spatial dataset.
- Sources such as road transport are similar to area sources, but are portrayed as "**line sources**" on the emission map, or a series of lines that make up a network.

Table 4-1 provides general guidance on the way in which different emission sources could be combined with commonly available spatial information to generate emission maps.

Table 4-1 Matching emission estimates with a suitable spatial coverage

Emission Source	Mapped Dataset
Electricity generation	The locations of these sources are typically known, so they can be mapped directly as point sources.
Industrial Combustion and Processes (point sources)	The locations of these sources are known, so they can be mapped directly as point sources.
Industrial Combustion and Processes (area sources)	Land cover datasets sometimes include "industrial" or "commercial" land cover. If this is not available, then it is difficult to accurately map small industrial sources. It may be necessary to assign the emissions to urban centres, which is not ideal.
Residential Combustion	Using mapped population is the obvious choice. However consideration should be given to the patterns of different fuel use. It may be that gas and electricity are used much more in urban areas. So it is best to map the emissions from different fuel uses separately – for example, assigning wood to more rural locations and gas to more urban areas.
Solvent use – industrial	See Industrial combustion (area sources) above
Solvent use – domestic	Mapping by population typically gives suitably reliable emission maps.
Road transport	Road transport emissions are typically the most important source in urban centres. The detail to which the emissions can be mapped very much depends on the information that is available. Emissions can be assigned to roads, but as a minimum, the traffic flow along each road

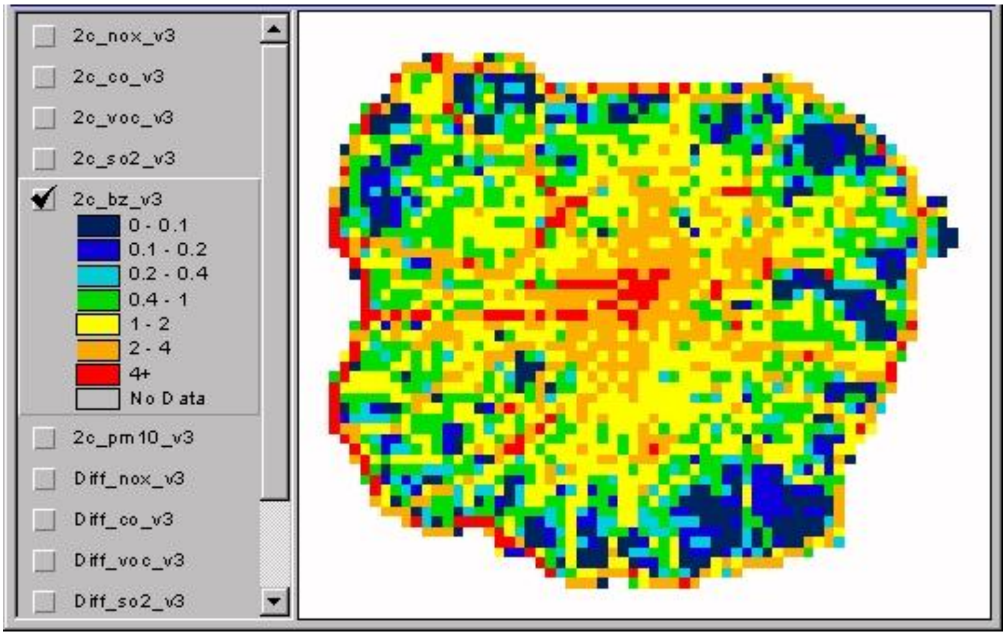
Emission Source	Mapped Dataset
	should be taken into account. It is also desirable to take into account the vehicle speeds and the fleet mix to reflect factors such as, for example, fewer HGVs in urban centres.
Rail	This is usually a relatively small source, and mapping emissions uniformly across the rail network is sufficiently accurate.
Aviation	Emissions from landing and take-off should be assigned to grid cells that contain airports.
Shipping	Obtaining information to assign shipping emissions to different shipping routes can be difficult. There are ways of simplifying this, for example, by spreading emissions uniformly along the coastline.
Off-road machinery – farming and industrial	Land cover data allows farming machinery to be assigned to agriculture land. Industrial machinery should be mapped in the same way as industrial combustion areas sources.
Off-road machinery – domestic	If land cover datasets that provide urban “green space” are available then they are ideal for mapping domestic garden machinery. Maps of house numbers also provide a useful coverage. If housing or population maps are used, then care should be taken to avoid assigning emissions to highly densely populated areas – because in city centre dwellings typically have smaller or no gardens.
Landfill	The locations of authorised landfill sites for waste disposal should be known, allowing these to be mapped as point sources. If this information is not available, for example if waste is dumped or disposed on unauthorised sites then it is difficult to map waste disposal sites with any certainty. It may be necessary to assign the emissions to some kind of green land cover type.
Waste water treatment	The locations of wastewater treatment sites ought to be known, allowing these to be mapped as point sources.
Agriculture - livestock	Emissions from livestock can be assigned to suitable land cover types. It may also be that some point source data are available (because some poultry housings are reported under IPPC). If land cover makes a distinction between pasture and arable land then this is a significant improvement on classifying land as simply “farmed”.
Agriculture – fertiliser application	Emissions from fertiliser application should be assigned to arable land if this is available from land cover datasets. If this is not available, then assigning to a general farmed land coverage provides an alternative.
Natural sources	Land cover datasets typically include several types of natural land cover. For emission inventories the most common source under “nature” are emissions from forests, and these should be easily assigned to a suitable land cover type.

4.2.4 Generating city level mapped emissions

It is explained above how each emission source can be mapped across the extent. This next step is simply to sum the measured or estimated emissions in each cell of the grid to obtain a map of total emissions. Typically these data are handled in GIS systems, which have the advantage of being able to handle the datasets so as to easily include or exclude different “layers”.

Figure 4-1 is an example of an emissions map. It is a 1km x 1km map of benzene emissions in London, England. This map was built by summing the different “layers”, and as a result there are emissions data for each source included within each of the grid cells, rather than just a total.

Figure 4-1 Map of benzene emissions in London



4.2.5 Converting from annual to shorter time resolutions

Most large scale emission inventories are compiled with a time period of one year. This is sufficient for many applications, but it may be that a shorter time period is required for modelling purposes.

Annual emissions can be divided into emissions across shorter time periods by using time profiles. This considers how each of the sources might vary across the period of one year, and allows the total to be divided up into smaller time periods. The approach is similar to the way in which a total emission might be distributed across an area.

Some emissions are considered to be approximately constant with time – for example some industrial sources and solvent emissions in the domestic sector. Other sources, such as emissions from domestic heating, will arise almost exclusively during the daytime. The following should be considered for each source:

Monthly variation: Some emissions such as domestic heating and electricity generation are much higher during winter months.

Weekly variation: Emissions for some sources – road transport and some industries for example - are very different on the weekend and on week-days.

Hourly variation: Some sources may be constant across a typical day, but most are higher during the daytime than at night.

Assigning portions of the emissions to different time periods is uncertain unless real data are available to support decision making. However, the above process does allow an annual emission to be converted into short time periods, which is considerably better than assuming that all emissions are constant with time.

5 DISPERSION MODELLING

5.1 Introduction

5.1.1 Scope of this Section

Air dispersion models are numerical tools that are commonly used to determine concentrations of air pollutants in the atmosphere. A dispersion model provides estimates of air pollution for a defined spatial area given the location and strength of one or more emissions sources. The concentrations of pollutants that are produced by a model are based on a mathematical treatment of the relationship between emissions, meteorology, existing pollution concentrations, topography, deposition and other factors. An important step in the modelling process therefore involves characterisation of these factors in some detail so that they can be input to the chosen model.

This Section introduces a number of factors that need to be taken into consideration when embarking on emission dispersion modelling. This introduction is supported by Part 2 Report which provides further detail, as indicated below:

- Dispersion modelling process
- Types of dispersion model
- Uses of dispersion modelling – see Part 2 also
- Limitations of dispersion modelling
- Characterisation of sources of emissions
- Meteorology in dispersion modelling – see Part 2 also
- Gaussian vs advanced models
- Modelling the local and urban scales – see Part 2 also
- Dispersion modelling tools – see Part 2 also
  A number of commonly used models are identified and their application in local and urban scale studies is described. The list is not exhaustive but the models identified are representative of those in use throughout the world for air dispersion studies
- Minimising model error and uncertainty – see Part 2 also
- Model calibration - combining monitoring and modelling data – see Part 2 also.

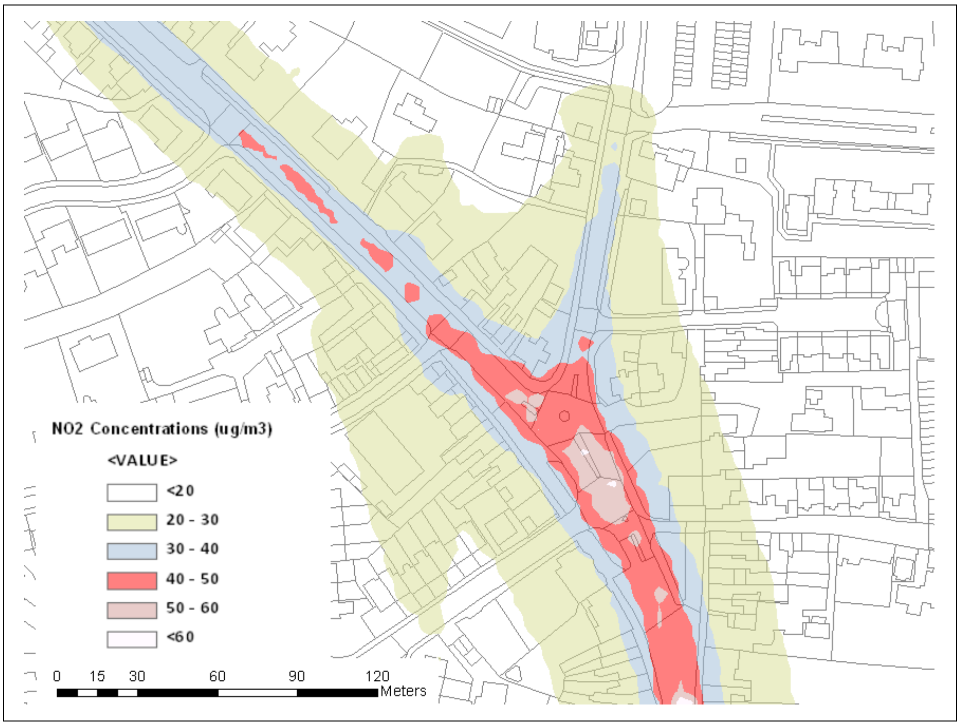
Section 6 illustrates the above considerations through presenting a number of case studies involving dispersion modelling and provides some conclusions.

5.1.2 Applications: local scale and urban scale

An important consideration is the geographical scale of the air quality management plan to be investigated using modelling techniques. Description in this Report focuses on local scale and urban scale models. These are defined according to the definitions used by the European Environment Agency (EEA)[3]. The EEA definitions suggest that local models should cover spatial areas of up to 1000 metres square, urban scale models should cover areas up to 300 km square. Figure 5-1 provides example output of applying dispersion modelling at a local level, whilst Figure 5-2 illustrates its application in larger urban-scale air quality management planning.

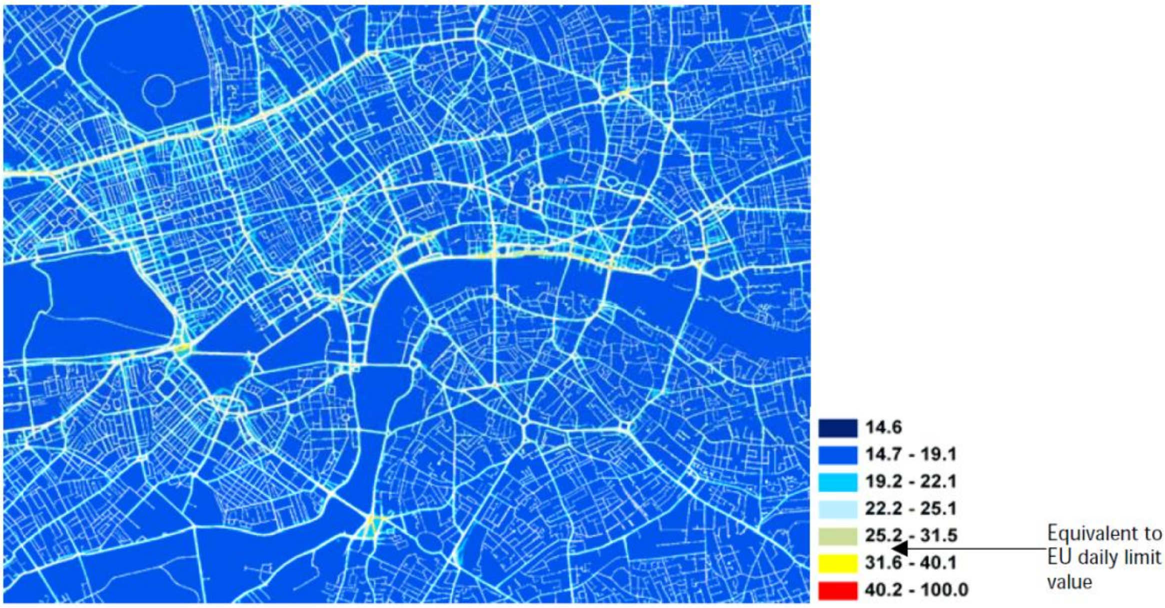
[3] European Environment Agency, 2011. The application of models under the European Air Quality Directive: A technical reference guide

Figure 5-1 Example output of local scale dispersion model (NO_2 – annual mean, Henley, England)



Source: Assessment of air quality in Henley, South Oxfordshire UK, AEA Technology, 2011

Figure 5-2 Example larger-scale urban area dispersion model (PM_{10} annual mean in London)



Source: Clearing the air: The Mayor's Air Quality Strategy, Greater London Authority, 2010

Dispersion modelling may also characterise the consequences of past and future emissions scenarios and is therefore a key tool in design of air pollution abatement strategies. It can provide a key input to the cost benefit analysis of specific alternative possible measures.

The technique has been applied to a wide range of air quality problems including PM_{10} and NO_2 in European urban areas[4],[5], investigation of public health impacts of reduced emissions from the energy sector in China[6], and SO_2 concentrations arising from domestic and industrial fuel combustion in Izmir, Turkey[7]. There are numerous studies assessing the ambient effects of road transport emissions both in the scientific literature and in local and national government reports from around the world.

Modelling can provide a thorough characterisation of the air quality problem, including an analysis of factors and cause; for example analysing the contribution to concentrations from different source types or meteorological states. Pollutants typically modelled in point source studies include NOx/NO_2, PM_{10}, SO_2, and VOCs. For road traffic the main pollutants of interest are PM_{10}, and NO_2.

5.1.3 *Requirements and Limitations*

Before a modelling assessment is carried out the following data and tools are generally required:

- air quality monitoring data for validation or assimilation;
- meteorological monitoring data for validation, model input parameters for use with diagnostic wind field models;
- emissions inventory;
- other relevant input data dependent on model type, such as background concentrations, land use or traffic data;
- modelling tools for carrying out the assessment (i.e. the air quality model and meteorology);
- analytical tools for model validation and assessment.

Modelling, however, cannot provide all the answers, and there are a number of limitations associated with the use of dispersion models - see later.

5.2 The dispersion modelling process

Most modern air pollution models are computer programs that calculate the pollutant concentration arising from a source given information on the:

- contaminant emission rate
- characteristics of the emission source
- local topography
- meteorology of the area
- ambient or background concentrations of pollutant.

Figure 5-3 provides a generic overview of how this information is used in a computer-based air pollution model. The modelling process comprises four stages (i) data input (ii) dispersion calculations (iii) deriving concentrations and (iv) analysis.

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[4] Calori *et al*., 2011. Air quality integrated modelling in Turin urban area, Environmental Modelling and Software, 468-476

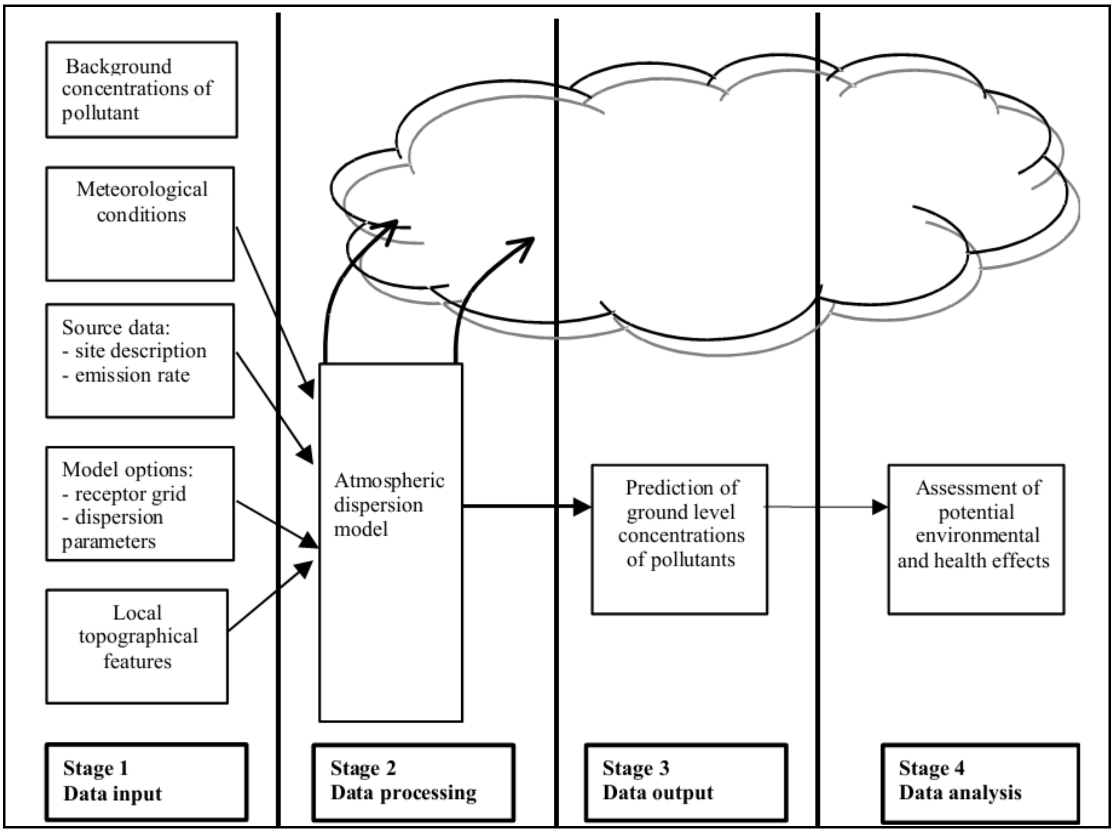
[5] Moussiopoulos *et al*., 2009. Air quality status in Greater Thessaloniki Area and the emissions reductions needed for attaining the EU air quality legislation, Science of the Total Environment, 1268-1285

[6] Kan *et al*., 2004. An evaluation of public health impact of ambient air pollution under various energy scenarios in Shanghai, China, Atmospheric Environment, 95-102

[7] T. Elbir., 2003. Comparison of model predictions with the data of an urban air quality monitoring network in Izmir, Turkey, Atmospheric Environment, 2149-2157

Care must be taken to ensure the accuracy and minimise uncertainty of each stage to ensure a reliable assessment of the significance of potentially adverse effects. Where dispersion modelling is used for CBA, mistakes at any of these stages will carry through the modelling process and affect the results of economic analysis.

Figure 5-3 Overview of the air pollution modelling procedure



5.3 Types of dispersion model

5.3.1 Introduction

Modelling tools are usually based on one of the three treatments of dispersion noted below. Currently, the most commonly used dispersion models are steady-state Gaussian plume models. These are based on mathematical approximation of plume behaviour and are the easiest models to use. They incorporate a simplistic description of the dispersion process, and some fundamental assumptions are made that may not accurately reflect reality. However, despite these limitations, this type of model can provide reasonable results when used appropriately.

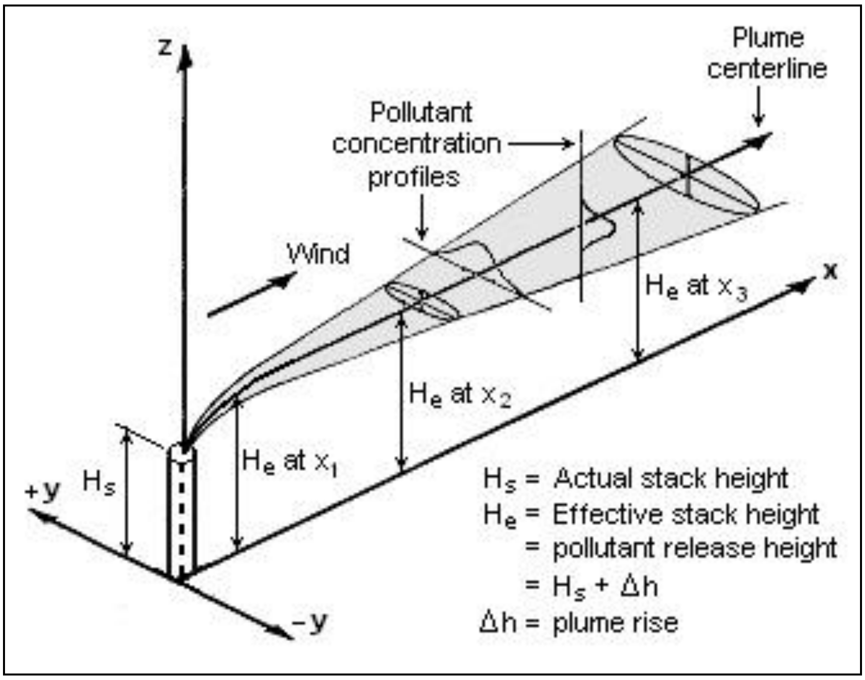
New generation dispersion models adopt a more sophisticated approach to describing diffusion and dispersion using the physical properties of the atmosphere rather than relying on general mathematical approximation. This enables better treatment of difficult situations such as complex terrain and long-distance transport.

5.3.2 Gaussian models

These are built on the Gaussian (normal) probability distribution of wind vector (and hence pollutant concentration) fluctuations. These are a subset of Eulerian models (see below) but are normally treated as a group in their own right.

The simplest realisation of this idea is to consider an instantaneous release of a pollutant leaving a point source. As the "puff" leaves the source it expands in volume taking in dilution air from around it thereby reducing the concentration. A continuous emission is described as an infinitely rapid series of small puffs. Gaussian dispersion theory allows us to calculate concentration downwind of the release at a given point. Figure 5-4 illustrates Gaussian dispersion theory.

Figure 5-4 Schematic of Gaussian dispersion



5.3.3 Eulerian models

These numerically solve the atmospheric diffusion equation and work on the measurement of properties of the atmosphere as it moves past a fixed point.

5.3.4 Lagrangian models

These treat dispersion processes in a moving air mass, or represent the processes by the dispersion of fictitious particles or pollution plume parcels.

5.4 Uses of dispersion modelling

Models can be set up to estimate downwind concentrations of contaminants over varying averaging periods – either short term (several minutes) or long term (annual). In Europe, the most common use of dispersion modelling is to assess the potential environmental and health effects of discharges to air from industrial premises and road traffic.

5.5 Limitations of dispersion models

Even the most sophisticated atmospheric dispersion model cannot predict the precise location, magnitude and timing of ground-level concentrations with100% accuracy. However, most models used today (especially the USEPA approved models) have been through a thorough model evaluation process and the modelling results are usually reasonably accurate, provided an appropriate model and input data are used.

Errors are introduced into results by the inherent uncertainty associated with the physics and formulation used to model dispersion, and by imprecise input parameters, such as emission and meteorological data. The most significant factors that determine the quality and accuracy of the results are:

- the suitability of the model for the task
- the availability of accurate source information
- the availability of accurate meteorological data.

The sources of uncertainty and some broad methods by which they can be minimised when using dispersion models are discussed in more detail in Part 2.

5.6 Characterisation of emission sources

Of the parameters required for dispersion modelling, the strength of the emissions source(s) being studied is the most important determinant of the ambient concentrations that result. The scale of the modelling study will determine the types and number of sources for which one require emissions data.

In a city, for example, there are sources over which local authorities have some form of control (e.g. traffic), but there are longer-range pollutant transfers over which they have no control. For most model applications, contributions from anthropogenic sources are recorded by addressing sectors e.g. traffic, industry or component sub-sectors e.g. heavy-duty or light-duty vehicles.

In localised urban settings an important source of NOx (and hence NO_2) and PM_{10} is road traffic. In order to model the ambient air impact from road traffic it is necessary to understand the number of vehicles using the road(s), the types of vehicles, and sometimes the age of the vehicles. All of these traffic characteristics have an impact on the emissions produced by the vehicles, and hence the emissions for the road that will be input to the dispersion model. Therefore any inaccuracy in the traffic parameters will lead to inaccurate emissions, and hence a poor dispersion model.

If the subject of the modelling study is an industrial facility, perhaps with high emissions of SO_2, it is necessary to understand both the emission mass-load but also the physical parameters associated with the emission e.g. stack height, stack width, efflux velocity and temperature. Characterising a complex industrial site can be challenging due to the presence of several point sources but also because it can be difficult to obtain data for other on-site sources such as fugitive emissions and vents. Emissions from the main stacks at such sites are normally calculated from measurements taken inside the stack for the pollutants of interest or from surrogate calculations based on properties of the fuel.

If the study aims to characterise the impacts of domestic combustion it is necessary to know the fuel use of the individual properties so that emission rates can be derived that can be applied to wide areas.

A dispersion model requires the calculated (or measured) emission rate of the source to be provided in units such as grams per kilometre per second (g/km/s) for local roads, or grams per second (g/s) for point sources. Most models can account for temporal variation in emission rates if these can be calculated, e.g. to reflect the diurnal nature of road sources, or an increase/decrease in activity at an industrial site.

In urban scale studies the number and type of sources is likely to be greater and will probably include contributions from transport, industrial and domestic sectors. This can lead to a significant data gathering exercise prior to modelling and normally would require compilation of a reasonably detailed and spatially disaggregated emissions inventory.

It follows that where dispersion modelling is being used to test potential abatement scenarios, it is important to understand the baseline sources very well so that the scenarios for air quality management are based on a sound estimate of existing emissions and their contribution to ambient concentrations. This can result in the data collection part of the modelling process involving significant effort. However, it should be remembered that robust estimation of the source is the best way to reduce uncertainty in a dispersion model predictions, and hence any cost benefit analysis of abatement policies.

5.7 Meteorology in dispersion modelling

5.7.1 Major meteorological factors

Three main meteorological factors influence atmospheric dispersion and, therefore, have to be taken into account in the modelling process – see Part 2 Report for further, detailed information:

- Wind direction, which is rarely constant. The medium-term average value of wind direction is fundamental in determining the area that will be affected by an emission from a particular source.
- Wind speed, which varies with height above ground level, is important in three ways: (i) dilution, which is proportional to wind speed (ii) turbulence increases with wind speed and (iii) buoyant emissions are 'bent over' more at higher wind speeds.
- Atmospheric stability (or turbulence). Where atmospheric conditions are unstable, turbulence causes air pollutants to mix more with dilution air, thereby encouraging relatively rapid dispersion and dilution.

5.7.2 Meteorological data for dispersion models

Most regulatory air dispersion models for local scale modelling based on Gaussian dispersion require measured meteorological data from surface stations and sometimes upper-air stations. In contrast, larger scale Eulerian or Lagrangian models usually require the use of modelled meteorological data. The Part 2 Report provides a fuller account.

5.8 Gaussian versus advanced models

The fundamental technical difference between advanced (three dimensional Eulerian and Lagrangian) models and Gaussian-plume models is that the advanced models require three-dimensional meteorological data. The other main issue is that costs are inevitably higher when employing advanced models.

A number of factors may deter a potential user from applying an advanced dispersion model to air quality assessment. They relate to the high level of expertise and extra investment of effort required. These factors include:

- a high-specification desktop PC with more memory, disk space and processing time than required for Gaussian-plume models. Even with high specification computers, run times can reach several days.
- a complex user interface because of more input parameters, which means that the visualisation of output can also require post-processing software to handle large output files.

- an increased risk of model misuse resulting from fewer people using advanced models compared to Gaussian-plume models and, as a consequence, a more limited experience base.
- a fully three-dimensional, time-dependent meteorological data set is usually required.

5.9 Modelling the local and urban scale

Different models are available to characterise local scale problems such as high traffic volumes on a congested road, or emissions from an industrial facility, through to urban, regional and even national scale issues e.g. whole city models with many source types.

A local scale model (usually based on Gaussian dispersion) allows the use of detailed input data (emissions for example) and the outputs (concentrations) can be defined in a very resolved manner. Use of larger scale advanced models usually necessitates a coarser approach, both in terms of the way emissions are described within the model, and also in the resolution of the output predictions. The larger-scale advanced models often do not provide results at the fine resolution that would allow authorities to compare conditions with health protection standards at the local level.

Table 5-1 notes the differences between local and urban scale modelling approaches in terms of model type, treatment of meteorology and chemistry, and approach to emissions determination.

Table 5-1 List of typical model characteristics for the local and urban scales

Description	Local (1-1000 metres)	Urban (1-300 kilometres)
Model type	Gaussian and non-Gaussian models Statistical models Computational Fluid Dynamics models Lagrangian particle models	Gaussian and non-Gaussian models Eulerian chemical transport models Lagrangian particle models
Meteorology	Local meteorological measurements Computational Fluid Dynamic model Diagnostic wind field model	Mesoscale meteorological models Localised meteorological measurements Diagnostic wind field models
Chemistry	Parameterised or none	Ranging from none to comprehensive, depending on the application
Emission modelling	Bottom up traffic emissions Source specific emissions	Bottom up and/or top down emissions model
Source: The application of models under the European Air Quality Directive: A technical reference guide, European Environment Agency, 2011		

Additionally, the pollutants of interest may be treated in quite different ways in local and urban scale models. Factors include whether or not the model calculates secondary particle formation from precursors in the atmosphere, whether deposition to land is considered, and treatment of photo-oxidant chemistry. Table 3-1 in the Part 2 Report lists how local and urban scale models can be configured to treat some common air pollutants.

It is important to consider whether the model options available will adequately characterise the problem under investigation. For a local study involving an industrial source or road traffic, for example, a Gaussian dispersion model that assumes steady state meteorology and chemistry would likely suffice and other issues like deposition to land or secondary inorganic particle formation can be ignored. For urban scale assessments the same Gaussian model may suffice depending on the needs of the study, but it may also be appropriate to choose a chemical transport model which will consider complex chemical interactions and secondary issues. The Part 2 Report provides further detail on alternative dispersion model types.

5.10 Dispersion modelling tools

It is important to select a suitable model. In most applications, Gaussian models provide a sufficient basis to undertake CBA but sometimes their limitations necessitate a switch to an advanced model based on Lagrangian or Eulerian treatments of dispersion.

The present Report does not provide an exhaustive list of available models - there are at least a few hundred in use. A comprehensive catalogue of dispersion models used by the European community can be found at the Model Documentation System[8] run by the European Topic Centre on Air Pollution and Climate Change Mitigation (EIONET). However, Table 5-2 notes a number of commonly used air dispersion models that could be employed to support future CBA at the local and urban scale in Turkey: the Part 2 Report provides descriptions of each.

If any of these models were applied in Turkey for policy analyses, both spatially disaggregated emissions data and robust air monitoring data would be required to validate the model outputs. Otherwise, the model results might be subject to unacceptable levels of uncertainty that would carry forward into CBA.

Table 5-2 Summary of models available for local and urban scale dispersion

Model name	Model type	Scale	Model developer	Comments
ADMS ADMS-Roads ADMS- Urban ADMS-Airport	Gaussian	Local/ urban	CERC UK	• Straightforward to operate though for larger scale studies the compilation of emissions data can be resource intensive • Emissions can be precisely aligned to receptors • Simple meteorology • 2D model outputs
AERMOD	Gaussian	Local/ urban	USEPA	• Not directly applicable to road transport studies. • Free source code for the advanced user although many third party interfaces available • Emissions can be precisely aligned to

[8] http://acm.eionet.europa.eu/databases/MDS/index_html

Model name	Model type	Scale	Model developer	Comments
				receptors • Simple meteorology • 2D model outputs • Treatment of road emissions possible with modified emission inputs
AUSTAL2000	Lagrangian	Local/ urban	TA Luft	• Advanced model but still applicable to local studies. • Free source code for the advanced user although many third party interfaces available • Simple meteorological data sets can be applied • 3D model outputs if required
CAL3QHC and CAL3QHCR CALINE3 CALINE4	Gaussian	Local	USEPA CDT	• Simple operation though limited to road sources and at local scale • 2D model outputs
OSPM	Gaussian with modification s	Local	NERI	• Simple operation though limited to road sources and at local scale • 2D model outputs • For urban studies can be employed nested within another model
CALPUFF	Lagrangian	Local/ urban/ regional	SRC	• Advanced model • Requires intermediate expertise • 3D modelled meteorology can be used, or single station measurements • Requires spatially disaggregated emissions data • Applicable to range of scales • Treatment of road emissions possible with modified emission inputs
CAMx	3D Eulerian	Urban/ regional	USEPA	• Advanced model • Requires significant user expertise • 3D meteorology required from e.g. WRF model • Requires spatially disaggregated gridded emissions model • Not applicable at local level • Computationally demanding
CMAQ	3D Eulerian	Urban/ regional	USEPA	• Requires significant user expertise • 3D meteorology required from e.g. WRF model • Requires spatially disaggregated gridded emissions model • Not applicable at local level • Computationally demanding

5.11 Minimising model error and uncertainty

5.11.1 Scope

The outputs of dispersion modelling are sometimes criticised for being inaccurate and a simplistic reflection of reality. To avoid such criticisms it is important to follow some simple principles to reduce model uncertainty, as listed below:

- All modelling studies should be designed so as to be as accurate as possible for the purpose of the study.
- Allow the accuracy of the modelling study to be assessed easily by:
  - stating the objectives of the study
  - demonstrating that the model inputs are as correct as possible
  - knowing and stating the chosen model performance limitations
  - demonstrating via the methodology that the modelling process has been conducted appropriately, including any validation with available monitoring information.

If any of these factors are not given appropriate consideration, the results may be meaningless and misleading; leading to a faulty CBA and any decisions based on the analysis. There are three main general sources of error and uncertainty in dispersion modelling:

- Input data uncertainty – see Part 2
  - Source or emission characteristics
  - Meteorological data
  - Terrain and local features data
- Inappropriate use of the model - or expecting too much from it
- Poor performance of the model.

The total uncertainty contained in the model results is the cumulative effect of these sources. It is useful to distinguish between 'reducible' and inherent uncertainty. Reducible uncertainty includes the accuracy of the input data and the way in which the model is set up and run. The inherent uncertainty is the fundamental limitations in the way a model works. This is beyond the control of the model user but is an issue they must be aware of. The uncertainties are normally investigated through an iterative model calibration process described in Part 2.

5.11.2 Model performance

After input data uncertainty, the fundamental limitation for dispersion modelling accuracy is the way the model works. It should be possible to evaluate any model's performance by a formalised evaluation scheme where its outputs are compared with actual monitoring results (with all other things being equal i.e. emission rates, meteorology and terrain).

Model validation has been addressed by the initiative on the Harmonisation within Atmospheric Dispersion for Regulatory Purposes[9]. One of the outcomes of this

[9] http://www.harmo.org

has been to produce the “Model Validation Kit”. This is a collection of three experimental data sets accompanied by software for model evaluation.

Most dispersion models have undergone some form of validation of their performance. Model users should be aware of the validation studies carried out for their chosen model. It is typically accepted that when accompanied by good input data, dispersion modelling may be used to predict concentrations within a factor of two. Model calibration is considered below and further in Part 2.

5.12 Model calibration

Comparison of modelled values with measurements is a fundamental step in assessing the performance of a model for a given situation. A dispersion model that is prepared in the absence of monitoring data for calibration is subject to significant (and even unknowable) uncertainty. Indeed it is the comparison with local monitoring data that is the key driver for model improvement which can proceed through several iterations before a satisfactory output is produced.

Model calibration is an iterative process where the model is run for the baseline situation, then gradually calibrated by checking the output of subsequent model runs against measurement data. The calibration process is essentially an exercise in reducing uncertainty and improving the model agreement with measurements. Model calibration can involve improving any of the parameters described above.

The Part 2 Report addresses this issue further, considering a number of statistical measures of model performance.

6 Dispersion Modelling Case Studies

A selection of case studies is now presented which outline how some of the modelling tools described in Section 5 have been applied to dispersion modelling in the local and urban settings. The Beijing study reported by Guo et al took the impact pathway approach from compilation of an emissions inventory, through modelling to economic analysis.

6.1 Local Example - Dispersion of vehicle emissions, UK Local Authority

In the UK Local Authorities participate in a process called Local Air Quality Management (LAQM). This mandates them to carry out air quality assessment work throughout their areas using a combination of monitoring and modelling (where necessary). The work is carried out on a 3-yearly cycle and is designed to identify any areas where air quality standards are at risk of being breached.

Modelling is key to this process and authorities carry out two types of modelling assessment - the "Detailed Assessment" and the "Further Assessment". "Detailed Assessment" is the first step in modelling an area of poor air quality and usually leads to the designation of an Air Quality Management Area (AQMA) which then becomes the focus of local policy. "Further Assessment" revisits the modelling of the area and assesses the likely impact of management interventions to inform an air quality action plan for the area.

While the Further Assessment does not require a full CBA its outputs are normally used as part of the feasibility assessment that is undertaken for a subsequent air quality action plan.

The Department of Environment, Farming and Rural Affairs (DEFRA) in the UK appraise every modelling study that is carried out for LAQM and publish examples of best practice[10].

In the example below, the ADMS-Roads dispersion model was used to assess the existing air quality in a small area of Chepstow, UK[11] though it should be noted that no CBA was undertaken (CBA is not common for small studies). A combination of monitoring and modelling was used to assess the risk of exceedance of the UK air quality standard for NO_2. Figure 6-1 shows the modelling output for NO_2. UK modelling studies involving urban areas and road traffic are required to use monitoring data to verify the outputs of any model. In the example below, road emissions would be prescribed very precisely as lone sources in the model - probably using a GIS tool to ensure accuracy.

The study found a small area that exceeded the NO_2 standard and the next stage of the assessment was to assess which local sources were causing the exceedance. This is done through a process called source apportionment and normally seeks to estimate which classes of road vehicles contribute most to the problem. Each vehicle class has a different emission rate of NOx and so it is possible to estimate the relative contribution to ambient concentrations of NO_2. Figure 6-2 shows the apportionment of NO_2 between different vehicle types in Chepstow.

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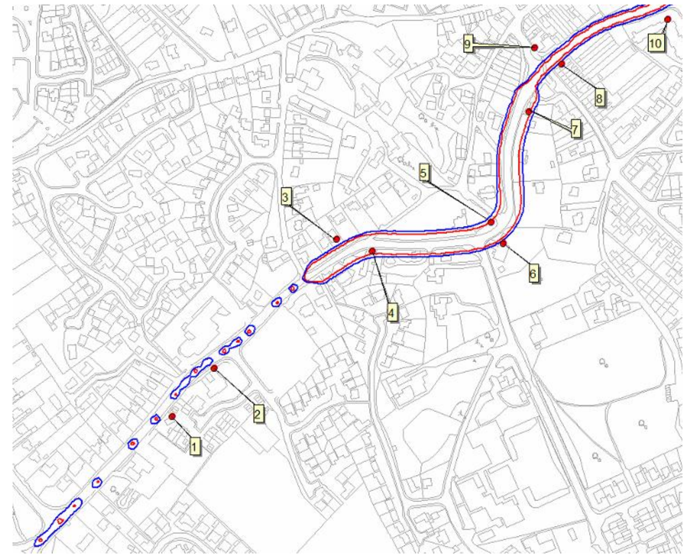
[10] http://laqm.defra.gov.uk/review-and-assessment/good-practice/examples.html

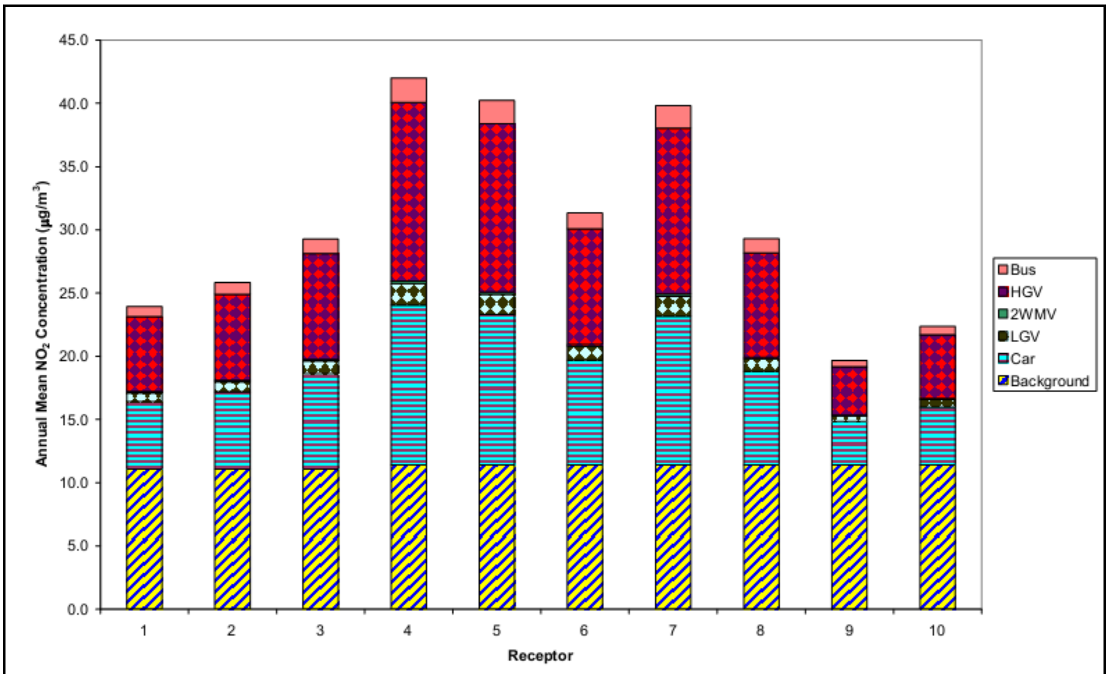
[11] http://laqm.defra.gov.uk/documents/Chepstow_FA_August2008.pdf

Figure 6-1 NO_2 modelling results in Chepstow, UK

NO_2 concentrations in Chepstow, UK (red line= 40µg/m3, blue line= 36µg/m3)	Site	Modelled NO_2 value
	1	24.0
	2	25.8
	3	29.0
	4	**42.0**
	5	**40.2**
	6	31.3
	7	39.8
	8	29.3
	9	19.7
	10	22.4
	UK NO_2 annual mean standard =40µg/m3	

The output of the source apportionment in Figure 6-2 shows quite a large contribution to NO_2 from heavy vehicles and cars. The apportionment assessment points the way to targeted intervention in the subsequent air quality action plan.

Figure 6-2 Source apportionment study for Chepstow, UK



Whilst the study above did not include a CBA, it would be reasonably straightforward to do this using the modelling outputs to feed into damage cost calculations. Local (and usually hypothetical) policy measures are also compared with the base case and could therefore be subject to CBA as well. These techniques could be readily applied in Turkey for local assessments but would require monitoring data for model validation and availability of quite resolved vehicle activity data.

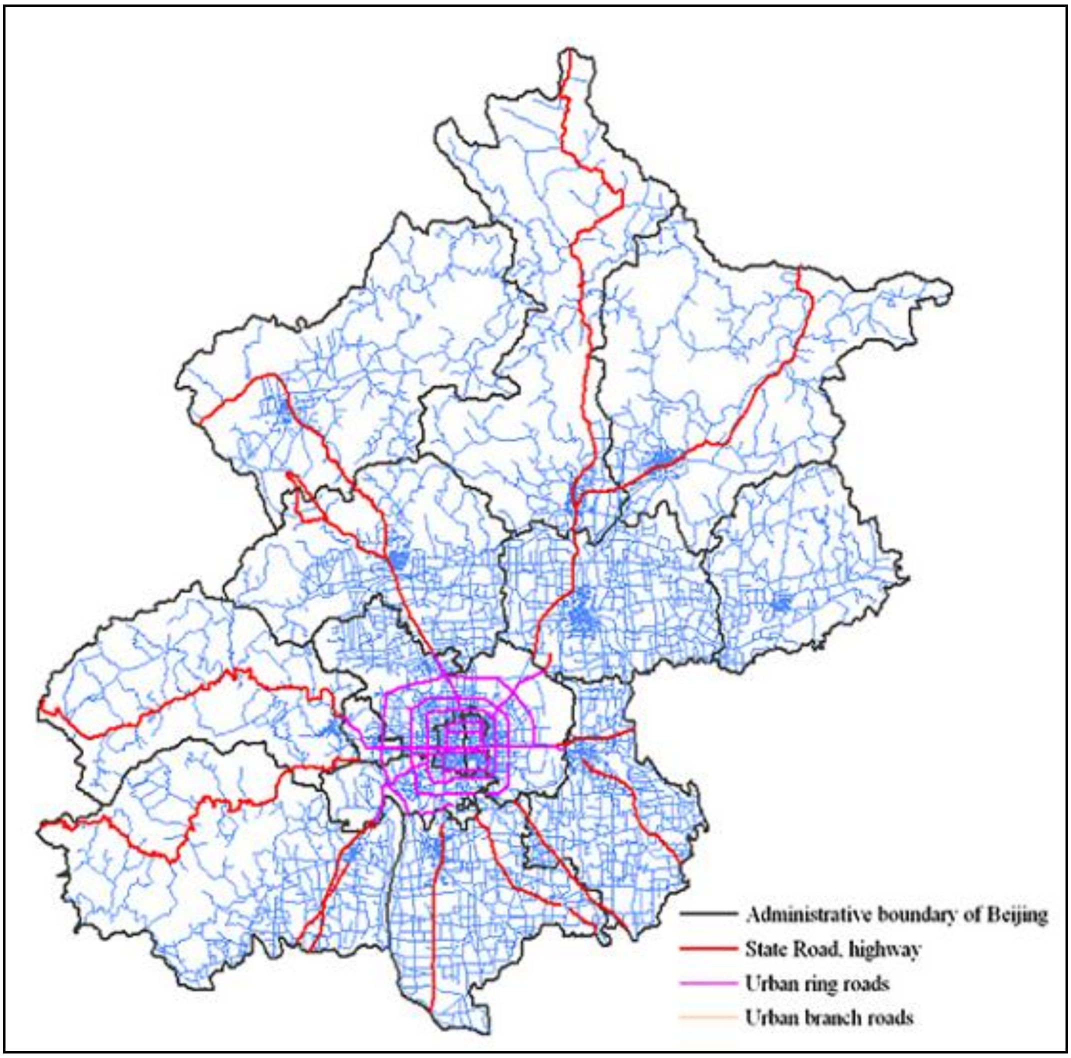
6.2 Urban Example - Exposure modelling in Beijing using CMAQ

In recent years an increasing number of studies have utilised more sophisticated “one atmosphere” models such as CMAQ to characterise urban scale pollution problems. These sophisticated models are only readily applicable to the urban scale but can yield very useful results to feed into CBA when the pollution data is assessed in the context of population density data.

In the example below (Guo et al) a dispersion modelling approach was applied to assessing the value of health impacts associated with road traffic particulate in Beijing, China[12]. A full CBA was carried out so this example is very useful in illustrating the use of models for such applications.

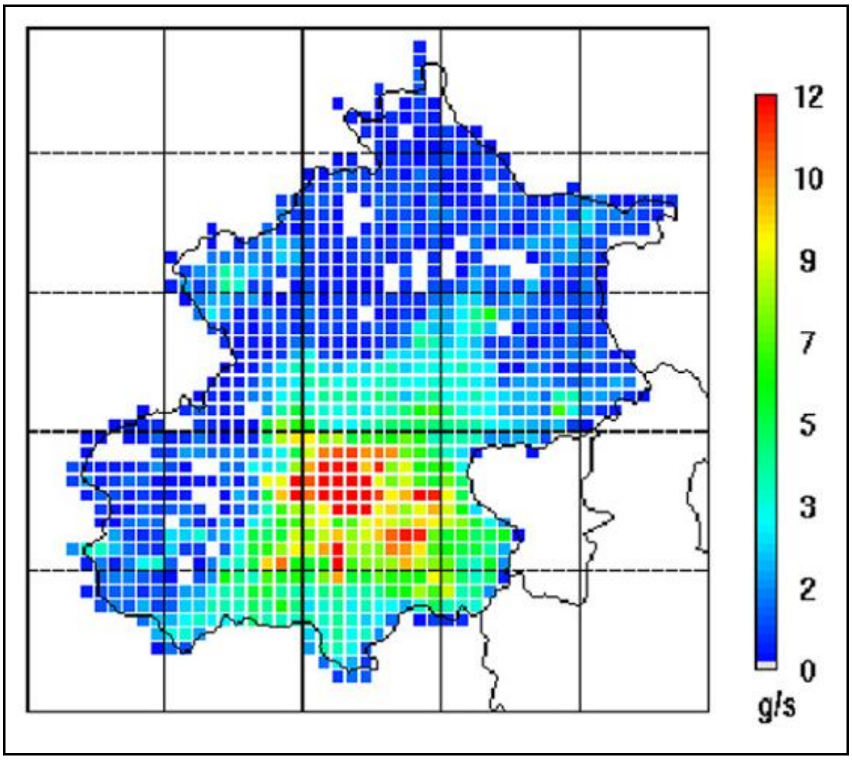
The study split the city into 4km^2 areas and defined the emissions from road transport based on average vehicle activity on the roads within each area - see Figures 6-3 and 6-4.

Figure 6-3 Beijing Road Network (after Guo et al)



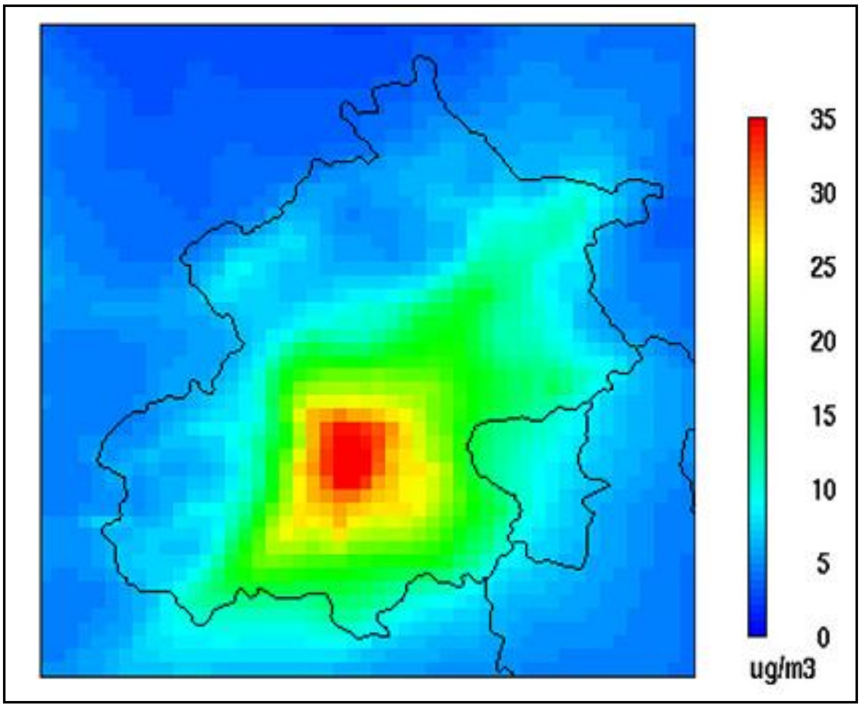
[12] Guo *et a*l., Estimation of economic costs of particulate air pollution from road transport in China, Atmospheric Environment, 3369-3377

Figure 6-4 PM_{10} emissions from road transport in Beijing (after Guo et al)



The group used the CMAQ model to estimate concentrations in each of the $4km^2$ gridded areas and checked the performance of the model at thee monitoring stations in the city. It was found that the highest PM_{10} concentrations coincided with the most densely populated areas of the city - see Figure 6-5.

Figure 6-5 PM_{10} concentrations from road transport in Beijing (after Guo et al)



The study used population density data in combination with the concentrations predicted by the model to assess exposure to PM_{10}. When the overall population weighted exposure was calculated, dose-response functions for PM_{10} were used to assess the overall impact on human health of the predicted concentrations. An economic assessment was carried out which used the “willingness to pay” concept as well as the Value of a Statistical Life (VOSL) – see Section 9 and Part 2 Report.

In addition the modellers were able to assess the impact of the vehicle control measures introduced during the 2008 Olympic Games.

The study estimated that PM_{10} from road transport in Beijing was responsible for 3143 early deaths with 16030 cases of acute bronchitis, 598 cardio-vascular hospital admissions, 545 respiratory hospital admissions, 19159 asthma attacks, and over 2 million reduced activity days.

Using the unit values of the various health endpoints, the economic costs of transport-related air pollution were calculated. The total economic cost of health impacts due to air pollution contributed from transport in Beijing during 2004 to 2008 was $298 million US (mean value). From this estimate, it was concluded that the vehicle control measures in 2008 saved about $ 5 million.

The economic valuations were compared with Beijing’s annual GDP. The economic costs of road transport accounted for 0.52, 0.57, 0.60, 0.62, and 0.58% of Beijing’s annual GDP from 2004 to 2008.

The Beijing study is a very good example of urban scale dispersion modelling being used to support economic assessment of existing air quality and policy interventions in a city of comparable population to Turkey’s biggest urban areas. Such an approach would be directly applicable in Turkey but would require significant modelling effort/expertise and availability of monitoring data to support model validation efforts.

6.3 Urban Example - Use of dispersion modelling to support air quality management in Istanbul

Reported recently (Elbir et al) this example is especially pertinent given that its subject is the largest city in Turkey, Istanbul[13]. The example does not include a CBA but is very useful in demonstrating the potential of modelling techniques for this application in the Turkish context.

One of the world's biggest cities with a population in excess of 12 million, air pollution in Istanbul is problematic due to rapid population growth, improper site selection for industry, use of poor quality fuels, use of old combustion technologies in industry, lack of control technologies for stack gases and traffic emissions[13].

Metropolitan Municipality Directorate of Environmental Protection has monitored the urban air quality in Istanbul since 1998 in 10 monitoring stations that were located at various sites across the city (the availability of monitoring data is crucial to verification of the modelling). These stations continuously measure CO, NO_X, SO_2, PM_{10}, and O_3 and measurements are recorded on an hourly basis.

Studies have indicated that the city experienced high concentrations of SO_2 due to domestic combustion of poor quality lignite fuel with high sulphur content[14].

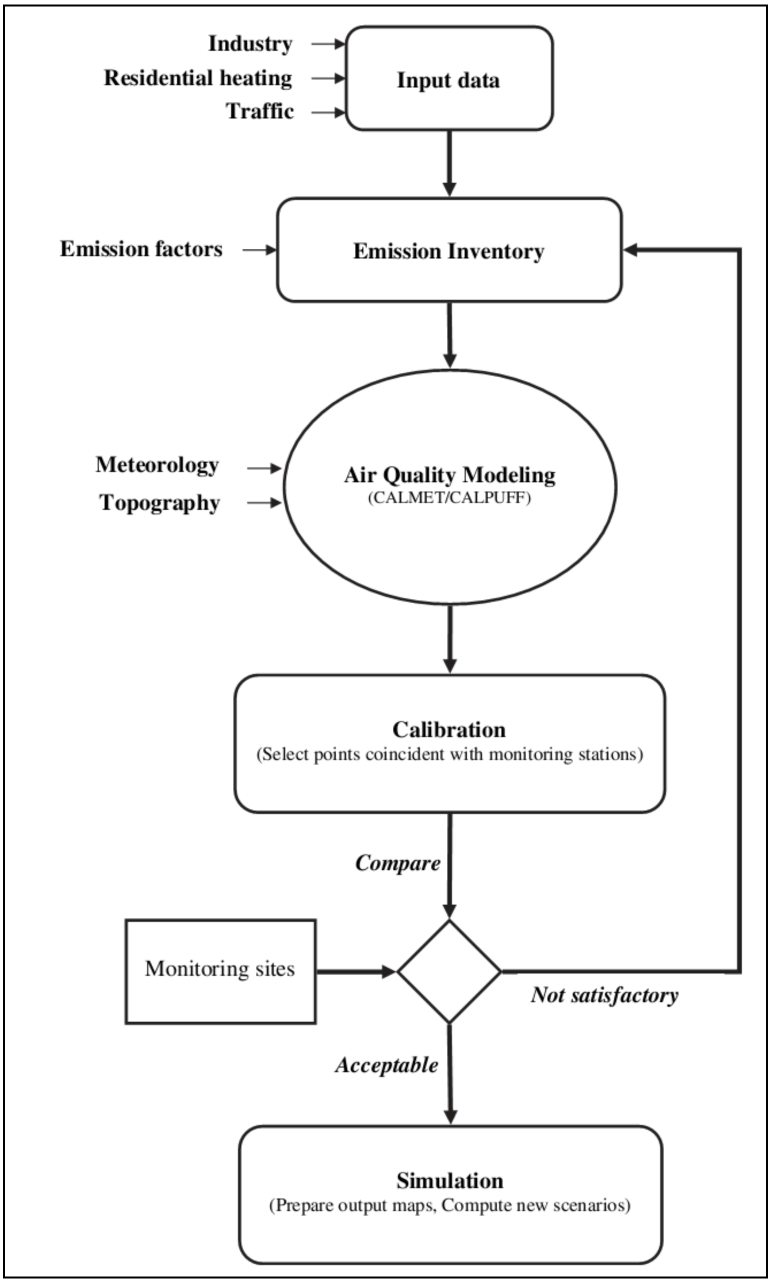
[13] Elbir et al., 2010. Development of a GIS-based decision support system for urban air quality management in the city of Istanbul, Atmospheric Environment, 441-454

[14] Tayanc, M., 2000. An assessment of spatial and temporal variation of sulphur dioxide levels over Istanbul, Turkey. Environmental Pollution 107, 61-69

However later studies indicate that levels of SO_2 were reducing due to the introduction of higher quality coal and gas for domestic combustion[15]. In addition to local sources, it appears that Istanbul is affected also by long range transport of pollutants from Eastern Europe under certain meteorological conditions[16].

A detailed emissions inventory was compiled which was followed by air quality modelling using CALPUFF, air quality mapping in a GIS system, and scenario analysis for air quality management. Figure 6-6 shows a schematic of the process.

Figure 6-6 Process diagram for dispersion modelling in Istanbul (after Elbir et al)



[15] Akkoyunlu, A ., Erturk, F., 20 03. Evaluation of air pollution trends in Istanbul. International Journal of Environment and Pollution 18, 388-398.

[16] T. Kindap., 2008. Identifying the trans-boundary transport of air pollutants to the city of Istanbul under specific weather conditions, Water, Air & Soil Pollution 189, 279-289.

For compilation of the emissions inventory for Istanbul the researchers collected industrial source specific information on production capacities, raw materials used, manufacturing processes, fuel consumptions, stack characteristics (i.e., stack height, diameter, flue gas temperature and exit gas velocity). Domestic combustion was characterised using information like number of inhabitants, types of fuels used, fuel consumptions and population densities. The road traffic sector was characterised by estimating the numbers, types and ages of vehicles, fuels used and fuel consumption. This data was then used in combination with emission factors to prepare an emission inventory stored in GIS databases.

The emission inventory was then input to the dispersion model, CALPUFF, to allow calculation of concentrations arising in the city. Large point sources were input to the model separately according to their location and other sources were combined into area sources. Sources were assigned to 1km^2 grid squares covering the whole city and surrounding areas- see Figure 6-7.

Figure 6-7 SO_2 source allocation in Istanbul (after Elbir et al)

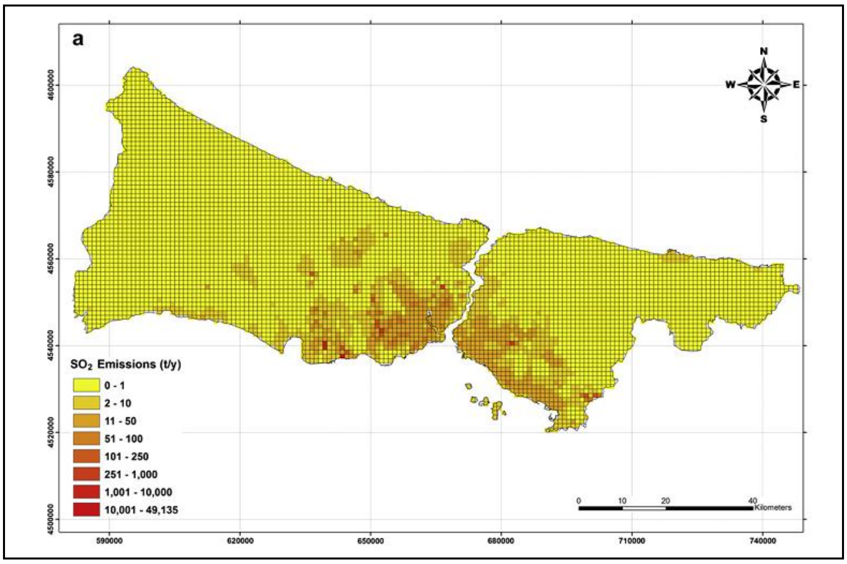
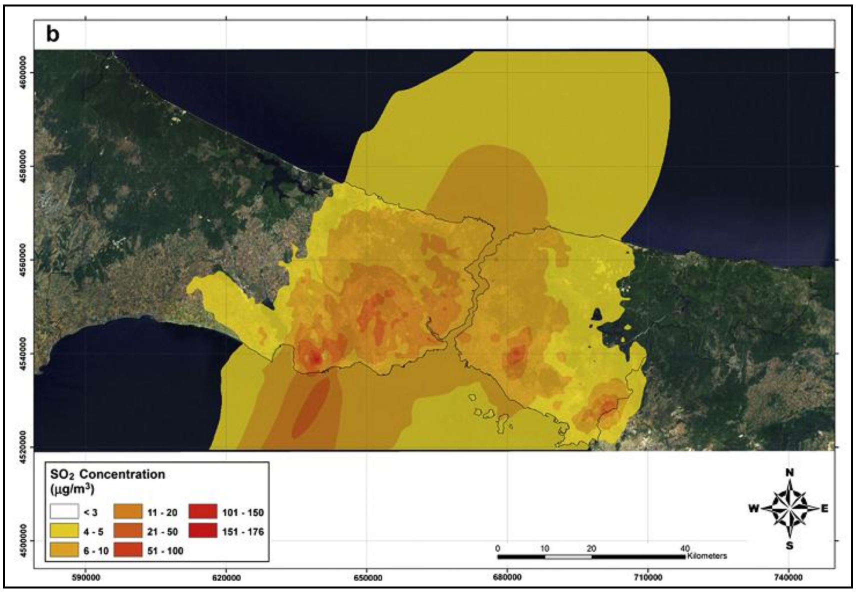


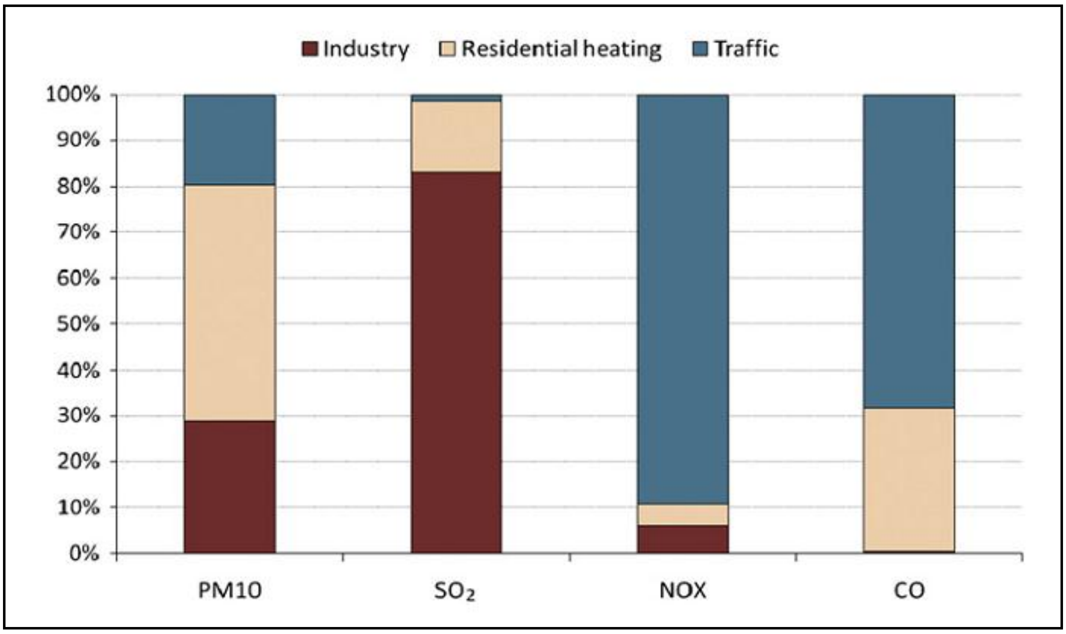
Figure 6-8 SO_2 predicted concentration in Istanbul (after Elbir et al)



PM GROUP

Figure 6-8 above shows the gridded concentrations of SO_2 predicted for Istanbul: similar plots were provided for PM_{10}, NO_2 and CO in the study. Figure 6-9 shows the results of the sectoral emissions characterisation. The study found that industry was the most polluting sector for SO_2 contributing to about 83% of total emissions while residential heating was the most polluting sector for PM_{10} contributing to 52% of total emissions. However, traffic was the most polluting sector for NO_X and CO emissions with contributions of 89% and 68%.

Figure 6-9 Sectoral emissions in Istanbul (after Elbir et al)



When the emissions of individual industries were examined, a public power and cogeneration plant was found to be the largest source of air pollution in the study area due to its high fuel oil consumption. It contributed 84% to the total industrial SO_2 emissions and 70% to the overall SO_2 emissions.

Several sand and gravel processing plants and one big cement plant (having the highest PM_{10} emission) were the major contributors to PM_{10} emissions. Several different dust control technologies varying from electrostatic precipitators to fabric filters are used in the cement plant depending on the regulatory requirements. Therefore, control technologies and their efficiencies were taken into account in calculating the emissions from large facilities, such as this cement plant.

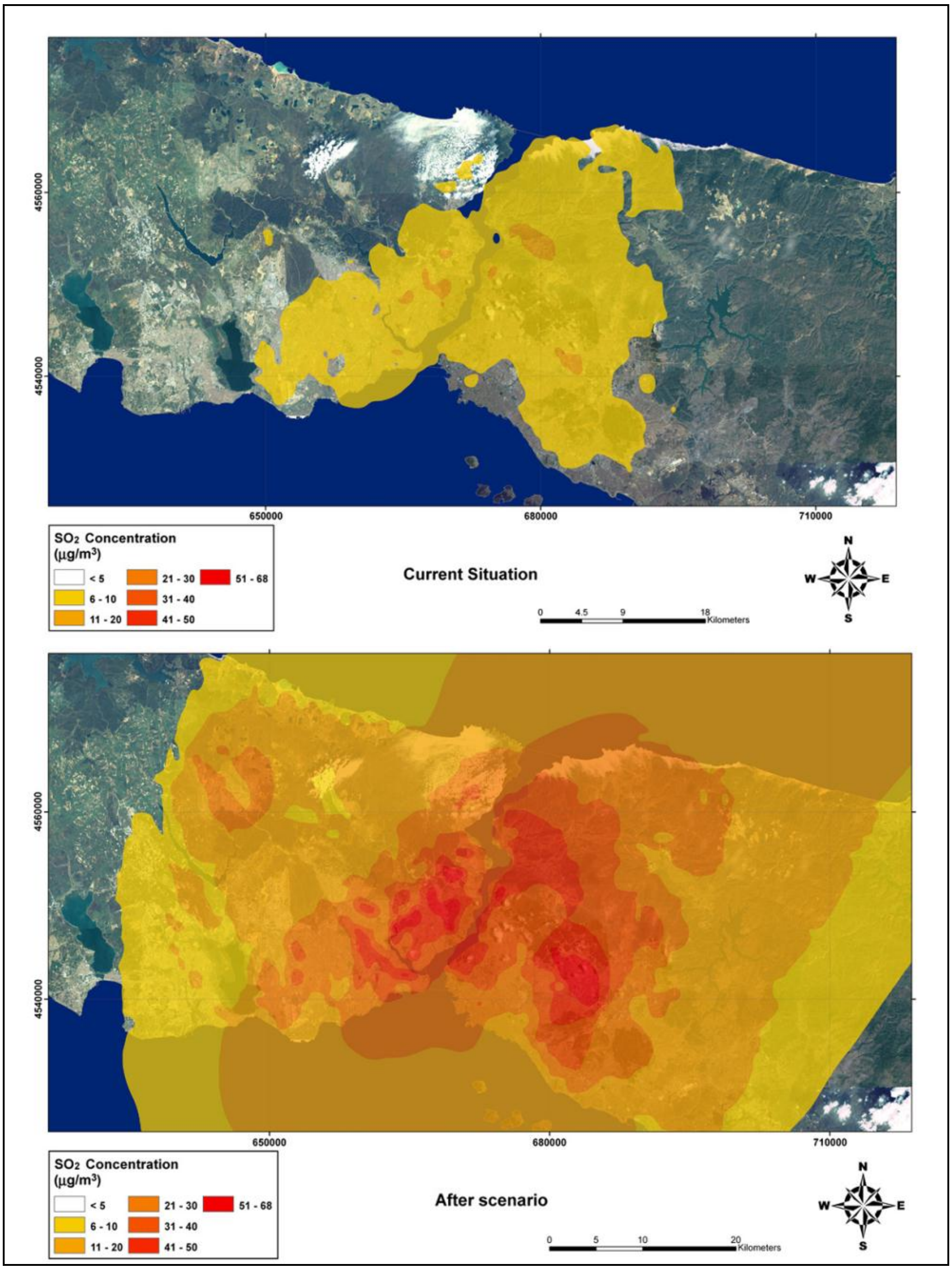
Maximum concentrations of SO_2 were located quite close to the large power station and PM_{10} was highest near the cement plant and sand/gravel facilities. The highest NOx concentrations were located near highways.

The model was used to assess the impact of fuel switching in the residential sector back from gas to lignite, thereby reflecting situations where security of supply is compromised (Turkey imports all of its natural gas). The researchers estimated that this would increase SO_2 emissions in the domestic sector by a factor of 4 and this scenario was run through the dispersion model of the city. The results of the scenario analysis are shown in

Figure 6-10.

The scenario analysis shows that concentrations of SO_2 would increase in most areas of the city as a result of increased domestic combustion of poor quality lignite.

Figure 6-10 SO_2 scenario analysis in Istanbul (after Elbir et al)



Whilst the economic costs of the existing air pollution climate in Istanbul were not calculated in this study, it would be reasonably straightforward to do so using a similar methodology as the Beijing study. Though in this instance multiple pollutants were assessed and in the Beijing example only one was assessed.

The Istanbul study is obviously directly relevant to Turkey and the methodologies contained therein may be useful for other large urban areas in the country. The process outlined in **Error! Reference source not found.** could be applied anywhere in the country but it is likely that the challenges in characterising the sources in Istanbul will be repeated elsewhere in Turkey. Spatial disaggregation of sources is fundamental to producing good quality modelling that can be used to support policy making. If this can be coupled to a reasonably high resolution monitoring campaign in Turkish cities there is good potential for modelling to be applied usefully in the country for the purposes of CBA of air quality management interventions.

6.4 Conclusion

There is undoubtedly a potential role for dispersion modelling in air quality management and CBA of policy interventions in Turkey; the technique is already widely applied across the international community. Depending on the scale of the modelling that would be undertaken there are a range of well-established tools and methodologies available, though their application requires greatly varying degrees of modelling experience, computational resources and input data.

If local scale modelling is to be carried out for the purposes of CBA, it is reasonably straightforward to gather the majority of the required input data and uncertainties can be managed quite effectively. The case of urban scale modelling is much less straightforward and would as a minimum require a detailed, spatially disaggregated emissions inventory for the area of interest, as well as a network of monitoring stations for model verification.

The use of dispersion modelling supported CBA appears to be more applicable to large urban areas as it is likely that the approach would only be applied to air quality management interventions with a wide “reach”. Dispersion modelling at the urban scale could yield useful data to support CBA in Turkey. The methodologies outlined in the Istanbul and Beijing studies summarised in this report are directly applicable to Turkish conurbations though this is subject to a requirement for significant effort to establish the input data required.

A useful first step could be to establish the availability of monitoring data in Turkey in the context of the requirements of urban scale dispersion models. It might be that spatially disaggregated emissions data are not currently available but this should be examined with regard to the present and the future.

7 MAPPING OF AIR QUALITY DATA

7.1 Introduction

Generating maps of air pollutant concentrations requires a substantial amount of spatial information. In addition, a GIS platform needs to be constructed to handle the spatial information. Spatially disaggregated information provides a very powerful tool in assessing the impacts of air quality. Combination with population density maps (or maps of sensitive ecosystems etc.) allows impacts on health to be determined. This in turn allows the quantification of the impact in monetary terms, which is then used as input into cost-benefit analysis. Consequently generating concentration maps is a key step in providing support to policy formation.

7.2 Mapping of Air Quality Model Data

After an air quality simulation has successfully run, it is often desirable to have a graphical representation of air pollutant concentrations. This visualisation can be useful in determining the impact of a plume of pollutants on a nearby community and in determining if the simulation was a reasonable representation of the real world.

If a project involves producing very accurately contoured data or high-quality graphical output, or if we need to know the relationships between the simulated plume and several other types of data, it is useful to import the modelling results into a GIS workspace. This is a straightforward process if the modelling output is given with the coordinates and concentrations at each receptor used in the simulation.

GIS systems are now very widely used in air pollution modelling studies. The data manipulation and post processing options available make model set up and interpretation of results significantly more straightforward. The GIS environment allows the user to have great control over how the ambient concentration field is displayed, and the user can also overlay many other types of data along with the plume. It also allows the user to easily import the locations and measurements from monitoring locations for comparison with the plotted concentration fields.

Since air quality models are often used to predict the impact of pollutants on people at ground level, two-dimensional (2D) representations of concentration are often sufficient. However, it may be useful to have a three-dimensional (3D) visualisation of a plume if the simulation is over a very large area.

For example, for a local study a model domain may be 1km x 1km, and may include a grid of ground level points within the domain spaced at 10m intervals. This fine resolution allows the modeller to compare the predictions with ambient air quality standards at the level of single buildings or receptors. The use of fine resolution spatial datasets enables the modeller to visualise small areas of high concentration and, importantly, to assess these concentrations in light of the presence of sensitive groups - in the UK this approach is used to declare “Air Quality Management Areas” which can become the focus of local policy.

For larger urban scale or regional studies the model domain is obviously much larger and the grids are normally less resolved (large high resolution model domains are very computationally demanding). In larger studies it is common to map the concentration fields as averages in 1- 5km^2 grid squares, which can then be used in combination with population density data within each square to estimate exposure. The concentration, exposure and population data can therefore be manipulated in the GIS environment to yield inputs to CBA calculations. This approach allows the effect of changes in emissions across a large area to be tested and their benefits to be expressed in economic terms...

The Beijing study outlined in Section 6 is a good example of the use of spatially disaggregated datasets for air quality modelling and subsequent economic analyses.

7.3 The BENMAP tool

A promising tool which takes concentration field data and produces spatially disaggregated economic data to support CBA has recently been produced by the Environmental Benefits Mapping and Analysis Program (BENMAP) of the USEPA.

BENMAP is a Windows-based computer program that uses a GIS system to estimate the health impacts and economic benefits occurring when populations experience changes in air quality. The tool can perform a comprehensive benefits analysis but is straightforward enough for non-technical users to estimate benefits. Analysts have used BENMAP to estimate the health impacts from air quality changes at the city and regional scale, both within and beyond the U.S.

BENMAP includes nearly all of the information users would need to start performing a benefits analysis; advanced and non-U.S. analysts can customise the program to address any policy question. Because BENMAP is based on a GIS, the results can be mapped for ease of presentation. Some of the purposes for which BENMAP is used include the following:

- Generation of population/community level ambient pollution exposure maps;
- Comparison of benefits across multiple regulatory programs;
- Estimation of health impacts associated with exposure to existing air pollution concentrations;
- Estimation of health benefits of alternative ambient air quality standards;
- Performance of sensitivity analyses of health or valuation functions, or of other inputs; and
- Hypothetical, or “what-if,” type analyses.

Recent research has applied the tool to an analysis of health benefits from air quality policy interventions in Spain[17]. The researchers used the outputs from a CMAQ model as inputs to BENMAP.

The BENMAP tool has potential applications in Turkey for policy analysis although it does not include a dispersion model. Provided the outputs of the model can be formatted in such a way as to allow importing into BENMAP, the software is reasonably straightforward to use.

More information is available from the USEPA[18].

[17] E. Boldo et al., 2011. Health impact assessment of a reduction in ambient PM2.5 levels in Spain, Environment International 37, 342-348

[18] http://www.epa.gov/air/benmap/index.html

8 QUANTIFYING THE PHYSICAL IMPACTS OF AIR POLLUTION

8.1 Introduction

8.1.1 Scope

Previous sections have described the relationships between the emissions inventory, the air quality dispersion model, ambient air quality and the spatial mapping of pollutant concentrations and populations. This section explains how that information is used:

- To quantify the exposure of the population to pollution
- To translate these exposure data into health effects, and
- To quantify the number of cases of premature mortality and morbidity (illness) attributable to the exposure.

Section 9 then explains how monetary values can be placed on these impacts.

It is hard to know how effective an air pollution abatement measure is (or the benefits it provides) without information on the magnitude of the economic damage that air pollution causes. In order to help policy makers identify or create more effective measures to counter air pollution it is necessary to assess the costs of air pollution to public health and welfare.

The effects of air pollution exposure extend beyond damage to human health, and can include damage to crop yield and appearance, the disappearance of trees and reduced timber yield, a reduction in ecological diversity, and the loss of fish from lakes and rivers. In addition, other effects of air pollution exposure include the degradation of building materials (such as steel, stone, cement, concrete, paint, etc) and damage to other materials, such as rubber and textiles.

Each of these impacts can be factored into analysis of the costs and benefits of reduction measures. The focus of this report, though, is solely on health impacts. This is not because other impacts of exposure are not important – they are – but because the health benefits of air pollution control tend to significantly outweigh other benefits, as indicated in Table 8-1.

It is also the case that most of the policy measures introduced to reduce the impacts of air pollution exposure on human health also lead to a reduction in damage to other receptors, such as building structures and ecological systems.

Table 8-1: Health Benefits as a Percentage of Total Benefits in Recent Cost-Benefit Studies

Study	Title and subject area	Health benefits as a % of total benefits
Holland and Krewitt, 1996	Benefits of an acidification strategy for the EU: reductions of SO_2 and NH_3	86 – 94%
AEA Technology, 1998a	CBA of proposals under UNECE Multi-Effect Protocol: reductions of SO_2, NO_x and NH_3	80 – 93%
IVM, NLUA and IIASA, 1997	Economic evaluation of AQ for SO_2, NO_x, fine and suspended particulate matter	32 – 98%
AEA Technology, 1998b	Economic evaluation of the control of acidification and ground level ozone: reductions of NO_x and VOCs	52 – 85%

8.1.2 Approach

Advances over recent years now mean it is possible to assess the health and environmental effects of air pollution. Assuming that monitored and modelled ambient air quality have been mapped – see Section 7– the steps involved in quantifying and monetising the health benefits of air pollution control measures are:

1. Assess the exposure of the population to pollutants by combining data on concentrations with population density maps
2. Use exposure-response functions that link changes in pollution levels to health effects to quantify the number of cases of various effects
3. Apply accepted valuations to monetise the effects calculated.

The calculation of exposure and effects serves two inter-related purposes:

- It enables the baseline conditions towards which air quality improvement policy and objectives are directed to be assessed. Baseline conditions are measured in terms of current pollution concentration levels and health effects (assessment phase).
- It enables the effects on the baseline conditions of a range of pollution abatement measures to be analysed in terms of reduced pollution levels and health impacts (options analysis phase). From this the relative effectiveness of each of the measures can be gauged.

Estimating the contribution of specific air pollutants to specific health effects involves four steps:

1. Determine the spatial concentrations of an air pollutant using either fixed-site monitoring data or dispersion model-based estimates
2. Determine the size of the population groups exposed to the air pollutant concentrations
3. Measure the total exposure of the population to the pollutant
4. Determine the exposure-response functions to be used in relating air pollutant concentrations to the selected health effects
5. Use the exposure-response functions to calculate the number of cases of each health effect that are attributable to the air pollutant being examined

8.2 Pollutants Assessed

A single index pollutant is commonly selected to estimate the health impacts of air pollution, though it is recognised that selecting only one ambient air pollutant as the main pollutant may underestimate the magnitude of the health effects. However, as the ambient concentrations of many air pollutants are highly correlated, simply aggregating their impacts may overestimate the total health effects.

PM_{10} has been identified as the most useful indicator because of the large amount of epidemiological literature on its health effects and also because PM_{10}

(which includes finer particulate material $PM_{2.5}$) contributes most to the total health costs of air pollution[19].

For example, in the Beijing transport study[20] – see Section 6 also - PM_{10} was selected as the index pollutant for estimating the overall health effects of traffic-related air pollution. Reasons for this were:

- The focus of controlling mobile source emissions had shifted in recent years from NO_x and VOCs to PM_{10} emissions after researchers found that PM_{10} damage values are far higher than those for VOCs and NO_x.
- The health effects of PM_{10} can be quantified using the current literature. Although the smaller particles (e.g. $PM_{2.5}$) have potentially larger health effects, epidemiological evidence on those effects was limited.

Two recent Australian studies adopted similar approaches:

- A study to estimate the health costs of ambient air pollution in the Greater Sydney Metropolitan Region[21] identified health effects due to the following ambient air pollutants: particulates; carbon monoxide; sulphur dioxide; hydrocarbons; nitrogen dioxide; ozone; lead; air toxics (benzene and 1,3-butadiene).

  However, PM_{10} was used as the index air pollutant to quantify the health costs of ambient air pollution[22], because a broad and sound epidemiological literature was available for PM_{10}.

- A second study, to quantify the economic costs of the health effects of transport related ambient air pollutants in Australia[23], quantified health effects using PM_{10} as the index pollutant. PM_{10} was used because of available epidemiological and monitoring data.

  It was recognised that by selecting only one air pollutant health effects may have been underestimated, but the approach was consistent with the conservative approach used by Kunzli et al[24].

  Only transport-related outdoor ambient air pollution was assessed and monetised. Transport sources of ambient air pollution included motor vehicles, aircraft, shipping and boating, rail, paved and unpaved roads (dust) and related industrial activity.

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[19] Jalaludin, B, et al, Methodology for Cost-Benefit Analysis of Ambient Air Pollution Health Impacts, Australian Government Department of the Environment, Water, Heritage and the Arts through the Clean Air Research Program (2009).

[20] Guo. X.R., et al, Estimation of economic costs of particulate air pollution from road transport in China, Atmospheric Environment 44 (2010).

[21] Australian Department of Environment and Conservation, Health effects and monetary value of health effects, 2005.

[22] Kunzli et al, Health Costs Due to Road Traffic-related Air Pollution - An Assessment Project of Austria, France and Switzerland. London, Prepared for the Third Ministerial Conference for Environment and Health (1999)

[23] BTRE, Health Impacts of Transport Emissions in Australia: Economic Costs. Canberra, Australian Government, Department of Transport and Regional Services, Bureau of Transport and Regional Economics (2005)

[24] Kunzli et al, Public Health Impact of Outdoor and Traffic-related Air Pollution: A European Assessment. Lancet, 356, 795-801 (2000)

8.3 The Health Effects to be Quantified

8.3.1 Premature mortality and morbidity (illness)

The health effects of air pollution range from minor, reversible respiratory symptoms that affect almost everyone exposed to it, to aggravation of symptoms among asthmatics, and to hospitalisations for cardiopulmonary disease and premature death of some with pre-existing disease (Jalaludin et al, 2009[25]).

A key factor in deciding on the health effects to be quantified is the availability of good quality published quantitative data on exposure-response functions – see Section 8.5. Another is the ability to express the health effect in monetary values. Advice from air pollution epidemiologists should be sought when deciding on the health effects to be examined.

Health effects associated with ambient air pollution are divided into two broad categories: premature mortality and morbidity (illness).

Air pollution has both acute and chronic impacts for premature mortality. Earlier studies reported the relationship between air pollution exposure and human health by time-series data. More recently, cohort studies have become recognised as the best evaluation method for establishing the long-term impacts of air pollution. The outcomes of such studies reflect a combination of acute and chronic effects, as these accumulate over long-time periods and could be triggered by either cumulative or short-term peak exposures (Dominici et al., 2003; Kunzli et al., 2001; cited in Jalaludin et al, 2009[26]).

Cohort studies are therefore considered to represent more accurately the full effects of air pollution than time-series studies. However, very few of the current morbidity studies have focused on chronic morbidity. Time-series studies therefore tend to be relied upon for most morbidity effects, although this is likely to lead to an underestimate of total morbidity (Jalaludin et al, 2009[27]).

8.3.2 Examples

Some recent examples of health effects quantified are:

- The Beijing transport study referred to above (Section 8.2) relied on published studies on air pollution and health. Exposure-response functions derived from epidemiological studies conducted in China were used whenever possible. International studies were used only when local studies were not available.

  The assessment considered chronic mortality; acute and chronic bronchitis; and acute cardiovascular hospital admissions, respiratory hospital admissions, asthma attacks, outpatient hospital visits, and restricted activity days.

  The incidence rates of health endpoints were obtained from the year books of the Beijing Municipal Bureau of Public Health (BMBPH) and Chinese Ministry of Health (CMH) and other references.

- The health effects quantified for the Greater Sydney Air Pollution Study (Section 8.2) were: long-term mortality, respiratory hospital admissions,

[25] *Ibid*.

[26] *Ibid*.

[27] *Ibid*.

cardiovascular hospital admissions, asthma attacks in adults and children, restricted activity days, acute bronchitis and chronic bronchitis.

- The Australian Transport Study (Section 8.2) quantified the health effects of chronic mortality, all respiratory hospital admissions, all cardiovascular hospital admissions, chronic bronchitis, bronchitis episodes in children, restricted activity days, and asthma attacks in adults and in children.

8.4 Population Exposure

8.4.1 Pollutant Emissions-Concentrations Relationships

Figures 8-1 and 8-2 show the relationships established between emissions sources, pollutant types and pollutant concentrations for four cities in Europe, Asia and South America. For four sectors (transport, small sources, large industry and power and heat) and three pollutants (PM_{10}, SO_2 and NO_2) they show, respectively:

- The contribution made by each emissions source to total emissions from fuel combustion, and
- The contributions made by each emissions source to ambient air pollutant concentrations.

Pollutant concentration data sets provide the basis for establishing the level of exposure of the population to pollution.

The relationship between emissions and concentrations is far from being a direct one. The purpose of the dispersion model is to project these relationships for the different sources and pollutants, thereby allowing abatement policy to be focussed on the sources with the greatest impact on human health. The quality of the relationships established depends to a significant degree on the quality and comprehensiveness of the data available from the emissions inventory.

Figure 8-1: Sector Contribution to Emissions from Fuel Combustion

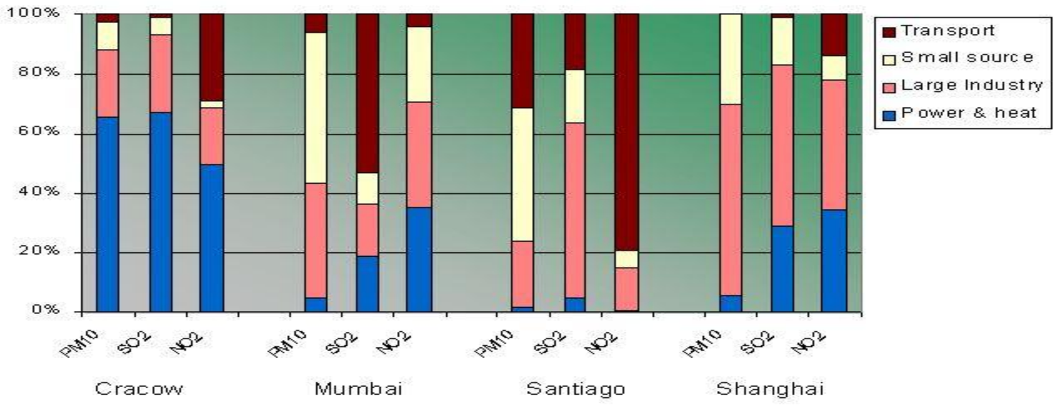
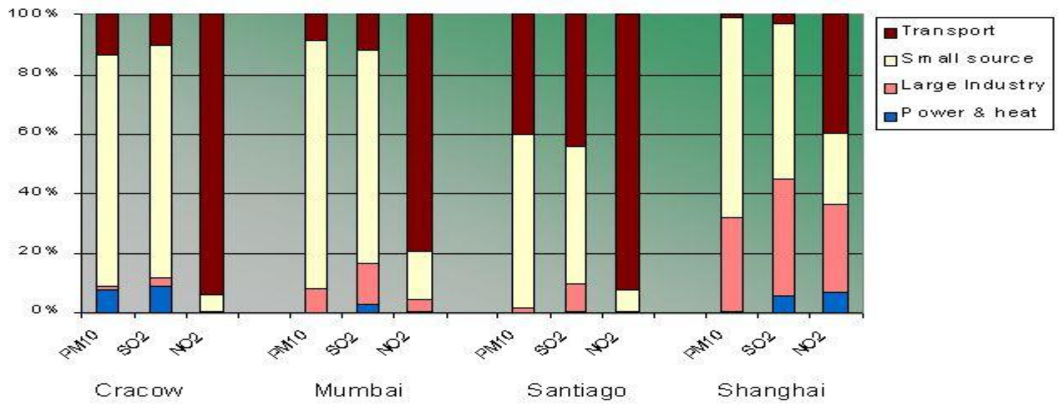


Figure 8-2: Sector Contribution to Ambient Air Concentrations from Fuel Combustion



8.4.2 Measuring Exposure

Exposure is the product of the pollutant concentration and the population affected. Each is mapped spatially and digitally on an identical grid system. Exposure is measured for each cell as person.mg/m^3 (the product of the concentration and the estimated population for each cell. An exposure index (total impact) is calculated as the sum of exposures across the entire grid system.

The exposure-index is the population weighted index of *total* pollutant concentrations. Dividing the emissions index by the total population gives the population-weighted *average* pollutant concentration. A simplified example of the exposure index calculations is shown in Table 8.2.

The data shown in Table 8-2 are a small sub-set from a much larger data set from which the base-case exposure conditions for the heavily industrialised town of Fushun in NE China were estimated. The total exposed population was 1.5 million people, with a total reported exposure index of 145,276 person.mg/m^3.

Table 8-2 Calculating the Exposure Index

Grid Reference		Population	Concentration	Exposure
X	Y		mg/m3	**person.mg/m3**
58500	29500	100	0.0869	8.6914
59500	29500	100	0.0871	8.7145
60500	29500	100	0.0874	8.7424
61500	29500	100	0.0876	8.7631
62500	29500	100	0.0878	8.7849
63500	29500	100	0.0870	8.6962
64500	29500	100	0.0886	8.8629
65500	29500	100	0.0866	8.6639
66500	29500	100	0.0853	8.5306
67500	29500	100	0.0847	8.4680
Exposure Index				86.9180

Source: Liaoning Air Quality Management Benefits and Damage Assessment Model (2002) for the town of Fushun, China. The data are a small sample from a grid represented by 476 rows of data.

The significance of a potential pollution control measure on the base-case conditions is determined by establishing the change in the base case conditions resulting from implementing the measure. This is achieved by:

- Modifying the input and output characteristics of the emissions inventory for the expected changes in pollutant emissions resulting from the implementation of a specific air pollution control measure
- Running the air-quality dispersion model with the changes made to the emissions inventory data
- Running the economic model to calculate the exposure index for the changed concentration levels
- Calculating the difference in exposure indices between the base case and the “with abatement” case
- Using the difference in exposure index to estimate the changes in health effects resulting from the abatement measure

The exposure index is linked to an appropriate exposure-response function (see below) to estimate the effect of the abatement measure on the number of cases of a particular health outcome. The monetary value of the damages avoided can then be quantified. Translating changes in pollution exposure into population health outcomes and monetary values enables the economic impact of the changes to be quantified and the value to society of specific pollution control measures to be assessed and compared.

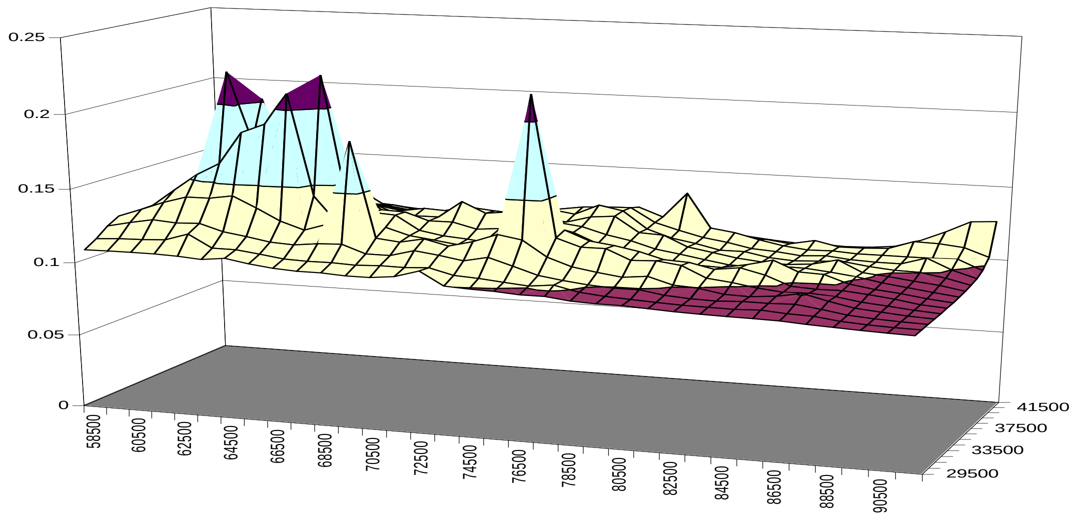
8.5 Exposure-Response Functions

Exposure-response functions are causal links between air pollution and damage to human health. They measure the change in the incidence of a health-related outcome relative to a change in pollutant concentration levels. The change is measured in terms of the number of cases avoided. For example, changes in the number of premature deaths or in the incidence of lower respiratory infections. The functions specify the expected change in the incidence of an effect (e.g. the number of people dying prematurely) in response to each 1µg/m^3 change in pollutant concentrations. The approach to the calculation is:

- Changes in the concentrations of air pollutants in the atmosphere give rise to changes in health effects.
- Exposure-response functions relate changes in pollutant concentrations (e.g. PM_{10}) to changes in the damage caused. This is reflected in changes to the number of cases of health effects.
- The physical impact depends on the size of the population exposed and the pollutant concentrations (the exposure index).
- Data on the costs of illness and the value of premature death are then applied to the physical impact (e.g. number affected by a specific illness) to estimate the change in damage costs.

Figures 8-3 and 8-4 provide an example of the relationship between pollutant concentrations, as projected by a dispersion model, and the population exposed to the pollutants[28].

Figure 8-3 Concentrations (mg/m^3 PM_{10})



The exposure peaks are highest in areas suffering both high pollution concentration levels and high population densities. Because high levels of pollution do not necessarily coincide with high levels of population, the peaks in the exposure graph are different from the peaks in the concentration graph.

Exposure-response functions are typically drawn from primary research undertaken elsewhere. The underlying epidemiological research can, however, sometimes form part of the overall study. For example, extensive research into the epidemiology of severe air pollution in the north-eastern province of Liaoning in China was undertaken by Professor Xu[29] and used directly in developing air-pollution control strategies for major towns in the province[30]. The exposure-response functions estimated by Professor Xu are given in Table 8-3.

[28] The charts are from the air quality dispersion model constructed for the Air Quality Improvement Strategy prepared within the framework of the EU-China Liaoning Integrated Environmental Programme

[29] Xu Zhaoyi (2002) Assessment of economic loss due to health impact of air pollution in Fushun. Paper produced for LIEP.

[30] The work was undertaken as part of the Air Quality Improvement component of the EU-China Liaoning Integrated Environmental Programme (Lot D)

Figure 8-4 Exposure (person.mg PM_{10}/m^3)

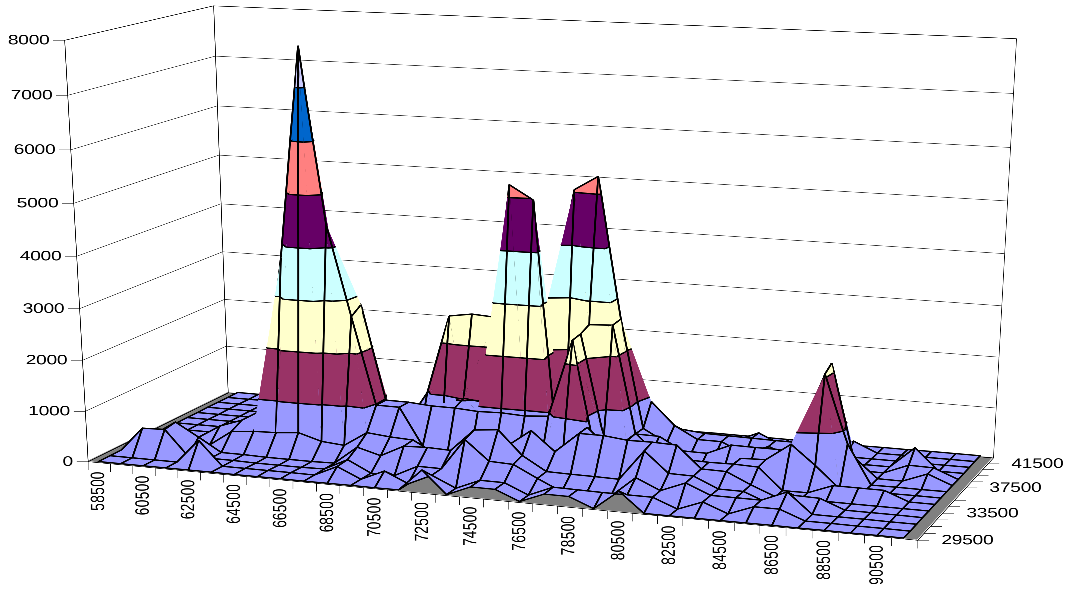


Table 8-3 Exposure-Response Functions Calculated for Liaoning Province, China

Pollutant	Exposure	Health Effect	Exposure-Response Function
			cases/person/(µg/m3)/year
PM_{10}	Acute	Respiratory hospital admissions	0.000001
PM_{10}	Acute	Asthma	0.000239
PM_{10}	Acute	Acute upper respiratory infection	0.000231
PM_{10}	Acute	Acute lower respiratory infection	0.000311
PM_{10}	Chronic	Bronchitis	0.000286
PM_{10}	Chronic	Emphysema	0.000029
		Premature mortality	
PM_{10}	Chronic	- life years lost	0.000012
PM_{10}	Chronic	- labour years lost	0.000012

Source: Xu Zhaoyi (2002) Assessment of the economic loss due to the health impacts of air pollution in Fushun. Paper produced for LIEP.

Taking the health effect 'acute upper respiratory infection' as an example, an exposure response function of 0.000231 cases/person/(µg/m^3)/year means that a 1 µg/m^3 change in PM_{10} concentrations will lead to a change of 0.000231 in the number of cases of the health effect for each person exposed.

The total number of cases of a specific health effect is thus the product of the 'exposure index' and the 'exposure-response function'. Using the same example and the exposure index of 86.918 person.mg/m^3 calculated in Table 8.2, the total number of cases of upper respiratory infection per year is: 86.918 * 0.000231 * 1000[31], or 20 cases.

The example figure is based on a very small population (1000 people). When the exposure index of 145,276 person.mg/m^3 for the entire study area is used, the number of cases for each µg/m^3 change in PM_{10} concentration becomes significant, at 33,560 cases.

Table 8-4 gives examples of exposure-response functions calculated for changes in acute mortality in response to a 1µg/m^3 change in PM_{10} concentrations. The

[31] To convert from mg to µg

APHEA meta-analysis, for instance, indicates a change of 435 in the number of people / million dying in response to a 1μg/m^3 change in PM_{10} concentrations.

Table 8-4: Acute Mortality and Particles (PM10)

Group	Area	Source	Function	
All	Europe	APHEA Meta-analysis	0.0435%	Per 1μg/m^3 PM_{10}
All	Western Europe ExternE	Mean of Spix et al; Verhoeff et al, 1996	0.0399%	Per 1μg/m^3 PM_{10}
All	WHO	Joint estimate (WHO 1996)	0.0737%	Per 1μg/m^3 PM_{10}
All	Chile	Ostro et al 1996	0.0669%	Per 1μg/m^3 PM_{10}
Elderly (65 +)	Brazil	Saldiva et al 1995	0.122%	Per 1μg/m^3 PM_{10}

In another example, Table 8-5 summarises the estimated health damages in China in response to increases in ambient concentrations of $PM_{10.}$ The term 'Function' is the number of additional cases per million people each year for every 1μg/m^3 increase.

Table 8-5: Estimated Health Damages from Air Pollution in China (World Bank, 1997[32])

Endpoint	Function	Cases/year in China
Premature mortality	6	178,000
Respiratory hospital emissions	12	346,000
Emergency room visits	235	6,800,000
Restricted activity days	57 500	4,500,000
Lower respiratory infections	23	661,000
Asthma attacks	2608	75,000,000
Chronic bronchitis	61	1,800,000
Respiratory symptoms	183 000	5,270,000,000

Note: Functions are in terms of additional cases per 1 million people for every 1μg/m^3 increase in ambient concentrations of PM_{10}

8.6 Summary

Table 8-6 consolidates data from the Fushun air quality study referred to earlier. It shows relationships between exposure index, exposure-response functions and the change in the number of occurrences of each health effect resulting from a 1μg change in the population-weighted average concentration of PM_{10}. The study area population was 1.5 million persons.

[32] World Bank, Clean water, blue skies: China's environment in the new century (1997)

Table 8-6: Summary Calculation of Health Effects from the Fushun Study

Health Effect	Exposure Index	Exposure-Response Function	No of Cases
	person.mg/m3	cases/person/(µg/m3)/year	
	(A)	(B)	(A*B*1000)
Respiratory hospital admissions	145,276	0.000001	145
Asthma	145,276	0.000239	34,768
Acute upper respiratory infection	145,276	0.000231	33,525
Acute lower respiratory infection	145,276	0.000311	45,197
Bronchitis	145,276	0.000286	41,507
Emphysema	145,276	0.000029	4,151
Premature mortality (life years lost)	145,276	0.000012	1,680

NB: Differences between the calculation and the number of cases are due to rounding errors

The main advantages of this approach are that it:

- Allows the use of state of the art environmental and health models
- Can easily be updated
- Permits spatial dependence of impacts relative to the sources of emissions to be taken into account.

It can use data relevant to individual technologies and locations that are important in the context of optimising measures to reduce urban energy-use improve vehicle emissions efficiency and improve air quality cost-effectively.

Understanding the relationship between exposure (exposure index) and health effects (exposure-response functions) enables the physical impacts of changes in pollutant concentrations on health to be quantified. Placing monetary values on the physical impacts and comparing these values with the costs of pollution abatement measures are the subjects of the next two Sections.

9 VALUING THE BENEFITS OF AIR POLLUTION CONTROL

9.1 Introduction

Placing monetary values on health effects has become a critical component in assessing the social costs of air pollution. By enabling a cost-benefit analysis of pollution control measures to be undertaken it provides a basis for setting priorities for action.

This section introduces the techniques used to calculate the economic cost to society of the damage caused by air pollution, whereby monetary values are established and assigned to the health effects identified in Section 8. For example, the financial cost of a period of hospitalisation resulting from contracting an upper respiratory infection or the value placed on premature death. The concepts and measurement methods that underpin this analysis are described below. An overview of the economic analysis is given in the Part 2 Report.

The main steps involved in undertaking an economic assessment of either a specific air pollution control measure or a combination of measures are:

- Quantify the monetary value of the benefits from implementing the air pollution control measure (i.e. improvements in human health)
- Quantify the financial costs of implementing the air pollution control measure (e.g. switching to more emissions-efficient cars or a bus fleet retirement and replacement programme)
- Compare the size of the benefits to the size of the costs and determine whether the proposed measures add value to society
- Rank alternative strategies according to their net economic benefit to society

The Section closes with an illustration of how the concepts introduced here were applied in Fushun (Section 8.5) and an illustration of the significance of different health impacts found in a Canadian study (Section 8.6).

9.2 Valuing Health Benefits

There are a number of reasons why it is useful to translate health benefits from physical values (such as the number of cases of an illness) into monetary terms:

- There are a large number of effects (different forms of illnesses, for example) which cannot be directly compared. Placing a monetary value on these effects enables such comparisons to be made.
- Also, monetary valuation allows direct comparison of benefits with costs, thereby improving the identification of the most effective control strategies and the quality of the decision making process.

A number of approaches can be used to value air pollution effects. Approaches to valuation most commonly used in air pollution control analysis are:

- Cost of illness (morbidity)
- Human capital (mortality)
- Willingness to pay (mortality, morbidity).

9.3 Valuing Illness (Morbidity)

One approach is simply to value pollution-related illness (morbidity) in terms of lost productivity (measured in terms of wages lost) plus medical and hospitalisation costs, etc. Data on the costs of illness can either:

- Be transferred from research studies that have been undertaken elsewhere (known as secondary valuation or benefits transfer). If this approach is used it is necessary to adjust the values transferred for differences in income levels between those in the source country to which the research relates and those of the country undertaking the CBA e.g. Turkey.
- Alternatively, primary research into health costs can be undertaken directly for the city (or country) in which the air pollution control strategy is being prepared.

Primary valuation methods generally require extensive research and are expensive and time-consuming to apply. The use of 'secondary' methods is therefore often more practical for project assessments. However, the approach used in the example of Fushun referred to in Section 8 was based entirely on primary methods, using the actual costs encountered in the study area rather than on adjusted figures from other places.

A quite different approach to determining the value of the costs of pollution-related illness is to ask people directly for their willingness-to-pay (WTP) to reduce morbidity *risks* using an approach based on the theory of contingent valuation. Significant research has been undertaken to elicit peoples' willingness to pay to reduce risks of illness, research which can be transferred for use elsewhere (subject to the income adjustments referred to above).

This is also the primary method used in approaches to placing monetary values on premature death.

9.4 Valuing Premature Mortality

It is clear that some people die earlier than they otherwise would as a result of exposure to some forms of air pollution and that this has a cost for society. If this cost is to be reflected in the economic analysis then, somehow, a value must be placed on the effect of premature death. Understandably, significant controversy surrounds the valuation of effects on human health, and particularly on mortality (premature death). This relates to two principal issues:

- The 'moral' issue of 'valuing life'; and
- The methodology used for valuing the effects on mortality.

Part 2 of the Report provides an introduction to these issues and explains a number of the terms used in placing an economic value on premature mortality. Suffice to say here that economic values can be placed on premature death though the estimates can vary considerably and there is not unanimity on the approach to be adopted. Key terms explained in Part 2 are:

- Value of statistical life (VOSL) – estimates are provided in Part 2
- Value of life-year (VOLY) – estimates are provided in Part 2
- Willingness to pay (WTP)
- Contingent valuation (CV).

9.5 Example of Health Impacts Valuation: Fushun, China

Table 9-1 summarises the estimated health impacts of air pollution in Fushun, China, continuing the analysis of Table 8-6 in Section 8. It may be seen that, whilst the number of premature deaths was much lower than the aggregate number of cases of illness, premature death represented just over 50% of the total estimated health damage costs.

Table 9-1: Summary Calculation of Health Effects from the Fushun Study

Health Effect	No of Cases	Monetised Value / Case	Health Damage Costs	
		(Yuan)	Total (Yuan m)	%
	(A)	(B)	$(A*B)/10^6$	
Respiratory hospital admissions	145	3,665	0.53	0.2%
Asthma	34,768	133	4.62	1.8%
Acute upper respiratory infection	33,525	214	7.17	2.7%
Acute lower respiratory infection	45,197	276	12.47	4.7%
Bronchitis	41,507	2,230	92.56	35.2%
Emphysema	4,151	2,960	12.29	4.7%
Premature mortality (life years lost)	1,680	79,540	133.65	50.8%
Total	160,973		263.30	100%

NB: Differences between the calculation and the number of cases are due to rounding errors

9.6 Significance of the Different Health Effects

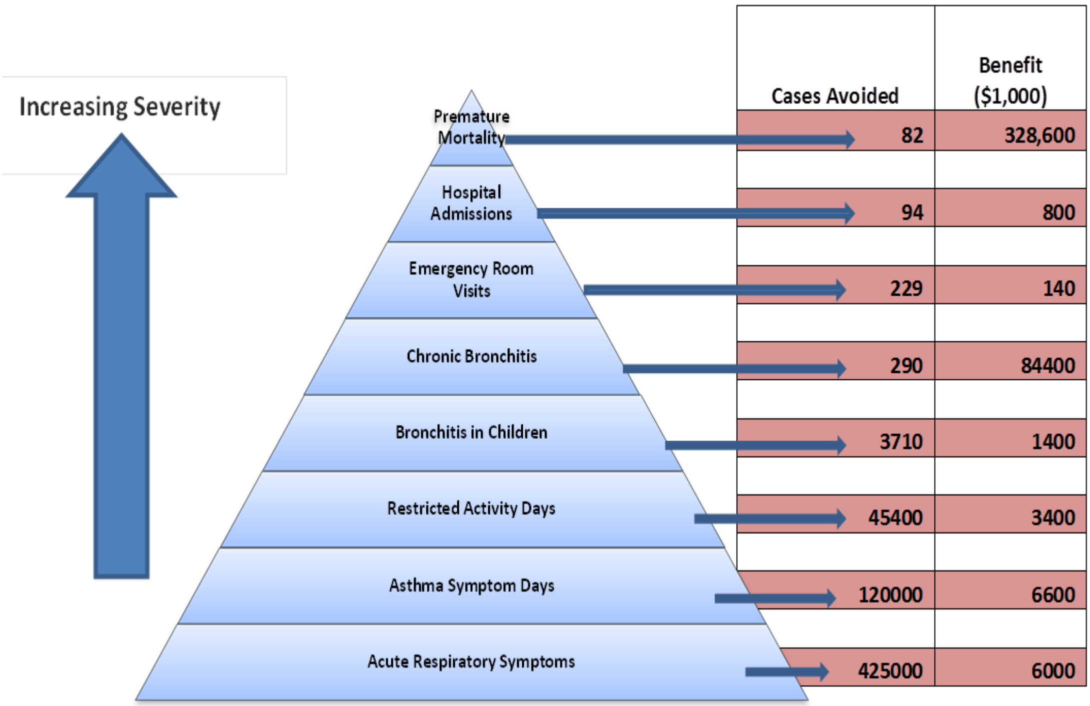
A difficulty in quantifying the health impact of air pollution is that a very large number of people are exposed to relatively low levels over long periods of time, often resulting in mild or rare health problems that are difficult to value or attribute to a given emission source (Jalaludin, 2009[33]).

Figure 9-1 illustrates this issue regarding the health effects of air pollution. It shows as a pyramid the expected effects of reduced sulphur in gasoline in the year 2020 in Canada. The pyramid shows the following:

- The lowest numbers of cases of avoided adverse effects are in the most severe health effects categories (hospital admissions and premature mortality).
- Over 90% of the estimated adverse effects avoided consist of reductions in acute respiratory symptom days.
- In contrast, the monetary valuation of the outcomes avoided is dominated by the valuations ascribed to the most severe outcomes, despite representing only a small percentage of the total number of health effects avoided.
- For example, air pollution-related premature deaths, which have a central estimate valuation of approximately Canadian $4 million each, account for only 0.001% of the total number of adverse health effects avoided, but about 75% of their associated monetary valuation.

[33] *Ibid*

Figure 9-1: Pyramid of Expected Health Effects and Estimated Benefits



Source: Health Canada, Study of the Effects of Reducing Sulphur in Gasoline, Health and Environment Impact Assessment Panel Report, 1997

10 Pollution Abatement Costs and Cost-Effectiveness

10.1 Introduction

This chapter focuses on the assessment of pollution abatement costs and their cost-effectiveness. Cost-effectiveness is a measure of how *effective* an abatement measure is in abating emissions or in achieving specific air pollution management objectives relative to the financial cost of the measure. It is a measure of the cost per tonne of pollutant emission abated or the cost per µg/m^3 reduction in the ambient concentration of a pollutant.

The analysis is developed further in Section 11, which explains how the costs of abatement measures are linked to the monetary values of their associated health benefits (Section 9) to define an optimal programme of measures for air pollution control that can be implemented progressively over time.

Steps involved in selecting and evaluating the performance and effectiveness of potential pollutant abatement measures are:

1. Establish the investment and operating costs of each measure
2. Determine the contribution of each measure towards specified air pollution control objectives
3. Combine Steps 1 and 2 to assess the cost-effectiveness of each measure.

10.2 Measures of Cost-Effectiveness

Assessing the cost-effectiveness of a measure involves two elements:

- The emissions reduction or air quality improvement potential of the measure: note that the cost-effectiveness of emissions reduction is not the same as the cost-effectiveness of air quality improvement (which results from reductions in the concentration of a pollutant).
- The cost of implementing the measure, where costs reflect both initial capital costs and any ongoing operational costs (including any operating cost savings arising, for example, from reduced fuel consumption).

For many measures it is possible to define quite clearly what the *emissions reduction potential* is. This normally conforms to design standards and, when properly installed and maintained, the measure will achieve this level of reduction wherever it is located. Cost-effectiveness in terms of its emissions reduction potential (e.g., TL/tonne SO_2 reduced) is therefore a generic characteristic of a particular abatement option.

The cost-effectiveness of an abatement option in terms of its potential to *improve ambient air quality* depends, however, on a variety of factors:

- The location of the source
- The nature of the source (particularly stack height), and
- The location at which its effects are being measured.

This measure of cost-effectiveness (e.g., TL/µg/m$_3$) is therefore *location and plant dependent* and must be calculated separately for each city/area being examined.

Also, the cost-effectiveness of a measure in terms of its *potential to reduce the economic impact* of air pollution (e.g. reduced incidence of respiratory disease) is different again from its cost-effectiveness in improving ambient air quality.

In this case, cost effectiveness is determined in terms of the density of the population affected by ambient air quality in a particular area. The analysis can thus be extended to assess in monetary terms the population-weighted improvement of the measure; for example, the cost/unit value of health benefit achieved.

10.3 Costs

When considering the costs of different measures it is important that information is expressed in equivalent terms. There is often a tendency to ignore important differences in information that can have major effects on the relative costs of measures. In interpreting cost data it is important to know: when the cost information was compiled; whether the costs have been discounted over time; and what assumptions are included in the estimates of project life. Standard methods should be used in the cost analysis.

Costs are usually expressed in terms of the equivalent annualised cost of a measure. This includes (i) the annual operating costs plus (ii) the initial capital cost amortised over its expected economic life at an appropriate interest rate. The annualised approach to cost estimation enables comparisons to be made between different measures on an equivalent basis, in terms of cost per tonne of pollutant emission abated or cost per $\mu g/m^3$ reduction in pollutant concentration.

Note, however, that a number of emissions reduction measures can involve no capital costs, minimal operating costs and can generate net financial benefits to an enterprise. Most cost data in the literature centres on stationary abatement measures, with far less being available on the costs of non-technical measures, such as transport options and economic and fiscal instruments. The costs of such measures are more difficult to assess and tend to be strongly site specific.

10.4 Cost Curves

Once the costs of individual abatement measures have been established, the following parameters can be calculated for each measure:

- The cost per tonne of emissions abated
- The cost per $\mu g/m^3$ reduction in the ambient concentration of a pollutant
- The net benefit to society in terms of improvements to human health. This is considered in Section 11.

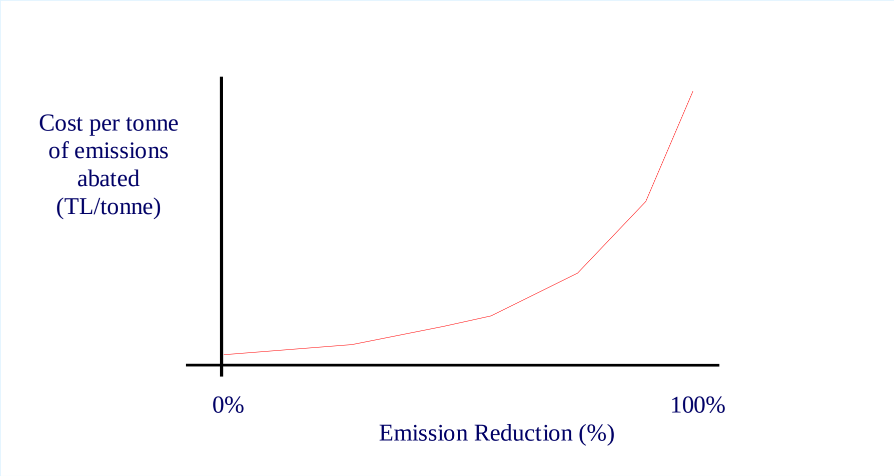
Consistent assessment of cost-effectiveness across a range of measures allows direct comparison to be made between the measures. More importantly, by enabling measures to be ranked it provides the basis for developing a cost-effective action plan.

Cost-curves are used to compare and rank the costs of different measures for reducing pollution levels. These typically show that the costs of pollution abatement rise exponentially: that is, the costs of pollution reduction are initially very low but begin to rise steeply at some point as additional measures are introduced (the law of diminishing returns).

10.4.1 Generic Cost-Curves

Generic cost curves are first prepared by sector (e.g. power sector, large industry sector, transport sector) and pollutant to show the relationship between cumulative emissions reduction (measured as a percentage of the total reduction potential) and marginal costs (measured in terms of the marginal cost per tonne of pollutant abated). Such a cost curve is illustrated in Figure 10-1. These cost-curves then form the basic building blocks for the cost analyses done in all cities.

Figure 10-1: Example of a Generic Cost Curve



The cost curve shows that costs rise as the level of technical abatement increases. Measures at the bottom-left of the graph are the most cost-effective and should, in the interest of economic efficiency, be introduced first.

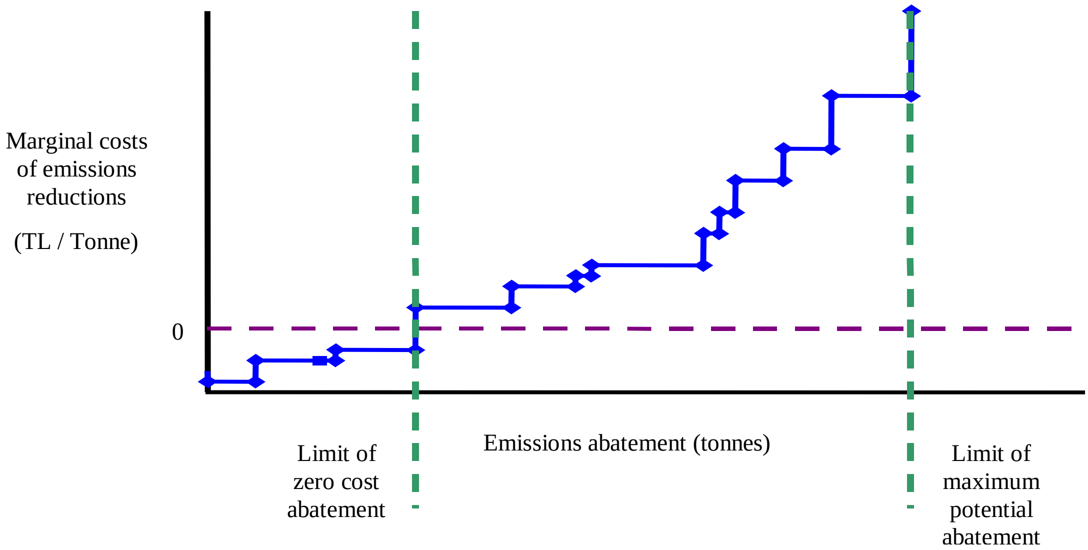
10.4.2 Area-Specific Emissions Cost-Curves

Once generic emissions cost curves have been prepared for a specific sector and pollutant they can then be translated into area-specific emissions cost curves, taking into account the actual emissions circumstances of the area in question. This cost-curve relates directly to the emissions inventory,

The curve (it is in fact more like a step function) then shows the cumulative emissions reduction (in tonnes) relative to the marginal costs of achieving these reduction levels (in TL/tonne). This is illustrated in Figure 10-2. Preparation of this chart depends on having information about the characteristics of the actual levels of emissions in the subject area attributable to each sector.

The cost-curve is prepared using data generated from an ‘emissions model’ – i.e. a mechanism for estimating the emission reduction expected from each measure. The ‘emissions model’ is run separately for each of the pollution abatement measures being assessed and the effect of each measure on total pollutant emissions recorded. From this the marginal cost/tonne of emissions abated by each measure may then be calculated. The measures are then ranked according to their cost-effectiveness and the cost-curve is prepared showing cumulative emissions abatement against the rising marginal costs (falling cost-effectiveness) of the measures. Pollution reduction effects of the measures are assumed to be additive.

Figure 10-2: Area-Specific Emissions Cost Curve



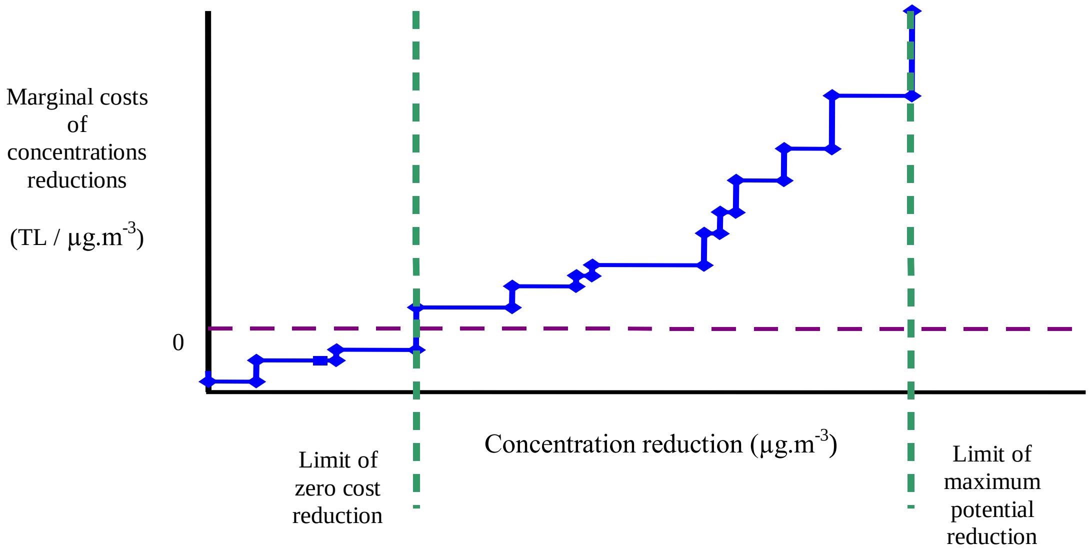
10.4.3 Area-Specific Concentration Cost-Curves

The cost-effectiveness of one measure relative to another in reducing emissions is not necessarily the same when measured in terms of its ability to reduce ambient pollutant concentrations. This is because of the dynamics of emissions after they have left the originating source: emissions at a low height above ground can have a disproportionately larger impact on ambient air quality than emissions released at a greater height because of dispersion effects. In order to establish the cost-effectiveness of a particular measure in terms of its *impact on ambient air quality*, it is necessary to relate emissions reductions to air quality improvements. Establishing these relationships involves using an air quality emissions dispersion model.

Air quality improvement cost curves for a specific location and sector show cumulative air quality improvements (in µg m^{-3}) relative to marginal costs (in TL/µg m^{-3}). They relate directly to the linked emissions inventory and dispersion model. They take explicit account of the contributions of each of the source categories to the final ground level pollutant concentrations across the city. This process effectively weights the marginal costs shown in the emissions cost curve according to the impact these emissions have on air quality, and enables a link between costs and benefits to be more directly established. An example of such a curve is shown in Figure 10-3.

In a manner similar to the emissions cost-curve, the concentrations cost curve is prepared using data generated from the dispersion model. The model is run separately for each measure, the effect of each on total pollutant concentrations is recorded, and the marginal cost per µg/m^3 of concentration reduction is calculated. The measures are ranked and the cost-curve prepared showing cumulative concentration reductions against the rising marginal costs of the measures.

Figure 10-3: Area-Specific Concentration Cost Curve



10.4.4 Population-Weighted Concentration Cost-Curves

A third category of cost curve, not shown here, illustrates the marginal costs for the various abatement measures in achieving progressively higher reductions in population-weighted ambient air pollutant concentrations (the exposure index).

Compilation of this cost-curve draws on the linked emissions inventory, dispersion and economic models to generate for each abatement measure the exposure index, the number of cases of each health effect, and the monetary value attributable to the health effects.

This comes closest to defining the most cost-effective set of abatement measures. This is because the basis for the calculation (reductions in the exposure index) is directly analogous to reductions in monetised health effects. This can help identify the abatement measures that bring maximum benefit to society (in terms of reduced health effects) at least cost.

By directly linking the costs of an abatement measure to its monetised benefits, this measure of cost-effectiveness leads to the concepts of cost-benefit analysis. CBA – the final component of the impact pathway approach to air quality management – is considered in the following chapter.

11 Cost-Benefit Analysis in Pollution Control Planning

11.1 Introduction

Section 10 has looked at how the cost-effectiveness of air pollution abatement measures can be assessed and ranked in terms of their relative costs in achieving ambient air quality objectives. It also showed how different measures of cost-effectiveness can affect the rankings of control measures (the order in which some control measures are preferred over others).

It showed that the effectiveness of an abatement measure can vary depending on how it is measured: whether in terms of emissions abated, concentrations reduced or the value of health effects avoided. For example, the effectiveness of an abatement measure in reducing pollutant emissions at source can be quite different to its effectiveness in reducing pollutant concentrations in ambient air.

A limitation of the first two measures of cost-effectiveness (cost/tonne of emissions abated and cost/µg m^{-3} of pollutant concentration reduced) is that they offer no information on whether an abatement measure is of *net benefit* to society. That is, whether the value of the health benefits realised by the measure exceed the costs of implementing it. They focus solely on costs.

The third cost-effectiveness measure, however, by focusing on the exposure index, enables the costs of an abatement measure to be directly related to its monetised benefits (TL / TL of health benefit realised). From this, the optimal combination of abatement measures that will achieve the maximum benefit to human health at least cost can be determined. This is the role of CBA.

11.2 The Purpose of Cost-Benefit Analysis

CBA is an economic tool used to help decide whether or not an abatement measure or proposed programme of measures is beneficial to society. That is, whether its benefits (Section 9) outweigh its costs (Section 10).

CBA can be used to determine:

- If the benefits of an abatement measure are larger than its costs
- If the benefits of a proposed programme of abatement measures outweigh its costs
- The cut-off point at which inclusion of an additional abatement measure in a programme leads to a reduction in the programme's net benefits.

CBA is concerned with time:

- A programme of air pollution control measures is introduced progressively over time, with the more cost-effective measures coming first and the less cost-effective measures later
- The costs and benefits of an abatement measure occur at different points in time:
    - Investment costs are usually incurred when the measure is introduced
    - Operating and maintenance costs are incurred annually over the operational life of the investment

  - Likewise, monetised air quality benefits also arise annually over the operational life of the investment.

- The population density of a city or city area can change over time. This can result in a larger future population benefiting from air pollution control measures. This, in turn, can lead to a progressive increase in the size of the monetised benefits over time.
- Continued economic growth in Turkey can be expected to progressively increase average real incomes over time, thereby raising residents' ability and willingness to pay for improved air quality conditions in the future.

11.3 The Elements of Cost-Benefit Analysis

11.3.1 Introductory Comments

CBA relates the costs of implementing an emissions abatement measure to its benefits expressed in monetary terms. Measures can then be ranked according to their net benefits and the optimal combination of measures defined that maximises social net benefits.

This is important for planners and policy makers responsible for preparing and implementing local air pollution abatement plans intended to achieve maximum social benefits within strict financial constraints over specified periods of time.

In order to compare the costs of a project or programme with its benefits it is necessary to find a common and unambiguous parameter by which to represent the uneven flows of future costs and benefits (cash flows) that are to be compared. The concept of 'discounting' is used to calculate such a parameter.

Discounting converts future flows of costs or benefits into an equivalent single value, known as the present value[34] (PV). The financial costs or monetised benefits projected for each year of a cash flow are discounted to their present values using a specific factor, called the discount rate. This gives more weight to present costs and benefits than to future ones.

The appropriate discount rate to use is typically defined by the national finance agency, but is likely to be within the range of 3 – 5%. Table 11-1 gives a simple example of the discounting process using a higher discount rate of 10%.

Table 11-1: Discounting Example

TL '000	Year	0	1	2	3	4	5
Costs		50,000					
Benefits		0	30,000	30,000	30,000	30,000	30,000
Net Cash Flow		-50,000	30,000	30,000	30,000	30,000	30,000
Discount factor @ 10%		1.000	0.909	0.826	0.751	0.683	0.621
Discounted Cash Flow		-50,000	27,270	24,780	22,530	20,490	18,630
NPV of Cash Flow	63,700						

Cost-benefit analysis is undertaken in constant values. That is, all costs and benefits, whether incurred today or in the future, are expressed in today's prices. The effects of inflation are not considered at this stage. The purpose of the discount rate is to convert future costs and benefits (expressed in today's prices) into their equivalent present values today (not to remove the effects of inflation).

[34] The process of discounting can be thought of as the converse of compound interest, whereby the future value of a given amount of money today is calculated through the compounding effect of the interest rate. In discounting, the present value of a known future amount of money is calculated via the discounting effect of the discount rate.

An economic model is created using MS Excel, for example, to capture and analyse all data needed to compare the costs and benefits of a measure or programme of measures over time. Key initial decisions are:

- The projected implementation period for the local area air quality action plan (e.g. all elements of the plan to be implemented within10 years)
- The assessment period over which the analysis is to be conducted (a realistic time-frame of 15-20 years is typical)
- The discount rate to be used

Further details of the data requirements and the steps involved in preparing a CBA are given in the Part 2 Report.

Note that:

- For the analysis of a single measure, cost-effectiveness and net benefits are calculated as if the measure were to be implemented today.
- For the analysis of a programme of measures, each measure is incorporated progressively over time, in order of its cost-effectiveness.

  This means that capital expenditures are incurred throughout the implementation period as additional measures are introduced. O&M costs and benefits rise accordingly over the assessment period.
- It is quite possible that some investments will need to be replaced over the assessment period.

An air pollution control project or programme is considered to be economically acceptable if the PV of its projected benefits is greater than the PV of its projected costs.

The parameters typically calculated in the cost-benefit analysis are:

- Net present value (NPV)
- The benefit-cost (B/C) ratio
- The profitability index.

11.3.2 Net Present Value

NPV is a powerful indicator:

- It is the difference between the PV of the benefits and the PV of the costs. A positive NPV means that the benefits exceed the costs and that the measure or programme can be accepted on economic grounds. The measure should be rejected on economic grounds if NPV is negative.
- NPV is a measure of the value added to society by implementing the abatement measure. That is, it is a measure of the value added in addition to that needed for the benefits to cover the costs.
- Abatement measures should be added incrementally to the analysis with total NPV being checked after each measure is added. A rise in total NPV means that additional value is added by an additional measure. A fall means that the additional measure has destroyed value, its costs exceed its benefits, and it should therefore be excluded from the programme.

11.3.3 The Benefit-Cost Ratio

The B/C ratio is defined as the PV of the benefits divided by the PV of the costs. A project is considered to be economically viable if the B/C ratio is greater than 1 but should be rejected if it is less than 1.

The ratio relies on the same parameters as used in the calculation of NPV, it leads to identical conclusions and provides minimal additional information.

11.3.4 The Profitability Index (PI)

The profitability index is calculated by dividing NPV by the initial investment. It is particularly valuable for ranking abatement measures for which the net benefit is shown to be positive (NPV > 0 or B/C ratio > 1).

- It provides a measure of the value added to society by an abatement measure relative to its investment costs. For example, a PI of 3 means that the value added to society by the measure after covering its operating costs is three times the size of the initial investment.
- Pollution abatement measures can be ranked according to the profitability index: that with the highest ratio being ranked first.
- This can help establish – from an economic perspective – the priority order in which the measures should be implemented.
- This can be particularly valuable when – as is common – the air pollution control programme faces fixed capital constraints.

11.4 Concluding Comments

The above is a simple outline of the theory behind cost-benefit analysis. In practice, it is more complicated, forming part of the overall framework within which air pollutant emissions, concentrations, health effects and their monetised values are determined and analysed. This framework may itself be only part of a wider programme of measures intended to improve many aspects of city life.

The economic component has a key role in the development of an air pollution control strategy but can be undertaken only after large effort has been put into defining the building blocks of the strategy development process:

- The emissions inventory and model
- The dispersion model
- The digital mapping of pollutant concentrations and population densities
- The relationships between pollutant concentrations and the incidence of health effects
- The monetary values to be placed on those effects
- The costs and cost-effectiveness of abatement measures
- Comparing and ranking abatement measures according to their relative costs and benefits.

A high degree of interdependence exists between all of these components. For example, the emissions inventory and model must be adjusted each time the effects of an alternative abatement measure are to be examined; this, in turn, triggers responses in each of the other components. Air quality strategy planning can be an intricate and time consuming process. The level of complexity will, of course, depend on the nature of the system being examined.

If it concerns a tightly defined problem within a small area – possibly a stretch of road subject to high traffic-related ambient pollution levels – then the level of complexity may be low with only a very limit economic assessment required (perhaps excluding the need for a CBA).

If, on the other hand, the planning area covers an entire city, then the scope of the analytical framework can become very wide indeed. In particular, it is likely that air quality planning will form part of far more wide-reaching plans regarding the future physical and socio-economic development of the city and which may themselves bring about improvements to current levels of air pollution.

Under these circumstances it may be necessary to:

- Define a number of scenarios that reflect as closely as possible the projected development of the city
- Assess the impact of those scenarios on ambient air quality in the absence of specific pollution abatement measures being introduced
- Define a set of potential air pollution control measures to be implemented *in tandem* with the wider plans for the city.

Air quality planning then becomes a much larger task and one that needs to be integrated very closely into the wider planning structures of the city.

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