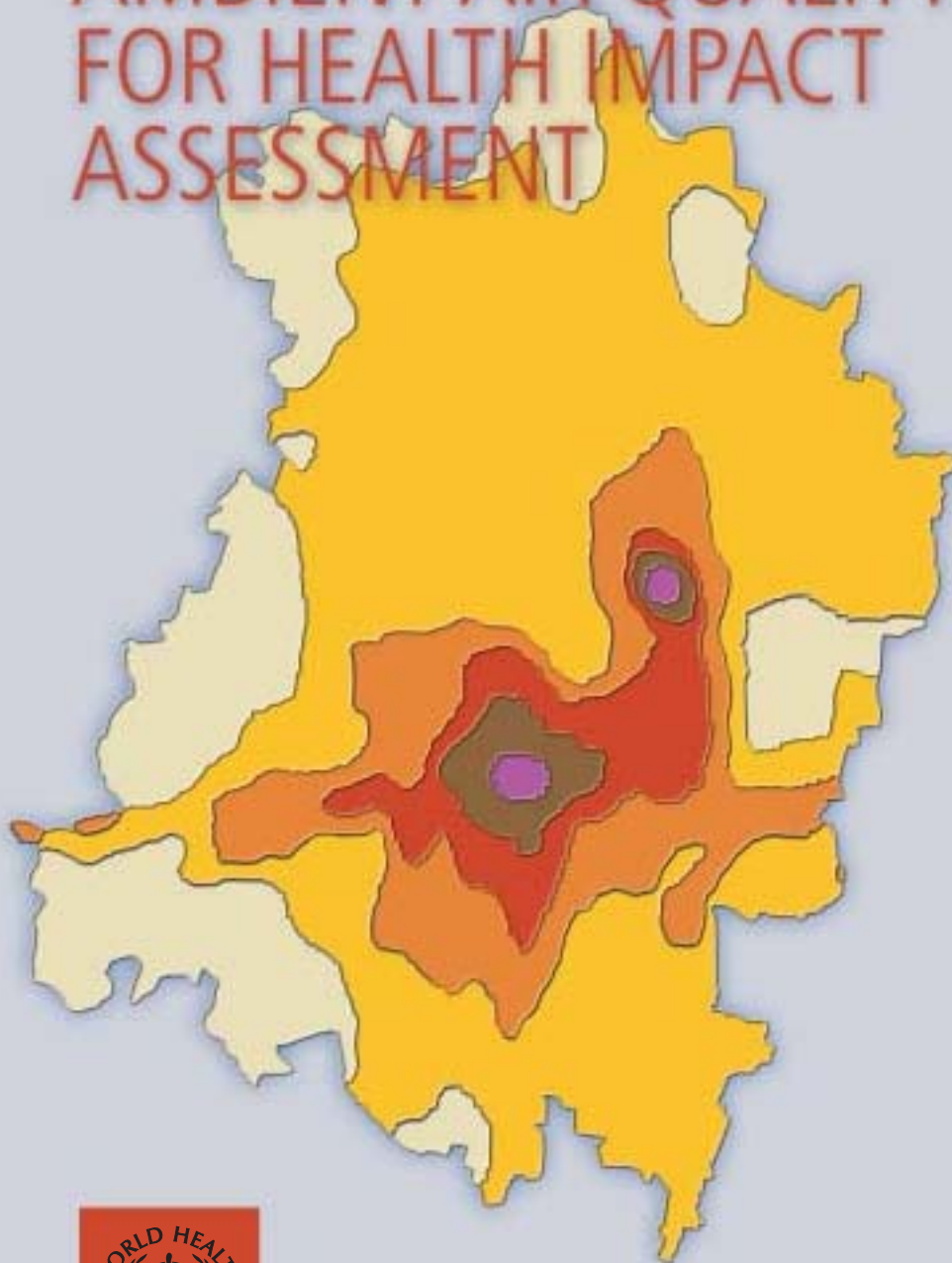


MONITORING AMBIENT AIR QUALITY FOR HEALTH IMPACT ASSESSMENT

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Foreword

The World Health Organization, including the Regional Office for Europe, has a long tradition in setting policies and promoting action to improve the environment and thereby support health. The European policy for health for all guided the regional action in the 1980s and 1990s. HEALTH21, the health for all policy for the European Region, poses new challenges. These policies recognize the physical environment as an important determinant of health, both through its direct effects on health and safety and through its significance in ensuring the sustainable development of society.

Ministers for health and for environment of all European Member States have met in 1989 in Frankfurt, 1994 in Helsinki and 1999 in London to maintain the momentum for improving regional and national policies on environment and health. National environment and health action plans, which have been developed in most countries of the Region, consolidate multisectoral efforts at the national level. Action in accordance with these plans must be preceded by thorough assessment of the current effects of the environment on health and of the benefits, or hazards, to health associated with the planned developments. When the action is implemented, population exposure should continue to be monitored to ensure that the risk prevention is effective and to provide operational feedback to all sectors and agencies involved in the action, including the public.

Clean air is necessary for healthy life. Nevertheless, many essential human activities related to economic and social development pollute the air where most people live. The recently revised WHO air quality guidelines for Europe identify the hazards related to a number of common air pollutants. Preventing the risks caused by these pollutants is most effective if the pattern of population exposure is recognized and monitored simultaneously with the implementation of action to prevent or reduce pollution.

This publication can assist national and local agencies responsible for protecting public health from the adverse effects of ambient air pol-

lution and for monitoring ambient air quality. It continues a series of monographs prepared by the Regional Office that present an overview of modern methods and allow optimum efforts to create environments supporting the health of the population of the European Region.

The working group that prepared this publication, convened by the WHO European Centre for Environment and Health, included individuals with a wide range of expertise and professional experiences. I am grateful to them for their creative and constructive discussion at the working group meetings, for their contribution to the text and for their work on revising and editing consecutive drafts of the publication. I gratefully acknowledge the financial contribution of the Government of Germany, which made this project possible.

J.E. Asvall
WHO Regional Director for Europe

Executive summary

Air quality assessment is frequently driven by the need to determine whether a standard or guideline has been exceeded. This overshadows another objective of air quality assessment: providing the information needed to estimate population exposure to air pollution and the effects on the health of the population. Consequently, most existing air quality monitoring systems do not fully address population exposure to toxic air pollution. Given the importance of these data for air quality management, this report describes strategies and methods for providing information on ambient air quality that is adequate for health impact assessment.

Human exposure to air pollution may result in a variety of health effects, depending on the types of pollutants, the magnitude, duration and frequency of exposure and the associated toxicity of the pollutants of concern. People are exposed to air pollutants both indoors and outdoors depending on the activities of individuals. It is important to assess the exposure levels of different population groups, especially sensitive or susceptible individuals such as children, elderly people and chronically ill people. Health impact assessment combines estimates of population exposure with information on toxicity or the relationship between exposure and response.

Information on the relationship between exposure and response is necessary to estimate the potential health risks. The estimates of health effects for a population base are typically calculated in terms of predicted excess negative health effects (such as increases in hospital admissions or mortality) caused by exposure to a certain level of air pollution. This involves combining the information on the response to certain concentrations derived from epidemiological or toxicological studies with the number of people exposed to each concentration of air pollution in the community being assessed.

The report discusses the overall requirements for designing and operating networks for monitoring ambient air and a range of air quality models used in comprehensive programmes for air quality assessment

and management. The report does not detail monitoring practice and methods. However, it is intended to prompt good practice in monitoring and health impact assessment. To this end, several important issues are identified and recommendations are made on each step of the process of monitoring air quality. Specific details of these issues are discussed in the body of this report and in the other reports cited in it.

The following points are addressed to decision-makers and designers of systems for air quality assessment.

Designing and planning a monitoring system

- Existing air monitoring systems often do not fully address the evaluation of population exposure to toxic air pollutants and the assessment of the resulting health effects.
- The design of new monitoring programmes or refinement of existing systems should therefore consider the need to use the data measured for the purpose of assessing the effects on population health.
- The pollutants studied, measurement time scales and locations should be relevant to assessing human exposure and the expected health effects. Local conditions and pollution climates will determine the pollutants and methods to be given priority.
- Monitoring can have many objectives besides health impact assessment. These objectives, together with resulting data quality objectives, need to be clearly defined when monitoring systems are designed or updated.
- Monitoring is only one of a range of tools for assessing air quality; monitoring, emission inventories and predictive models are complementary components in an integrated approach to assessing exposure and health effects.

Cost-effectiveness of assessment programmes

- Investment in monitoring, assessing and controlling pollution helps to avoid outcomes to health and ecosystems that are usually more costly than preventive action.
- Monitoring programmes need to be cost-effective, have stable financial, material and personnel resources and be adjusted to local needs and conditions.

- Always use the simplest technologies and procedures that are consistent with fulfilling the overall objectives of monitoring.
- In addition to initial capital expenditure, financial planning of monitoring programmes needs to take full account of ongoing expenditure for system maintenance, operation, data management and quality assurance and control.

Quality assurance and control

- Comprehensive quality assurance and control of monitoring programmes is essential to ensure that measurements are accurate, reliable and fit for the intended purpose.
- Harmonization of measurement quality – at both a national and international level – should be promoted through national quality assurance and control coordination, laboratory accreditation and international validation programmes.

Managing and disseminating information

- Raw measured data are of limited utility; these need to be transformed by appropriate analysis and interpretation into useful information, targeted at the needs of a wide community of end-users. These activities require special expertise, infrastructure and funding.
- Possible end-users may include scientific and health communities, policy and planning decision-makers at the local or national level, the mass media and the general public.
- Data and information from monitoring programmes should be communicated to users in the scientific and health communities in a form and time frame appropriate to their specific needs.
- Every person has the right to know about the quality of the air he or she breathes. Disseminating information on air quality to the public serves to inform, educate and raise awareness of important environmental and health issues.
- An informed and aware public can also contribute and assist in a meaningful way to improving the environment. Public communication and education schemes are therefore recommended.
- Free international exchange and dissemination of air quality information is recommended, using such freely and openly available communication media as the World Wide Web.

The report discusses numerous technical issues. The following points are addressed primarily to network managers and public health scientists who use air quality data.

Assessing population exposure

- Ambient air quality should be considered an indicator of true population exposure since factors related to buildings and indoor sources of air pollution influence personal exposure.
- To serve the needs of health impact assessment, air quality assessment should cover the full range of population exposure to ambient air pollution.
- Assessment of population exposure should address the differences in air quality within a city by considering a typical time–activity pattern and involving a time-weighted average calculation.
- Determining population exposure correctly requires knowing the population distribution and location of air monitoring stations to identify the pollution concentrations to which the population is exposed. Not only hot spots or areas where maximum concentrations are expected but also representative community sites where most of the population lives should be monitored.

Applying relationships between exposure and response

- The mathematical form of the relationships between exposure and response can vary depending on the toxicity of the pollutant.
- The concentration and exposure data for a given pollutant should be consistent with the corresponding averaging time specified by the relationships between exposure and response or other information on health effects.

Relationships between emission sources and air quality

- Ambient pollution sources in the urban environment include various stationary, mobile and area sources of emission. These sources must be identified to enable effective management action to reduce exposure.

-
- Air quality dispersion and statistical models are used to complement existing monitoring networks and are essential components of a comprehensive programme of air quality management.
 - Dispersion models can be used to select the location of representative or hot-spot monitoring sites to measure either typical or high pollution concentrations.
 - Air quality models are often used in developing optimum and cost-effective source-specific programmes to reduce emissions with the aim of protecting human health.

Introduction

Much of the population of the WHO European Region lives in areas where the concentration of air pollution reaches levels that affect health (1). Reducing the risk to health is the main objective of the pollution abatement programmes undertaken by authorities at various levels and promoted by most of society. This is also part of the regional strategy for health for all (2). Designing an effective pollution abatement strategy requires identifying the basic characteristics of the pollution causing the most serious or the most prevalent health problems in the exposed populations. The location of the pollution hot spots, the changes in pollution over time as well as the chemical composition of the pollution are of concern. The existing networks for air quality monitoring describe these characteristics and provide a general picture of the state of and trends in air quality in Europe.

Designing a strategy for reducing pollution aimed at effectively protecting the public health requires more information than knowledge of the locations where the adverse health effects may occur. Information on the severity and magnitude of the effects, in terms of the type and expected number of cases attributable to the pollution, may be necessary to justify and support decisions that may be costly and require various efforts from society. The health benefits expected from pollution abatement may outweigh these costs, and the programmes may more easily gain public support. The quantitative estimates of the outcomes provide methods for health impact assessment. These methods require information on population exposure to the pollutants that affect health as well as knowledge on associations between exposure and health established based on epidemiological and toxicological studies.

Systems to monitor air quality do not always adequately assess the population exposure to air pollution. The spatial variation in the pollution concentration and the differences between the areas covered by the monitoring and the areas where the population is located create problems in using the air quality data generated by routine monitoring net-

works. In addition, the method used to routinely report air quality monitoring data restricts the availability of the collected information for assessing exposure to air pollution and, subsequently, its health effects.

Objectives and scope

Given the importance of the availability of valid information on population exposure to air pollutants, the WHO European Centre for Environment and Health organized a working group with the objective of defining the features of monitoring networks that allow their use in assessing the potential exposure of the population to air pollution from ambient air. The air quality assessment must include links with population exposure and with the pollution sources.

The principles outlined in this report are intended to promote progressive modification of the networks monitoring air quality to improve their usefulness for health impact assessment.

The scope of this report was determined by a meeting of the Preparatory Group in June 1997, which also designated the participants in the Working Group, outlined this publication and designated the authors of the first draft. The draft was revised at a meeting of the Working Group in Vienna in November 1997. Following the discussion at the meeting, the authors of the original drafts or additional writers have modified the text, which was edited at a meeting of the editorial group in Bilthoven in January 1998.

The methods discussed in this report are relevant for the components considered by the revised and updated WHO air quality guidelines for Europe (Annex 1.1). Since the set of pollutants monitored depends on the specific situation in the investigated city, this report specifies the methods for selecting components in a given situation.

This report focuses on pollution at the local level, with due consideration for the contribution of air pollutants transported over long distances. Harmonization between the local networks to achieve a national picture is considered as well. Although it is well recognized that indoor sources of air pollution contribute to the total exposure of individuals and populations to air pollution, this report concentrates on monitoring the pollution in ambient air. This is justified by the intrinsic

differences in monitoring exposure to indoor air pollution (3) as compared with assessment of ambient air quality and by the specific need to provide tools for managing the health risks related to ambient air pollution. This is also the reason why the links between ambient air quality and the pollution sources are discussed here.

The target groups of this publication are:

- network managers who undertake design of new networks or modify existing ones;
- policy-makers at various administrative levels;
- the people who influence policy.

The emphasis is on strategic issues and general approaches. The reader is therefore referred to other publications for more technical details of the methods discussed here.

Structure of the publication

The report consists of four general parts after the introduction. The first (Chapter 2) specifies the location and role of air quality assessment in the causal chain between the emission of pollution and health effects. The second part (Chapter 3) formulates a general concept of air quality assessment, including principles of designing a monitoring network, interpreting and reporting data and solving problems with quality assurance. The third part (Chapter 4) provides specific approaches to monitoring selected air pollutants to illustrate the principles listed in previous parts. It also provides some examples for the format and contents of reports disseminating the results of air quality assessment (Chapter 5). The fourth part, consisting of five annexes, technically describes the issues addressed briefly in the previous sections. The conclusions and recommendations emerging from the working group discussion are presented as Chapter 6.

To emphasize the long tradition of WHO programmes related to air quality assessment, this introduction briefly summarizes the previous activities that are complemented by the present report.

Previous related activities of WHO

GEMS/AIR and the Air Management Information System

A number of past and ongoing WHO activities have addressed air quality assessment and the health effects of air pollution. The Global Environment Monitoring System (GEMS)/AIR programme of WHO and the United Nations Environment Programme, which operated from 1973 until 1996, focused on upgrading air quality assessment throughout the world and providing important, consistent information on global trends in pollution concentrations. The tasks of the GEMS/AIR programme included:

- improving the quality of data;
- helping cities to develop strategies for abating air pollution; and
- using quality control and quality assurance procedures to ensure that data are comparable and compatible.

National or city authorities operated GEMS/AIR monitoring stations and voluntarily contributed their data to the GEMS/AIR database. The participating cities represented a wide range of climatic, topographic and socioeconomic conditions and capability in air quality management. GEMS/AIR was the only global programme that provided long-term monitoring data on air pollution for cities in developing countries. Thus, the programme enabled the levels of and trends in urban air pollution and air pollution management capability to be assessed globally.

The preparation of methodological documents and publications was an important part of the GEMS/AIR activities (4–14). The five volumes of the GEMS/AIR Methodology Review Handbook Series (4–8) are the most relevant documents related to the present one.

In 1996, the GEMS/AIR programme ended. A WHO project developed under the umbrella of the Healthy Cities Programme is the Air Management Information System (AMIS), which can be considered as a successor to the GEMS/AIR programme. Part of this system is an urban air quality database disseminated by WHO on CD-ROM. The information collected by the AMIS has already been used to analyse the status of and trends in urban air quality in developing countries (15).

WHO air quality guidelines

The WHO air quality guidelines for Europe have provided a basis for characterizing the health effects of major air pollutants (16). The update and revision of the guidelines has expanded this evaluation by estimating the exposure–response relationships for a number of pollutants. The revised guidelines are briefly summarized in Annex 1.1 of this report.

One of the important contributions of the updated guidelines is recognition of the continuity of the effects of some pollutants and of their effects on part of the population even at relatively low concentrations that are common in the environment. This emphasizes the need for a more complex approach to air quality management than avoiding occasionally exceeding a certain air quality guideline, which might be an appropriate approach for pollutants with a threshold effect level. Understanding the extent to which air quality influences population health helps in selecting the most effective strategy for preventing risk and contributes to improving population health. Methods of health impact assessment combining information on population exposure with data from toxicology and epidemiology have been developed to support this task.

The guidelines are meant to provide a scientific background to tools for air quality management such as national standards. The European Commission has used the revised air quality guidelines to prepare its directives on air pollution control (17). The directives specify, among other issues, the methods to be used in assessing air quality and in verifying whether air quality complies with the defined objectives. These methods are summarized in Annex 1.2, to provide an example for the comprehensive approach to be taken in a coordinated way by the 15 European Union countries, which are part of the European Region of WHO.

Assessing the health impact of air pollution

The methods used in assessing the health impact of air pollution have been discussed at workshops and conferences organized by or with participation of the WHO European Centre for Environment and Health (18,19). Estimating the impact of air pollution by solely monitoring population health status is not practical because the expected health outcomes are not specific. The preferred approach is therefore based on well designed epidemiological studies that can distinguish the contribution of air pollution to the range of other health determinants and can

provide data for deriving the relationships between exposure and health. Since epidemiological studies cannot be widely applied because they are complex and require extensive time and resources, the risk assessment approach is applied instead (20). This method combines information on exposure-response relationships with data on population exposure to estimate the extent of health effects expected to result from the exposure in the population. This report presents an example of this on pages 23–27.

Valid exposure data are crucial in applying this method. A consultation on health-related air quality indicators in 1995 (21) recommended a format for collecting data from air quality monitoring networks aimed at assessing population exposure to selected and most commonly monitored air pollutants. The consultation also recognized that the design of the network for air quality monitoring in a specific country or region may determine how useful the data generated by the network are for assessing population exposure and, consequently, health impact. Extracts from the consultation report are included as Annex 5.

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Information on air quality required for health impact assessment

Otto Hänninen, Alexander Economopoulos
& Halûk Özkaynak

Introduction

Air quality evaluation is important for assessing the nature of population exposure to air pollution. Assessment of population exposure is necessary for health impact assessment, which in turn is crucial for developing plans for air quality management and protecting the public health.

Human exposure to air pollutants may result in a variety of health effects depending on the type of pollutant; the magnitude, duration and frequency of exposure; and the associated toxicity of the specific pollutant. People come in contact with pollutants in the air both indoors and outdoors during their daily activities. Consequently, the differences in the sources and composition of indoor and outdoor pollutants and their relative contribution to total personal exposure should be recognized. This chapter describes the various factors that affect personal exposure and the role of ambient pollution in determining population exposure and health impact.

Since numerous epidemiological studies have already shown that exposure to elevated levels of various ambient pollutants is associated with either acute or chronic health effects, methods for quantifying population exposure to ambient pollution are emphasized here. In particular, monitoring methods and modelling techniques useful for estimating population exposure to pollutants of outdoor origin are discussed. Air quality monitoring, along with other methods, is crucial for assessing population exposure to air pollution and predicting the magnitude of the health risks to the population.

Air quality monitoring is often used to determine the air pollution levels in urban or rural environments. A monitoring network produces concentration measurements that can then be compared with the national and international guideline values. The prevalence of risk factors can be used to estimate population health risk from exposure data. One of the most important functions of the monitoring is to provide basic information for estimating:

- the level and distribution of exposure in the population;
- the population groups with high exposure; and
- the risks of potential health effects.

On-line air quality monitoring can be used in warning and alert systems during episodic pollution events. These systems use mass media to inform people of the current air quality and, if necessary, to give instructions aimed at reducing excess pollution and minimizing exposure. Monitoring ambient air quality also provides important input data for epidemiological studies, which are crucial in establishing associations between health outcomes and concentrations of ambient air pollution.

In general, exposure assessment requires both monitoring and modelling to identify target sources for reducing emissions and to implement an effective programme of air quality management for protecting human health.

Air quality and health impact assessment

Purpose of air quality monitoring and management

The activities of a human society – the economy, production of goods, transport and consumption – all affect the environment. All stages of these activities contribute directly or indirectly to creating air pollution.

Air quality management includes all activities aimed at managing air quality in the environment. The aim of air quality management is to keep the ambient air clean enough so that it is safe for the public health and the environment.

This process leading from the functions of the economy and society in general to the health effects of air pollution is described in the driv-

ing force–pressure–state–exposure–effect–action chain (Fig. 2.1) (1). The chain emphasizes the action society can take at each link of the chain to minimize the adverse health effects. The role of air quality monitoring is to provide information on the concentrations of pollution in the environment. These are then used to assess the population exposure and adverse health effects caused by the pollution. If the health risks are considered to be too high, action is needed to control the emissions and to improve the environment. Box 2.1 depicts the rationale for air quality monitoring.

Two approaches can be applied to prevent the adverse effects of pollution on health. The established method is to work to improve the environment, to control the emissions and to monitor and control the

Box 2.1. Rationale for air quality monitoring

Air quality assessment: what is the air quality in the city?

- Air quality monitoring
- Application of modelling tools
- Assessing and monitoring emissions



Exposure assessment: what is the level of population exposure?

- Where are different population groups located?
- What are the concentrations at these locations?
- What are the target populations, including sensitive groups?



Health impact assessment: what health effects are caused?

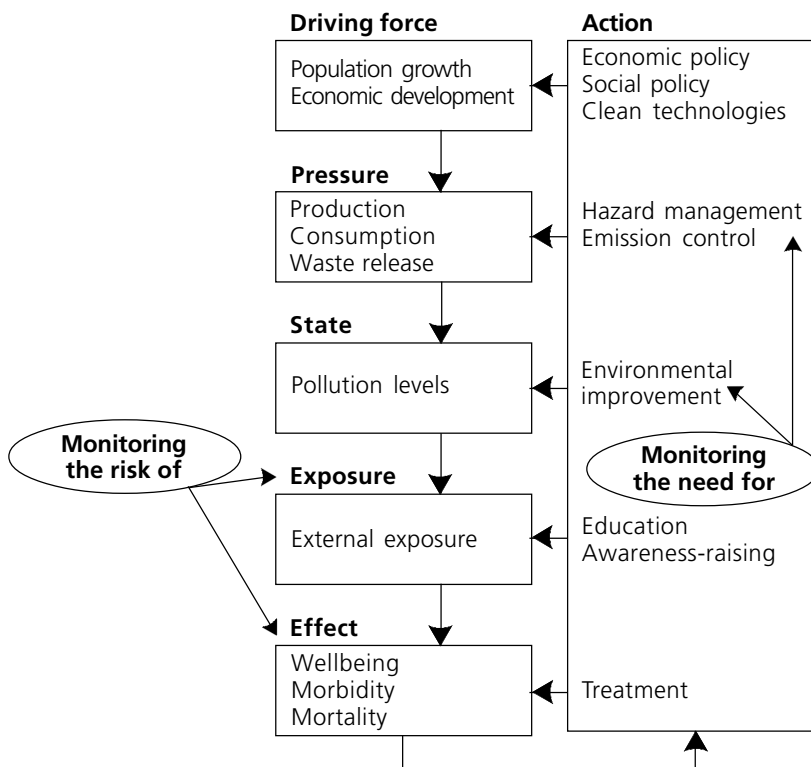
- How many cases are associated with the exposure?
- How severe are the effects?



Air quality management: what action is needed?

- Which sources, pollutants and areas should be targeted for reducing emissions?
- What strategies will be most cost-effective?

Fig. 2.1. Role of monitoring in the driving force–pressure–state–exposure–effect–action chain



pollution levels in the environment. This will lead to a cleaner environment in general. A more cost-effective approach in protecting human health is to start from the adverse effects of the pollution and plan the environmental protection activities aiming at reducing the exposure that leads to these effects as much as possible.

Relationship between air pollution and human health effects

The health effects caused by air pollution are realized through a lengthy chain that includes physical, chemical, behavioural and physiological processes. The chain starts with emissions into the atmosphere, where the pollutants are dispersed and diluted, forming a variable spatial and

temporal distribution of concentrations. Photochemical and other reactions transform the pollutants as they are transported in air.

At the same time, people conduct their daily activities. Most of people's time (about 80–90% in Europe) is spent inside buildings. These buildings are mostly homes, workplace and industrial environments, which may contain air pollution sources of their own. Even homes have internal sources of air pollutants, including the building materials and human activities in the home, such as tobacco smoking, cooking, heaters and personal hobbies.

The monitoring of ambient air quality focuses on the concentrations of outdoor pollutants. Different time–activity patterns and microenvironments determine personal exposure. The exposure to outdoor pollutants determines the dose to human lungs and, subsequently, the dose delivered to the different target body organs. The pollutant dose received by the different biological systems along with the toxicity of the pollutant or its metabolites, as well as individual susceptibility, then determine the individual health effects. Thus, the monitoring results at best only indirectly indicate the risk of potential health effects. Even though exposure is a reasonable measure of risks to health, different people exposed similarly may receive different doses of the same pollutant and may experience dissimilar health effects. To account for the exacerbated reaction of the sensitive people, the exposure levels of various population groups should be assessed, especially vulnerable ones such as children, elderly people and handicapped people. This involves evaluating human time–activity patterns and microenvironmental concentrations for different population groups.

Community studies of air pollution have shown a number of different types of adverse health effects from exposure to ambient pollution. The WHO air quality guidelines for Europe (Annex 1.1) comprehensively review health effects. The expected health effects depend on the type of pollution, the level of exposure and the personal susceptibility of an individual. Typical health effects observed by studies include:

- reduced lung functioning
- asthma attacks
- respiratory symptoms
- restricted activity

- increased medication use
- increased hospital admissions
- increased emergency room visits
- development of respiratory diseases
- premature death.

People living in polluted areas tend to get sick more often or for longer periods of time than do people in areas with less pollution. Exposure to elevated levels of pollution has also been linked to premature mortality. The higher mortality or morbidity rates require public and private expenditure and create human suffering that can be avoided in the future by directing societal development towards cleaner air and especially reducing exposure levels.

Definition of exposure

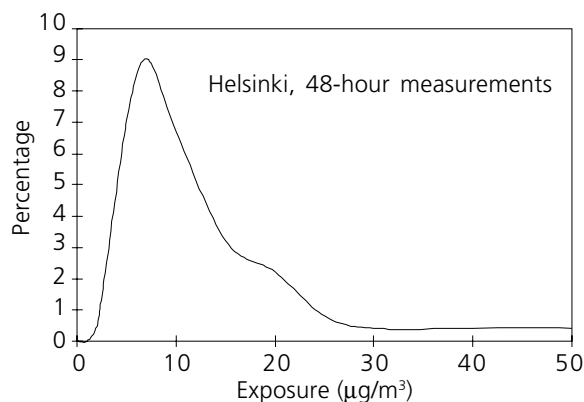
Air quality monitoring is the main source of information in assessing the exposure of the population to ambient air pollution. Exposure is determined by the concentrations people experience in their living environments. Thus, the monitoring should measure the concentrations in the places the population is, taking into account both the areas with maximum concentrations and the areas with high population density.

The word exposure has different meanings in different contexts. Personal exposure relates to true integrated concentrations experienced by individuals. When a person stays in one place, the air at that place determines his or her exposure. When a person moves from one place to another, his or her total personal exposure is determined by the time-weighted average air quality of all the places the person visits.

Population exposure summarizes the exposure of everyone in the population. Population exposure can be presented as the distribution of estimated personal exposure (Fig. 2.2). The true population exposure distribution cannot easily be measured because individual behaviour is complex.

The ambient air quality can be considered to be an indicator of population exposure. If the measured air quality described the spatial and temporal variation in concentration, the population were outdoors all day (or the pollution fully penetrated indoors without filtering) and indoor sources of pollution were not taken into account, the monitoring

Fig. 2.2. The distribution of personal 48-hour $\text{PM}_{2.5}$ exposure among the working-age population of Helsinki in 1996–1997 measured by the EXPOLIS study



Source: Jantunen et al. (2).

results would describe the true population exposure. The word indicator is used to reflect these complexities between the outdoor air quality and the true population exposure. The relationship between outdoor concentrations and personal exposure is discussed more extensively in Annex 4.

Activities to manage air quality

If ambient concentrations are considered to be too high, the state of the environment needs to be improved. Preventing pollution requires reducing emissions. Emission reduction should be targeted so that the population exposure is decreased effectively. It is usual to control the major sources of emission, such as implementing changes in industrial processes or exhaust filtering. More comprehensive strategies may include changing the demand for certain highly polluting activities. Increasing energy efficiency is an example of reducing emissions by controlling the demand for the product.

Besides controlling emissions, another measure to reduce exposure levels is urban development planning: how the emissions should be placed in the community in relation to the areas where people live and work. Transport systems are critical here, as the population density and the density of road traffic are closely connected. Moreover, motor vehicle exhaust is emitted close to the ground level where people are located.

The distribution of the emissions in the community, the height at which the emissions are released and the location of the population can often be altered during community planning processes, which reduces the potential health effects of pollution.

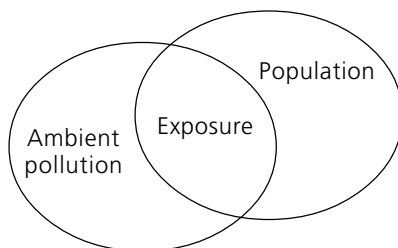
Certain techniques other than preventing pollution can also be undertaken to reduce exposure, such as releasing the emissions of diesel trucks at the top of the truck instead of at the engine level. This reduces the exposure of the occupants of the vehicle immediately behind the truck but not necessarily of others driving on the highway. Care must be taken in such procedures to ensure that such measures to reduce the peak personal exposure of a few people do not increase the total population exposure or result in unexpected exposure that harms people or the environment. Thus, the use of tall stacks historically reduced the peak exposure of the nearby people and environment to SO₂ but resulted in the exposure of much larger number of people to acid sulfate particles and a much larger area of the environment to acid rain.

Public awareness of air quality can contribute to both reducing emission levels and decreasing exposure. This is especially true in transport behaviour. The most polluted cities in the world restrict the use of private cars. Adverse health effects may be further reduced by avoiding unnecessary exposure of susceptible individuals. In extreme conditions the most vulnerable people are instructed to stay inside during days with severe pollution. These means are not a desired goal but are necessitated by the environmental conditions, often caused by a combination of human activities and natural phenomena, such as weather or natural disaster.

Relationship between information on air quality and exposure

Air pollution causes health effects when the population is exposed. Conceptually, exposure is the intersection between the ambient pollution concentrations and the location of the population (Fig. 2.3). Air quality assessment in general and specifically air quality monitoring should produce information that can be interpreted to indicate population exposure. This requires taking into account the location of different population subgroups and assessing the quality of the air prevailing at those locations.

Fig. 2.3. Exposure is the intersection between the variable concentration field and the dynamic population density



Variability of ambient pollution concentrations in time and space

Air pollution is often unevenly dispersed in the environment. In many cases the areas with higher concentrations are near the sources of pollution. As the volume of road traffic has increased in urban areas, the highly polluted areas have moved from industrial areas towards the areas where the population is. This trend is tending to increase exposure by bringing the pollution and people together.

The time of year affects the emission levels of many sources. Besides the emission levels, the dispersion properties of the atmosphere vary between seasons and especially between normal and extreme conditions. Estimating the range of possible concentrations at a site normally requires 6–12 months.

This variability of the emissions in both time and space as well as variable dispersion conditions in the atmosphere produce a complex varying pollution concentration field in the environment. Monitoring results represent only the point and time where and when the sample was taken or the measurement was made. The sampling strategy must be carefully planned to make the best use of the available time and the number and location of monitoring sites.

The concentrations of pollutants in indoor locations are not the same as those outside these same places. Ventilation systems filter away a portion of some pollutants, but certain indoor sources might also increase the indoor concentrations of these pollutants.

In the simplest case, one ambient measurement site is used to indicate the concentrations in the whole city and the exposure of the whole population. For pollutants with high spatial variability, this is not very accurate. When the monitoring network is refined, more measurement stations can be added and the exposed population can be assigned to the concentrations obtained at different monitoring stations. Also, the use of other methods such as dispersion models can be valuable in assessing the variation of air quality in both time and space. Further ideas related to processing and presenting spatial variation in air quality as well as pollutant-specific concentration differences between indoors and outdoors are discussed in Annex 4.

Population density and time–activity profiles

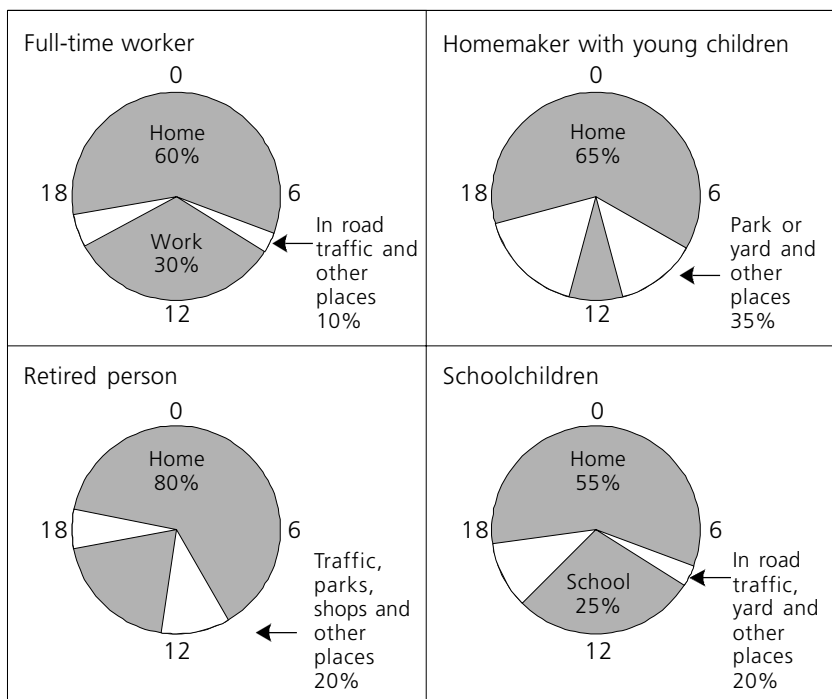
Population density is determined by where the population lives. The location of the population also varies according to the time of day, day of the week and the season. Most of the population is located in homes at night. During the day, many people go to work, school, a day care centre or other places. The centres of cities might be thinly populated at night compared with the number of people working there or visiting them during the day. People move through fields with varying concentrations, thus being exposed to varying levels of air pollution during their normal daily lives. During the day, people are exposed to both higher and lower levels of air pollution. The actual level of average exposure depends on the individual behaviour of the person.

The behaviour and time usage of the population can be measured using time–activity diaries (2) or direct observation. Time–activity pie charts for some typical days are sketched in Fig. 2.4. In addition to the dynamic nature of the true population density, most people spend much of their time in indoor environments, whereas some population groups spend more time outdoors.

Ambient concentrations and population exposure

Monitoring ambient air quality means outdoor air, and the monitoring sites are more or less fixed at selected locations. The population moves into, out of and across a community every day. The exposure estimated by using the outdoor concentration levels is the potential exposure of the population. If the person spent all his or her time outdoors at the monitoring site or at a similar concentration, the monitoring result would be his or her true exposure.

Fig. 2.4. Examples of time–activity profiles for typical 24-hour days



Most of the difference between potential and true exposure arises because a large portion of people's time is spent indoors. For some pollutants (such as ozone), buildings act as filters, removing some pollution from the air taken in. The ventilation system also smooths out the variation in outdoor pollution by reducing the peak concentrations indoors. These processes mean that the potential exposure overestimates the actual personal exposure. Differences in building ventilation systems produce great variation in the removal rate of pollutants. When windows are open and the wind speed is moderate or high, the indoor concentrations of pollutants may reach the same level as outdoors.

In other cases, there may be significant indoor pollution sources, such as volatile organic compounds (VOC) or particulate matter. Even relatively modest indoor emissions of air pollution can produce high concentration levels inside a closed indoor environment. The relation-

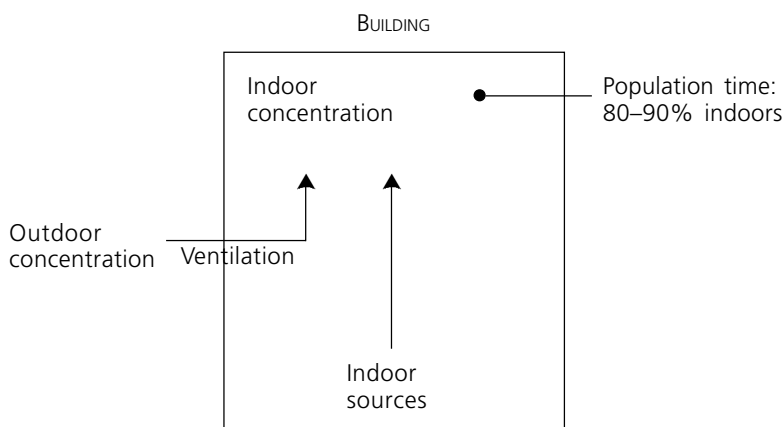
ship between outdoor concentration (the potential exposure) and the actual exposure indoors is shown in Fig. 2.5 and discussed further in Annex 4. When the indoor sources of air pollution are substantial, using outdoor concentrations alone would underestimate actual personal exposure.

The most important indoor sources of air pollution include tobacco smoking, fuel-burning heaters and dryers, cooking (even with electric stoves) and chemicals used indoors or emitted from the household equipment or building materials. These sources can affect the indoor concentrations of carbon monoxide, nitrogen dioxide, fine particles, many VOC and other compounds.

Spatial variation in concentration

Urban areas with no local pollution sources represent the baseline pollution concentration in a city. For many pollutants, the lowest concentrations are observed in these types of areas. Local sources in other parts of the city will add to the baseline and produce higher concentrations. Higher concentrations exist in road traffic, in the centres of large cities or in industrialized areas. Information on pollution is most urgently needed in the areas with maximum concentrations. The first measure-

Fig. 2.5. The relationship between the ambient concentration and the actual exposure given that most people spent most of their time indoors



ment sites of a monitoring network should probably be selected from the areas in which the concentrations are predicted to be at a maximum (hot spots). When the monitoring network is expanded later, the suburban areas with substantial housing and inhabitants should also be included as well as the areas with very low local emissions.

Ozone and some other chemically reactive pollutants are an exception to this scheme. Ozone is depleted in chemical reactions if other pollutants are present, and the presence of local pollution sources therefore tends to adjust the baseline concentration downwards.

Methods of assessing population exposure using monitoring data

Health impact assessment uses relative risk estimates from epidemiological studies and requires the population exposure as input. A simple way to assess population exposure from monitoring data is to use one monitoring station only or to use some selected or all monitoring stations in a city and to take an arithmetic mean of selected concentrations. This average value, such as daily or yearly, is then used to indicate the exposure of the whole population.

A more complex method is to divide the population into groups and to assess the exposure of the groups separately. Then the different amounts of time spent on various kinds of activities can be assessed as well as different areas of working and residence. This kind of procedure is sketched as an example in Table 2.1. The population of the target city is divided into groups according to factors that differentiate the exposure of these groups. The factors used in this example are time–activity data and the areas of residence. Three groups have been created: 1) the suburban population not working outside the area of residence, 2) the suburban population commuting to work in the centre of the city and 3) the population living, and possibly working, in the centre of the city.

In this example, four monitoring stations are available: one in the centre of the city, another one along a main road oriented towards the road traffic and two others in different types of suburbs. Table 2.1 shows examples of time proportions that the selected three population groups spend in environments where the pollution concentration for a given time unit, such as a given day, can be approximated with the measurements collected by each monitoring station.

Table 2.1. Assigning population exposure to PM₁₀ using the results of air quality monitoring and approximate residence location and time–activity data

Population group (size)	Fraction of time assigned to each station (average PM ₁₀ concentration in (µg/m ³))				Calculated exposure (µg/m ³)
	Centre (50)	Road traffic (70)	Suburban (30)	Suburban (20)	
Not employed, including children (100 000)	0	0	0.5	0.5	25
Employed and commuting (500 000)	0.3	0.1	0.3	0.3	37
Centre residents (400 000)	0.9	0.1	0	0	52
Total population (1 000 000)	0.51	0.09	0.20	0.20	42

In this simple example (Table 2.1), the suburban population that commutes to work in the centre of the city lives in different types of suburban areas around the city. The suburban exposure is calculated as the average of the two available suburban monitoring stations. The same procedure is used for people not employed, including children. The average exposure of the groups differs substantially. The average PM₁₀ (see section on particulate matter for definition) exposure for the non-working population group is predicted to be 25 µg/m³ versus 52 µg/m³ for the city centre residents and 37 µg/m³ for the commuting population. The entire city-wide time-weighted population average PM₁₀ exposure is predicted to be 42 µg/m³. This example shows that PM₁₀ concentrations can vary within a large urban area and that exposure to different population groups can also vary depending on their locations and their patterns of activity.

This approach based on population groups enables the exposure of and health effects on the whole population of the city to be assessed more accurately. The health impact in different population groups can also be assessed separately. This improves the understanding of the dis-

tribution of adverse health effects in the population and allows for more efficient action in air quality management.

Relevance of exposure to health effects

Exposure is the key factor in potential health impact. High concentrations do not harm people if they are never present, and even low levels become relevant when many people are present all the time. Thus, the most important factor in assessing the risk of adverse health effects is the population exposure.

Time response of health effects

The health effects caused by some air pollutants are related to the total cumulative exposure during people's lifetime or over a very long period. An example of this kind of pollutant is radon, which produces high-energy radiation and causes lung cancer. The health outcome does not depend on whether the cumulative exposure (radiation units times hours) come from a brief high-level peak or from low-level continuous lifetime exposure.

Other pollutants cause mainly acute health effects. In these cases the peak exposure is more relevant than the cumulative lifetime exposure. An example of this kind of pollutant is carbon monoxide (CO). Short-term exposure to high levels of CO causes clear health effects, even death, but the same cumulative exposure spread over a much longer period of time has no visible effect.

The time response of the health effects of pollutants has been taken into account in defining the WHO air quality guidelines for Europe (see Annex 1.1). A separate guideline level is given for different exposure times. An air quality monitoring network should assess the population exposure for all the time spans the pollutants have guideline values. Shorter measurement interval results can be combined into longer averages, but caution should be taken when assessing peak exposure from long-term integrated measurements.

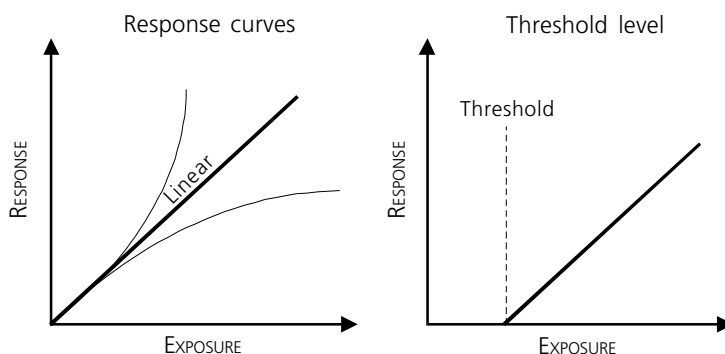
Relationships between exposure and response

Some health effects are chronic, such as lung cancer from high radon exposure. Chronic health effects take a long time to develop and may occur years after the exposure has been discontinued. Other effects are

acute, such as carbon monoxide poisoning. Short-term peak exposure is normally responsible for the acute effects.

For some pollutants a threshold level is assumed. If the exposure is below the threshold level, no health effects occur (Fig. 2.6). In other cases, such as particulate matter, evidence shows that, even at very low concentrations, health effects can be observed at the population level. Thus, these pollutants have no safe level at which no risk to health can be assumed.

Fig. 2.6. The health response to exposure to an air pollutant is often assumed to follow the exposure level linearly. For some pollutants a threshold is assumed: if the exposure is below the threshold level, no response is expected.



Estimating health impact – an example

The impact can be estimated by calculating the attributable proportion (AP), which indicates the fraction of the health effects that can be attributed to the exposure in a given population (provided that the exposure and the health effects are causally associated). If the population distribution of exposure is determined in the exposure assessment stage and the function between exposure and effect has been identified, the attributable proportion can be calculated using the formula:

$$AP = \frac{\sum \{ [RR(c) - 1] \times p(c) \}}{\sum [RR(c) \times p(c)]} \quad [1]$$

where: $RR(c)$ is the relative risk for the health effect in category c of exposure and $p(c)$ is the proportion of the target population in category c of exposure.

If a certain underlying frequency of the effect in the population, I , is known (or, often, assumed), the rate (or number of cases per unit of population) attributed to the exposure in the population can be calculated as:

$$I_E = I \times AP$$

Consequently, the frequency of the outcome in the population free from the exposure can be estimated as:

$$I_{NE} = I - I_E = I \times (1 - AP) \quad [2]$$

For a population of a given size N , this can be converted to the estimated number of cases attributed to the exposure, $N_E = I_E \times N$.

If the (estimated) incidence in the unexposed population and the relative risk at a certain level of pollution are known, an excess incidence ($I_+(c)$) and excess number of cases ($N_+(c)$) can be estimated at a certain category of exposure:

$$I_+(c) = (RR(c) - 1) \times p(c) \times I_{NE} \quad [3]$$

$$N_+(c) = I_+(c) \times N \quad [4]$$

The example presented in Table 2.1 illustrates this method. The objective of the exercise is to estimate the number of hospital admissions for respiratory diseases of the 400 000 inhabitants of the city centre that can be attributed to ambient PM_{10} air pollution during 1 year of air quality monitoring. For each day of the year analysed, an exposure estimate is calculated according to the method illustrated in Table 2.1. The 365 calculated estimates form a frequency distribution that provides information on the frequency of days in a given category of pollution level ($p(c)$) shown in Table 2.2. The mean annual PM_{10} concentration that can be calculated from the data presented is $50.3 \mu\text{g}/\text{m}^3$.

The increase in the relative risk of hospital admission caused by the pollution is taken from the updated WHO air quality guidelines for Europe (Annex 1.1, Table A1.2). The risk changes 1.008 times for each $10 \mu\text{g}/\text{m}^3$ increase in PM_{10} concentration; the 95% confidence limits of the estimate are 1.0048 and 1.0112. It is assumed that a concentration

Table 2.2. Calculation of the number of hospital admissions in a hypothetical population of 400 000 people caused by respiratory diseases attributable to pollution of ambient air with PM₁₀ during 1 year

PM ₁₀ concentration (µg/m ³)	Number of days with pollution <i>c</i>	Proportion of days (1.00 = 365 days)	Relative risk in category <i>c</i>	Estimated excess number of cases in category <i>c</i> ^a
<i>c</i>		<i>p(c)</i>	RR(<i>c</i>)	<i>N₊(c)</i>
< 20	10	0.027	1.000	–
20–29	30	0.082	1.008	3
30–39	71	0.195	1.016	15
40–49	83	0.227	1.024	27
50–59	76	0.208	1.032	33
60–69	50	0.137	1.040	27
70–79	20	0.055	1.048	13
80–89	10	0.027	1.056	8
90–99	8	0.022	1.064	7
100–109	5	0.014	1.072	5
110–119	2	0.005	1.080	2
120–129	0	0.000	1.088	0
Total	365	1.000		139

^aThe numbers do not add to the total because of rounding.

of PM₁₀ below 20 µg/m³ is the background level and that the risk in this category is 1.00. It is also assumed that the increase in relative risk is linear.

The attributable proportion AP(*c*) can then be calculated according to equation 1. In this example AP = 0.0276: 2.76% of all hospital admissions for respiratory diseases of the residents of the city centre can be attributed to the pollution of ambient air with PM₁₀ experienced by these people.

An estimate for the average frequency of hospital admissions for respiratory diseases is based on national statistics. With *I* = 126 cases per 10 000 people per year, about 5040 hospital admissions can be expected among 400 000 people annually. Of this number, 139 (2.76% of

5040) can be attributed to the PM_{10} pollution. When the limits of the confidence interval for the relative risk estimate are used in equation 1, the corresponding limits of the attributable proportion estimate are 1.67% and 3.82%, and the range for the attributable number of cases in the 400 000 population is 84 to 193.

Besides the total number of attributable cases, the analysis provides the estimates of the distribution of the cases by the concentration (exposure) category, calculated according to equation 4. In the example above (the last column of Table 2.2), less than half (62) of the 139 expected extra cases could be attributed to relatively high exposure, exceeding $60 \mu\text{g}/\text{m}^3$. This results from the shape of the association between exposure and response and the increased risk in the low, but the most frequent, exposure categories. The information for individual exposure categories indicates that a strategy for pollution control focused on eliminating days with a high pollution level (such as over $80 \mu\text{g}/\text{m}^3$) will have limited effect. Instead, a comprehensive policy aiming at shifting the entire pollution distribution to lower levels is advisable.

Calculations similar to those presented in Table 2.2 can be repeated for each population for which a reasonable estimate of exposure can be derived. However, the adequacy of the assumptions used in the calculations must be always considered separately. For instance, the national estimate of hospitalization frequency may be not correct for the commuting population, which is likely to have better average health than the (older) people staying at home, who are more likely to be hospitalized for respiratory diseases. The possible limitations of the relative risk estimate (such as specifically addressing children or elderly people) should be also scrutinized before it is applied in estimating the health effects.

The impact assessment methods are more broadly discussed elsewhere (3). The calculations presented in the example can be performed with a software tool AIRQ developed and distributed free on request by the WHO European Centre for Environment and Health (ecch@who.nl).

Air quality and sources: from assessment to management

Introduction

The sources of ambient air pollution, especially in urban environments, are often quite diverse. These sources can be broadly classified as stationary, mobile and area emission sources. Emissions of pollution from these three classes of sources typically result in complex spatial and temporal distributions of ambient air pollution concentrations. Emission inventories form a cornerstone for planning a monitoring strategy. Many air pollutants have the highest ground-level concentrations near the local sources. Emission inventories indicate the areas where the emissions of air pollutants are highest, and thus, where the concentration hot spots might be expected. Emission inventories are described in more detail in Annex 2.

Estimating reliably the exposure profiles of a dynamic urban population requires complete information regarding the seasonal and diurnal variability of pollutant concentrations at representative locations at which the population is exposed. For pollutants that strongly depend on the specific location of dominant sources of emissions, such as sulfur dioxide (SO₂) from power plants or CO from motor vehicles, pollution must be measured at many different sites of high impact. Since achieving this goal through monitoring alone is difficult for both technical and practical reasons, modelling methods are necessary to indirectly generate a more comprehensive profile of concentrations for different cohorts of the population. The air quality models are designed to estimate both the short- and long-term average concentrations relevant to the pollutant-specific health impact assessment being conducted. These models are either deterministic or statistical in nature. Dispersion models (or air quality models) use the emission and meteorological data as input and use physical, statistical and empirical equations to calculate ground-level concentrations. These models are further discussed below and in Annex 3.1. The deterministic or physical models are also used in either designing or modifying the monitoring networks by identifying hot spots or areas of greatest potential population exposure.

The results of air quality monitoring are quite important for evaluating the dispersion model results against the observed ambient levels. The utility of dispersion models and air quality monitoring is compared in Table 2.3.

Table 2.3. Comparison of air quality monitoring and dispersion modelling as tools for assessing air quality

Task	Relevance to:	
	Monitoring	Modelling
Assessing true concentrations	High	Low ^a
Alert systems	High	Low
Assessing variability in time	High	High
Assessing variability in space	Low ^b	High
Assessing concentrations in future	Low	High
Source apportionment	Low	High

^a Modelled results should always be compared with some measured values to assure that the model is reliable and the input data correct.

^b Increasing the number of monitors or samples can improve the spatial resolution and coverage of the monitoring network.

Semi-empirical or statistical models, such as the receptor models, are commonly used to determine the relative contributions of different emission sources to measured pollutant concentrations at selected monitoring sites. These models allow the emission reduction activity to be directed towards the sources that contribute significantly to the total concentrations. Receptor models are described in more detail in Annex 3.2. Statistical models are used to build air quality forecasting systems, which can be used to increase public awareness and to help people to adapt their daily lives to the environmental conditions. Statistical models are described in Annex 3.3.

Both the physical air dispersion models and the statistical receptor models are essential components of a comprehensive programme for air quality management. In conjunction with air monitoring data, these models are important for developing an optimum and/or cost-effective source-specific plan for reducing emissions. Various models of each type mentioned above have been developed, and many models are periodically modified and improved. The European Environment Agency manages a Model Documentation System on atmospheric dispersion models. This contains information about more than 80 models, their area of application and their status with respect to evaluation and validation as submitted by the modellers (<http://aix.meng.auth.gr/database/index.html>, accessed 12

August 1999). The system allows users to search for specific models or for models appropriate for preselected applications.

The description and various uses of these air quality models are further presented here.

Application of dispersion models

The application of air quality models complements and supports ambient air monitoring. The monitoring is necessarily limited to few stations, not necessarily covering all critical locations, and thus provides a limited picture of the air pollution situation. The air quality models can provide a more complete picture of the pollutant concentration profiles in space and time and can be used for assessing the contribution of various sources and the impact of alternative control measures. They are thus valuable for management and for health impact assessment. The output of the models can be used to improve the siting of monitoring stations, and the monitoring results are necessary for validating and calibrating the predictions of the model.

Estimating the pollutant concentrations tends to be a demanding task, as it involves the use of appropriate models, along with terrain and meteorological data, and detailed data from an air emission inventory. These data comprise information not only about the load of released pollutants but also about the release conditions and their distribution profiles in space and time. It also involves data on ambient air quality for verifying and calibrating the model predictions.

The data required depend on the model to be used. In practice, however, the model is often selected based on the data available. This is because collecting the required input data normally constitutes the most painstaking part of modelling.

The use of air quality models is related to the tasks of air pollution management. As such, the models to be used should be able to generate concentration predictions that are compatible with the applicable air quality guidelines or standards. These can generally be placed in two categories. One category aims at protecting human health from the acute effects of air pollution, and the relevant guideline values or standards refer to short averaging periods, spanning from a few minutes up to 24 hours. The other category aims at protecting human health from

the long-term effects of air pollution, and the relevant guideline values or standards refer to annual and/or seasonal average concentrations.

Based on this, most air quality models can be classified into two broad categories: those capable of predicting short-term and long-term concentrations. For each such category, suitable dispersion models are available, some of which, but not all, can fulfil the requirements of both. Short-term concentrations can be also predicted from stochastic models, which are generally simple to use. Receptor models fall into a supplementary category, as they consider the observed pollutant concentrations in assessing the contribution from various sources (Annex 3.3).

These models are described here in relation to their management tasks and input data requirements. Annex 3 provides a more detailed description focusing on the model formulation and computational principles.

A key input in all models, except perhaps for the stochastic ones, is source emission inventory information. The relevant techniques are discussed in Annex 2, and actual models are presented elsewhere (4).

Short-term models

Short-term models for air pollution management are often applied for analysing specific episodic events or for performing critical impact analysis. The basic difference between these tasks is that the former type of analysis studies episodes that have taken place and the conditions leading to it are generally known. In critical impact analysis, the objective is to predict critical episodic events under credible conditions that have to be searched for or otherwise defined.

Depending on the objectives of the analysis (such as analysis of primary or photochemical pollutants or analysis of simple or complex terrain), appropriate models have to be selected and multiple models normally have to be combined.

Models for analysing episodic events

Models for analysing episodic events can again be distinguished into ones that perform rigorous analysis by using fairly sophisticated packages and simpler ones that are used routinely in environmental management studies.

The meteorological and terrain data required differ substantially from model to model. However, most models require similar source inventory data. For each point source, data are normally required about the stack location, physical height and internal diameter. In addition, hourly data over the simulated period are required about the pollutant emission rates, the exit gas volume or velocity and the exit gas temperature.

For each area source type, data are required about the average release height and about the hourly pollutant emissions within each square of a grid, typically sized from 500 m to 2 km on one side, that covers the entire study area.

For the rigorous analysis of episodic events, a meteorological model can be combined with a compatible dispersion model and/or a photochemical one. The input data for the above models, besides the inventory ones, are topographic and meteorological. The topographic data include ground elevation, which is widely available with a resolution of about 1 km, as well as vegetation and soil data, generally available with lower resolution. Vegetation and soil data are used to calculate evaporation and heat exchange. The meteorological data are needed to initialize the calculations. On a local basis, daily information on the wind speed, temperature and humidity profile on the surface and in the upper air may satisfy the needs of a region with a size of up to 50 by 50 km or even 100 by 100 km.

For typical analysis, meteorological pre-processor models are used first for converting the surface and upper air data into the meteorological input accepted by the dispersion models. The meteorological data in this case include surface measurements (wind direction, wind speed, dry bulb temperature, total cloud cover and opaque cloud cover) as well as mixing height data obtained twice daily. For complex terrain models, terrain data are also required.

Several dispersion models are available; selection depends on the type of sources and the terrain being considered. In addition, photochemical models are also available that can be used not only for simulating an episodic event but also for assessing the required reduction in VOC and nitrogen oxides (NO_x) emissions to attain the applicable standard for hourly ozone concentration.

Stochastic models

Stochastic models can be used for predicting episodic events based on anticipated weather conditions. They thus help planners in deciding which temporary restrictive measures have to be instituted. These models are semi-empirical and need to be calibrated in each study area by using the relevant air quality data gathered by the monitoring network. After calibration, such models are easy to use, requiring mostly meteorological information.

Models of critical impact analysis

Models of critical impact analysis can be distinguished into ones that perform typical analysis and cover point and area sources and ones that yield rapid assessment and cover only point sources. The latter type is important in formulating strategy, as it allows emission limits and/or stack height requirements to be defined directly.

Rapid assessment models for analysing critical point sources require fewer input data and provide rapid predictions (Annex 3.3) (4). Local meteorological data are not required, as the computed concentrations are maximized against all credible combinations of the meteorological parameters. These models, in addition to assessing the maximum ground concentrations under current conditions, allow direct computation of the emission limits, the minimum stack height required and the impact of flue gas compounding from multiple adjacent stacks.

Most models performing typical analysis of episodic events can be used for performing typical critical impact analysis. In this case the models are run over extended periods, such as 1 or more years, to compile the most adverse conditions and the corresponding air quality levels.

Long-term models

Long-term models predict the seasonal and annual average concentrations of pollutants and can be distinguished into various categories according to how sophisticated the analysis is. Several dispersion models that compute the hourly pollutant concentration in the specified recipients can also produce seasonal or annual averages.

The climatological models are somewhat simpler to use. Application requires a suitable meteorological pre-processor model that transforms the surface meteorological data into joint frequency functions.

A recent rapid assessment model for area sources requires reduced input data and is simple to use (4). This model, assuming round cities and area sources with normal profiles of emission density, yields the maximum concentration in the city centre and spatial-average concentrations over the urban area. These concentrations are related to the air quality standards and to typical population exposure to ambient pollution. Each area source is assessed separately, and the compounded impact from all sources can then be easily computed. This provides valuable information on the contribution of each source and on the impact of control measures.

Pollution emission characteristics and air quality

Practical experience shows that, if point sources are made to operate without violating any short-term air quality standards, their long-term impact on any recipient will normally be nearly negligible. This means that observing the short-term guidelines or standards largely dictates the allowable emission rates and release conditions from point sources.

In relation to primary pollutants, practical experience also shows that, if the area sources without excessive seasonal variation are made to operate without violating the long-term (seasonal or annual) guidelines or standards, their short-term impact tends to be well within the relevant guidelines or standards. This rule does not include highly variable sources, such as space heating boilers, which often cause excessive violations of the 24-hour standards, or secondary pollutants, such as ozone, for which the short-term standards (such as 1-hour or 8-hour) tend to be critical for human health. Based on this, observing the long-term guidelines or standards largely dictates the control measures required in relation to primary pollutants from area sources without excessive seasonal variation.

The practical ramifications of this are significant for management, allowing complex air quality problems to be effectively decoupled into a number of simpler ones, each of which can be tackled separately from the rest. In other words, separate analysis can be carried out for each point source or for each group of adjacent point sources, as well as for certain common types of area sources. For these tasks, the models for the rapid assessment of point sources described previously can be used. Besides providing strong support to pollution management, these models may contribute to the appropriate design of air quality monitoring and assessment of population exposure.

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Design, operation and quality assurance and control in a monitoring system

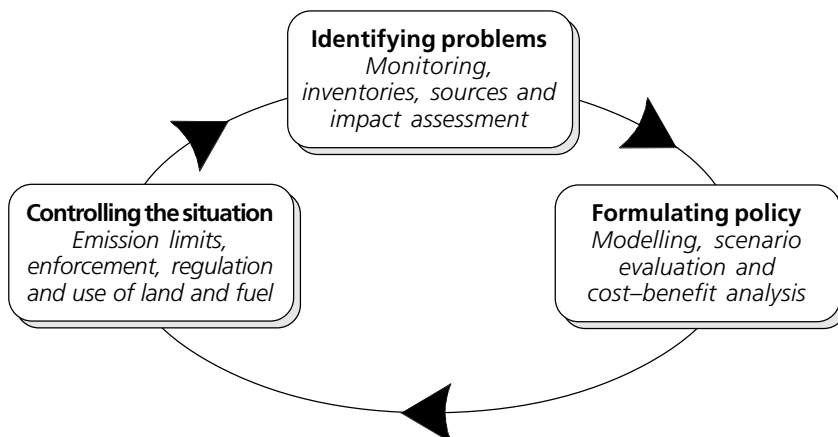
Jon Bower & Hans-Guido Mücke

The role of monitoring

This chapter reviews some of the requirements for the design and quality assurance and control of ambient air monitoring networks for assessing population exposure to ambient pollution. Issues related to data management and dissemination are also addressed. In accordance with the revised air quality guidelines for Europe, the pollutants of concern are CO, ozone (O₃), NO₂, SO₂, suspended particulate matter, lead (Pb) and cadmium (Cd). The relevant averaging periods range from 10 minutes (SO₂) to 1 year (NO₂, SO₂, Pb and Cd).

The ultimate purpose of monitoring is not merely to collect data but to provide the information required by scientists, policy-makers and planners to enable them to make informed decisions on managing and improving the environment. Monitoring fulfils a central role in this process, providing the necessary sound scientific basis for developing policies and strategies, setting objectives, assessing compliance with targets and planning enforcement action (Fig. 3.1).

Nevertheless, monitoring has limitations. In many circumstances, measurement alone may be insufficient or impractical for the purpose of fully defining population exposure in a city or country. Therefore, as emphasized previously, monitoring often needs to be combined with other objective assessment techniques, including modelling, measuring and inventorying emissions, interpolation and mapping. Measurement of ambient air quality and modelling-based assessment are useful complementary activities. No monitoring programme, however well funded and designed, can hope to comprehensively quantify patterns of air pol-

Fig. 3.1. The role of monitoring in air quality management

lution in both space and time. Conversely, reliance on modelling alone is equally unsound. Although models can provide a powerful tool for interpolating, predicting and optimizing control strategies, they are effectively useless unless properly validated by real-world monitoring data. Accordingly, ambient monitoring and modelling should be inter-related components in any integrated approach to exposure assessment.

The European Union directive on ambient air quality assessment and management (1) explicitly recognizes the utility of using a broad range of monitoring, modelling and objective estimation techniques for assessing air quality in Member States. The technique of choice depends on the air quality status of the area under study. Explicit guidance on detailed monitoring strategies and selection of sites and instruments for criterion pollutants will be included by working groups preparing various European Union daughter directives, in accordance with the brief description presented in Annex 1.2.

Objectives of monitoring and quality assurance and control

The first step in designing or implementing any monitoring system is to define its overall objectives. Setting diffuse, overly restrictive or ambitious monitoring objectives will result in cost-ineffective programmes

with poor data utility. Such circumstances will not allow optimal use to be made of the available personnel and resources.

The relationships between the data collected and the information to be derived from them must be taken into account when a monitoring programme is planned, executed and reported. This emphasizes the need for users and potential users of the data to be involved in planning surveys, not only to ensure that the surveys are appropriate to their needs but also to justify committing the resources.

Box 3.1. Typical monitoring objectives

- Population exposure and health impact assessment
- Identifying threats to natural ecosystems
- Determining compliance with national or international standards
- Informing the public about air quality and establishing alert systems
- Providing objective input to air quality management and to transport and land-use planning
- Identifying and apportioning sources
- Developing policies and setting priorities for management action
- Developing and validating management tools such as models and geographical information systems
- Quantifying trends to identify future problems or progress in achieving management or control targets

Assessment of exposure and health impact is only one of the possible range of monitoring objectives. In practice, networks are invariably designed for a variety of functions. National statutory requirements will always be paramount; others include developing policies and strategies assisting local or national planning, assessing progress against international standards, identifying and quantifying risk and promoting public awareness. Typical monitoring objectives are summarized in Box 3.1. Every monitoring survey or network is therefore different, being influenced by a unique mix of local and national issues and objectives.

Clear, realistic and achievable monitoring objectives must be set. This enables appropriate data quality objectives to be defined (Box 3.2). In turn, this enables a targeted and cost-effective quality assurance programme to be developed. The requirements for this are also addressed

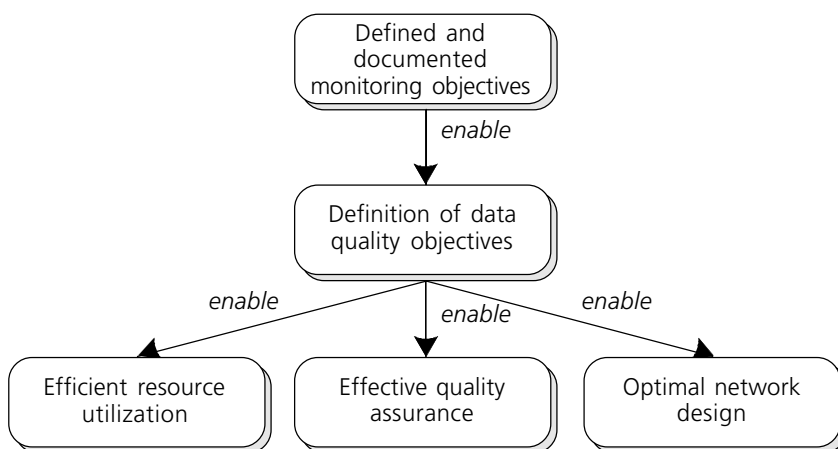
elsewhere (2,3). A clear definition of overall monitoring objectives and data quality objectives is therefore essential to enable networks to be optimally designed, priority pollutants and measurement methods to be selected and requirements for data management and reporting to be identified (Fig. 3.2).

Box 3.2. Data quality objectives

The essential requirements to be met by measurement if overall monitoring objectives are to be achieved:

- measurement accuracy and precision
- adaptable to metrology standards
- temporal completeness (data capture)
- spatial representativity and coverage
- consistency from site to site and over time
- international comparability and harmonization

Fig. 3.2. The importance of setting objectives



Role and functions of quality assurance and quality control

Quality assurance and quality control are an essential part of any air monitoring system. Quality assurance and quality control comprise a programme of activities ensuring that measurement meets defined and appropriate standards of and objectives for quality, with a stated level of confidence. The function of quality assurance and control is not to achieve the highest possible data quality. This is an unrealistic objective that cannot be achieved under practical resource constraints. Quality assurance and control ensure that data are fit for a purpose.

The major objectives of quality assurance and control are summarized in Box 3.3; the functional components of a quality assurance and control programme are identified in Box 3.4.

Quality assurance activities cover all pre-measurement phases of monitoring, including determining the objectives of monitoring and data quality, designing the system, selecting sites, evaluating equipment and training operators. Quality control functions directly affect measurement-related activities such as site operation, calibration, data management, field audits and training. In other words, quality assurance relates to the measurement process, whereas quality control is concerned primarily with output.

Each component of a quality assurance and control scheme needs to be implemented successfully to ensure the success of the complete programme.

The main principles of designing a quality assurance and control system apply to most types of network or instrument. However, there are

Box 3.3. Quality assurance and control for air monitoring: overall objectives

- Measurements should be accurate, precise and credible
- Data should be representative of ambient conditions
- The results should be comparable and traceable
- The measurements should be consistent over time
- The rate of data capture should be high and distribution even
- Resources should be used optimally

Box 3.4. Major components of quality assurance and control for air monitoring*Quality assurance*

- Defining the objectives of monitoring and data quality
- Designing a network and systems for management and training
- Selecting and establishing sites
- Evaluating and selecting equipment

Quality control

- Controlling routine site operations
- Establishing a chain of calibration and traceability
- Auditing and intercalibrating the network
- Maintaining and supporting systems
- Reviewing and managing data

often characteristic differences in the emphasis and practical implementation of the system. A common oversight is placing too much emphasis on laboratory-based quality assurance activities, as these are often easier to control and monitor. Although such tasks are vital, especially for sampler-based measurement programmes involving substantial laboratory analysis, considerable emphasis in any network quality system needs to be focused on the point of measurement. Mistakes or problems at the start of the measurement chain cannot be readily corrected afterwards. Design and maintenance of the sampling system, regular site visits, audits and intercalibration therefore play an important role in network quality assurance.

In addition to internal site-specific procedures, overall network intercalibration must be performed regularly (usually every 6 to 12 months). Intercalibration exercises generally consist of circulating carefully prepared standards: submitting them to a large number of laboratories or measurement locations, which are conducting the particular analysis. Further, comparison workshops may be conducted at a central sample air manifold for various participating laboratories.

An example of comparison workshops at the national level is in Germany; the North Rhine–Westphalia State Environmental Agency has been conducting such workshops for all agencies responsible for monitoring air pollution in the German *Länder* at least once a year since 1984 (4). At the international level, WHO has undertaken five comparisons of

air quality monitoring in the European Region between 1994 and 1996 (5–7).

Another unifying feature of network quality systems is the need for effective screening and validation of data. In any measurement programme – however well designed or operated – equipment malfunction, human error, power failures, interference and a wide variety of other disturbances may result in the collection of spurious or incorrect data. These disturbances must therefore be identified and removed before a final, definitive dataset that maximizes data integrity and utility can be generated or used.

The design of an effective and targeted quality assurance and control programme is only the first step in the process of managing for quality. The programme needs to be fully documented and compliance with its procedures and requirements actively monitored. Monitoring programmes often evolve over time as objectives, legislation, resources or air pollution problems change. Quality assurance programmes therefore also need to be regularly reviewed to ensure that they remain properly targeted and fit for the purpose.

A step-by-step model for developing and implementing quality assurance and control programmes for air monitoring is depicted in Fig. 3.3. Quality assurance and control systems are considered in greater detail elsewhere (2,8).

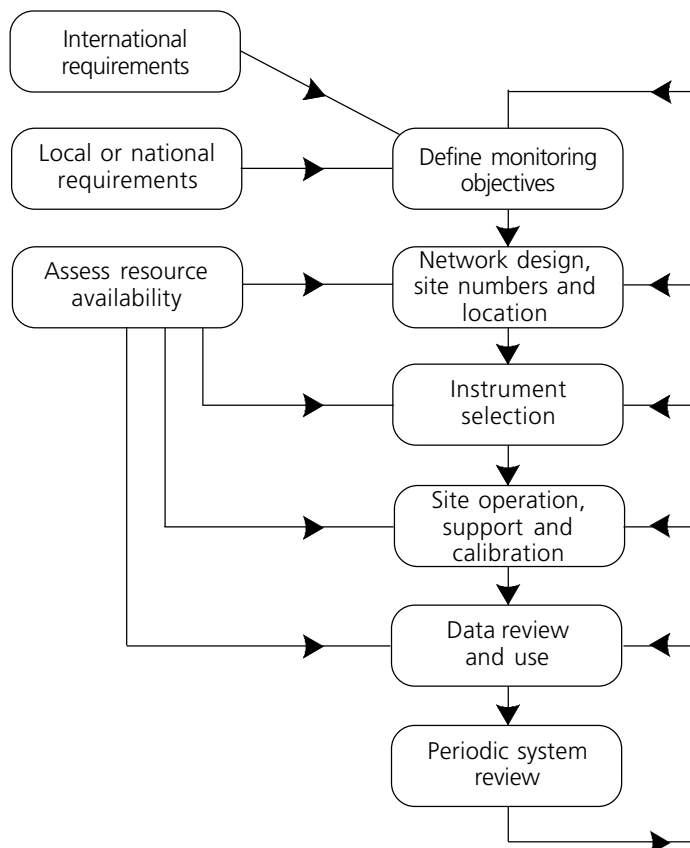
Regarding the process of harmonizing measurements and data at international level, a manual on a quality assurance programme was prepared for participants of the GEMS/AIR network (2). This manual gives an overview of different aspects that should be considered and provides basic information for anyone managing air quality.

Developed strategies on quality assurance for air monitoring have already been implemented in part in national (8,9) and international (2,10–13) programmes.

Network design

There are no universal rules for network design, since any decisions made will be determined ultimately by the overall monitoring objectives

Fig. 3.3. Quality assurance and quality control for air quality monitoring: a step-by-step approach



(including assessing exposure and health effects) and the availability of resources.

Although monitoring systems can have one specific objective, they more commonly have a broad range of targeted programme functions. No survey design can help to completely address all the possible monitoring objectives listed in Box 3.1. However, the design of surveys to meet these individual requirements often has common features and can

Box 3.5. Network design: what resources are available?

- \$£ money (start-up funds and operating funds)
- ☺ skilled personnel
- 🕒 time pressure and the time available to develop the network

use common data (to avoid duplication of effort) and overlapping data to verify the credibility of results and conclusions. The overall design goal is to ensure that maximum information can be derived from a minimum of effort.

Resource availability and constraints

Resource availability is a key issue that needs to be addressed very early in the process of designing a network (Box 3.5). In practice, this is usually the major determinant in network design and strongly influences the numbers of sites chosen, the pollutants to be monitored and the instrumentation selected.

A wide range of commitments and costs is likely to be incurred in any air monitoring programme. Some are listed in Box 3.6. Before any capital or other resources are firmly committed, the survey must be planned, resource availability assessed, the most appropriate equipment selected and monitoring sites chosen.

Any equipment must be purchased based on its long-term operational or financial sustainability. Local sustainability requires the continuing availability of agents (or an in-house capability) for repair and maintenance, together with the necessary skill base for routine operation and

Box 3.6. Costs of air monitoring

- Initial purchase of analysers, samplers and site infrastructure
- Servicing, maintaining and repairing equipment
- Staff and subcontractor costs – operations and management
- Quality assurance and control audits, intercalibration and training
- Data management: collection, analysis, archiving and reporting
- Collecting samples, processing and laboratory analysis
- Operating costs: site rental, electricity, consumables, spare parts, calibration gases, telephone, transport and other items

Box 3.7. Basic issues in assessing exposure

- Where is the population?
- What are the concentrations of pollutants to which they are exposed and for how long?
- What are the priority pollutants (relative to standards)?
- In what areas and microenvironments is exposure important?

calibration of equipment. Financial sustainability recognizes the need for an ongoing budget for equipment operation, typically amounting to approximately 10% per year of the initial capital expenditure.

An ongoing resource commitment to quality assurance and control is also required in any monitoring survey or network, in order to ensure that its measurements comply fully with programme data quality objectives and are therefore fit for the purpose. Typically, a budget of 20–50% of the total annual operating costs may be appropriate for quality assurance and control, depending on the complexity of the programme and the stringency of its data quality objectives.

Numbers of sites and selection

For the purposes of designing a network to assess population exposure, a number of basic issues first need to be addressed, as discussed extensively in Chapter 2 (Box 3.7).

In practice, the number and distribution of air quality monitoring stations required in any network, the samplers required in a survey, also depend on the area to be covered, the spatial variability of the pollutants being measured and how the data are to be used (Box 3.8).

Box 3.8. Network design: numbers of sites

- The number of sites required depends on:
- the intended use and objectives of the data
 - the area to be covered
 - the spatial variability of pollutants
 - the availability of resources
 - the instruments deployed.

There are a number of approaches to designing a network and selecting sites. Exposure assessment often needs to target both source-oriented monitoring sites (often synonymous with worst-case or hot-spot environments) and background locations optimized for quantifying general population exposure (see the example in Table 2.1). Depending on the pollutants being assessed, data from a wide variety of location types may therefore be necessary to build up a reasonably complete picture of ambient exposure patterns (Table 3.1). Relevant site location types for the purpose of health impact assessment or compliance monitoring are reviewed in further detail elsewhere (14,15) and are also presented in Annex 5.

Although the overall requirement of any network or survey is to maximize spatial coverage and representativity, in practice this goal is only approached by grid-based monitoring strategies. These can be opti-

**Table 3.1. Possible monitoring locations
relevant to exposure assessment**

Site classification	Description
City or urban centre	An urban location representative of general population exposure in towns or city centres, such as pedestrian precincts and shopping areas
Urban background	An urban location removed from the sources of pollution and therefore broadly representative of city-wide background conditions
Suburban or residential	A location type situated in a residential area on the outskirts of a town or city
Kerbside or near a road	A site sampling within 1–5 meters of a busy road
Industrial	An area where industrial sources make an important contribution to long-term or peak concentrations
Rural	An open countryside location as far away as possible from roads and populated and industrial areas
Other	Any special source-oriented or microenvironment site or one located at a targeted receptor point, such as a school or hospital

mized to provide detailed information on the spatial variability and exposure patterns of priority pollutants. However, this approach is highly resource-intensive and is therefore not widely used. To reduce resource requirements, a grid approach can be used in conjunction with intermittent or mobile sampling. However, use of this technique is not consistent with the need to maximize temporal representativity as well as spatial coverage.

A more flexible approach to network design, appropriate over a city-wide or national scale, involves siting monitoring stations or sampling points at carefully selected representative locations, chosen based on the required use of the data and the known patterns of emission and dispersion of the pollutants under study. Some factors to be considered in site selection are detailed in Box 3.9.

This approach to network design requires considerably fewer sites than grid strategies and is therefore less expensive to complement. However, sites must be carefully selected if the data measured are to be useful. Moreover, modelling and other objective assessment techniques may need to be used to fill in the gaps in any such monitoring strategy. Numerous factors should be considered in selecting a site location.

The overall monitoring objectives usually determine the target areas for the study, priority pollutants and the number of sites required.

Compiling data on sources and emissions can assist substantially in selecting sites. These will help to identify the most polluted areas as well as other location types where population exposure may be significant. If a full emission inventory is not available, then surrogate statistics

Box 3.9. Network design: factors to consider in site location

- Major sources or emissions of pollutants in the area
- Target receptors and environments
- Weather and topography
- Model simulations of dispersion patterns in the area
- Existing air quality information (such as from screening studies)
- Data on demography, health and land use

such as population density, road traffic flows and fuel consumption may be of use in estimating likely pollution hot spots.

If the area of interest has already been monitored, the existing air quality data from previous studies may be useful in targeting problem areas for the purpose of health impact assessment. If no such studies have been carried out, special screening surveys may be designed to provide area-wide or local information on pollution problems. These often involve passive samplers and/or mobile monitoring laboratories.

The results of dispersion modelling simulation can be used to predict patterns of pollutant dispersion and deposition, thereby helping to identify areas where exposure may be greatest. To be of real use, the data on emissions and meteorology must be reliable and combined with an appropriate and validated model.

The prevailing weather conditions and local topography strongly influence the dispersion of air pollutants and affect the production of secondary pollutants in the atmosphere.

Other information, such as that related to demography, health, population and land use, can be valuable in targeting locations representative of both baseline and worst-case exposure. The use of geographical information systems, in particular, allows both ambient measurements and other geographically coordinated data sets to be used for exposure assessment, epidemiological studies and a range of air quality management activities.

The process of site selection must also take into account the spatial distribution and variability of gaseous and particulate pollutants within urban environments. For example, concentrations of primary traffic pollutants such as CO are highest at roadside locations, whereas ozone levels are more spatially uniform but are lowest in near-road locations because of scavenging by vehicle NO emissions. For this reason, optimizing the measurements for all pollutants at any one site location is not usually possible. In such circumstances, some degree of compromise is often required.

In general, the spatial variability of secondary pollutants such as NO₂ and O₃ tends to be more homogeneous than that of primary pollu-

tants such as CO and SO₂. This greater variability of primary pollutants, especially near the sources, influences the density and numbers of monitoring sites required in any survey.

Small-scale siting considerations are also important in ensuring meaningful and representative measurement. If baseline concentrations are to be assessed, then monitoring sites should be adequately separated from local pollutant sources (for example, roads or small boilers) or sinks (such as dense vegetation). Probe aerodynamics and site sheltering are also often important. Free airflow around the sampling inlet is necessary to ensure representative sampling; for this reason, sampling in a stagnant or highly sheltered microenvironment should also be avoided.

For the purpose of health impact assessment, sampling heights need to approximate, as far as is practicable, the breathing zone of relevant population subgroups.

A properly designed air sampling system is extremely important. In order for samplers or automatic analysers to reliably measure ambient pollutant concentrations, these pollutants must be transferred unchanged to the instrument reaction and collection chamber.

The sampling manifold is a crucial and often overlooked component of any monitoring system that strongly influences the overall accuracy and credibility of all the measurements made. Some design requirements common to all gas sampling systems for gaseous air analysers are summarized in Box 3.10.

Box 3.10. Key features desired in sampling system design

- Inertness to pollutants being sampled
- Minimum air residence time
- Low interaction between the airstream and sample line
- Flow above total analyser demand
- Minimum pressure drop
- Removal of interference such as water vapour or pollutants
- Avoidance of sudden temperature changes when air is sampled
- Ease of cleaning and maintenance

A variety of practical considerations also apply to selecting monitoring sites. They must be accessible for site visits, but potential public interference or vandalism must also be avoided. Mains electricity must be available for pollutant analysers and station infrastructure together with a telephone link if data telemetry is being used (Box 3.11).

Sampling strategies

Monitoring involves qualifying pollutant behaviour in both space and time. A good network design should therefore seek to optimize both spatial and temporal coverage within the constraints of available resources.

The previous section focused on maximizing spatial coverage and the representativity of measurement. Achieving good time-domain performance is not a problem for most methods commonly used in air monitoring. However, once priority pollutants are identified, the measurement technologies selected must be capable of time resolution consistent with the pollutant averaging times specified by the WHO air quality guidelines for Europe.

Continuously operating automatic analysers may be used to assess compliance with short-term or long-term guidelines. Well recognized semiautomatic methods such as SO₂ samplers based on ultraviolet fluorescence measurement or high-volume particulate samplers are perfectly adequate for measurement against daily standards or criteria. Integrating measurement methods such as passive samplers, although fundamentally limited in their time resolution, are useful in assessing long-term exposure and are valuable for a variety of area-screening, mapping and network design functions (11).

Box 3.11. Small-scale considerations in site selection

- Public safety
- Site visibility
- Security and vandalism
- Access to utilities
- Planning permits
- Local sources or sinks
- Aerodynamic clearance or sheltering

Problems can arise, however, when manual sampling methods are used based on a strategy of intermittent, mobile or random deployment. Such an approach is usually adopted for operational or instrumental reasons or simply because the sample numbers or data produced by continuous operation could not be analysed.

Intermittent sampling is still surprisingly widely used world-wide, for instance in parts of eastern Europe and China. In the United States, high-volume samplers for suspended particulate matter are frequently deployed in a sampling schedule comprising one day in six. Such a sampling strategy may be of limited utility in assessing diurnal, seasonal or annual pollutant patterns or, indeed, in reliably assessing patterns of population exposure.

Instrumental issues

The capabilities of air monitoring methods, as well as the inevitable resources required, strongly influence network design. This section reviews some of these issues. Specific monitoring methods applicable to individual criteria pollutants are reviewed in Chapter 4.

Air monitoring methods can be divided into four main generic types, covering a wide range of costs and performance levels. The main advantages and characteristics of these techniques are summarized in Table 3.2. An additional technique, involving the use of bioindicators, is essentially qualitative in nature and is therefore not widely used for assessing regulation, monitoring compliance or assessing health impact.

Choosing the simplest technique that will do the job is good practice. Inappropriate, too complex or failure-prone equipment can result in poor network performance, limit data utility and waste money. Although monitoring objectives are the major factor to consider, resource constraints and the availability of skilled personnel must also be considered. There is a clear trade-off between equipment cost, complexity, reliability and performance. More advanced systems can provide increasingly refined data and greater temporal resolution of measurement but may be more skill-intensive in terms of operation and support, more expensive to operate and maintain and less reliable in service.

Table 3.2. Air monitoring methods

Method	Advantages	Disadvantages	Capital cost
Passive samplers	Very low cost Very simple No dependence on mains electricity Can be deployed in very large numbers Useful for screening and baseline studies	Unproven for some pollutants In general, only provide monthly and weekly averages Labour-intensive deployment and analysis Not a reference method for monitoring compliance Slow data throughput	US \$10 to \$70 per sample
Active samplers	Low cost Easy to operate Reliable operation and performance Historical dataset	Provide daily averages Labour-intensive sample collection and analysis Laboratory analysis required Slow data throughput	About US \$1000 to \$3000 per unit
Automatic analysers	Proven High performance Hourly data On-line information	Complex Expensive High skill requirement High recurrent costs	About US \$10 000 to \$15 000 per analyser
Remote sensors	Provide path or range-resolved data Useful near sources Multi-component measurements	Very complex and expensive Difficult to support, operate, calibrate and validate Not readily comparable with point measurements Not a reference method for compliance monitoring	About US \$70 000 to \$150 000 per sensor or more

Sampler methods are not necessarily less accurate than automatic analysers. For instance, co-located measurements can show excellent agreement, to within plus or minus 10%, between chemiluminescent NO_x analysers and diffusion tubes, providing that both techniques are subject to high standards of quality assurance and operational practice (16).

Measurement accuracy is not discussed at length here. However, many of the determinants of the accuracy of an overall measurement chain are extrinsic to the analyser itself. These include the sampling system, calibration technique and traceability, operational and quality assurance practice. The way any measuring technique is used therefore often more strongly influences the final accuracy than does the nature of the technique itself.

In practice, the combined use of samplers and automatic analysers in a hybrid monitoring programme can offer a versatile and cost-effective approach to network design on a municipal or national scale. Such a network design will use passive or active samplers to provide good spatial coverage and area resolution of measurements. Automatic analysers, deployed at carefully selected locations, can provide more detailed time-resolved data for assessing peak concentrations or comparing with short-term standards.

In some circumstances, integrating samplers may also be used. Reasonably robust statistical relationships can often be derived between peak, upper percentile and long-term average pollutant concentrations (16,17). Although these semi-empirical relationships may differ from pollutant to pollutant and among generic site types, they may enable long-term datasets from sampler surveys to be used to assess broad compliance with short-term guidelines or, at least, to identify areas where concentrations are likely to exceed the guidelines. This sort of indirect assessment technique should, however, always be used with caution.

Indicator pollutants can also be useful. The levels of one pollutant may be deduced by measuring the levels of another one when the local air pollution is dominated by emissions from one source sector and when robust and well established emission ratios exist for the pollutant in question. For example, traffic-related NO_x, benzene and lead concentrations may be estimated in near-source locations (for example, at

kerbsides) from the corresponding CO concentrations. However, surrogate measurements of this kind must always be used with caution.

System review

No network design is static. Any monitoring programme must evolve over time as objectives, resources and the air pollution situation change (Box 3.12).

For example, throughout much of Europe, national measurement programmes have had to shift from a historical emphasis on smoke and sulfur dioxide (arising from domestic coal combustion) to a completely different problem: primary and secondary and air pollution from motor traffic in cities.

Networks, priority pollutants, instrumentation and quality assurance procedures must be reviewed regularly to ensure that programmes remain properly targeted, cost-effective and fit for the purpose of meeting overall monitoring objectives. However, when networks are reviewed, it should also be recognized that a long-term commitment to uninterrupted monitoring at selected fixed locations (typically over 5 years) is required to deduce meaningful long-term trends for most air pollutants.

System operation: automatic networks

Site visits

Automatic networks often use telemetric data transfer systems, which can provide an efficient and cost-effective method for acquiring data from sites. However, automatic networks still require regular inspection

Box 3.12. Networks are not static!

Periodic review is required to assess:

- changes in monitoring objectives and priorities
- new priority pollutants and measurement methods
- changes in site conditions
- new local or national legislation
- changing patterns and sources of emission
- changes in resource availability.

of the equipment by operators. Frequent, documented site visits are therefore an essential component of any quality assurance and control system for air monitoring, although the frequency of visits required varies from network to network. The visits should be performed as frequently as operational needs, geographical constraints and available personnel permit.

Many operations essential to maximize data integrity and capture rate must be carried out on site. These are summarized in Box 3.13.

To enable these functions to be carried out efficiently and systematically, a site visit schedule should be drawn up that provides for all monitoring sites to be visited regularly at specified intervals, typically between weekly and monthly. A comprehensive calibration record and instrument checklist should be completed after each site visit and retained for subsequent quality assurance and control checking.

Equipment support and maintenance

Maintenance procedures for air quality analysers are extremely important. Only through proper instrument support can monitoring systems be relied on to operate satisfactorily and for extended periods in the field. Maintenance schedules for replacing consumable parts, making diagnostic checks and overhauling equipment should always follow the manufacturer's recommendations. Call-out procedures and repair turnaround times also need to be specified if equipment fails in the field. In many networks, equipment maintenance and support are sub-contracted to local equipment service agents.

Box 3.13. Site visit functions

- Weekly to monthly frequency
- Ensure smooth running of equipment
- Calibrate and perform diagnostic checks
- Anticipate future problems
- Change filters and consumables
- Check sampling system and pumps
- Clean sampling systems
- Install, replace and repair equipment
- Check external site conditions

Complex air monitoring technologies require resource commitment well beyond the initial capital investment costs. Ongoing expenditure is required for the lifetime of the equipment to support the monitoring effort, maintain the equipment in an operational state and ensure that meaningful data are being acquired.

Calibrating equipment

Automatic monitoring equipment must be properly calibrated to obtain accurate and traceable air quality data. For most common gaseous air pollutants, this involves the use of on-site transfer gas cylinders or permeation sources to generate a reproducible equipment span point, thereby determining the system response to an accurately predetermined concentration of the air pollutant under analysis.

An additional determination of the corresponding zero point, or system response when no pollutant is present (a measurement made using zero-gas cylinders or suitably scrubbed air), suffices to give a two-point calibration, which is adequate for many purposes. Multi-point calibration involving several different span concentrations is required in some circumstances: for instance, after equipment servicing or repair or if linearity problems are suspected.

A calibrating cylinder from a commercial source or permeation tube determination cannot always be relied on for field calibration; these sources should, wherever possible, be independently verified in the laboratory before use on site. They also need to be checked frequently during their operational lifetime, to identify drift or degradation.

The quality assurance plan for the network should define the frequency and type of field calibration required for any analyser. A typical scheme would include automatic calibration every 24 hours, using on-site permeation tube ovens or gas cylinders, and manual calibration using independent sources performed during every site visit. Regular intercalibration involving all the network's analysers should also be performed (see page 59) in large networks.

Rigorously certified gas mixtures, or sources produced in house, should always be used as primary laboratory references for the field transfer standards. Such primary standards must be directly traceable to absolute measurements or to accepted national or international metrology standards.

A number of proven laboratory-based techniques are available for preparing or verifying primary gas standards (Table 3.3). In practice, preparing gases with one technique and verifying or cross-checking with others are often desirable.

A sound primary calibration base and a clear traceability chain for all measurements are very important. A sound capability for generating primary standards is necessary to prepare or verify secondary standards. These can then be used at the point of measurement and for producing transfer standards used for harmonizing measurements throughout a monitoring programme. Reliable primary standards are therefore fundamental determinants of measurement quality and accuracy in any monitoring network (18).

Table 3.3. Primary gas calibration methods and traceability
(+: appropriate method; -: not applicable)

Method	Pollutant					Comments on method traceability
	CO	SO ₂	NO	NO ₂	O ₃	
Commercial cylinder	+	-	-	-	-	Concentrations not assumed; must be checked by independent methods as appropriate
Permeation tubes	-	+	-	+	-	Absolute (weighing); commercial tubes may be traceable to standards
Static dilution	+	+	+	+	-	Absolute method (volume)
Dynamic dilution	+	+	+	+	-	Dependent on cylinder and mass flow controller performance
Gas phase titration	-	-	+	-	+	Not absolute but comparative technique (O ₃ /NO)
Ultraviolet photometry	-	-	-	-	+	Absolute method (ultraviolet absorption)

Intercalibration and audits

In large-scale national networks, on-site calibration procedures need to be supplemented by regular audits and intercalibration (Box 3.14). Audits are typically arranged at least once a year. These provide an opportunity for direct and qualitative assessment of operator procedures, site performance, infrastructure and instruments. They also allow data or instrument anomalies to be investigated on site.

Intercalibration may be performed every 3–6 months, depending on the network type. This involves a quantitative assessment of the full measurement system at each site, using common reference standards transported sequentially to each site in a network. In large-scale national networks, intercalibration is essential to ensure data comparability and to establish a direct measurement traceability chain to primary standards.

Both audits and intercalibration also offer a powerful tool for harmonizing network performance and measurements across large international networks such as GEMS/AIR (now re-launched as the Air Management Information System (AMIS)) or the emerging EUROAIR-NET (14).

System operation: sampler networks

Many of the generic quality assurance and control and operational methods discussed in previous sections are equally applicable to automatic and sampler-based monitoring networks. However, some additional considerations, discussed here, are especially appropriate to sampler monitoring programmes.

Box 3.14. Site audits and intercalibration

- Usually performed by an external organization
- Ensure data comparability within networks
- Check site conditions or anomalies
- Establish a traceability chain to national standards
- Check consistency of site operations
- Investigate systemic measurement anomalies

Although active and passive samplers are relatively simple to use, careful attention must be paid to quality assurance and control procedures to ensure that the data obtained are of defined and adequate quality. As they use relatively simple techniques to collect samples, any quality assurance and control programme should largely emphasize the subsequent laboratory analysis of the samples collected.

Equipment evaluation and selection of sampling equipment

Passive samplers are very simple to construct and can be constructed or purchased commercially from manufacturers. For instance, a number of institutes and manufacturers supply NO₂ tubes and analytical services. In selecting a suitable analytical service, evidence of its quality assurance system should be carefully examined.

Active samplers can be built from their constituent parts (such as sampling inlet and tubing, absorption medium or filter, gas meter or flow controller and pump) or purchased as units from commercial suppliers.

Analysis of samples

Quality assurance of the analysis of diffusion tubes includes activities such as the use of analytical and field blanks, checking the dimensions of the tubes and preparing quality control charts for the analysis.

All samples must be handled and analysed carefully by trained, competent technicians; samples must be chemically analysed by a competent analytical laboratory. It is not possible, for reasons of space, to discuss in detail the analytical methods used. However, laboratories normally have their own methods that have been developed and validated in house. These may be colorimetric or instrumental: for example, ion chromatography. In all analyses, reliable analytical-grade chemicals should be used.

As a further quality assurance measure, staff should document and adhere to all analytical procedures. They should follow procedures laid down in International Organization for Standardization (ISO) standards or others where available and applicable. Methods and procedures should be independently audited and reviewed regularly.

Passive samplers can be used to measure pollution in rural areas, where pollutant concentrations are much lower. However, where sam-

plers are exposed in areas with very low ambient concentrations, extra care needs to be taken with analysis and the use of analytical and field blanks.

Calibrating sampling systems

For active sampling systems, the volume or flow-metering device and the sampling or analytical procedures must be calibrated. Gas meters used to determine the sample volume or flow meters used to check the flow rate should be calibrated against primary standards before use and regularly thereafter. Primary standard methods for flow calibration include a mercury sealed piston volumeter, a soap film volumetric calibrator, a spirometer and the displacement bottle technique. The sampling volume should be checked at every site visit, to ensure that the flow rate is maintained at a stable level within expected bounds.

Ideally, the sampling and analytical procedures should be calibrated by introducing a known concentration of the pollutant of interest into the sampling system for a measured period of time and subsequently analysing the sample following normal procedures. The known concentration of a pollutant can be generated from a permeation device from bottled span gas or by static or dynamic dilution.

However, this may be logistically difficult for networks with large numbers of sites, and the sampler is generally checked by visual inspection and checking the flow rate. Sample lines also need to be checked and replaced as required. Leaks are a common problem with active samplers, and careful attention needs to be paid to the sample line and to the joints within the sampler.

For passive samplers, the dimensions of the diffusion tube samplers should be accurately measured and checked for consistency between different batches of samplers. A small error in the determination of the dimensions may lead to substantial error in the final gas concentration calculated.

Harmonizing analytical laboratories

In large or multiple-organization programmes where several laboratories may be conducting analysis, laboratory intercomparison must be performed at regular intervals to check the consistency of analytical procedures and results. This can be implemented by arranging for each

laboratory to analyse a standard solution, gas sample and/or doped solutions. These samples are prepared centrally by one laboratory and sampled and analysed following normal procedures. This will check the accuracy of the calibration standards used in the analytical laboratories. In addition, doped passive samplers can be used to intercompare the full procedure for passive sampler analysis.

Data management

Even if all the guidelines for network operation described so far have been successfully implemented, further quality control measures are necessary to maximize data integrity. In any air monitoring network – however well implemented or operated – equipment malfunction, human error, power failures, interference and a wide variety of other disturbances can result in spurious or incorrect data. These must be filtered out before a final, definitive, database can be generated or used.

For convenience, the data review process can be regarded as a two-stage process: data validation followed by ratification. Data validation involves a rapid front-end screening process to identify or remove clearly spurious data prior to initial use; ratification refers to a long-term review of databases prior to final archiving, analysis and reporting.

Data validation

Rapid front-end screening of measurements is especially important in networks with a commitment to real-time data dissemination to public or technical users. Nevertheless, the screened data disseminated in this way are provisional and may be subject to change as a result of subsequent ratification (see pages 63–64). Box 3.15 reviews some basic ground rules for data validation.

Box 3.15. Data validation: some ground rules

- An ongoing front-end screening process
- Review all data
- Do it quickly
- Use printouts and graphs
- Common sense and experience are required
- Avoid excessive dependence on automatic systems

Many commercially available data telemetry and turnkey monitoring systems allow out-of-range or suspect data and calibration factors to be identified. Adhering to strict data acceptance criteria and automatically rejecting flagged data, however, do not necessarily guarantee high data quality. For instance, following such a pre-established data rejection routine often invalidates extreme (but valid) pollutant measurements simply because they lie outside pre-set or accepted limit values.

Although leading-edge software tools such as neural networks offer promise of reducing routine data validation workloads, they can only be regarded as a useful tool rather than a complete solution at the present time.

Active examination of data throughput by skilled personnel can provide a more flexible approach to data validation. This review process is an important component of network quality assurance programmes; it serves both to identify possibly erroneous or invalid data and to inform field operators in good time of any equipment malfunction or problem requiring attention.

Data can be summarized daily and graphical data and calibration control charts compiled regularly (monthly and/or seasonally) to assist front-end data review. The experience, common sense and initiative of data-screening staff are prerequisites for the review process to be implemented successfully.

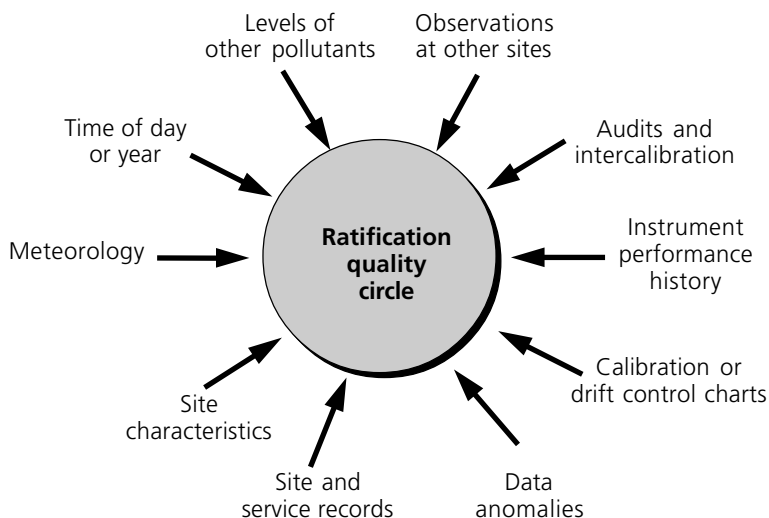
Data ratification

Data ratification is not a mechanistic process and does not readily lend itself to automation. Although software-based expert systems may be able to assist in future, human judgement is the only reliable method. Ratification is a high-skill exercise involving considerable knowledge of pollutant behaviour and dispersion, instrument characteristics, field experience and judgement (Box 3.16). Some of the many inputs to the ratification of national network datasets are summarized in Fig. 3.4.

A regular intercalibration dataset is often a major type of input to the ratification process. Ratification is therefore typically based on databases covering 3–6 months, allowing long-term performance drift, site and instrument anomalies to be reliably identified.

Box 3.16. Data ratification

- Periodic review, perhaps every 3–6 months
- The final stage of data acceptance
- Usually carried out by a separate quality assurance and control unit
- Based on judgement and experience
- Many types of input

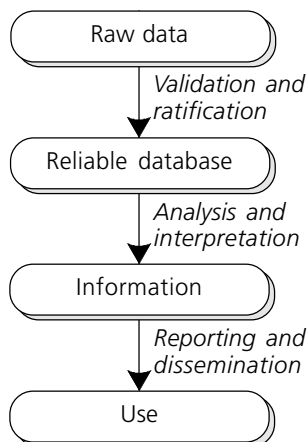
Fig. 3.4. Input to the data ratification process

Less frequent data ratification frequencies are not usually recommended. These may allow errors to propagate or worsen, leading to invalid and rejected data in the long term.

Turning data into information

The purpose of monitoring is not merely to collect data but to produce information useful for technical, policy and public end-users (Fig. 3.5).

Raw data, in themselves, are of very limited utility. These first need to be screened (by validation and ratification) and collated to produce a reliable and credible dataset. In typical information systems for air quality management, the ratified measurements are placed in a database to-

Fig. 3.5. Data flow in a monitoring programme

gether with corresponding emission datasets, model predictions and other input relevant to decision-making.

The next stage in data management is appropriate analysis and interpretation, designed to provide useful information in an appropriate format for end-users. A variety of proven analytical methods are available for air quality datasets (19). In the final analysis, however, the appropriate level and method of data treatment will be very much determined by the ultimate end-use.

A minimum level of data management could be the production of daily, monthly and annual summaries, involving simple statistical and graphical analysis. The use of geographical information systems should be considered, especially for combining pollution data with those from epidemiological and other geo-coordinated social, economic or demographic sources.

The information thereby derived from measured data must be reported or otherwise disseminated, in a timely manner, to end-users. This can be in the form of bulk datasets, processed summary, peak or average statistics, cases in which standards or targets are exceeded, analytical results, graphs or maps. Chapter 5 provides examples of forms for presenting data.

Formats for information transfer should be designed that are appropriate to both the capabilities of the network and to the requirements of the users. Communicating data or information may involve a number of transmission methods, including paper, computer media and electronic media.

Paper. The longest-established method of communicating air quality information is a written paper report. Annual summary reports of air quality for technical end-users, together with information brochures for the general public, remain widely used channels for data dissemination. Some argue, however, that this method is rapidly becoming obsolete in the light of advances in electronic data communication methods.

Computer media. Based for many years on the transfer of floppy disks, this method has now been rapidly superseded in many developed networks by the use of on-line or broadcast methods. The systems are supplemented by the annual publication of CD-ROMs containing both data and/or summary statistics from national or municipal air monitoring programmes.

Electronic media. Electronic media will be the dominant communication medium in the future. For public and technical users in many countries, they are already the source of information on air quality that is easiest to use and most accessible.

Public information systems play an increasingly important role in many countries in raising awareness, warning of pollution episodes and advising susceptible population subgroups. The Air Quality Information Service in the United Kingdom provides a good example of such a system. Hourly updated data on all pollutants monitored in national automatic networks in the United Kingdom, together with 24-hour regional forecasts, are available via teletext, television weather bulletins and a free telephone service. These hourly data, together with ratified datasets, analyses, maps and information on current air quality issues, are also available on the Internet (<http://www.aeat.co.uk/netcen/airqual/welcome.html>, accessed 12 August 1999).

In the most recent development, the entire archive of air quality and emission data for the United Kingdom has been made globally available, in a user-friendly, interactive and easily downloadable format, on

a Web site (<http://www.aeat.co.uk/netcen/airqual>, accessed 12 August 1999). Similar methods of disseminating air quality information are applied in other countries, such as in Austria (<http://www.ubavie.gv.at>, accessed 12 August 1999) or Germany (<http://www.umweltbundesamt.de/uba-info-daten-e/daten-e/hid.htm>, accessed 12 August 1999). The Swedish Environmental Research Institute maintains a Web site presenting the analysis and interpretation of the collected information (<http://www.ivl.se/proj/urban>, accessed 12 August 1999). For the present, these Web sites may be regarded as the state of the art in dissemination of air quality data. On a European scale, EUROAIRNET, the Air Quality Monitoring and Information Network of the European Environment Agency (14), implements the AIRBASE system, allowing access to the data from several countries through the Internet (<http://www.etcaq.rivm.nl>, accessed 12 August 1999).

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Strategies for monitoring selected pollutants

Following the discussion of generic requirements for air monitoring systems in the previous chapter, this chapter focuses on methodological, quality assurance and monitoring strategy requirements for specific pollutants. The pollutants addressed are CO, O₃, SO₂, NO₂, PM₁₀ and PM_{2.5} (see definitions under particulate matter), benzene, polycyclic aromatic hydrocarbons (benzo[*a*]pyrene), lead and cadmium. The rationale for selecting these species is that they are:

- ubiquitous in urban air
- widely recognized as posing a potential risk to population health
- commonly regulated at a national or international level.

A consistent, harmonized approach has been adopted in considering the individual components. For each of the pollutants, the following issues are considered:

- health effects, which provide a rationale for assessment (why monitoring is carried out);
- sources and exposure patterns, which suggest where to monitor;
- monitoring methods, which indicate how to monitor using available technologies;
- monitoring and assessment strategies: general principles for a cost-effective, targeted approach; and
- a selected example, intended to highlight phenomena related to specific pollutants or to outline a national approach to monitoring.

Both gravimetric ($\mu\text{g}/\text{m}^3$ or mg/m^3) and volume ratio (parts per billion volume) concentrations are used in this chapter for gaseous pollutants, whereas only gravimetric concentrations are appropriate to particulate matter. The relationships between the gravimetric and volume ratio measures of gaseous concentrations depend on the molecular weight of the species and only apply to a standard temperature and pressure.

At a standard temperature and pressure, the ratios between mg/m³ and ppm presented in Table 4.1 apply.

Table 4.1. Conversion factors for selected gaseous air pollutants at a standard temperature of 20°C and pressure of 1.01325×10^5 Pa

Component	Gravimetric to volume ratio	Volume ratio to gravimetric
CO	1 mg/m ³ = 0.86 ppm	1 ppm = 1.17 mg/m ³
O ₃	1 mg/m ³ = 0.5 ppm	1 ppm = 2 mg/m ³
SO ₂	1 mg/m ³ = 0.37 ppm	1 ppm = 2.67 mg/m ³
NO ₂	1 mg/m ³ = 0.52 ppm	1 ppm = 1.91 mg/m ³

Carbon monoxide

Päivi Aarnio, Tarja Koskentalo & Kari Hämekoski

Health effects

The binding of carbon monoxide (CO) with haemoglobin to form carboxyhaemoglobin (COHb) reduces the capacity of blood to carry oxygen, and the binding with other haem proteins is directly related to changes in the functions of affected organs, such as the brain, cardiovascular system, exercising skeletal muscle and the developing fetus. At very high concentrations, well above normal ambient levels, CO causes death.

A COHb level of 2.5% should not be exceeded to protect middle-aged and elderly people with documented or latent coronary artery disease from acute ischaemic heart attacks and to protect the fetuses of pregnant women from untoward hypoxic effects. Based on this, WHO (see Annex 1.1) has suggested the following guideline values for CO: 100 mg/m³ for 15 minutes, 60 mg/m³ for 30 minutes, 30 mg/m³ for 1 hour and 10 mg/m³ for 8 hours (1).

Sources and exposure patterns

The most important source of carbon monoxide in ambient air is motor vehicle traffic. CO emissions peak when traffic movement is restricted: in crossroads with traffic lights and in traffic jams. As the use of three-way catalytic converters in cars is increasing, the emissions and thus

also the concentrations in ambient air will decrease. The use of reformulated fuels also reduces CO emissions. In the future, most CO emissions will be caused by driving with a cold engine, which retards the oxidation of CO in the catalytic converter. In some areas, industrial sources may also cause elevated CO concentrations.

The natural background levels of carbon monoxide range between 0.01 and 0.23 mg/m³. In urban road traffic environments of larger European cities, 8-hour mean concentrations are generally less than 20 mg/m³, and 1-hour peak values less than 60 mg/m³.

In underground and multistorey car parks, road tunnels, ice arenas and various other indoor microenvironments, the mean levels of CO can rise above 115 mg/m³ for several hours, with much higher short-term peak values. In these situations, excessive concentrations are caused by combustion engines operated with insufficient ventilation. In homes with gas stoves, peak concentrations up to 60 mg/m³ have been measured. Smoking in dwellings, offices, shops and restaurants can significantly raise the mean CO concentration in indoor air (1).

Monitoring methods

The most commonly used method for measuring CO in urban atmospheres is non-dispersive infrared spectrometry. It is an automated continuous method with a measurement range of about 0.5 to 115 mg/m³ (2). The alternative gas-chromatographic method is manual or automated and continuous only insofar as several discrete air samples can be drawn and analysed each hour. It is not commonly used in monitoring networks (3). A hot mercury oxide method is also available for automated CO monitoring, but this method has disadvantages because it uses mercury and may produce interference.

As discussed in Chapter 3, the advantage of automated methods is that they give continuous real-time data and thus their time resolution is good. Nevertheless, the instruments are relatively complex and expensive and require skilled personnel for operation. In 1997, the European Commission gave the European Committee for Standardization a mandate for establishing standard measuring methods for, among others, CO in ambient air. The European Union will probably propose non-dispersive infrared spectrometry as a reference method.

A sensitive diffusive sampler has been developed for determining carbon monoxide concentrations (4). The method is cost-effective but has the disadvantage that the measurements are time consuming and the time resolution is poor; it provides only long time averages. The sampler has been shown to successfully measure ambient CO concentrations when the product of the concentration and the sampler exposure time is 30 to 1600 ppm-hours and to be unaffected by environmental factors such as relative humidity, temperature and wind velocity. Personal exposure monitors have been developed, enabling population exposure to be measured directly at levels that are found in ambient air.

Monitoring and assessment strategy

The assessment aims at, for example, checking whether the concentrations are below guideline values, supporting air quality management, providing public information and assessing exposure. The design of a monitoring network is always a compromise between theoretical considerations and practical restrictions. The measurements can be complemented with additional assessment, especially with dispersion models.

CO concentrations in ambient air vary substantially over time and space. Concentrations are a function of emissions (traffic density), weather conditions and street configuration; they are highest near emission sources and decrease rapidly as the distance from the sources increases. Because CO is primarily released near the ground, the vertical and horizontal dispersion is different from that of emissions from elevated sources. CO is a primary pollutant, and concentrations do not therefore directly depend on atmospheric reactions that form secondary species.

A few fixed monitoring sites cannot provide a representative picture of the average population exposure to CO. Many other pollutants are more evenly distributed throughout an urban area. Moreover, ambient monitoring stations cannot describe many sources of exposure, such as tunnels, garages, ice arenas and indoor environments. Fixed monitoring stations poorly describe exposure for a short time period. However, personal exposure is better related to ambient levels when 8-hour or longer time averages are used.

Emission data and the location of the most important sources give valuable information on the hot spots for the siting of monitors. If a

comprehensive emission inventory is not possible, some evaluation may be conducted using, for instance, data on the volume of road traffic. Information on the population distribution and land use and on urban and transport plans can help to identify the major present and future exposure environments.

Weather conditions and topography strongly influence the dispersion of air pollutants. In conjunction with emission data, dispersion models may be used to initially assess likely concentrations and to identify possible hot spots. Other assessment methods such as passive samplers may also be used to map the concentrations and can thus give preliminary information on the spatial distribution of concentrations.

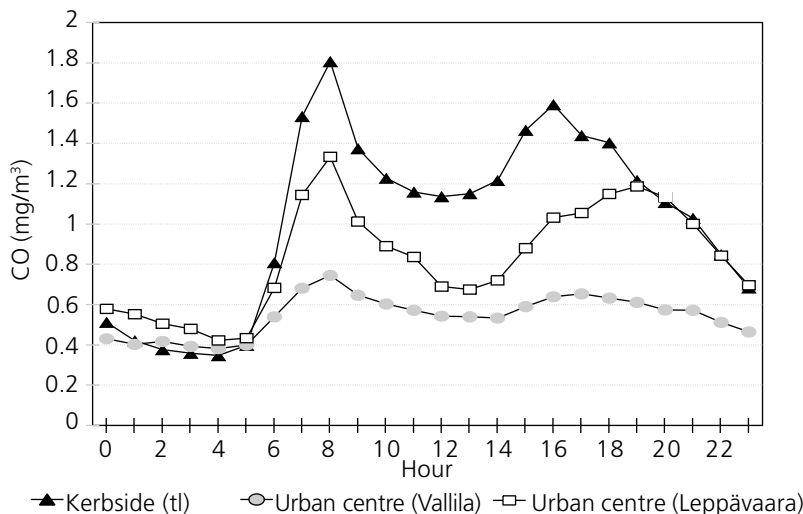
Approaches to site selection for CO differ depending on the objectives of the monitoring. If the main purpose is to monitor compliance with guidelines or standards, then only different types of hot spots might be chosen. If the purpose is to evaluate population exposure, then more widespread siting is needed. The monitoring sites should represent specific exposure environments covering kerbside, urban centre, urban background, suburban or residential and industrial environments.

The results from the monitoring network can be supplemented with dispersion models to improve spatial coverage when adequate emission and meteorological data, needed as model inputs, are available.

The siting of monitoring should be representative so that the results can be generalized to all environments of the same type. More emphasis should be given to the areas where the number of people exposed is large and to areas with weak dispersion conditions, such as street canyons. The results of ambient monitoring can be used for evaluating population exposure in a few different ways: relying on the measurement at fixed sites alone, which is widely used although it does not give an full picture of exposure; combining CO dispersion models with monitoring data; or establishing CO exposure models that combine CO concentrations in microenvironments with time–activity data.

Diurnal variations of CO in the Helsinki Metropolitan Area in Finland show clear differences in concentrations between stations (Fig. 4.1). The hourly concentrations are mean values from January to March 1996. The kerbside monitoring station is situated in a busy road traffic

Fig. 4.1. Diurnal variation of CO concentrations in the Helsinki Metropolitan Area, Finland from January to March 1996



environment near a major crossroad; the Urban Centre in Vallila is about 20 metres from a street, in an environment with much less traffic than in Töölö. The urban centre monitoring site in Leppävaara is situated in a suburban traffic environment near a highway. The monitoring height is about 4 m.

If mobile monitoring units are to be used, the monitoring period at one location should be long enough to cover diurnal, annual and random variation of concentrations as well as all the essential weather conditions within a year. The frequency of sampling should be adequate to make comparisons with the guidelines appropriate to CO. This means that averages for 15 and 30 minutes, 1 hour and 8 hours should be obtained from the collected data. In practice, the best way to obtain all these data is to use fixed monitoring sites with continuous analysers.

Selected example

The personal exposure of preschool children to CO was studied in Helsinki, Finland (5). Road traffic is the major outdoor source of carbon monoxide. Indoor sources of CO include, for example, gas stoves, garages and smoking. The children in this study were from two down-

town day care centres that were located closest to the fixed ambient air CO monitoring sites (Töölö and Vallila) of the Helsinki Metropolitan Area Council.

The median CO level at the Töölö monitoring site was twice that of Vallila. Both the medians and frequency distributions of one-hour personal exposures in Töölö and Vallila (homes with electric stoves) were practically indistinguishable. The median personal CO exposure levels of children from homes with gas stoves, however, were about twice those of children from homes with electric stoves both in Töölö and Vallila.

The personal CO exposure levels of children from homes with gas stoves were usually higher than corresponding levels measured at the closest fixed-site monitors (personal exposure monitoring mean 2.79 mg/m³, ambient air monitoring mean 1.36 mg/m³). The indoor exposure of children living in houses with electric stoves was also higher than the CO levels measured outdoors, but the difference of the mean values was smaller (personal exposure 1.91 mg/m³, ambient air 1.52 mg/m³).

It was concluded that data on fixed-site ambient air monitoring alone should not be used to estimate the personal exposure of children to CO, neither average levels, the time distribution for exposure nor even the relative differences between different areas. Gas stoves using human-made town gas may strongly increase personal exposure to CO.

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Ozone

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Health effects

Ozone (O₃) is a secondary photochemical pollutant formed from the precursors volatile organic compounds, NO_x and CO in the presence of short wavelength solar radiation. Ozone can enter the body through inhalation and can reach the respiratory system because it is not very soluble in water. Acute exposure to high ozone levels can induce changes in lung function, airway inflammation and increased airway responsiveness to bronchoconstrictors. Ozone exposure has also been associated with increased numbers of hospital admissions from respiratory diseases, including asthma.

Existing data from field studies and controlled exposure studies lead to the recommendation of a guideline value of 120 µg/m³ as an 8-hour mean value (Annex 1.1). Besides providing the guideline values, the updated air quality guidelines for Europe specify the risk of various health outcomes at certain exposure levels close to the guideline concentration. Measurement data collected by the EMEP (the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (<http://projects.dnmi.no/~emep/index.html>, accessed 12 August 1999)) (1–3) and the European Topic Centre on Air Quality (4) indicate that this WHO air quality guideline is currently exceeded in almost every European country.

Sources and patterns of exposure

As mentioned before, ozone is a secondary pollutant that is not directly emitted into the air but is formed by photochemical reactions. As a consequence and in contrast to primarily emitted pollutants, short-term

ozone concentrations tend to peak some distance from the emission sources of ozone precursors.

The temporal and spatial distribution of ozone concentrations can be complex and depends on various effects such as photochemical ozone formation, horizontal and vertical ozone transport, ozone depletion by reaction with NO and dry deposition. These effects are strongly influenced by weather conditions, topography and the presence of other pollutants, mainly the precursor substances volatile organic compounds, NO_x and CO. These processes are responsible for distinct European, subregional and local patterns of ozone levels. Since weather conditions not only influence the propagation of ozone but also its formation, ozone levels vary greatly from year to year in many countries.

Elevated short-term peak values often occur at locations different from those with high long-term averages. Peak short-term values (1 hour to a few hours) usually occur in several parts of Europe in the plumes of large emission sources. Consistently high long-term average values (8-hour or daily mean values) may often be recorded at elevated stations (especially in mountainous regions), where neither dry deposition nor NO depletion plays a major role and exchange processes with the free troposphere may occur.

The rural background concentrations of ozone in Europe can be crudely estimated from the calculations with the photochemical EMEP model (5). These calculations show that long-term average ozone values increase from northwestern Europe towards central Europe. However, these model results are not suitable for assessing the exposure of the population in non-rural areas.

Ozone patterns in or near urban areas can be quite complex. Near major sources of NO such as roads with heavy traffic, ozone levels are usually quite low. In cities in central and southern Europe, ozone can be produced quite rapidly in episodes of high photochemical activity, and suburban regions often have the highest peak values. By contrast, the photochemical formation of ozone can be slower in northern Europe, leading to peak concentrations far outside cities. Ozone levels in urban areas in these countries are therefore often below those in rural background areas.

Monitoring methods

In principle, continuous and discontinuous measurement can be distinguished. Continuous measurement is preferred for the assessment of health effects using dose-response relationships and compliance with WHO air quality guidelines for Europe. Discontinuous measurement using, for example, passive samplers can, however, be used for indicative measurements in the process of designing a network, area screening or elucidating spatial patterns of exposure.

Automatic measurement can be divided according to three separate criteria: sampling techniques, method of measurement and method of calibration.

The chemical reactivity of ozone has important implications in designing sampling systems for ambient ozone measurement using any of the techniques listed in Table 4.2. Interaction between the wall and the sample should be minimized, together with gas residence times in the system. Careful attention to maintaining and regularly cleaning sampling systems is also vital, especially in polluted urban areas.

The first two measurement methods listed in Table 4.3 are point measurements and are therefore not directly comparable to measurement using the differential optical absorption system, since these give information on the ozone concentration along an optical path. Ozone concentrations for health assessment are currently measured based on the first two methods, which ensures that the results are comparable. The ultraviolet photometry method is most widely used in European networks.

An ultraviolet calibration photometer is recommended for calibration; this is widely used throughout Europe. In principle, each country should have its primary calibration standard, which should be traceable to an ap-

Table 4.2. Commonly used techniques for sampling ozone

Method	Description	Advantage
Laminar flow	Flow 150 l/min, tube diameter 0.15 m	Isokinetic sampling
Turbulent flow	Modular construction	Low cost
Sampling without manifold	Direct connection to analyser	Low cost

Table 4.3. Commonly used automated methods of measuring ozone

Method	Reference	Disadvantage
Chemiluminescence	ISO 10313: 1993 (6)	Use of ethylene
Ultraviolet photometer	ISO 13964: 1998 (7)	Possible interference
Differential optical absorption system (DOAS) spectrometer		Measurement disturbed by fog, field calibration complicated, expensive

appropriate national metrology standard or to an international reference such as a US National Institute of Standards and Technology standard.

Strategy for monitoring and assessment

In principle, measurement should be carried out where ozone levels are highest and where it is likely that the receptor is exposed. This implies that the design of the network should primarily be determined by the pattern of ozone levels and the distribution of the population. Measurement should therefore be carried out in densely populated urban areas not directly influenced by local sources of emissions. As mentioned previously, additional measurement may be necessary in urban background regions, suburban regions (which may readily lie outside the city boundaries) affected by urban plumes and in rural areas to ensure a complete overall picture of the exposure of the population. Sites in rural areas should preferably be located in small towns and villages.

Since ozone concentrations are measured at a limited number of monitoring sites, the spatial representativity of the monitoring sites has to be ensured. Monitoring stations should therefore be representative in urban regions for at least 10 km², in large conurbations for 100 km² and in rural regions for even larger areas, depending on the complexity of the terrain. With respect to emission sources, ozone monitoring sites should be located at background locations: not in the immediate vicinity of heavily frequented roads or industrial sources, since local NO emissions may cause ozone depletion and thus distort the spatial representativity of the ozone measurement.

Careful siting should ensure the maximum possible comparability of results obtained from continuous measurements at one site to those

from similar locations in a network (Table 4.4). Consequently, information can be made available not only on the measured ozone levels but also on the spatial representativity of each monitoring site and thereby on the population potentially exposed to these ozone levels. This is especially important, since most countries have existing ozone monitoring networks with limited flexibility in removing or relocating stations.

Existing ozone measurement should also be considered in designing a measurement network. Very few sites suffice in regions with either very low or spatially uniform ozone levels.

Examples

Austria. Austria has a dense network to monitor ozone (8). One primary task of the network is to provide data to inform the public about current ozone levels, as required in the Austrian Ozone Act and in the European Union's 1992 ozone directive (9). To obtain information on the spatial distribution of ozone, a sophisticated model was developed to interpolate ozone levels between monitoring stations, using a daytime- and altitude-dependent function (10). The results of the model calculations are updated and published every few hours during ozone episodes. In principle, these data can be used in combination with population data for health assessment in Austria.

European Union. The European Topic Centre on Air Quality publishes an annual report on ozone concentrations exceeding the threshold values set in the European Union's 1992 ozone directive (4,9). These

Table 4.4. Site characterization for ozone measurements

Site classification	Comments
Road traffic	Not suitable for measurement
Urban	Preferentially at representative residential sites not directly influenced by emissions of precursors
Suburban	Sites in the outer fringes of cities where maximum ozone concentrations can be expected, influenced by urban plumes
Rural	These sites might be located in or at small settlements
Industrial	Not a high priority for ozone measurement

reports provide a comprehensive overview of ozone levels in urban and non-urban sites scattered over the territory of the European Union. In addition, a consolidated report on the ozone situation including some health assessment is expected to be published soon (11).

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Sulfur dioxide

Bohumil Kotlik & Jon Bower

Health effects

A range of chronic and acute health impacts may result from human exposure to sulfur dioxide (SO₂) or related species.

In a gaseous form, SO₂ can irritate the respiratory system; in case of short-term high exposure, a reversible effect on lung functioning may occur, according to individual sensitivity.

The secondary product H₂SO₄ primarily influences respiratory functioning. Its compounds, such as polynuclear ammonium salts or organosulfates, act mechanically in alveoli and, as easily soluble chemicals, they pass across the mucous membranes of the respiratory tract into the organism.

Particulate aerosol formed by the gas-to-particle formation has been found to be associated with numerous health effects, as mentioned in the section on PM₁₀.

The WHO air quality guidelines for SO₂ are:

- 500 µg/m³ for 10 minutes of exposure
- 125 µg/m³ for 24-hour average exposure
- 50 µg/m³ for annual average exposure.

Sources and exposure patterns

Although natural sources (volcanic or geothermal) may exist in some localities, anthropogenic sources, involving the combustion of fossil fuels containing sulfur, dominate in most urban areas. These include:

- point sources (power plants, incineration plants, mining and metal processing);
- area sources (domestic heating and district heating); and
- mobile sources (diesel engines).

Exposure patterns and duration often show significant regional and seasonal differences, depending on the dominant sources and their spatial

distributions, weather and dispersion patterns. Extended episodes of elevated concentrations, which may persist for several days during cold, stable winter months when dispersion is restricted, still occur in many parts of the world where coal is used for space heating. Area sources usually dominate in such episodes, resulting in spatially homogeneous patterns of concentration and exposure.

In contrast, short-term episodes ranging from minutes to hours may occur as a result of fumigation, dispersion or plume-bending downwind of point sources. The resulting exposure patterns vary substantially, depending on emission heights, plume buoyancy and weather conditions. The temporal variability of ambient concentrations is also often high in such circumstances, especially for local sources.

Because both the acute and the chronic health effects of SO_2 are important, both episode types – together with long-term exposure – need to be fully assessed in any monitoring strategy.

Monitoring methods

Automatic analysers need to be used if compliance against short-term guidelines is to be determined; a variety of active samplers can be appropriate for comparing with daily or annual guidelines or for estimating the effects of daily variation of the pollution. Passive samplers may be used to provide data for comparison with the long-term annual guideline.

Passive samplers. No national or international standard governs the application of SO_2 diffusion tubes to ambient air monitoring or their laboratory preparation and analysis. Protocols for sampler preparation and analysis by spectrophotometry and ion chromatography have, however, been published (1).

Many passive sampling techniques are available. The most widely used include:

- the triethanolamine/glycol/spectrophotometry method (2)
- the KOH/glycerol/spectrophotometry method (3)
- the Na_2CO_3 /glycerine/ion chromatography method (4).

These techniques are widely hybridized. In the United Kingdom, for instance, KOH or NaOH is used as an absorbent but with the tube mem-

brane proposed by Ferm (4) and using ion chromatography as the analysis method. In practice, the ion chromatographic technique has been informally accepted as the standard method for SO₂ diffusion tube analysis.

The typical measurement precision associated with this hybrid technique is $\pm 8.5 \mu\text{g}/\text{m}^3$: some under-reading against automatic analysers has been observed (about 30%), although agreement with active samplers is better (5).

Active samplers. The sampling equipment required to determine the concentration of gaseous sulfur compounds in ambient air is described in full in ISO 4219: 1979 (6). This standard gives details of the equipment necessary to sample gaseous pollutants by absorption in a liquid bubbler. The standard also includes guidance for siting and installing the apparatus.

The principle of active sampling methods is to draw ambient air through a collecting medium (typically a liquid bubbler) for a specified time, typically 24 hours. The volume of air is metered. The collecting medium is subsequently analysed and the concentration of pollutant in the sampled air determined. This method is well established and proven and has been used in many monitoring networks worldwide for a number of years. In consequence, a long history of SO₂ measurements with active samplers is available to assess the trends.

Several methods of SO₂ monitoring are based on this principle. They can be carried out using the apparatus specified in ISO 4219: 1979. They differ in the solutions used in the bubblers for absorption of SO₂ and the method of analysis. The four most widely used methods are described below.

The acidimetric (total acidity) method, described in ISO 4220: 1983 (7), is used to determine a gaseous acid air pollution index. Although this method measures total acidity and is not specific to SO₂, it is adequate for general use. The simplicity of the method and the fact that the reagents are relatively safe makes it a popular choice for routine monitoring (8).

Ion chromatography is a variation on the above technique. The exposed peroxide solutions are analysed for sulfate ions by means of ion

chromatography, rather than titration. This has the advantage of being sulfate-specific but requires the use of an expensive ion chromatograph.

The tetrachloromercurate method is also known as the pararosaniline method described in ISO 6767: 1990 (9). This is the reference method specified in the European Union directive on sulfur dioxide and suspended particulate (10). However, the reagents used are very toxic, and the method is therefore not widely used.

The Thorin method is presented in ISO 4221: 1983 (11). The reagents used – perchloric acid, barium perchlorate, dioxane and thorin – are hazardous and must be handled and disposed of with care. Accordingly, this method is not commonly used.

Accuracy of $\pm 10\%$ has been estimated for SO_2 measurements using the total acidity method, taking account of all contributory factors. Precision of $\pm 4 \mu\text{g}/\text{m}^3$ is achievable for this widely used method (8).

Automatic analysers. Well established automatic monitoring techniques are available. The most widely used method for automatic SO_2 measurement is ultraviolet fluorescence. SO_2 molecules in the sample airstream are excited to higher but unstable energy states by ultraviolet radiation at 212 nm. These energy states decay, causing emission of secondary fluorescent radiation with an intensity proportional to the concentration of SO_2 in the sample.

The accuracy of data from automatic SO_2 analysers depends on a range of factors encompassing the entire measurement chain. These include accuracy of calibration standards, analyser stability and sample losses in the measurement system. An accuracy of $\pm 10\%$ has been estimated for SO_2 measurements in national automatic networks in the United Kingdom, taking account of all contributory factors. The precision of SO_2 measurements as determined from long-term variation in the baseline response of in-service analysers is estimated to be $\pm 3 \mu\text{g}/\text{m}^3$ (12).

Remote sensors. Remote optical sensor systems, such as the differential optical absorption system, use a long-path spectroscopic technique to measure the real-time concentration of a pollutant integrated along a path between a light source and a detector. Long-path monitoring

systems can be used to measure SO₂, but the methods are less well established than those for automatic point monitors. The accuracy and precision of the data from these instruments are, therefore, much more difficult to determine. The method does not conform to ISO 7996: 1985 (13). Especially careful attention needs to be paid to instrument calibration and quality assurance to obtain meaningful data from remote sensing instruments.

Monitoring and assessment strategy

In some respects, designing a cost-effective, targeted and appropriate strategy is especially complex for SO₂. This is because SO₂ can have both acute and chronic health effects. In consequence, measurements over a variety of time scales may often be required. In addition, SO₂ concentrations are often highly variable in both space and time, resulting in heterogeneous exposure patterns.

In designing an overall monitoring and assessment strategy for SO₂, the relative importance of area space heating and point sources related to industry, incineration and local generation of district heating and power should first be assessed for the area under study. This will enable spatial hot spots, peak concentration seasons and target populations to be identified. The time scales of the relevant effects will also need to be considered, as these will enable the most appropriate monitoring technologies to be selected.

A hybrid approach to the design of networks may be applicable, especially when resources are limited and cost-effectiveness must be maximized. Such an approach uses automatic analysers:

- to identify short-term concentration peaks
- to evaluate short-term temporal variability
- to enable comparison with short-term guidelines
- to provide input to alert or public information systems, if used.

Matching sampler networks (active or passive type) are used:

- to fill in the gaps in the automatic network
- to assess broad spatial patterns and variability, and
- to enable comparison with long-term standards.

Intermittent or spot sampling is often still widely used in many parts of eastern Europe; in this case, careful survey design is necessary to

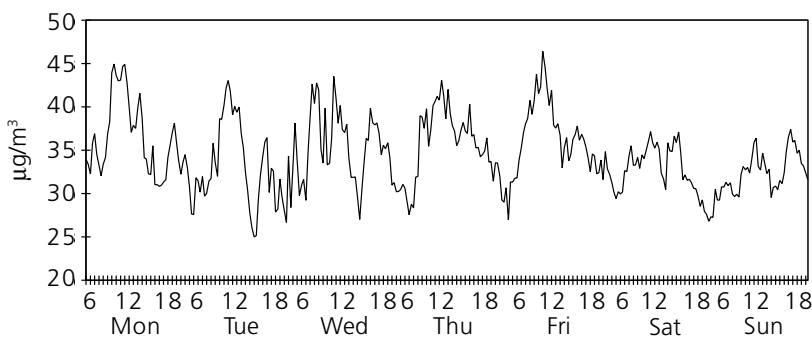
maximize temporal representativity. The results of such surveys also need to be analysed, interpreted and used with care.

Example

The typical temporal variation of SO_2 can be illustrated by the results from an impact study in a small city (less than 250 000 inhabitants) in January 1996. The major outdoor sources of sulfur dioxide investigated were area sources (combustion of fossil fuels) and small factories. The city is situated in a poorly ventilated river valley in the Czech Republic. A mobile measuring system and central automatic station were used for obtaining results, which were based on 30-minute values. The correlation between the results from the mobile and fixed monitors ranged from 0.5 to 0.8 on individual days.

The measured concentration of SO_2 was well below the guideline level (Fig. 4.2). The results varied substantially during the day and were highly correlated with the intensity of human activity. The high mid-day pollution peaks were much lower on Saturday and Sunday than during the rest of the week.

Fig. 4.2. SO_2 concentration (30-minute average) measured over 1 week in a small city in the Czech Republic



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Nitrogen dioxide

Jon Bower

Health effects

Nitrogen dioxide (NO₂) is an air pollutant produced in combustion processes. Whenever nitrogen dioxide is present, nitric oxide (NO) is also found; the sum of NO and NO₂ is collectively referred to as nitrogen oxides (NO_x). Only the health effects of NO₂ are considered here.

At very high concentrations, which may only be encountered in serious industrial accidents, NO₂ exposure can result in rapid and severe lung damage. Health effects may also occur at the far lower ambient concentrations likely to be observed during pollution episodes in cities. The available evidence suggests that ambient exposure may result in both acute and chronic effects, especially in susceptible population sub-groups such as people with asthma.

NO₂ primarily acts as an oxidizing agent that may damage cell membranes and proteins. At high concentrations, the airways may become acutely inflamed. In addition, short-term exposure may predispose towards an increased risk of respiratory infection (1). Although many controlled exposure studies have been undertaken, evidence for clearly defined relationships between concentration or dose and response is lacking.

For acute exposure, only very high concentrations (>1880 µg/m³, 1 ppm) affect healthy people; however, people with asthma or chronic obstructive pulmonary disease are more susceptible at lower concentrations. Based on the best available clinical evidence, a 1-hour guideline of 200 µg/m³ (110 ppb) has been set (2).

Ambient epidemiological studies and animal toxicology investigations have demonstrated that prolonged NO₂ exposure may reduce lung host defences and significantly change lung structure. To protect the general public at large from such chronic effects, therefore, an annual average guideline value of 40 µg/m³ has been set (2).

Sources and patterns of exposure

Development of a cost-effective and targeted NO₂ monitoring strategy requires not only full knowledge of potential health effects but also con-

sideration of the pathways and patterns of human exposure. These, in turn, depend on patterns of emission, population distribution and behaviour.

The main human source of NO_x in the atmosphere is road traffic. This is responsible for about half of total emissions throughout Europe (3,4). Other major sources include power stations, heating plants and industrial processes (5).

Much of the NO_x is emitted as NO , which is subsequently oxidized to NO_2 by ozone or other oxidants. The conversion chemistry of NO to NO_2 , their photostationary equilibrium with O_3 together with the presence and relative importance of other oxidation routes strongly affect the spatial distribution of NO_2 . In general, NO_2 acts as a secondary pollutant and therefore tends to be spatially more homogeneous than primary road traffic pollutants such as CO .

Although motor vehicles account for about 50% of overall NO_x emissions, the proportion is higher in cities. In London, for instance, 74% of NO_x emissions result from road traffic (6); this figure is typical for much of Europe (4). As vehicle emissions are ground-based, moreover, their impact on air quality in the breathing zone tends to be proportionally greater, at least nearby, than corresponding effects from elevated point sources.

A monitoring strategy for NO_2 should therefore take into account the fact that spatial patterns and population exposure are mostly dominated by road traffic. This has a number of consequences. For instance, long-term average city-centre NO_2 concentrations should be higher (typically by a factor of 2 : 1) than corresponding suburban levels (3); the expected urban-rural ratios may be about 4 or 5 to 1 (7,8).

Near-road concentration gradients are also important in network design. The results from large-scale (more than 1200 sites) diffusion tube surveys of cities in the United Kingdom suggest broadly consistent relationships between NO_2 levels at kerbside environments (1–5 m from a busy road), intermediate locations (20–30 m distance) and urban background areas (>50 m from roads). Kerbside levels are typically 35–40% higher than intermediate levels and 60–70% higher than background levels (9).

The temporal variability of NO₂ may also substantially influence monitoring strategy. As indicated in Chapter 3, good semi-empirical relationships may often be derived between peak and mean levels for NO₂ (10). Although these relationships are not universally applied and must be derived locally, using them can enable the likelihood of exceeding short-term guidelines to be inferred from measured long-term average concentrations.

When human exposure to NO₂ is assessed, monitoring data from city centres, near roads and suburban or residential areas are especially relevant, together with source-oriented data from locations likely to be affected by industrial emissions. Microenvironmental data from hot spots are also useful. There is significant potential for NO₂ exposure indoors, especially in domestic environments using such combustion sources as gas cookers. However, indoor exposure is outside the scope of this discussion.

In summary, the following pollutant characteristics should be considered in designing NO₂ monitoring programmes:

- the concentrations are largely determined by road traffic emissions;
- this is a spatially homogeneous, secondary pollutant;
- the concentrations are highest in city centres and near roads; and
- the ratios of peak to mean concentration are statistically robust and useful.

Monitoring methods

Chapter 3 introduced the four most widely used generic air monitoring methods – passive and active samplers, automatic analysers and remote sensors. This section reviews specific techniques based on these generic methods applicable to measuring NO₂ concentrations in ambient air. Many well established and proven methods are available (Box 4.1).

Passive samplers. A number of types and different shapes of passive sampler – tube or badge – are available (11). The most widely used is the tube sampler employing triethanolamine as an absorbent, the Palmes tube (12). This sampler type has been used and proven in a number of large-scale urban and rural surveys, both in Europe and the rest of the world.

Although no national or international standard governs the use of diffusion tubes, protocols for preparing the samplers and photospectroscopic analysis have been published and are widely accepted (12,13).

Recent comparisons of NO₂ diffusion tube measurement with simultaneous, co-located chemiluminescence analysers show good agreement (14). Over typical urban concentration ranges (about 10 to 40 ppb), diffusion tubes tend to over-read analysers by about 10%. Similar studies quoted 5–8% as the precision of the measurement technique.

Quality assurance and harmonizing the laboratory analysis of tubes are essential for best results (8).

Active samplers. A variety of technologies for active sampling are available (10). The best known of these is the Griess-Saltzman method, covered by ISO 6768: 1985 (15). Although this method is sensitive and requires a relatively simple, inexpensive sampling apparatus, there are a number of disadvantages. It uses corrosive chemicals and is not readily usable for sampling periods above 1–2 hours. There is also doubt about calibration methods, collection efficiency and possible side reactions. Thus, this method cannot be recommended for general baseline monitoring applications.

Automatic analysers. The widely accepted reference method for automatic analysers, as defined in the European Union directive on air quality standards for NO₂ (16) and described in ISO 7996: 1985 (17), is the chemiluminescence analyser. A number of models are commercially available and widely used worldwide. These analysers can provide high-resolution real-time data. However, ensuring that the measurements are accurate and reliable requires a high standard of maintenance and operational and quality assurance support (18,19).

The accuracy of measurement from automatic analysers depends on a number of factors encompassing the active measurement chain. These include the accuracy of calibration standards, analyser stability and response and losses in the sampling system. In-service tests show that an overall accuracy of about $\pm 10\%$ may be expected for NO₂ measurements in a well run network, taking into account all these factors. The corresponding precision estimates are about ± 3.5 ppb.

Remote sensors. Remote sensor systems, such as those using differential optical absorption spectroscopy, can be used for real-time measurement of NO₂ – and other pollutants – integrated along a path between a light sensor and detector. Although the technique is well validated for NO₂, it does not conform to ISO 7996: 1985 and cannot be used to determine compliance with NO₂ limit values in accordance with the European Union directive (16).

Differential optical absorption spectroscopy systems may be useful if path-integrated measurements are required or in near-source situations. However, the technique is relatively complex and considerably more expensive than the other NO₂ measurement methods discussed here. Moreover, especially careful attention to calibration and quality control is required if meaningful measurements are to be made.

**Box 4.1. The most widely used methods
of measuring NO₂ concentrations**

- Passive samplers, such as the triethanolamine/Palmes diffusion tube
- Active samplers, such as the Griess-Saltzman method
- Automatic analysers, such as chemiluminescence
- Remote sensors, such as differential optical absorption spectroscopy

Strategy for monitoring and assessment

When an appropriate monitoring strategy for NO₂ is being designed, the generic sites locational and network design criteria introduced in Chapter 3 are applicable. The concept of the hybrid network design, which uses both sampler and automatic monitoring techniques, is especially applicable and well proven for NO₂. This approach can be used at both the municipal and national scales. Hybrid networks optimize both spatial and temporal coverage, thereby ensuring that maximum information can be derived with minimum effort. Such a network design uses automatic analysers, deployed at a few carefully selected representative types of location, to provide detailed time-resolved datasets enabling direct comparison with short-term guidelines.

Because these systems are expensive and complex, however, a monitoring strategy designed solely based on automatic analysers would be extremely resource-intensive. A hybrid network will therefore supple-

ment these measurements with a matching, dense network of passive samplers. Because they are inexpensive and independent of mains electricity, they can be deployed in relatively large numbers, filling in the gaps between automatic analysers and providing both good overall spatial coverage and area resolution of measurements.

The main features of such a network design concept, also using short-term targeted measurement campaigns and the coordinated application of dispersion models, are summarized in Box 4.2.

**Box 4.2. Recommended components of a strategy
for NO₂ monitoring and assessment**

A hybrid measurement network uses samplers and automatic analysers to maximize spatial and temporal coverage.

Automatic analysers are used:

- to identify peak concentration episodes
- to assess compliance with short-term guidelines
- to provide on-line public information systems and alerts
- to model input and calibration.

Passive samplers are used:

- to identify geographical hot spots
- to screen areas and map pollution patterns
- to design and optimize automatic networks
- to assess compliance with long-term guidelines
- to infer compliance with short-term guidelines
- to model input and calibration.

These may be supplemented by short-term measurement campaigns for microenvironmental assessment, plus models and objective assessment for:

- formulating a control strategy
- air quality management
- land-use and transport planning
- forecasting and prediction
- assessing the effects of road traffic and industry.

Example

The ambient networks in the United Kingdom provide a good example of the practical implementation of the monitoring strategy outlined

above. The national networks have more than 80 automatic NO₂ measurement sites, with more operating in a range of local, municipal and city monitoring programmes. The national automatic sites are primarily located in city or urban centre, urban background, near-road and suburban or residential types of location, as defined in Box 3.9 in Chapter 3, with emphasis given to both background and hot spot location types. The automatic sites provide a good picture of temporal variability and of peak concentrations in relation to both short-term WHO air quality guidelines for Europe and European Union limit values. Hourly data are also widely disseminated to the public and mass media by the Public Air Quality Information Service.

Networks in the United Kingdom also measure NO₂ at approximately 1200 passive sampling sites. There are, therefore, about 15 passive sampling sites for every automatic station. Passive samplers are located at near-road, intermediate and background location types. Although it only provides data of limited (monthly) time resolution, this national dataset provides information on near-road gradients as well as a detailed picture of spatial patterns of NO₂ concentrations in urban areas throughout the country.

The automatic and passive sampler data are complementary in practice. Taken together, they provide a good picture of both spatial and temporal pollutant variability throughout the United Kingdom. This hybrid approach to network design has proved extremely cost-effective in practice. Use of measured data in conjunction with the National Atmospheric Emission Inventory has also enabled high-resolution, interpolated maps of NO₂ concentrations to be produced. These are of considerable relevance and utility to assessing both exposure and health effects.

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Particulate matter (PM₁₀ and PM_{2.5})

Michal Krzyzanowski

Airborne particulate matter represents a complex mixture of organic and inorganic substances. Because of the complexity of particulate matter and the importance of particle size in determining exposure and human dose, multiple terms are used to describe particulate matter. Some terms are derived from and defined by sampling and/or analytic methods, such as suspended particulate matter, total suspended particulate and black smoke. Other terms refer more to the site of deposition in the respiratory tract, such as inhalable particles that pass beyond the upper airways (nose and mouth) and thoracic particles that deposit within the lower respiratory tract. Other terms, such as PM₁₀, have both physiological and sampling connotations. The International Organization for Standardization (ISO) and the European Committee for Standardization (CEN) have provided definitions for various types of particles (1–3).

For monitoring purposes, the designations PM₁₀ and PM_{2.5} as defined in Box 4.3 are commonly used.

Box 4.3. Definition of PM₁₀ and PM_{2.5}**PM₁₀**

Particles with aerodynamic diameter 10 µm or less, or, more strictly, particles that pass through a size-selective inlet with a 50% efficiency cut-off at 10 µm aerodynamic diameter. The upper cut-off of aerodynamic diameter is about 30 µm, which means that no particles greater than 30 µm enter the inlet. PM₁₀ roughly corresponds to the thoracic fraction of the particles, that is, those that penetrate beyond the larynx.

PM_{2.5}

Particles with aerodynamic diameter 2.5 µm or less, or, more strictly, particles that pass through a size-selective inlet with a 50% efficiency cut-off at 2.5 µm aerodynamic diameter and have upper cut-off at 7 µm. PM_{2.5} corresponds to the fraction of respirable particles in high-risk populations (children and adults with certain pulmonary diseases) that can penetrate to the unciliated airways.

Health effects

Most of the quantitative information available on the health effects of particulate matter comes from studies in which particles in air have been measured as PM₁₀. There is now substantial information on PM_{2.5}, and the newest studies are showing that fine particulate matter (PM_{2.5}) generally better predicts health effects than does PM₁₀. Evidence is also emerging that constituents of PM_{2.5} such as sulfates and particle strong acidity (measured as the concentration of H⁺ ions) are sometimes even better predictors of health effects than PM_{2.5} *per se*.

The large body of information on studies relating day-to-day variation in particulate matter concentrations to day-to-day variation in health provides quantitative estimates of the effects of particulate matter that are generally consistent. The available information does not allow the concentration below which no effects would be expected to be determined. Effects on mortality, respiratory and cardiovascular hospital admissions and other health variables have been observed at levels well below 100 mg/m³, expressed as a daily average PM₁₀ concentration. For this reason, the updated WHO air quality guidelines for Europe (Annex 1.1) do not include a guideline value for short-term average concentrations.

Less quantitative information is available on the long-term effects of particulate matter. Some studies have suggested that long-term exposure to particulate matter is associated with reduced survival. Other recent studies have shown that the prevalence rates of bronchitis symptoms in children and of reduced lung function in children as well as adults are associated with exposure to particulate matter. These effects have been observed at annual average particulate matter concentrations below 20 mg/m^3 (as $\text{PM}_{2.5}$) or 30 mg/m^3 (as PM_{10}). For this reason, no guideline value is recommended for long-term average concentrations. Instead, the WHO air quality guidelines for Europe provide tables indicating an increase in the risk of health effects connected with certain increments of PM_{10} or $\text{PM}_{2.5}$ short-term (24 hours) or long-term (1 year) average concentrations to guide decision-making on standards for particulate matter.

Based on the WHO air quality guidelines for Europe, the European Commission proposed limit values to be adopted by the 15 European Union countries (4). Independently, national ambient air quality standards have been adopted in the United States (Box 4.4).

Box 4.4. Selected standard levels for particulate matter

European Union directive on limit values for PM_{10} , April 1999 (4)

Stage 1. 1 January 2005

24-hour average: 50 $\mu\text{g}/\text{m}^3$ not to be exceeded more than 35 times per year
1-year average: 40 $\mu\text{g}/\text{m}^3$

Stage 2. 1 January 2010

24-hour average: 50 mg/m^3 not to be exceeded more than 7 times per year
1-year average: 20 $\mu\text{g}/\text{m}^3$

Current United States national ambient air quality standards, adopted in July 1997

PM_{10}

24-hour average: 150 $\mu\text{g}/\text{m}^3$ (99th percentile) average over 3 years
1-year average: 50 $\mu\text{g}/\text{m}^3$

$\text{PM}_{2.5}$

24-hour average: 65 $\mu\text{g}/\text{m}^3$ (98th percentile) average over 3 years
1-year average: 15 $\mu\text{g}/\text{m}^3$

Sources and exposure patterns

The larger particles in the PM₁₀ mix, the coarse fraction (between 2.5 and 10 mm aerodynamic diameter), are mechanically produced by breaking up even larger solid particles. These particles can include wind-blown dust from agricultural processes, uncovered soil, unpaved roads or mining operations. Road traffic produces road dust and air turbulence that can re-entrain road dust. Near the coasts, evaporation of sea spray can produce large particles. Pollen grains, mould spores and plant and insect parts are all in this larger size range.

Smaller particles, called the fine fraction, are largely formed from gases. Particles smaller than micron size can be produced by the condensation of metals or organic matter, which are vaporized in high-temperature combustion processes. The particles produced by intermediate reactions of gases in the atmosphere are called secondary particles. Combustion of fossil fuels such as coal, oil and petrol can produce coarse particles from the release of non-combustible materials (fly ash); fine particles from the condensation of materials vaporized during combustion; and secondary particles through the atmospheric reactions of sulfur oxides and nitrogen oxides initially released as gases.

The main sources of total anthropogenic emissions of primary PM₁₀ are road traffic (10–25%), stationary combustion (40–55%) and industrial processes (15–30%). However, the contribution of road traffic to ground-level urban concentrations and to human exposure would be considerably larger than the contribution of road traffic to emission (5). For an urban area in the central part of the Netherlands, the contribution of domestic road traffic to PM₁₀ concentrations has been estimated to be about 24%, with an additional 17% attributed to the long-range pollution generated by road traffic (6). The other major contributor to PM₁₀ concentration (26–34%) was estimated to be long-range pollution from sources other than transport.

Several studies conducted in Europe, including the central European study on air pollution and respiratory health (CESAR) (7), have indicated that the PM_{2.5} constitutes, on average, about 70% of the PM₁₀ mass. The seasonal variability of PM₁₀ was entirely accounted for by the changes in PM_{2.5} concentration. The CESAR study has also indicated relatively small spatial variation in the concentration of PM₁₀ between locations with markedly different local sources and, historically, signif-

icantly different concentrations of total suspended particulate. The PEACE study (pollution effects on asthmatic children in Europe) collected PM_{10} information in 14 areas throughout Europe and found rather small differences between (background) urban and rural sites: the median urban-rural ratio was 1.22 (8).

Monitoring methods

A GEMS/AIR document (9) comprehensively reviews the monitoring methods. Several types of equipment with various technical characteristics and in various price ranges can be used for the monitoring. It has been proposed that every sampler should be equivalent to those approved according to CEN standard EN 12341 (3), which requires a target accuracy of $<10 \text{ mg/m}^3$ and precision of $<5 \text{ mg/m}^3$ for daily average concentrations below 100 mg/m^3 . The PM_{10} sampling inlet should be tested to ISO standard 7708: 1995 to ensure accurate size fractionation at the point of sampling (1).

According to a number of recent large-scale instrument comparisons, several commercially available high- and medium-volume samplers have been shown to be equivalent to a reference wide ranging aerosol collector. Also, a low-volume Harvard impactor sampler has been already used worldwide as a part of numerous epidemiological and exposure assessment studies. Its good performance has been demonstrated in a comparison study conducted according to CEN requirements. The advantages of the Harvard impactor system include its low cost, ease of operation and maintenance and its ability to collect PM_{10} and $PM_{2.5}$ simultaneously.

The samplers collect particulate matter on a filter using high-volume (about $100 \text{ m}^3/\text{hour}$) or low-volume (about $1 \text{ m}^3/\text{hour}$) pumped sample flow. The weight of the particulate matter deposited on the filter is used to calculate a 24-hour average mass concentration. The appropriate aerodynamically designed inlets are essential for proper particle size cut-off points. Correct filter handling is also fundamental to obtaining valid data. The filters must be conditioned in a humidity- and temperature- controlled environment, typically at 20°C and 50% relative humidity, for at least 24 hours before and after exposure. The filters must be accurately weighted using a suitable microbalance calibrated using an accredited method.

Air quality monitoring networks also use automatic analysers of particulate matter. These instruments are based on the following principles of operation:

- a tapered element oscillating microbalance
- beta-ray absorption analysers
- light-scattering systems

Of these instruments, the tapered element oscillating microbalance and beta-ray systems have been operated widely for many years and are well tested in the field. The light-scattering type of instrument has been developed more recently and is therefore less well proven in service. Operating experience and co-located measurement campaigns indicate that measurements from the automatic analysers are not always equivalent to those from reference samplers.

Monitoring and assessment strategy

The basic information relevant to health is 24-hour average PM_{10} or $PM_{2.5}$ concentration. According to the updated air quality guidelines for Europe, the risk of numerous health effects increases linearly with daily pollution levels in the range of the most common particulate matter concentrations. The network should therefore be able to produce a complete series of daily data allowing the health impact of the daily variation in the pollution concentration to be assessed and the annual average calculated.

The examples from selected exposure assessment studies mentioned on pages 102–103 indicate that the monitoring should not be limited to urban areas, since the elevated levels of PM_{10} or $PM_{2.5}$ may be also found in areas remote from sources.

The spatial distribution of PM_{10} concentration within an urban area may also be quite uniform, as demonstrated by the analysis of daily monitoring data from Birmingham and the Bristol area in the United Kingdom. This analysis is consistent with the uniformity of the annual mean levels observed in the other urban background sites across the United Kingdom (10) as well as with the CESAR study (7). In such situations, a single monitoring station may be representative of a particular site type in a given urban area. However, as shown in an example from PM_{10} monitoring in Prague, a complicated topographic situation may substantially influence pollution concentrations. A preliminary sur-

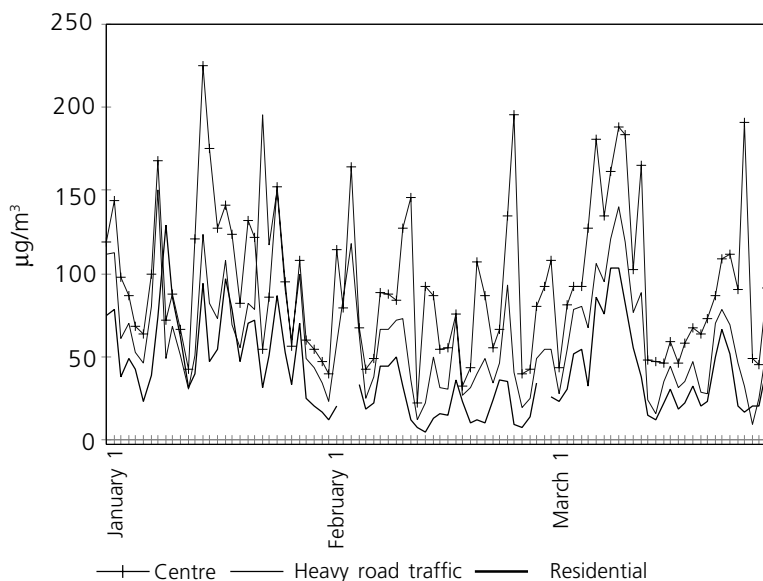
vey of pollution patterns may therefore be necessary to determine a minimum number of the monitors and their spatial distribution.

Example

In 1995, PM₁₀ monitoring was introduced in the Czech Republic. Beta-gauge instruments with PM₁₀ inlets are used. Daily average concentrations of PM₁₀ measured in three different locations in Prague from 1 January 1997 until 31 March 1997 are shown in Fig. 4.3.

Besides significant temporal variation of PM₁₀ concentrations, the monitoring results show significant differences between pollution concentrations in various locations in Prague. The average PM₁₀ concentration calculated for the period presented was 94 $\mu\text{g}/\text{m}^3$ in the city centre, 63 $\mu\text{g}/\text{m}^3$ at the road traffic station located near the main road outside the centre and 41 $\mu\text{g}/\text{m}^3$ in the residential area away from the centre. The

Fig. 4.3. Daily average PM₁₀ concentrations in Prague, Czech Republic, 1 January to 31 March 1997



daily measurements were not very well correlated (correlation coefficients 0.56 for centre versus residential and 0.68 for road traffic versus residential stations). These spatial differences in pollution levels may be explained by the diversified topographic structure of the city, with a number of hills and valleys, and with a large river crossing the town and providing a good ventilation channel.

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Benzene

Anne Lindskog

Health effects

Benzene has low acute toxicity, but repeated exposure to very high concentrations can cause severe effects on the blood and blood-forming organs in humans. Whether benzene also affects the immune or reproductive systems is not clear. Benzene is, however, known to be a human carcinogen. The most convincing relationship is found between benzene exposure and the development of acute non-lymphocytic leukaemia. In this case no concentration can be considered as safe, and the risk assessment is based on lifetime exposure. The WHO air quality guidelines for Europe do not specify a limit value for benzene. Instead they give the geometric mean of the range of estimates of the excess lifetime risk of leukaemia at an air concentration of $1 \mu\text{g}/\text{m}^3$, which is 6 per 100 000. The concentrations of airborne benzene associated with excess lifetime risk of 1 case per 10 000, 1 per 100 000 and 1 per 1 000 000 are, respectively: 17, 1.7 and $0.17 \mu\text{g}/\text{m}^3$.

Sources and exposure patterns

Most benzene in ambient air is emitted from the combustion and evaporation of benzene-containing petrol close to the ground. The only other regionally important source is the emission from small-scale wood combustion (1). Other sources include cigarette smoke and petrochemical plants.

Inhalation is the dominant pathway for benzene exposure in humans, and smoking is a large source of personal exposure. In addition, high short-term exposure can occur during refuelling of vehicles or in connection with commuting (2).

Long-term averages in areas where people live (urban and suburban background) are of interest in establishing the human exposure for risk

assessment. In addition, measurement with higher time resolution is sometimes needed to understand the underlying processes and to trace and evaluate industrial sources.

Measurement methods

A variety of methods is available. A CEN working group is working on a standard covering measurement of benzene concentrations in ambient air. Depending on what can be regarded as an acceptable risk, the required detection limit can be as low as much less than $1 \mu\text{g}/\text{m}^3$.

Two different principles are applied: sampling at the site followed by analysis performed at a central laboratory, and sampling and analysis at the site (automatic sampling).

Sampling at the site can be performed by passive, active or canister sampling and point or line measurement.

Passive sampling. Passive sampling uses sampling tubes with adsorbents such as Tenax, Chromosorb or graphitized carbon black (3,4). The sampler collects benzene by diffusion (following Fick's first law), the driving force being the concentration gradient established between ambient air and the adsorbent. The sampling time is 1–4 weeks. The optimal sampling period must be selected carefully. It would depend on ambient concentration and the need to minimize sample loss through thermal desorption before the analysis. In the laboratory, the trapped benzene is removed by solvent extraction or thermal desorption followed by analysis using high-resolution gas chromatography and a flame ionization detector. Thermal desorption is preferable to avoid the use of a toxic solvent, to increase the sensitivity and to enable the use of an automatic injector. The detection limits are about $0.5 \mu\text{g}/\text{m}^3$ and $0.03 \mu\text{g}/\text{m}^3$ respectively. A method using Tenax TA tubes and an automatic thermal desorption unit is accredited in Sweden (5,6). Several sites can be covered with one analytical instrument. The use of radial diffusion tubes (7) in combination with thermal desorption is being tested to improve time resolution.

Passive samplers are the only method capable of continuous measurement and are thus well suited for monitoring long-term limit values. In addition to benzene, the samplers also cover and assess other aromatic hydrocarbons as part of the standard procedure.

Active sampling. Active samplers use a pump and the same adsorbent tubes as in diffusive sampling. Desorption and analysis is performed as described above. Time resolution from 0.5 hours to 24 hours is possible. The detection limit is in the range of 0.14 to 0.4 $\mu\text{g}/\text{m}^3$. The method is reliable and, when performed with automatic devices (8), cost-effective. Several sites can be covered with one analytical instrument. A general CEN standard is available on the requirements and tests methods of sorbent tubes (9). German standards for pumped tube sampling both with solvent extraction and with thermal desorption are available (10,11).

Canister sampling can be performed in two ways: either as grab sampling or as pumped sampling. Analysis is performed using gas chromatography–flame ionization detection (12). The detection limit is 0.3 $\mu\text{g}/\text{m}^3$. The method is expensive and time consuming and can thus only be used in random sampling. The representativity of the result is therefore generally limited, which adds to the overall uncertainty.

Automatic sampling can be done with two methods: point and line measurement. Point measurement uses on-line gas chromatography–flame ionization detection, available either as relative simple BTX monitors, measuring benzene, toluene, ethyl benzene and xylenes, or more complex instruments capable of measuring C_2 – C_{10} hydrocarbons. Ambient air is sucked through a trap kept at a temperature of 40°C or at a subambient temperature as low as about –190°C depending on the other compounds to be trapped. Following sampling, trapped compounds are injected into the gas chromatograph by thermal desorption, either directly (BTX monitor) or via a secondary, cryo-focusing trap. The detection limit is 0.1–0.03 $\mu\text{g}/\text{m}^3$. The systems work with a time resolution of 30–60 min. However, the method is semi-continuous: for example, sampling 20 minutes of each 30 or 60 minutes.

The results obtained with on-line gas chromatography can be transmitted directly to the central laboratory, and on-line data are therefore available with these methods.

The use of BTX monitors is relative expensive. A monitoring hut with air condition and gas supply is needed, and each analytical instrument covers only one location. A German standard (13) with minimum requirements and a test for single-cycle analysers (= BTX analysers) is available. The monitors capable of measuring the whole range of C_2 – C_{10}

hydrocarbons are even more expensive and require trained personnel for maintenance. In some cases a supply of liquid nitrogen is also needed. This type of monitor is used within the hydrocarbon monitoring network in the United Kingdom and at several background sites.

Differential optical absorption spectroscopy is an open-path optical measuring technique applicable to line measurement for a number of gases, including benzene (14), based on the differential absorption of ultraviolet and visible light. The detection limit is about 3–5 $\mu\text{g}/\text{m}^3$, depending on the optical path, with a time resolution of 1 minute. The method measures along a typical optical path of 500 m. However, problems with measurement in low visibility (such as foggy conditions), with the zero point offset, with uncertainty because spectra are similar and with calibration using test materials (such as calibration gases) still have to be solved.

Monitoring and assessment strategy

As benzene is a primary pollutant, the concentration normally drops rather quickly with the distance from the source. Long-time averages are needed to evaluate the health risks, and the sampling strategy should provide a realistic image of the temporal and spatial distribution. Diffusive samplers could be the method of choice, provided that the accuracy and precision are proven to be adequate. Measurement performed at an carefully selected urban background site could give a value representative for a larger part of the town and thus also represent the average exposure of a great number of people. It is important that the site not be directly affected by road traffic or other benzene sources. The sites could preferably be located in residential areas and/or in the town centre in open squares or adjacent to pedestrian streets, 3–5 m above ground. The urban background site also enables trends to be followed and the outcome of different abatements applied to reduce the concentration in ambient air to be evaluated.

The hot spot sites include roadside (kerbside) measurement and industrial sites. Roadside (kerbside) measurement is important in determining the exposure of commuters. Again, the sites should be representative for the average exposure in areas where people are spending a substantial part of their time.

Since benzene is a relatively stable compound that can be transported long range, a national network should also include rural background sites.

Example

Children's exposure to benzene, toluene and xylenes has been studied in Denmark (16). Both the personal exposure and the front-door concentrations were measured using diffusive samplers. The front-door benzene concentrations were significantly higher in Copenhagen than in rural areas, a median value of $8.9 \mu\text{g}/\text{m}^3$ versus $1.9 \mu\text{g}/\text{m}^3$, but the personal exposure was only slightly higher, $5.4 \mu\text{g}/\text{m}^3$ versus $4.5 \mu\text{g}/\text{m}^3$.

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Polycyclic aromatic hydrocarbons

Eva Brorström-Lundén

Polycyclic (or polynuclear) aromatic hydrocarbons (PAH) are complex mixtures of hundreds of chemicals, including derivatives of PAH, such as PAH with a NO₂ group (nitro-PAH) and oxygenated products, and also heterocyclic aromatic compounds. PAH have a wide range of vapour pressure and are semivolatile, which means that they are transported in the atmosphere both in the gas phase and bound on particles. The partitioning of the compounds between the gas and particle phases depends on factors such as the vapour pressure of the compound, the ambient temperature and the concentration of particles in the air.

Health effects

The biological properties of most PAH are still unknown. Nevertheless, the available data, mostly from animal studies, indicate that several PAH may induce a number of adverse effects, such as immunotoxicity, genotoxicity, carcinogenicity and reproductive toxicity (affecting both male and female offspring). PAH may also influence the development of atherosclerosis. However, the critical endpoint for the health risk evaluation is the well documented carcinogenicity of several PAH. Benzo[a]pyrene (BaP) is the PAH most widely studied, and most information on the toxicity and occurrence of PAH is related to this compound.

WHO has recommended no specific guideline value for PAH as such in air. BaP, however, is used as an indicator for the carcinogenic PAH in air. Based on epidemiological data from studies in coke-oven workers, the lifetime risk of respiratory cancer using BaP as an indicator in air is estimated to be 8.7×10^{-5} per ng BaP per m³ (1). The corresponding concentrations of BaP producing excess lifetime cancer risks of 1 per 10 000, 1 per 100 000 and 1 per 1 000 000 population are 1.2, 0.12 and 0.012 ng/m³ respectively.

Sources and exposure patterns

PAH are frequently found in ambient air. They are formed during incomplete combustion of organic matter, and important PAH sources are transport and electricity and heating generation.

Most PAH are stable in the atmosphere, and considerable long-range atmospheric transport takes place. The current annual mean concentrations of BaP in major urban areas are in the range of 1–10 ng/m³. In rural areas, the concentrations are below 1 ng/m³. The atmospheric concentrations of PAH vary seasonally at background sites, with higher concentrations during the winter than during the summer. At the west coast of Sweden, increased PAH concentrations occur in connection with long-range transport, but there are also events with increased concentrations caused by local sources such as wood burning (2).

In the urban atmosphere, road traffic exhaust provides an increased local contribution of PAH, and emission from road traffic is the major source of PAH at the street level. A research programme was carried out in Sweden from 1984 to 1987 to improve the understanding of urban air pollution (3). The average PAH concentrations were 50% higher in

streets with high traffic than in streets with low traffic. The concentrations varied seasonally, with the highest concentrations during winter. However, only PAH in the particle phase were measured. PAH concentrations vary diurnally in urban air, with peaks during the morning and the afternoon rush hours and the lowest concentrations at night (4).

The introduction of catalytic converters on light-duty petrol vehicles as well as cleaner diesel fuels have reduced the PAH emissions from road traffic substantially during recent years. However, the results of long-term monitoring of PAH concentrations in urban air are not frequently reported in the international literature.

There are also indoor sources of PAH such as tobacco smoke or unvented heating sources. These may be important for the human exposure to PAH in the air.

Monitoring methods

Sampling methods. Since the concentrations of PAH in ambient air are low, large air volumes must be collected, which require long sampling times or a sampler with a large flow capacity. A high-volume air sampler equipped with a filter for trapping the particles from the air and a solid adsorbent for collecting substances in the vapour phase is the most frequently used technique for sampling PAH. A high-volume air sampler operates with an air flow rate of 20 to 25 m³/hour, and a volume of 1000–2000 m³ is normally used for collecting PAH in background air. This type of equipment has previously been recommended in measurements within EMEP (the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (<http://projects.dnmi.no/~emep/index.html>, accessed 12 August 1999)).

Glass fibre filters are frequently used for PAH in the particle phase. Other common filter materials are quartz fibres and Teflon-coated filters. The adsorbents commonly used for sampling PAH in the gas phase are polyurethane foam and Amberlite® XAD®-2 resin. Alternative adsorbents include Tenax TA (a porous polymer resin based on 2,6-diphenylene oxide) or Tenax GR (Tenax TA plus 30% graphite) and C₁₈.

In aerosol sampling, a possible source of error is that some of the PAH bound to particles may volatilize and then adsorb onto the adsorbent (blow off). On the other hand, organic compounds in the gas phase

may be adsorbed onto the filter surface. As a consequence, the distribution of the compounds between the adsorbent and the filter will differ from the true distribution between the vapour and the particle phases. Factors that may affect the distribution of PAH include the sampling duration, the ambient temperature and the vapour pressure of the compounds. Thus, a short sampling time reduces the risk for artefacts.

Chemical reactions during PAH sampling may also be a source of error. During sampling, retrained PAH are exposed to reactive gases such as NO_2 and O_3 during a relatively long time period. Although the choice of the filter media does not seem to affect the formation of chemical artefacts, the opposite can be true for the adsorbent media. A clear degradation of PAH has been observed in the case of Amberlite XAD-2, whereas PAH adsorbed on polyurethane foam did not seem to be affected.

Although Amberlite XAD-2 has been found to be more effective for trapping more volatile organic compounds than polyurethane foam, it cannot be recommended as an adsorbent for air sampling of PAH when even low concentrations of NO_2 are present. Polyurethane foam is a better choice for eliminating chemical artefacts even if there is a risk of breakthrough.

Low-volume air samplers have been tested recently. The advantage of this type of sampler is the high sampling capacity and possible high time resolution. However, the small sample volumes place high demands on the analytical methods. Filters from impactors can be used to collect particles for PAH analysis to obtain PAH concentrations on different particle size fractions.

Analysis of the collected samples. Ambient air samples are chemically complex and contain many organic species that require selective pre-treatment procedures as well as selective and sensitive analytical techniques for PAH. Two subsequent steps determine the effectiveness of the analysis: extraction and the analytical techniques used.

Prior to the analysis, the PAH collected on filters and adsorbents are extracted into a solvent. As PAH and other compounds with structures similar to that of the “coal sheet” in soot particles may be tightly bound to the particles, an effective extraction procedure is required. Soxhlet extraction with an organic solvent is the method most commonly used.

Pre-treatment procedures frequently used for PAH include fractionation on different columns, such as silica as well as liquid/liquid extractions. Internal standards must be added to the sample extracts to correct for losses during various pre-treatment procedures. Laboratory blanks as well as field blanks must also follow all procedures in the analytical work.

Gas chromatography–mass spectrometry (GC-MS) and high–performance liquid chromatography (HPLC) are analytical techniques used for PAH. HPLC with a fluorescence detector is a useful technique for PAH analysis. This analysis is performed with reversed-phase C₁₈ columns, which are specially developed for the analysis of PAH. The fluorescence detector gives a low detection limit for most PAH. Further, fluorescence detection is selective for PAH, because of their conjugated double bonds, which strongly adsorb light in the ultraviolet range. On the other hand, a gas chromatograph equipped with a capillary column gives high resolution among different PAH, and using mass spectroscopy as a detector reduces the risk for analytical interference.

The analysis should include PAH frequently found in air, which differ in volatility and reactivity. PAH with known biological effects should be included. The choice of the PAH also depends on the analytical method. The PAH measured with GC and HPLC are given in Table 4.5. In principle, HPLC is a good alternative to measure the population exposure of PAH, and GC-MS is the best method of estimating the sources.

Sample analysis for PAH is expensive and complicated. An especially careful protocol for sample collection and analysis as well as laboratory quality assurance and control procedures and interlaboratory harmonization are therefore necessary to assure data quality.

Monitoring and assessment strategy

The sampling sites should be located in the urban background air in order to measure the representative average exposure to PAH. In addition, PAH concentrations should also be measured at sites with increased concentrations, such as roads with heavy traffic and industrial areas. The PAH concentrations in urban air should be compared with the concentrations measured at background sites.

Table 4.5. Recommended analytical techniques for selected PAH according to the priority list of the US Environmental Protection Agency

PAH	GC is recommended	HPLC is recommended
Naphthalene	×	
Biphenyl	×	
Acentaphtylene	×	
Acenaphtene	×	
Fluorene	×	
Phenanthrene	×	×
Anthracene	×	×
Fluoranthene	×	×
Pyrene	×	×
Benso[a]anthracene	×	×
Chrysene	×	×
Benso[b]flouranthene	×	×
Benso[k]flouranthene	×	×
Benso[e]pyrene	×	
Benso[a]pyrene	×	×
Perylene	×	
Indeno[cd]pyrene	×	×
Dibenso[ah]anthracene	×	
Benso[ghi]perylene	×	×

The sampling frequency of PAH in a monitoring network is often limited by time-consuming and costly analysis. A weekly resolution of intermittent sampling may be sufficient for assessing long-term (annual) PAH concentrations, the most relevant for estimating health effects. However, a temporal resolution of less than 24 hours is preferred to estimate the sources, control traffic diversions and validate models. As PAH analysis is complex, other indicator pollutants emitted from the same sources as the PAH that are less costly to assess, such as CO and soot, may be more suitable for assessing air quality management. A long-term monitoring programme is important to allow trend studies.

Example

PAH concentrations are monitored within the Toxic Organic Micropollutants Survey(<http://www.aeat.co.uk/netcen/airqual/networks/tomps.html>, accessed 12 August 1999) in the United Kingdom. The measure-

ment in 1991 and 1992 was undertaken at four urban sites located at roof-top levels (25 m) in the centre of large cities (5). Both the gas and particle phases of PAH were collected using an air sampler equipped with polyurethane foam plugs and a glass fibre filter. The sampling was carried out weekly; the pump was timed to run 30 minutes every hour, and about 500 m³ was collected. The measurements showed seasonal variation both in concentration and in the gas versus particle distribution, with a greater share of PAH in the gas phases during the summer.

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Lead

Bohumil Kotlik

Health effects

Lead (Pb) toxicity can be explained by interactions with different enzymes, and that is why almost all organs can be considered as target organs for lead. A wide range of biological effects has been evidenced experimentally, including effects on haem biosynthesis, the nervous system, the kidneys, the reproductive organs, the cardiovascular system, the immune system, the liver, the endocrine system and the gastrointestinal tract.

For the purposes of assessing human exposure, the well established biomarker of exposure is the lead concentration in blood. The most sensitive people are young children, especially their nervous system. In this group, central nervous system effects, as assessed by neurobehavioural endpoints, appear to occur at blood concentrations below 200 µg/l. Consistent effects have been reported for global measures of cognitive functioning, such as psychometric IQ, to be associated with blood lead concentrations between 100 and 150 µg/l. Some epidemiological studies have indicated effects at blood lead concentrations below 100 µg/l. Based on this information and on the modelled relationship between blood lead concentrations and the long-term average concentration of lead in ambient air, the revised WHO air quality guidelines for Europe recommend that the annual average air lead concentration not exceed 0.5 µg/m³.

Sources and exposure patterns

The ambient concentration of lead in air shows regional and seasonal differences that depend on the dominant sources and their spatial distribution and on weather and dispersion patterns. Lead is present in outdoor air in the form of fine particles with a size distribution characterized by a mean aerodynamic diameter of less than 1 µm. The amount of lead bound in organic compounds (the unburned portions of tetraethyl and methyl lead) does not exceed 10%. Although lead is removed from the air via wet and dry deposition, mostly near the sources, tiny aerosol particles are involved in the long-range transport of this pollutant.

The main sources of lead are lead ore extraction and processing. Other sources are industrial production (lead is present as a secondary constituent in many minerals and sediments) and coal combustion (domestic combustion and heating and electricity plants). Motor vehicles (alkyl-lead additives in petrol) are an important mobile source of lead released into the air in countries where leaded petrol is used. As this source is near the population and widely distributed, road traffic is a major source of exposure.

Besides exposure through the air from point sources (industry), mobile sources (motor vehicles) and long-range transport of air pollution, the lead sedimented on the soil can contribute to total exposure, through food or through direct contact, especially in children. Another important exposure medium is paint containing lead or drinking-water delivered by lead-coated pipes.

Monitoring methods

Since lead in air is present as fine particles, its sample collection scheme follows the methods appropriate for particulate matter measurements described previously. The sampling time ranges from 24 hours to several days.

The lead concentration in the outdoor air is generally determined using either elemental analysis or non-destructive procedures on the particulate matter sample collected. The following sample collection systems are used, with the fraction to be studied defined by the sampling head determining the aerodynamic diameter of the particles entering the system (total suspended particulate, PM₁₀ or PM_{2.5}):

- systems based on sample aspiration through a membrane filter (acetyl-nitrocellulose, pore size 0.8 µm), as a rule, at a flow rate of 20 m³ per 24 hours;
- automatic analysers: beta-absorption techniques with filtration ribbon or gravimetric (tapered element oscillating microbalance) with bypass sampling and a sampling flow rate of 1–3 m³ per hours;
- special high-volume systems sampling at a flow rate over 1 m³ per minute;
- special sampling by personal or microenvironmental samplers dosimeters for suspended particles; and
- sampling of sedimented particles: interpretation of the data in terms of ambient lead concentration, which is difficult.

The collected samples must be subjected to pre-treatment procedures involving washing, microwave decomposition and chemical procedures aimed at decomposing acids under heating or basic melting.

For analytical determination of lead, the following methods can be used:

- non-destructive analytical procedures: X-ray fluorescence or particle-induced X-ray emission; or
- procedures for element analysis: atomic absorption spectrometry, inductively coupled plasma emission spectroscopy (ICP) or inductively coupled plasma mass spectroscopy (ICP-MS), differential pulse polarography with anodic stripping voltammetry, ultraviolet visible spectrophotometry and others.

The quality assurance and control protocol, besides implementing standard operating principles and good laboratory practice, should include interlaboratory proficiency testing. Commercial, national or in-house reference materials should be used, with lead concentrations near the ambient concentration ranges. The results should be periodically compared with those obtained by a method of known precision or with an independent method such as atomic absorption spectrometry with X-ray fluorescence or particle-induced X-ray emission.

Monitoring and assessment strategy

Long-term exposure to lead is relevant to health, and the spatial variation in lead concentrations depends on the long-term distribution of dominant sources. To recognize these spatial patterns, the active, pumped sampling systems can be complemented with passive samplers such as deposit gauges covering the whole population of interest. Emission inventory and modelling are an important part of the system for air quality monitoring, as they are useful for evaluating possible exposure to elevated lead concentrations.

Besides the monitoring aimed at assessing and controlling the impact of point (industrial) sources on population exposure, measuring the lead concentrations in the air may be of specific significance when lead is no longer used in petrol. Confirmation of the decline in lead concentrations in the ambient air of urban areas will demonstrate the effectiveness of the approach. Ambient air concentrations are expected to decline faster than the total exposure of the population because of possible exposure routes other than respiration and the persistence of sedimented lead in soil.

The blood lead concentration is a well established biomarker of the total exposure to lead in a recent period (approximately 6 months). This can be directly applied in assessing the health risk. However, lead concentrations must be monitored in ambient air as well as in other media (soil, indoor dust, drinking-water and food) to attribute the exposure to the main source. Ambient air monitoring also indicates the contribution of lead transport through air to the soil and (surface) water contamination and the resulting risk of indirect exposure.

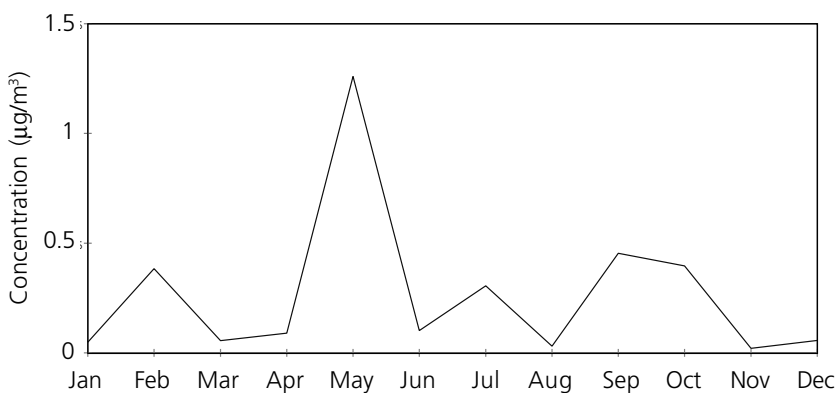
Example

The area around the lead smelter in Příbram, Czech Republic has been recognized to be heavily contaminated by lead. In the early 1970s, several

episodes of lead intoxication among livestock were reported in this area; thereafter, several epidemiological studies and ecological studies focused on the exposure of children. In contrast to earlier studies, an investigation from 1992 to 1994 revealed significantly lower exposure to lead (1). From 1986 to 1990, the recorded average blood lead concentrations were about 37.2 μg of lead per 100 ml in an elementary school population living in a neighbourhood within 3 km of the smelter. The 1992–1994 study (1), however, found mean blood lead concentrations of 11.3 $\mu\text{g}/\text{ml}$ (95% confidence interval = 9.3 to 13.8) among a comparable group of children. This study attempted to quantitatively assess the risk factors associated with elevated lead exposure in the Czech Republic. The concentration of lead in soil and in airborne particles, residential distance from the smelter, consumption of locally grown vegetables or fruits, drinking-water from local wells, the mother's educational level, cigarette consumption among family members and the number of children in the family were factors positively related ($P < 0.05$) to blood lead concentrations. The resulting blood lead concentrations were found to be inversely proportional to the child's age. In the 24-hour samples collected in 1993, the lead concentration in air ranged from 0.14 to 0.72 $\mu\text{g}/\text{m}^3$.

The monitoring results from 1996 (monthly means) are shown in Fig. 4.4. The enormous variation in monthly means indicates a need for

Fig. 4.4. Monthly mean ambient air concentration of lead in Příbram, Czech Republic, 1996^a



^a Annual mean = 0.30 $\mu\text{g}/\text{m}^3$.

long-term monitoring to assess the average concentration of lead in the long term, which is relevant for health impact assessment.

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Atmospheric cadmium

Laszlo Bozó

Effects on health

Cadmium (Cd), whether absorbed by inhalation or via contaminated food, may alter kidney functioning in various ways. However, the long-term exposure concentrations at which such effects can be expected exceed the concentration of cadmium in air of most urban or industrial areas by about 50 times. There is also sufficient evidence that cadmium can produce lung cancer in humans and animals exposed by inhalation, and the International Agency for Research on Cancer has classified cadmium as a class 1 human carcinogen, although no unit risk estimate can be derived from the existing data. In order to prevent any further increase of cadmium in agricultural soil likely to increase the dietary intake of cadmium by future generations, WHO has set an air quality guideline (revised) of 5 ng/m³ (annual average).

Sources and exposure patterns

Cadmium is emitted into the atmosphere by high-temperature processes such as non-ferrous metal industry and fossil fuel combustion in vapour form or very fine particles and usually disperses and mixes fairly quickly into the lower atmosphere. Within a few minutes or hours, the emitted gases condense into or adhere onto fine particles in the size range of 0.1 to 10 µm. Particles of this size are too fine to settle effectively and too coarse to be deposited by diffusion. Hence, they can be transported hundreds or more kilometres from their sources before gradually being removed from the atmosphere by dry or wet deposition. For this reason, cadmium pollution is a regional problem and not limited to the close vicinity of major emission sources.

The highest annual average concentrations (over 3 ng/m³) are estimated for Poland, Ukraine, northern Italy and the northern part of western Europe. Over the remaining rural parts of the European continent, the typical concentration range is 0.05–2.00 ng/m³ (1).

Monitoring methods

For sampling of aerosol particles, pumps of different capacity and various types of filters (Whatman 41, Teflon or Nuclepore) are recommended. The pump capacity should depend on the average ambient concentration of cadmium: remote areas far from huge sources require high-volume sampling (about 1000 m³/day) to reach the detection limit during the analysis. Monitoring sites in urban, rural or sub-industrial areas require only 20–30 m³/day. The type of filter required depends on the analytical methods used in the laboratory.

Washing the sample with any solvent destroys the sample, and the analysis cannot be repeated. Further, only species that are soluble in the specific solvent can be analysed in this way. This disadvantage is entirely avoided when non-destructive chemical methods are used that are capable of determining several elements simultaneously, independently of the solubility of the compounds. Another possible operation with samples is microwave decomposition or chemical procedures such as decomposing acids with heating or basic melting.

The procedures listed below are generally used to determine the composition of bulk aerosol samples, but they are also convenient for identifying the composition of individual particles.

X-ray fluorescence is based on the fact that samples irradiated by X-rays emit characteristic X-ray radiation: that is, each element emits X-rays at specific wavelengths. Their intensity is proportional to the quantity of the element present on the surface of the filter. Elements with an atomic weight higher than that of potassium can normally be analysed. The system can be automated if a continuous tape filter is used at the sampling site.

Neutron activation requires high-volume sampling. Aerosol samples are irradiated by neutrons, and the radiation of the radioactive elements generated is detected.

Graphite furnace atomic absorption spectrometry is a simple and relatively cheap method based on different elements absorbing light at different wavelengths.

Inductively coupled plasma emission spectroscopy is an expensive but very effective method for determining numerous elements during a single period of analysis.

Electrochemical methods include polarographic methods or anodic stripping voltammetry with cadmium-selective electrodes.

Monitoring and assessment strategy

Since the cadmium resides in the atmosphere for a few days, it can be transported several hundred kilometres from the emitting sources. This means that the relative contribution of long-range atmospheric transport to the ambient cadmium concentration is significant if the site is far from major cadmium sources. Thus, the concentration does not vary too much even in urban areas. However, temporal (daily and seasonal) variation could be significant because weather conditions vary, including wind speed, precipitation intensity, mixing height and stability.

As the spatial variation of cadmium concentrations near major sources is high, a network of aerosol samplers is recommended with a sampling period lower than 24 hours.

Cadmium concentrations are not directly related to road traffic sources, so the spatial variation in cities could be negligible. Depending on the size of the city, one or two sites seem to provide the necessary information for public health purposes.

High-volume sampling might be needed for rural background sites if the ambient concentration is low.

Atmospheric sampling periods over urban and background sites are controlled by practical considerations: although the annual average concentration needs to be measured, the most common strategy is to collect 24-hour samples in accordance with particulate matter monitoring.

Example

The highest cadmium concentrations occur near industrial sources emitting cadmium. The main thrust of a monitoring strategy in such areas is to pinpoint the effects of cadmium sources – as precisely as possible – on air quality and human exposure. Since weather conditions vary, the temporal variation of the cadmium concentration in air is substantial; this can be monitored by aerosol sampling periods of high temporal resolution.

In urban areas, the cadmium concentrations and spatial variation are relatively low, but a large population is exposed to the ambient cadmium load. As long-term exposure is relevant to health, the monitoring strategy should focus on long-term observation at selected sites with reliable and continuously controlled analytical techniques.

The National Survey of Air Pollution in the United Kingdom is an example of long-term monitoring of trace metal concentrations in ambient air. In central London, the annual mean concentrations of cadmium increased from 2.8 ng/m³ in 1984–1985 to 16.0 ng/m³ in 1988–1989. Then the concentrations decreased, reaching 0.4 ng/m³ in 1992–1993. Another study in the United Kingdom found mean cadmium concentrations of 2.0 ng/m³ in urban areas and 0.8 ng/m³ in rural areas not affected by local sources in 1986–1989. These values had declined by more than 60% compared with ten years earlier (2).

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Reporting and assessment

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A final, validated, air quality data set is of limited utility by itself. To be of use in health impact assessment, the data must be collated, analysed, interpreted and disseminated. This chapter deals with reporting, assessing and presenting air quality data to be used for health impact assessment.

Reporting

A minimum level of data evaluation is the production of annual and monthly data summaries. Simple methods of statistical and graphical analysis might be used.

Dissemination of validated air quality data represents a first step in the direction of health impact assessment by indicating a risk of health effects and by providing background information for decision-makers. Examples of such simple presentation are Fig. 4.1 or 4.2 in Chapter 4. Data communication may involve a number of transmission methods such as written reports or papers, floppy disks, cartridges or CD-ROM, broadcast media or direct database links. Within a harmonized data exchange system, such as within a national network, the data transferred may include annual validated data sets, processed summary and average statistics or analytical results, graphs and maps provided by geographical information systems. The methods of data dissemination are mentioned on pages 64–67.

Some forms of reporting are regulated by national (or international) legislation as standards, directives or conventions. They specify the subject, format and frequency of reporting. The amount by which a certain standard or threshold concentration established by the regulation is exceeded is often required to be reported. Reports from such compliance monitoring networks are of limited use for assessing population exposure and health effects. Certain health effects may be expected at

concentrations below the standard level, and these are not reported. In addition, the location of monitors in the compliance monitoring may be not optimal for assessing population exposure, and the reporting may give an incorrect picture of the exposure.

The European Environment Agency is developing the European Air Quality Information System (EUROAIRNET). This system aims at collecting and disseminating representative air quality information, including information on less polluted areas and on compounds that are not necessarily covered by European Union directives. The system is planned to extend beyond the 15 European Union countries and cover all of Europe. Information collected by the system is made available through the AIRBASE database mentioned on pages 64–67. (<http://www.etcaq.rivm.nl>, accessed 12 August 1999).

A WHO consultation in 1995 (1) proposed a format for reporting air quality data directly applicable to health impact assessment. Parts of the report on the consultation, including the format of data transfer, are presented in Annex 5. Following the consultation's recommendations, the WHO European Centre for Environment and Health has prepared a simple computerized tool for exposure data entry and analysis available on request: AIRQ. Besides allowing health impact estimates for individual populations to be estimated and graphically presented, the system allows the exchange of data and facilitates regional, national or multi-country analysis of the health effects of selected pollutants.

Assessment

One objective of an air quality network is to estimate statistical parameters such as annual arithmetic or geometric means, medians and percentiles, in each of the locations where the monitoring is conducted and to compare the estimates with the air quality standards or guidelines. Additional possibilities for data assessment include frequency distribution, histograms and (cumulative) frequency curves. For health impact assessment, the air quality information must be linked with the population subject to the exposure.

Knowledge of long-term (annual) mean pollution concentrations may be sufficient for assessing long-term health effects. Assessing short-term health effects requires more regular reporting, even daily or

hourly, and the availability of suitable data for this purpose depends on the network objectives. The desired method of summary presentation of data in a single population is the frequency distribution of mean values calculated over the averaging time of health relevance. For example, the frequency distribution of 24-hour average PM_{10} concentrations observed over 1 year allows the increase in the incidence of acute health outcomes (such as acute respiratory symptoms or hospital admissions) to be estimated in the population subject to the pollution on or after days with certain PM_{10} concentrations (1).

An example of data aggregated in the form of such a frequency distribution is shown on pages 24–27.

If air quality is monitored in several locations in one community, population exposure may be best approximated by the mean of observations from these locations. If appropriate data are available, the exposure estimate may be improved by calculating the weighted mean, with the weights determined by the population density near each of the monitors and the proportion of time people spend in each of the locations. The frequency distributions of such calculated means provide a population exposure profile, which can be used in analysing the acute health effects of exposure. Such a distribution summarizes the concentration data as information on the frequency of periods (such as the number of days) with certain concentrations and as the size of the population subject to certain pollution concentrations for certain time periods (Box 5.1).

Analysis of data collected by an air quality monitoring network also provides important input to reviewing and updating the system. The accumulated information indicates whether the spatial density of monitoring can be reduced or whether the changing composition of the pollution requires changes in the pollutants to be monitored.

Formats for presenting information

Collected data need to be presented to be used. A narrow group of experts dealing with air quality monitoring or modelling assessment can use unprocessed data, but not decision-makers or the general public. Interpretation of the temporal and spatial variation of the pollution is greatly facilitated when the numbers are converted to graphs or figures.

Box 5.1. Example of calculating a population exposure profile for ozone in the 15 European Union countries

According to European Council directive 92/72/EEC (2), the European Union countries report information on the 8-hour mean ozone concentration exceeding $110 \mu\text{g}/\text{m}^3$ to the European Union. The data from March to October 1995 were used for this analysis. It was assumed that the ozone concentration measured in each of the 400 monitoring locations is spatially representative for a circle with a 10-km radius. Data on population density in Europe and geographical information system techniques were used to calculate the number of people living in such circles around each of the monitors.

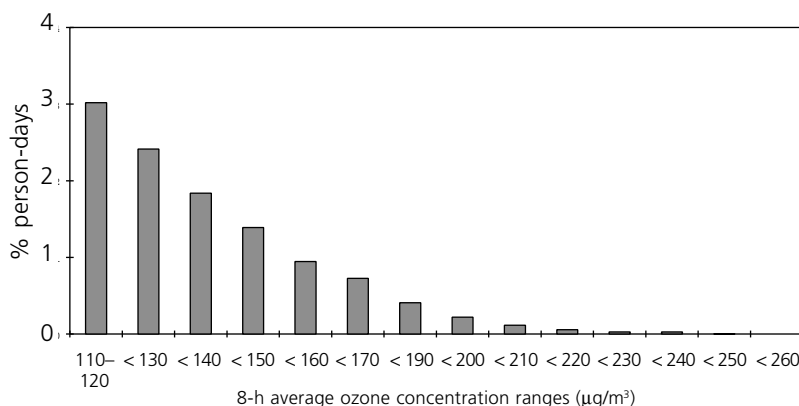
About 11.7% of the total population of the European Union live in such circles; the proportion ranges from 0.3% in France to 35.4% in Germany. The reported ozone monitoring data were used to calculate the frequency distribution of ozone concentration (over the $110 \mu\text{g}/\text{m}^3$ threshold). For each station, multiplying the number of days in a given category of concentration by the number of people in the circle surrounding the monitoring location estimates the number of person-days at a given concentration category. The concentration-specific person-days accumulated over all monitored populations provide a frequency distribution of the person-days in a country or in all 15 countries (Fig. 5.1). For all 15 countries, the $110 \mu\text{g}/\text{m}^3$ threshold value was exceeded for 11.2% of the person-days in summer 1995, and the level of $160 \mu\text{g}/\text{m}^3$ was exceeded for 1.6% of the person-days.

This frequency distribution is the input to the health impact analysis according to the methods presented on pages 24–27. The critical health outcome used in the analysis was emergency hospital admissions for respiratory diseases, for which the risk increases 1.043 times for each increase by $50 \mu\text{g}/\text{m}^3$ in 8-hour O_3 concentration. Some 700 emergency admissions were estimated to be attributable to the exposure in all 15 countries in summer 1995.

One limitation of this analysis is the fact that only values exceeding the $110 \mu\text{g}/\text{m}^3$ threshold value were reported. An increased risk of hospitalization exists even below this concentration.

The simplest presentation method involves tabular presentation of summary statistics calculated from the concentration measurements obtained by different monitoring stations during the reporting period. The relevant statistics include:

Fig. 5.1. Distribution of person-days by ozone concentration for the 15 European Union countries, March–October 1998



- arithmetic and geometric means;
- minimum and maximum concentrations (with the health-relevant averaging time);
- frequency by which the (specified) standard level is exceeded;
- selected percentiles (such as 95th or 98th percentiles); and
- data capture statistics, such as the number (or percentage) of days on which the daily measurements are available if 24-hour averaging time is of health relevance.

Tables may also contain the frequency distribution of measurement or the mean values calculated from several monitors operating in a community. The WHO consultation (1) recommended such a format for presenting and exchanging data; this is presented in Annex 5 and illustrated in Box 5.1.

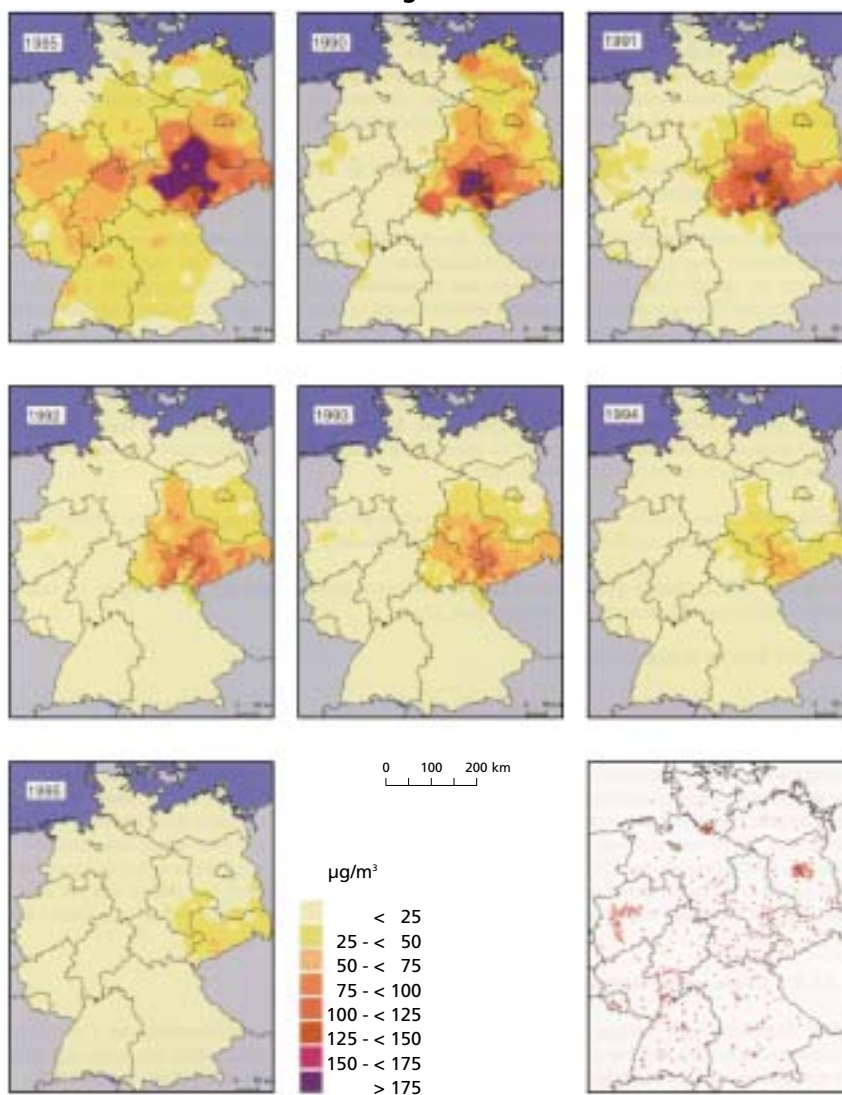
Tabular presentation of the data is not easy to interpret, but appropriately constructed tables may provide numerical data for further analysis. Information from tables transformed into graphs is easier to interpret, but further inference or data combination is more difficult and less precise.

The most frequently used graphs are:

- time series of measurements or their short-term means from one monitoring location (for one or more pollutants) or comparing various locations;
- bar charts or line graphs presenting long-term (annual) means over a longer period;
- the cumulative distribution of short-term means over a longer period;
- the distribution of person-days by concentration category;
- source apportionment pies; and
- wind roses.

More complicated spatial comparisons of pollution patterns are greatly facilitated by the use of maps. The (mean) pollution concentrations in various locations can be presented in a simple manner, such as dots with varying colour or size on the map of a city, country or continent (3–5). Modelling results can also be based on interpolation of monitoring data with geographical information system techniques (6–8). Estimates from pollution transport models are routinely presented on maps with varying grid size. The estimates obtained by the long-range transport models for continental scales and large grid sizes (150×150 km) are not very useful for health impact assessment, but more specific, local models may be sufficiently precise to provide valuable information. Mapped air quality data combined with population density in various geographical areas may be used to calculate population-weighted exposure indicators and be an important input to assessing health effects.

Fig. 5.2 presents an example of the use of maps for presentation of air quality data from a large monitoring network in Germany (9). The data presented on the maps are based on measurements from the air quality monitoring networks of 16 federal states (*Länder*) and the German Federal Environmental Agency. Annual average SO_2 concentrations were interpolated using a model that calculates the weighted mean concentration for each 2.5×2.5 km grid cell covering all of Germany. The weights are based on the nearest-neighbour function and the distance from the grid cell to the nearest monitoring site. Data from monitoring stations with an extreme location, such as hot spots related to road traffic or the top of a mountain, are not considered.

Fig. 5.2

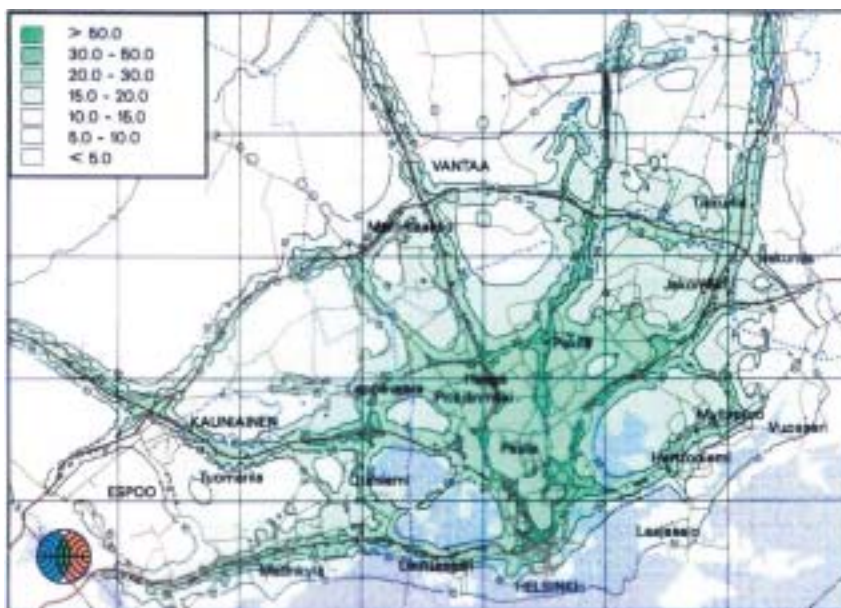
Source: German Federal Environmental Agency. *Daten zur Umwelt – Der Zustand der Umwelt in Deutschland*. Berlin, Erich Schmidt Verlag, 1997, p. 151

An interesting aspect of this presentation is its ability to demonstrate changes in the pollution concentration both in space and time. Combining such maps with the population density map may allow a similar presentation of the potential distribution of population exposure.

Fig. 5.3 presents an example of air quality on a smaller scale (25×25 km). This provides the model calculations for NO_2 concentration in the Helsinki metropolitan area in 1993.

The estimated NO_2 concentrations were obtained by using a modelling system containing the following models: the EMME/2 interactive graphic transport planning system, the computation of emissions using the EMME/2 and LIISA systems, a dispersion model for stationary sources, UDM-FMI (the urban dispersion modelling system of the Finnish Meteorological Institute (11)) and one for mobile sources, CAR-FMI (a model of the Finnish Meteorological Institute (12) for evaluating the dispersion of contaminants in the air from a road network) and a meteorological pre-processing model. The programs were executed on a Cray C94 supercomputer. The estimated concentrations of NO_2 are strongly increased in the vicinity of the main roads and streets and in central Helsinki. The figure shows the distinct influence

Fig. 5.3



Source: *Air quality in the Helsinki Metropolitan Area. Concentrations, emissions and trends.* Helsinki, Helsinki Metropolitan Area Council (YTV), 1998, p. 9

of the ring roads, situated at distances of about 8 km and 15 km from the city centre, and of the intersections of major roads and streets.

Storing data

Data should be stored in an electronic form in a protected database, such as on an optical disc. The database should be user friendly, easy to handle, flexible and open-ended with respect to the number of compounds monitored at different sites. Modern relational database software can be used and run on a personal computer. Two types of databases should be distinguished:

- a database with the validated raw data, such as 30-minute, 1-hour or 24-hour values; and
- a database with aggregated data for summary statistics such as the WHO Air Management Information System (AMIS).

Clear and fundamental documentation is necessary for storing both types of databases to enable further investigations.

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Conclusions and recommendations

Discussion in the meetings of the Working Group and the text presented in the previous chapters leads to the following conclusions and recommendations.

1. One of the main objectives of air quality management is to prevent effects on human health resulting from excessive exposure to ambient air pollution. Therefore, air quality monitoring networks should be able to provide information suitable for assessing exposure and health effects.
2. Air quality assessment based on a versatile set of tools – monitoring and models – is necessary for assessing population exposure to air pollution. Exposure assessment is, in turn, necessary for health impact assessment.
3. The pollutants studied, measurement time scales and locations should be relevant to the assessment of population exposure. Air quality monitoring should consider both the hot spot exposure, where a small part of the population is exposed to high concentrations, as well as the average concentrations to which most of the population is exposed.
4. When the population exposure is being assessed using the results of an air monitoring network, the population should be assigned to fields of ambient air concentration measured by individual monitoring stations. The information on the number of people exposed should be combined with the measured concentration. Reliable demographic data are also necessary for this purpose.
5. Availability of time–activity data for various population groups may further improve exposure estimates obtained by combining population density with data on ambient air pollution concentrations.

6. Monitoring systems should be designed based on clearly defined objectives and data quality requirements.
7. Monitoring programmes need to be cost-effective, sustainable and applicable to national and local priority needs and conditions. Always use the simplest technologies and procedures that are consistent with fulfilling overall monitoring objectives.
8. Comprehensive quality assurance and control of air monitoring programmes is essential to ensure that measurements are accurate, reliable and fit for the intended purpose. National and international harmonization of measurement quality should be fostered through national quality assurance and control coordination, laboratory accreditation and international validation programmes.
9. Raw air quality data are of limited utility; these need to be transformed by appropriate validation, analysis and interpretation into useful information that targets the needs of scientific, policy, planning, health and public end-users. This information should be rapidly communicated and should be easily available.
10. Monitoring is only one of a range of tools in air quality assessment; monitoring, emission inventories and predictive models are best regarded as complementary components of any integrated approach to assessing exposure and health effects and managing risk.

Major European activities related to air quality assessment

Annex 1.1. Overview of the updated WHO air quality guidelines for Europe

Rolaf van Leeuwen & Michal Krzyzanowski

Introduction

WHO has been concerned with air pollution and its impact on human health for more than 40 years. The first WHO conference on the public health aspects of air pollution in Europe was held in Milan in 1957 and was followed by several activities of the Regional Office for Europe in this field, all aiming at promoting international cooperation and harmonizing environmental policies in the European Region. These activities culminated in the development of the WHO air quality guidelines for Europe.

The first edition of the *WHO air quality guidelines for Europe* was published in 1987. This publication evaluated the health risk for 27 pollutants. Since then, new scientific data in air pollution toxicology and epidemiology have emerged and new methods of assessing risk have been developed. These developments have necessitated updating and revising the existing guidelines. The Bilthoven Division of the WHO European Centre for Environment and Health has undertaken the process of amending, updating and extending the existing guidelines. This process was carried out in close cooperation with the International Programme on Chemical Safety (IPCS) and the European Commission, Directorate-General for Environment, Nuclear Safety and Civil Protection.

The process of the update started in January 1993 with a planning meeting (1). WHO working groups on the following topics met to prepare the revised version of the guidelines:

- methodology and format, September 1993 (2)
- ecotoxic effects, September 1994 (3)
- “classical” air pollutants, October 1994 (4)
- inorganic air pollutants, October 1994 (5)
- certain indoor air pollutants, March 1995 (6)
- PCBs, dioxins and furans, May 1995 (7)
- volatile organic pollutants, October 1995 (8)
- “classical” air pollutants – second meeting, June 1996
- final consultation, October 1996.

In addition to these meetings determining air quality guidelines, a working group has prepared guidance on setting standards with regards to air quality (May 1997), which constitutes a part of the updated air quality guidelines.

Aim and scope of the WHO air quality guidelines for Europe

As in the first edition, the aim of the guidelines is to provide a basis for protecting the public health from adverse effects of environmental pollutants and eliminating or reducing to a minimum the exposure to the pollutants that are known or likely to be hazardous to human health or wellbeing. Although health effects were the major consideration in establishing the guidelines, ecologically based guidelines for preventing adverse effects on terrestrial vegetation were also considered, and guideline values for nitrogen and sulfur oxides and ozone have been established to protect vegetation.

The guidelines are intended to provide background information and guidance to national or international authorities in assessing risk and deciding how to manage it. The guidelines provide pollutant concentrations below which exposure for a lifetime or for a given period of time does not constitute a substantial public health risk and therefore form a basis for setting international and national standards or limit values for air pollutants.

In general, the guidelines address single pollutants, whereas in real life people are exposed to mixtures of chemicals that have additive, synergistic or antagonistic effects. In some cases, the updated guidelines have addressed these mixtures directly: environmental tobacco smoke and man-made vitreous fibres. The guidelines do not differentiate between exposure in indoor versus outdoor air because, although the site of exposure determines the type and concentration of air pollutants, it does not directly affect the exposure-response relationship.

Although the WHO air quality guidelines for Europe are considered to protect human health, they are maximum values, and air pollution levels should be kept as low as practically achievable. For some of the pollutants (such as PM₁₀, radon or benzene), no guideline value is provided; instead, a value (or values) of risk (or risks of different health outcomes) per unit concentration is given. This allows the risk to health of a certain exposure to be estimated.

The guidelines are based on health effects or environmental effects and are not standard per se. Prevailing exposure levels, technical feasibility, source control measures, abatement strategies as well as social, economic and cultural conditions must be considered in setting legally binding standards. Thus, international or national standards may be above or below the health-based WHO air quality guidelines for Europe. These aspects will be discussed in the second edition of the publication on the WHO guidelines.

Updated air quality guidelines for Europe

The guideline values and corresponding averaging times resulting from the revision process are summarized in Tables A1.1 and A1.2. As mentioned above, some pollutants have no guideline value. The reasons were different for different pollutants. If the information on exposure and response is available, it can be used to assess the risk, and its acceptability, in a population. Environmental tobacco smoke is associated with serious health effects at typical environmental exposure concentrations. For a group of chloroorganic pollutants (PCBs, PCDDs and PCDFs), direct health risk via inhalation is negligible in comparison with the total (oral) exposure to these compounds. For fluoride, it was concluded that concentrations that would protect livestock and plants would also protect human health adequately. For platinum, the derivation of a guideline value was not deemed necessary because ambient air concentrations

Table A1.1. Summary of the revised WHO air quality guidelines for Europe

Substance	Guideline value	Averaging time
Classical air pollutants		
Carbon monoxide	100 mg/m ³	15 minutes
	60 mg/m ³	30 minutes
	30 mg/m ³	1 hour
	10 mg/m ³	8 hours
Ozone	120 µg/m ³	8 hours
Nitrogen dioxide	200 µg/m ³	1 hour
	40 µg/m ³	1 year
Sulfur dioxide	500 µg/m ³	10 minutes
	125 µg/m ³	24 hours
	50 µg/m ³	1 year
Particulate matter (see Table A1.2)	exposure–response	24 hours
Indoor air		
Man-made vitreous fibres (refractory ceramic fibres)	1×10 ⁻⁶ (fibre/litre) ⁻¹	unit risk/lifetime
Radon	3–6×10 ⁻⁵ Bq/m ³	unit risk/lifetime
Environmental tobacco smoke	no guideline ^a	
Organic pollutants		
Benzene	6×10 ⁻⁶ (µg/m ³) ⁻¹	unit risk/lifetime
1,3-Butadiene	no guideline ^a	
Dichloromethane	3 mg/m ³	24 hours
Formaldehyde	0.1 mg/m ³	30 minutes
Polycyclic aromatic hydrocarbons (benzo[a]pyrene)	8.7×10 ⁻⁵ (ng/m ³) ⁻¹	unit risk/lifetime
Styrene	0.26 mg/m ³	1 week
Tetrachloroethylene	0.25 mg/m ³	24 hours
Toluene	0.26 mg/m ³	1 week
Trichloroethylene	4.3×10 ⁻⁷ (µg/m ³) ⁻¹	unit risk/lifetime
PCBs, PCDDs, PCDFs	no guideline ^a	
Inorganic pollutants		
Arsenic	1.5×10 ⁻³ (µg/m ³) ⁻¹	unit risk/lifetime
Cadmium	5 ng/m ³	1 year
Chromium (chromium (VI))	4×10 ⁻² (µg/m ³) ⁻¹	unit risk/lifetime
Fluoride	no guideline ^a	
Lead	0.5 µg/m ³	1 year
Manganese	0.15 µg/m ³	1 year
Mercury	1.0 µg/m ³	1 year
Nickel	3.8×10 ⁻⁴ (µg/m ³) ⁻¹	unit risk/lifetime
Platinum	no guideline ^a	

^a See text.

Table A1.1. (continued)

Substance	Guideline value	Averaging time
Ecotoxic effects		
SO ₂ critical level	10–30 µg/m ³ ^b	1 year
critical load	250–1500 eq per hectare per year ^c	
NO _x critical level (NO + NO ₂ , expressed as NO ₂)	30 µg/m ³	1 year
critical load	15–35 kg of N per hectare per year ^c	
Ozone critical level	0.2–10 ppm · h ^{b,d}	5 days to 6 months

b Depending on the type of vegetation.

c Depending on the type of soil and ecosystem.

d Accumulated exposure over a threshold of 40 ppb (daylight hours).

The unit risk is the excess risk of dying from cancer following lifetime exposure. For instance, for benzene six people in a population of 1 million will die when they are exposed for their lifetime to a concentration of 1 µg/m³, for polycyclic aromatic hydrocarbons, the unit risk means that 87 people in a population of 1 million will die from cancer following lifetime exposure to 1 ng/m³.

**Table A1.2. Updated WHO air quality guidelines for Europe:
summary of relative risk estimates for selected health
outcomes associated with a 10 µg/m³ increase
in the 24-hour average concentration of PM₁₀ or PM_{2.5}**

Endpoint	Relative risk for PM _{2.5} (95% con- fidence interval)	Relative risk for PM ₁₀ (95% con- fidence interval)
Bronchodilator use		1.0305 (1.0201–1.0410)
Cough		1.0356 (1.0197–1.0518)
Lower respiratory symptoms		1.0324 (1.0185–1.0464)
Change in peak expiratory flow rate (m ³ /kg) relative to mean		–0.13% (–0.17% to –0.09%)
Respiratory hospital admissions		1.0080 (1.0048–1.0112)
Mortality	1.015 (1.011–1.019)	1.0074 (1.0062–1.0086)

are at least three orders below the concentrations that induce sensitization reactions in a sensitive part of the population. No guideline was established for 1,3-butadiene, because cancer risk estimates in various species varied widely and no conclusion could be made as to which species should be used for human risk estimates.

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Annex 1.2. European Union framework directive and daughter directives

Frank de Leeuw

The framework directive on ambient air quality assessment and management of the European Union (96/62/EC) (1) sets a general framework for air quality measurement and assessment in the 15 European Union countries. The framework directive requires air quality limit values to be set in pollutant-specific daughter directives. According to the framework directive, measurements will be mandatory in the following cases:

- agglomerations, defined as a zone with a population concentration in excess of 250 000 inhabitants or a sufficiently high population density per km² to justify, in the opinion of the Member States, the need for ambient air quality to be assessed and managed;
- in zones with concentrations exceeding $x\%$ of the limit value, where x depends on the pollutant and is specified in the daughter directive; and
- other zones with concentrations above the limit value.

The measurements must be taken at fixed sites, either continuously or by random sampling, and the number of measurements must be sufficiently large to enable the levels observed to be determined. For air pollution approaching the limit value (that is, when concentrations are below $x\%$ of the limit value) a combination of measurements and modelling techniques may be used. At low concentrations (below $y\%$ of the limit value, where y depends on the pollutant and is specified in the daughter directive), modelling or objective estimation techniques may be used alone. For example, the $x = 60\%$ and $y = 40\%$ for SO₂.

European Union countries will have to inform the European Commission of the occurrence of levels exceeding the limit value, including the reasons, within 9 months after the end of each year. The countries must also annually forward a list of zones and agglomerations in which

air quality exceeds the limit values. The Commission must publish these lists annually.

Position papers are produced by working groups in preparing the daughter directives. These papers give recommendations for the limit values as well as the component-specific monitoring strategy. Criteria are specified for the location and the minimum number of sampling points and the reference measurement and sampling techniques. Recommendations for quality assurance and control are given. The position papers form the basis for the daughter directives prepared by the Commission. In 1997 and 1998, draft position papers were prepared for SO₂, NO₂, particulate matter and lead, benzene and CO and ozone. Preparatory work has also started for polycyclic aromatic hydrocarbons, cadmium, arsenic, nickel and mercury. The directive on SO₂, NO₂, particulate matter and lead was adopted in April 1999 (2).

The Commission requested that the European Environment Agency and the European Commission Joint Research Centre help to develop guidance on supplementary assessment in support of the requirements set in the framework directive. A guidance report (3) provides criteria, procedures and methods that lead to the most cost-effective implementation of monitoring obligations and to siting the monitors optimally to assure proper representation of the air quality in the entire zone for four assessment cases:

- preliminary assessment: when no representative measurements are available in zones or agglomerations, countries must undertake preliminary assessment to have data available in time for the implementation of the daughter directive;
- assessment supporting further optimization of station siting for (mandatory) measurements;
- assessment supporting generalization of mandatory measurements; and
- evaluation of existing assessment.

Three assessment methods or tools should be used in combination for the case mentioned above:

- preliminary air quality measurements
- air emission inventories
- air pollution modelling.

The guidance report (3) outlines the assessment procedures and assessment methods mentioned above.

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General methods for emission inventories

Alexander Economopoulos

Introduction

Reliable assessment of the air emissions generated by each source, or by groups of similar sources, in a study area is important in identifying the nature, magnitude and origin of the existing ambient pollution problems and in formulating rational pollution abatement strategies.

As each urban area has numerous pollution-generating activities, dealing with all sources may not be practical or even required. Indeed, most activities cause some pollution, but relatively few types of sources are responsible for the bulk of the emissions. An effective source inventory can be thus organized as follows:

- assessing on an individual basis the emissions from all large to medium-sized sources;
- assessing on a collective basis the emissions from groups of similar small sources with significant combined contribution to the total emissions, typically including space heating boilers, road vehicles, dry-cleaning shops and petrol stations; and
- ignoring all other sources with negligible combined contributions to the emissions (most economic activities).

Source inventory methods should provide guidance in organizing effective inventories and in assessing emissions. Appropriate methods for this purpose are rapid assessment, the simulation of sources and associated control systems and the direct monitoring of emissions. The former two methods are termed computational and are characterized by their ability to estimate both the current situation and alternative target situations that planners may wish to consider. Direct monitoring programmes yield valuable primary inventory data but can only deal with

the existing situation and are relatively difficult to implement, as they are expensive.

These methods are briefly described below. A detailed description, along with the required implementation models, can be found elsewhere (1). Goodwin et al. (2) produced an inventory for the entire United Kingdom by integrating top-down and bottom-up approaches, and Hutchinson & Clewley (3) prepared an inventory using a bottom-up approach on a city scale.

Rapid assessment

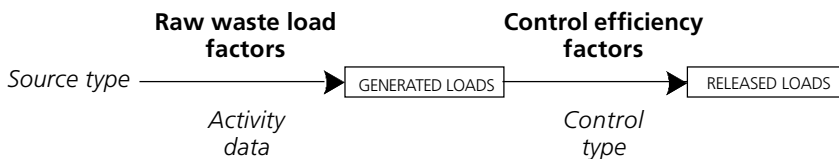
Rapid assessment is effective in assessing the emissions from a very wide spectrum of sources and the efficiency of alternative pollution control options.

This method is based on the documented and often extensive experience about the nature and quantities of pollutants generated from each kind of source, with and without associated control systems. As Fig. A2.1 illustrates, it makes constant use of this experience for predicting the anticipated loads from a given source.

Planners can thus compile comprehensive inventories for a large metropolitan area and analyse emission control scenarios rapidly and with modest resources. The results, however, are statistically valid, as the normalized emissions among similar sources vary significantly.

The released loads from a given source are calculated based on the use of appropriate emission factors, derived from the measured perform-

Fig. A2.1. Illustration of the rapid assessment approach for estimating the pollution loads on air, water and land



Bold: model components

Italics: field data

ance of similar sources. Each factor, e_j , is defined as the normalized emission of pollutant j per unit activity of the source under consideration. In principle, each factor can be expressed in the following mathematical form, as a function of several parameters:

$e_j = f'$ (source type, process or design particularities, source age and technological sophistication, source maintenance and operating practices, type and quality of the raw materials used, type, design and age of the control systems employed, type and design of control systems employed in other media, ambient conditions and other parameters)

The dependence of the waste factors e_j on parameters such as these included in the equation cannot be expressed as a continuous function because of the discrete nature of most parameters (such as the type of control systems employed) and the lack of sufficient information in relation to the remaining ones. A discrete functional form yielding a series of waste factor values, each valid under a specific set of common and important parameter combinations, is used instead.

This leads into a tabular construct of a comprehensive model for air emission inventory and control (1). This model introduces the impact of most major parameters into the assessment of the released loads while precisely defining the data requirements from the field surveys.

Modelling of pollution source and control systems

The use of mathematical models that simulate the performance of certain sources and their attached control systems constitutes the most advanced method for reliably assessing not only the current emissions but also the impact of possible modifications to design and operation.

The disadvantage of the modelling approach is the practical difficulty in developing such models for the great variety of existing sources and control systems. In addition, the data-processing requirements become fairly demanding, and manual implementation is subject to practical limitations. Nevertheless, selective applications offer significant improvements, and the development of graphical solution methods allow convenient manual application. Graphical solution tools for the following source and control systems are available (1):

- internal combustion sources (emissions and fuel consumption from motor vehicles);
- external combustion sources (flue gas volume for most fuel types);
- stacks (gas temperature drop);
- cyclone sampler (overall efficiency and outlet particle distribution);
- Venturi Scrubber (overall efficiency and outlet particle distribution); and
- electrostatic precipitators (overall efficiency).

The present approach extends and improves the rapid assessment method and can be complemented by the latter. Moreover, as it is computational in nature, it can be used to analyse the current as well as any target situation. It is thus suitable for environmental strategy purposes.

Direct waste measurement programmes

The direct monitoring of the waste sources through sampling and analysis is a rather obvious approach. This method is indispensable in many cases, especially when the waste discharges from large sources need to be monitored closely or when environmental services need to verify compliance with the applicable liquid effluent and air emission standards.

The measurement requirements for source inventory programmes are generally more relaxed than those for enforcement purposes. Thus, lighter and less expensive instruments can be used that are easier to transport and maintain and faster to operate. Additional parameters might have to be considered, and measured concentrations always have to be combined with waste volumes.

Source monitoring can improve the accuracy of inventories and should be pursued. However, significant difficulties are often encountered, such as variability of emissions, a lack of suitable sampling locations, high temperatures and noxious gases. Moreover, diffused or distributed sources are not easily measured and the number of individual sources is often excessive. Even for the sources that can be monitored, priorities must be set to cover the important sources. As the largest sources often account for the bulk of the released loads, accurate monitoring of their wastes contributes substantially to the accuracy of the overall inventory programme.

Despite the rational shortcuts that may and should be taken, source inventories based entirely on direct waste measurement tend to be time consuming and resource intensive, often cannot cover all sources and can only deal with the existing situation. Thus, regularly updating the inventory data tends to be impractical. Moreover, the inventory information has limited usefulness in planning, as the implications of alternative measures that planners wish to consider cannot be measured.

Combined approaches

Inventory programmes that can combine computational with direct measurement methods can be organized to produce the most credible results at reasonable cost. To achieve this, the predictions from the computational methods can be used to set priorities for the source monitoring programme, and the monitoring results can be used to verify and calibrate the predictions from the computational models. The computational models can then be used to compile credible inventories and to analyse control strategies.

More specifically, the computational source inventory procedures can provide information about the nature (polluting parameters of major interest) and the magnitude of the polluting loads released from each source. In most situations a few large sources account for the bulk of the released loads, and these can be easily identified and thus monitored.

Waste measurement programmes can be organized to have the maximum impact on the validation of computational methods. For this purpose, process and operating data need to be carefully documented, some extra measurements are required to produce the pollutant loads prior and after the treatment installation and typical plants from all major source and size categories need to be inspected. Representative sources need to be monitored for each important source type so that the computational models can be calibrated.

This combination offers very distinct advantages. The direct measurement programme, which is inherently time consuming and particularly costly, can be limited in extent since it does not have to be exhaustive. It is required to cover only a representative sample from the major source categories. Once the predictions of the computational methods are validated through the above information, their use can be extended to safely cover all sources under current as well as strategy

target conditions, and credible updates can be generated regularly. This way, the impact of the costly waste measurement programme can be long lasting and perpetuated through the use of the locally validated computational methods.

Independent execution of waste measurement programmes, only for inventory purposes, tends to be significantly more expensive. The programmes have to be exhaustive, they are of limited use in management as the data generated cannot be used for planning purposes, and their impact is short lived as new costly programmes have to be repeated every few years. Combined execution of waste measurement and computational programmes overcomes all these disadvantages, providing the best rationale for the waste measurement.

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Air quality modelling

Annex 3.1. Dispersion models

Jaakko Kukkonen

Introduction

Air pollution models are valuable tools for regulatory purposes, policy-making and research applications. The most efficient method in air pollution research is often the combined use of measurements and modelling. This section reviews atmospheric dispersion models, including the main objectives, the physical and chemical requirements of the models and their characterization for dispersion models at the local, regional and continental scales.

Atmospheric dispersion models can be used for a wide variety of purposes such as:

- establishing source-receptor relationships;
- evaluating the contribution to concentrations from various sources;
- estimating the distribution of spatial concentrations and population exposure to pollution;
- optimizing emission reduction strategies and analysing emission scenarios;
- predicting the concentrations over time;
- analysing the representativity of measurement stations; and
- as tools in research.

Many of these tasks cannot be accomplished using only air quality measurements.

The models require meteorological and geographical information together with source and emission data. Errors in the predictions of the models are caused by (1):

- inaccuracy in estimating the model input values;
- deficiencies in modelling the physical and chemical phenomena;
- numerical inaccuracies of the models; and
- random variability in the atmosphere.

Controlling the inaccuracy of the modelling methods requires subjecting models to continual quality assurance and quality control procedures. High priority should be given to evaluating models and validating them against high-quality databases.

This section attempts to present a general overview of dispersion models and therefore does not refer to specific models. Moussiopoulos et al. (2) have reviewed air pollution models, including brief reviews of selected individual models.

Classification of air pollution models

Zannetti (3) has classified air pollution models. The models include:

- 1) Eulerian models, which solve numerically the atmospheric diffusion equation;
- 2) Gaussian models, in which the concentration distribution is Gaussian in both the horizontal and vertical directions;
- 3) Lagrangian models, which either consider processes in a moving air mass or use fictitious particles to simulate dispersion processes;
- 4) semi-empirical models, which are mainly based on empirical parametrization;
- 5) stochastic models, which are based on semiempirical or statistical methods and seek to analyse the relationships of air quality and atmospheric measurements or to forecast air pollution episodes; and
- 6) receptor models, which consider the observed concentrations at a receptor point and attempt to apportion the contributions from various sources.

Clearly, these are only basic characteristic model types. For instance, both Eulerian and Lagrangian models include usually submodels for dry and wet deposition, chemical transformation and other processes. Gaussian models can also include submodels for, for instance, plume rise and downwash, deposition processes and chemical transformation.

Eulerian and Lagrangian models have been traditionally defined as describing processes relative to a fixed coordinate system and a moving fluid, respectively (4). This would classify, for instance, Gaussian models and most semi-empirical models to be sub-groups of Eulerian models. However, the application of these terms in air pollution commonly follows the above characterization.

The above list addresses air pollution models. Atmospheric dispersion models describe the turbulent diffusion processes in the atmosphere, and this includes categories 1–3 and partly category 4 in this classification. Dispersion models can be classified according to the scales of atmospheric processes:

- macro scale (length scale ≥ 1000 km), in which atmospheric flow is associated with synoptic phenomena;
- meso scale ($1 \text{ km} < \text{length scale} < 1000 \text{ km}$), in which air flow depends partly on synoptic phenomena and partly on hydrodynamic effects (such as surface roughness and obstacles) together with energy balance inhomogeneities; and
- micro scale (length scale $\leq 1 \text{ km}$), in which air flow depends largely on the surface characteristics.

Alternatively, the dispersion models could be classified as local (time scale less than a few minutes), local to regional (several hours), regional to continental (several days) and continental to global (weeks or more), in accordance with Moussiopoulos et al. (2). The following sections summarize the model classes.

Dispersion models for various spatial scales

This review presents, for dispersion models at the local, regional and continental scales:

- the main objectives of the modelling activities;
- the physical and chemical requirements of the models; and
- the characterization of commonly used models.

Air pollution models have also been developed for the global scale; these attempt to describe the changes of the chemical and physical composition of the global atmosphere.

Models of dispersion at a local scale. The main objective of models of dispersion at a local scale is quantifying the concentrations of pollutants that can cause adverse health effects for the population. In some cases, the objectives also include the deposition of pollutants and the influence of air pollution on vegetation. The models analysing the dispersion of hazardous materials from chemical accidents can also be classified mainly as local-scale models. Most of the local-scale models have been developed for regulatory purposes.

Models based on Gaussian concentration distributions have been very widely used for regulatory purposes historically. Traditionally, these models have been based on Pasquill-Gifford stability categories, and the dispersion parametrization has been very straightforward. However, the models should be able to allow for the structure of the atmospheric boundary layer and the various local-scale effects, such as the influence of buildings and obstacles, downwash phenomena and plume rise.

The latest generation of local-scale models is used in combination with meteorological pre-processing models, which are based on scaling theories of the atmospheric boundary layer. In this case, the dispersion processes are described in terms of scaling parameters for the atmospheric boundary layer and the boundary layer height. Some of these models include treatment of chemical transformation and deposition, plume rise, downwash phenomena and dispersion of particles.

Local-scale models have also been developed to describe specifically the dispersion of vehicular pollution. These models include road dispersion models, street canyon models and semiempirical models, to allow for the influence of various building configurations. Complex three-dimensional fluid dynamics models have also been used to model the flow in an urban environment.

Models of dispersion at a regional scale. The models of dispersion at a regional scale aim at quantifying the concentrations and deposition of acidifying and eutrophying compounds, such as sulfur and nitrogen compounds, and photo-oxidants such as ozone. Regional dispersion models also consider heavy metals, persistent organic pollutants, particulate matter, radioactive and hazardous materials and air quality episodes. The models have been developed either for policy-making or for research purposes.

Mesoscale dispersion models should include parametrization of the atmospheric boundary layer and allow for the relevant mesoscale weather conditions, as the influence of orography and the inhomogeneities of the surface energy balance. The models should also treat the dry and wet deposition processes and physico-chemical transformation.

Mesoscale air pollution models usually consist of a wind model (which describes the advection of air masses) and a dispersion model. The wind model can be a diagnostic or a prognostic model; for instance, part of a weather forecasting model. Both Eulerian and Lagrangian modelling systems are used.

Some regional dispersion models apply the results from meteorological trajectory models, together with a description of the dispersion processes. Modelling systems may include extensive parametrization of the dry and wet deposition processes and chemical transformation (photo-chemical dispersion models).

Models of dispersion at a continental scale. The aims of the models of dispersion at a continental scale are partly the same as for the regional scale. Most continental models (which are often called long-range transport models) aim at quantifying the concentrations and deposition of acidifying and eutrophying compounds, photo-oxidants and radioactive materials. The models have been developed either for policy-making or for research purposes.

Models of dispersion at a continental scale should include parametrization of the atmospheric boundary layer and allow for the relevant synoptic weather conditions. For long-range transport estimates, it is essential to allow for the advection (wind field) and atmospheric diffusion conditions, clouds and precipitation, surface properties and physico-chemical transformation. For instance, the dry and wet deposition processes depend on weather conditions, chemical transformation and surface structure.

The acidic loads or photochemical exposure commonly need to be assessed for periods of months or years. The policy-oriented models are therefore operated for extensive time periods, including large emission and meteorological databases. Clearly, this causes limitations concerning the model complexity. The research-oriented models can include

very extensive submodels to describe, for instance, photochemical transformation or dry and wet deposition. The model complexity also depends on whether the basic approach is Eulerian or Lagrangian.

Annex 3.2. Receptor modelling

Juhani Ruuskanen

Introduction

Developing management strategies for improving air quality requires understanding the relationship between pollutant sources and their impact at receptor sites. This requires identifying the sources emitting airborne pollutants, quantitatively estimating their emission rates and acquiring knowledge of the dispersion of pollutants in the atmosphere and the physical and chemical transformation processes taking place during this dispersion. Various receptor modelling techniques estimating the contribution of a given source at a given receptor site can form a potentially effective part of overall strategies for air quality management.

Since airborne pollutants in the atmosphere form a very complex system, mathematical or statistical methods must be available to identify the sources and apportion the pollutant concentrations observed at the receptor site to those sources. Source-oriented or dispersion models can be used to predict the concentrations of airborne pollutants at a receptor site using diffusion models comprising emission inventories and meteorological data. Receptor-oriented or receptor models are methods that focus on the behaviour of the ambient environment at a site, or receptor, as opposed to source-oriented models, which focus on transport, dilution and transformations from the source to the sampling or receptor. Receptor models estimate source contributions at receptors using statistical calculation procedures based on data collected at these sites.

The starting-point for receptor modelling is the reverse of that used in dispersion modelling. The main sources of airborne pollutants are resolved using the chemical composition of the pollutants observed at the sampling site and the chemical characteristics of possible source emissions. All receptor models are based on the assumption of mass constancy and the use of a mass balance analysis. The normal approach to

obtaining a data set for receptor modelling is to collect a number of samples and to analyse a large number of chemical constituents such as elemental, organic or gaseous concentrations in those samples. Most receptor modelling studies to date have used elemental tracers, but elements alone are not always sufficient to distinguish emitting sources. For example, as the use of leaded motor fuels has come to an end, lead and bromine have been disappearing as markers of road traffic pollution. This means that receptor models must be extended to chemical compositions other than elemental ones.

Several approaches have been successfully applied to receptor modelling (5–7). Methods have traditionally been divided into two main groups; chemical mass balance and multivariate methods. Chemical mass balance methods use data on the chemical composition of particles from both the measuring site and the potential sources, thus in principle permitting calculation of the proportions of one ambient air sample attributable to the various known sources. Multivariate methods, such as target transformation factor analysis and principal component analysis with multiple linear regression, normally use only chemical composition data on ambient air particles to ascertain the number of source categories, the chemical composition of their emissions and their relative contributions to the measured concentrations.

These methods require numerous ambient air samples, however, because source apportionment is based on statistical methods. The newest, very promising technique is the use of a neural network combined with a minimal spanning tree. The application of neural networks is still under development.

Sampling and analysis for receptor modelling

Most receptor models are based on particulate data handled by PM_{10} , $PM_{2.5}$ or (in early applications) total suspended particulate sampling to apportion the mass concentrations to elemental components related to a number of emission sources. The combinations of air sampler, filter material and analysis method are specified according to the application and the sources present in the area. Air samplers have to be selected so that an adequate particulate mass can be collected for the subsequent analysis. The filter materials must also be compatible with the requirements of both the air sampler and the analytical method. The choice of an elemental analysis method depends on the selection of elements to be

analysed, the amount of particulate mass collected, the accuracy required and costs of the analysis. Useful methods include X-ray fluorescence, inductively coupled plasma mass spectrometry, particle-induced X-ray emission and instrumental neutron activation analysis for elements and ion chromatography for sulfate and nitrate ions.

In many cases receptor modelling based only on inorganic species is not sufficient to ascertain all the sources. As emission regulations have placed limits on the use of fuel additives, such as lead and bromine in motor fuels, alternative markers are needed. Organic compounds are of particular interest in this respect. Since organic compounds constitute a major fraction of gaseous and particulate emissions, many organic compounds have been characterized for source identification. Many of them are also of direct concern for reasons of environmental health or climatic effects. The sources of these compounds can be apportioned directly instead of relying on surrogate elemental tracers. Organic compounds require their own sampling and analysis systems, which are more complicated than those for elements in many cases. The application of organic compounds to receptor modelling presupposes that the marker compounds are not reactive in the atmosphere. Hydrocarbons such as ethane, acetylene, propane, *i*-butane, *n*-butane, *i*-pentane and *n*-pentane are used as tracers for automobile exhaust, petroleum and its vapours and natural gas; hexane, toluene and xylene isomers are used to distinguish the contributions of solvents to air. Polycyclic aromatic hydrocarbons have been found suitable for identifying household wood combustion, motor spark ignition emissions and diesel engine emissions. The use of organic compounds is based in general on the chemical mass balance receptor model.

Chemical mass balance model

Chemical mass balance receptor models based on element spectra have been widely used for apportioning motor vehicles and stationary sources over the past decades. A standard chemical mass balance model uses the chemical and physical characteristics of gases and particles measured at both the source and the receptor site to identify the presence of pollutants and to quantify the contributions of the sources. Chemical mass balance models identify sources by comparing ambient chemical patterns, or fingerprints, with source chemical patterns. From the viewpoint of the receptor model, mass is assumed to be constant between the source and receptor, and the total mass of a given element is

taken to be the linear sum of the masses of the individual species that reach the receptor site from each source. The chemical and physical characteristics of the measured pollutants must be such that they are present in different proportions in different source emissions, the proportions remain relatively constant for each source type, and changes in the proportions between the sources and the receptor are negligible or can be estimated. Also, the compositions of source emissions should be constant over the period of source and ambient sampling. The chemical species chosen must not react with each other: that is, they must be linearly additive. All sources potentially contributing to the receptor must be identified, and the compositions of their emissions must be linearly independent of each other. Implicit in the above are the assumptions that only inert species can be accepted as tracers in chemical mass balance modelling. This makes it difficult to use organic compounds, as they may react or degrade during atmospheric transport, which will obscure their origin.

The source contribution estimates obtained by chemical mass balance modelling are not necessarily correct, because there is some variation depending on source activities and weather parameters. Source contributions for different periods need to be examined to determine whether or not the source contributions make sense. For example, contributions from household wood combustion are expected to be higher in winter than in summer, and samples collected upwind of the source should give lower contributions than those obtained downwind. Similarly, rush hours are expected to yield larger motor vehicle contributions than samples collected during off-peak driving periods.

The chemical mass balance model can also be used with other source and receptor models. A combination of models improves the ability to apportion pollutants to individual sources and provides a check on the results of each of the modelling systems individually.

Multivariate receptor models

The chemical mass balance model requires that the compositions of all the contributing sources be known, and in many cases this is not possible because the emissions are difficult to recognize and sample, or one is dealing with an area source such as road traffic. Also, the source compositions often fail to contain all the species observed in the samples.

These problems can be overcome by means of multivariate models, which are the only recourse if no source information is available.

Almost all multivariate receptor models first apply principal component analysis to the data set, which has usually been normalized in some way. The resulting principal components are then transformed. These are now treated as factors in a factor analysis. The basic purpose of principal component analysis is to reduce the number of variables in the data set so that new, latent variables are formed. In this way principal component analysis is used as a technical tool to reduce the dimensionality of a highly collinear data set or a qualitative tool to describe the interrelationships among variables. The principal factors resolved by principal component analysis represent different sources and indicate possible tracers for each source.

Factor analysis followed by multiple linear regression enables the estimation of source category contributions to samples taken at the receptor site. The initial factor analysis with Varimax rotation is used to identify the source categories and determine tracers for each. Multiple linear regression analysis of the samples against the tracers is then used to apportion the sample characteristics to the sources. Factor analysis followed by multiple linear regression is appropriate when the sources are known but their compositions are not available or not reliably elucidated. This method still requires tracer elements for each source category that are highly correlated with the source category and statistically independent of other tracers.

Multivariate methods require a great number of observations: the more the better. If the number of observations is low, the results given by the multivariate model will not be reliable. The number of observations is determined by experience, which shows that 100 samples is generally acceptable and 30 is usually too few. A simple rule of thumb can be developed by examining the degrees of freedom of the data set, hence the formula $N - 1 - (V - 1)/2$, where N and V are the numbers of observations and variables, respectively. Experience has shown that this formula should give a value greater than 30, and preferably 60 or more.

Multivariate receptor techniques can be applied to ambient concentration data and allow screening of the data to determine a possible

source structure, after which detailed source apportionment can be performed without any initial input of specific source profiles.

Neural mapping techniques

Neural mapping techniques have recently been applied to receptor site modelling (8–10). Appropriate techniques for this purpose are those based on Kohonen's self-organizing artificial neural network, which uses a projection technique in the case of multivariate airborne pollutant data. In this conceptual framework, the site is regarded as being affected by several sources of airborne pollutants that can contribute to different multivariate patterns of component species. Training the self-organizing artificial neural network by means of the measured data results in the formation of a base for pattern recognition. The effect of individual sources to receptors are estimated. This is similar to the results obtainable from hierarchical cluster analysis, non-linear mapping or principal component score plots. Thus, a self-organizing artificial neural network can be taken as a combination of cluster and factor analysis. The technique is still being developed, but it seems to be a useful tool for receptor modelling despite requiring the use of special computer programs.

Receptor model selection

Several considerations related to the characteristics of sources emitting airborne pollutants influence the selection of an appropriate receptor model, notably the availability of particle size data for the emissions, the chemical similarity and stability of the sources the need to identify individual sources and the time scale of interest. Chemical mass balance modelling is considered the most advanced of the receptor methods; the others, especially factor analysis followed by multiple linear regression and neural mapping techniques, are used when the data on source emissions are inadequate.

Annex 3.3. Statistical models of air quality

John Stedman

Introduction

There are a wide variety of statistical models of air quality. The term statistical model is used here to describe models that do not explicitly cover the dispersion and chemical transformation of air pollutants in the

atmosphere. Statistical models can be of several types, including rapid assessment models and empirical models.

Rapid assessment models generally consist of equations, look-up tables and graphs and provide a simplified, often non-computerized, method for modelling air quality. This type of model has often been developed by summarizing the key results from detailed computer modelling of the dispersion of air pollutants.

Empirical models of air quality do not describe the causes of the dispersion or transformation of air pollutants in the atmosphere. These models are based on empirical relationships between measured air quality and parameters related to weather and emissions. These relationships are often derived using regression or artificial neural network techniques.

Statistical air quality models are briefly introduced here in the form of three examples of very different types of models.

Rapid assessment models

An extensive WHO document (11) provides a range of rapid assessment models. These models are integrated with inventory techniques and can be used to study the following types of situations:

- the impact of the emissions from an individual point source on short-term air quality at a critical receptor: the location at which the emissions have the most severe impact;
- the impact of the emissions from an individual point source on a receptor at a known distance from the source;
- the impact of the emissions from an individual point source on the long-term average air quality in the vicinity; and
- the impact of area source emissions (such as road traffic and heating-related emissions) on the long-term average air quality of an urban area.

The impact of point source emissions on short-term air quality is modelled by assuming the most severe credible weather conditions, and local meteorological information is therefore not required. The modelling of long-term air quality generally requires local meteorological information such as the frequency patterns of wind direction, velocity and stability conditions.

These models simplify the modelling of the air quality impact of different sources by assuming that the impact of different sources can be decoupled. The models assume that chimneys are usually sufficiently far apart that the locations of the maximum impact of individual sources do not overlap. This is a reasonable assumption, with the exception of very large emission sources, and enables the short-term effects of sources to be treated independently. In addition, the weather conditions that lead to the maximum impact of high-level point sources are different from those that lead to the maximum impact of low-level area sources.

If applying this type of rapid assessment model indicates a possible air quality problem, then a more complete modelling exercise or monitoring is recommended. Assessing the air quality in situations in which pollutants are being photochemically transformed generally requires the use of a more complete air quality model and cannot be treated using these rapid assessment techniques.

Empirical models: air pollution forecasting

Statistical models are often used within national air quality forecasting systems. An example of this type of model has been developed by the National Institute of Public Health and the Environment in the Netherlands (12). This model forms part of a daily ozone forecasting service for the public, along with a causal trajectory model and expert opinion.

This type of computer model is generally rapid and inexpensive to operate but requires an extensive database of historical air quality measurement and meteorological data to formulate the model. The model computes the daily maximum ozone concentration for all sites in the national network for monitoring ozone in the Netherlands for 1 to 3 days in the future. The model requires the maximum concentrations of the monitoring sites of the previous day, statistics from the past and the maximum temperature of the previous day and the forecast temperature, both as averages for the Netherlands.

Such a statistical model has also been developed in the Russian Federation (13) and adopted for cities in the Russian Federation. Routine operation of this model is a key element of daily summary forecasting of relative increasing or decreasing air pollution levels for the next day.

The forecast gives a background for short-term emission control in 250 cities and allows a potential increase in exposure to be reduced.

The main limitation of this type of model is the requirement for historical data. Models need to be tuned using location-specific historical information, which is not available if measurements have not been made.

Empirical models: mapping air quality

Empirical statistical models have been used in the United Kingdom to derive maps of estimated air pollutant concentrations from a combination of measurement data and emission-related information (14).

The measured annual mean concentrations of air pollutants can be considered to be made up of two parts. The first is a contribution from relatively distant major source areas such as power stations or large conurbations. Measurements from monitoring sites well away from local sources, such as those in rural areas, indicate well the spatial variation of concentration caused by distant sources. The second part is a contribution from more local emissions. Estimates of emissions in an area of 25 km² centred on a background monitoring site have been found to provide the most robust relationships between emissions and ambient air quality.

The difference between measured ambient pollutant concentrations at automatic monitoring sites for urban background pollution (not roadside or industrial monitoring sites) and an underlying rural concentration field has been calculated where monitoring data are available. A regression analysis is then performed to find a coefficient, k , for the relationship between this difference and emissions in the vicinity of the monitoring sites:

$$\text{difference} = k \times \text{emissions}$$

This coefficient can then be used to calculate a map of annual mean concentrations from a combination of a rural map and emission inventory estimates. Thus, automatic monitoring data are used to calibrate the relationship between ambient air quality and emission inventories. Alternatively, measures of activity, such as traffic density, can be used as surrogates for emission inventories. Early work concentrated on estimat-

ing concentrations of NO_x and NO₂ for the United Kingdom, and maps with 1 km resolution have now been calculated for a range of species, including SO₂, PM₁₀, benzene, CO, lead and O₃. These maps have then been used to assess population exposure for studying health effects and monitoring network design.

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Personal exposure models

Halûk Özkaynak

Human exposure modelling for environmental contaminants has received considerable attention over the past decade. A number of human exposure assessment field studies, such as the Total Exposure Assessment Methodology (TEAM) studies of the US Environmental Protection Agency, have provided an important foundation for models of human exposure to CO, volatile organic compounds (VOC), pesticides and PM₁₀. The results from these field studies have produced greater understanding of the variation in the indoor, outdoor and personal pollutant concentrations. However, the measurements can usually be generalized and technically interpreted in terms of human exposure using exposure models.

Exposure models supported by adequate observations can be used to draw inferences about contaminant exposures and other endpoints for the population of interest. Exposure models provide an analytic structure for combining data of different types collected from disparate studies in a manner that may make more complete use of the existing information on a particular contaminant than is possible from direct study methods. The results can be used to evaluate the determinants of the magnitude of exposures and doses at various points with varying population distribution that cannot be measured directly because of limitations of monitoring methods or resources. The uncertainty about various components of environmental health assessment can be formally incorporated into such models to estimate uncertainty about the prediction endpoint (such as exposure, dose or health outcome), to identify the components that influence prediction accuracy and precision by comparing predicted values to those measured in the field. Validated models can then be used to investigate the efficacy of various strategies for managing the public health risks associated with exposure to doses of environmental contaminants.

Most of the exposure models, using the data gathered from these field investigations, however, have focused on inhalation pathway and

two key microenvironments: indoors at home and outdoors. Consequently, the accuracy and precision of the microenvironmental exposure models have mostly been somewhat limited to the availability of input data rather than the sophistication of techniques used to predict personal exposure. Additional monitoring and time–activity data would clearly help to improve the performance of current exposure models.

A variety of physical and statistical modelling methods have been developed for predicting exposure to gases and particles (1–5). In the framework of physical models, total personal exposure E_i is modelled as a sum of exposures encountered in various microenvironments (E_j).

Predicting personal exposure

The daily personal exposure (E_i) of an individual i is computed as the sum of microenvironmental exposure weighted by time–activity pattern:

$$E_i = \sum_{j=1}^m E_{ij} = \sum_{j=1}^m f_{ij} \times C_{ij}$$

where E_{ij} = exposure to individual i in microenvironment j ($\mu\text{g}/\text{m}^3$), f_{ij} = the fraction of time spend by person i in microenvironment j during the 24-hour prediction period, C_{ij} = the average pollutant concentration ($\mu\text{g}/\text{m}^3$) in microenvironment j when individual i is present and m is the number of microenvironments considered in the model. The key assumptions in this microenvironmental modelling approach are that: the concentrations of pollutants are uniformly distributed in each of the microenvironments; f_{ij} and C_{ij} are not correlated; and a few microenvironments are sufficient to characterize total personal exposure. The accuracy and precision of the microenvironmental exposure models depend on the number of different microenvironments that are needed to capture most of the variation in the concentrations affecting exposures. In addition, various exposure scenarios within these microenvironments (or submicroenvironments) also need to be considered. These may, for example, include: smoking or exposure to environmental tobacco smoke indoors and in a car; cooking at home; heating with kerosene space heaters; and use of consumer products emitting VOC. If activities or concentrations of studied pollutants do not vary much across the different locations that individuals visit over the course of a day, fewer number of microenvironments are sufficient to model personal expo-

sure. For example, five principal microenvironments may be included in the PM_{2.5} and PM₁₀ exposure models to represent the distinct locations and activities for particulate matter exposure: outdoors; indoors at home during the day; indoors at home during the night; in transit; and indoors not at home.

In practice, either measurements available from ambient air monitoring sites or modelled concentrations are used to estimate the profiles of outdoor pollution concentration across a community. The indoor residential or at-work pollutant concentrations are often estimated using semi-empirical methods that account for penetration of outdoor pollutants indoors and the contribution of indoor sources, such as smoking, cooking, heating and vacuuming, to indoor pollutant concentrations. Since most people spend large portions of their time indoors, measuring or modelling ambient and indoor pollutant concentrations is quite important.

A nationwide study of time budgets in the United States indicates that people spend an average of 87.2% of their time indoors, 7.2% in transit and 5.6% outdoors (6). The indoor environment plays a key role in personal exposure, since most of people's time during the day is spent indoors at home, at work or at school. In addition to examining the effects of pollutants generated indoors, it is important to understand how concentrations of pollutants are attenuated as they infiltrate indoors. Both indoor and outdoor pollutants contribute to indoor concentrations. The level of protection offered by building characteristics can directly influence the resultant exposure to ambient pollutants, such as ambient PM₁₀ and PM_{2.5}. Building-specific parameters such as volume, air exchange rate, filter efficiency, surface materials, levels of dust-loading activity, room use patterns and cleaning frequency all affect the cumulative particle concentrations. The air exchange rate contributes to diluting indoor source concentrations of particulate matter, such as from smoking or cooking, yet directly affects the penetration of ambient particles.

Indoor concentrations of particulate matter, NO₂ and VOC can be modelled using, for example, the methods developed from the PTEAM study by Özkaynak et al. (7,8). A semi-empirical physical model that assumes contributions to indoor pollution from outdoors, environmental tobacco smoke, cooking and other unaccounted indoor sources is specified (9):

$$C_{\text{in}} = \frac{P \times a}{a + k} C_{\text{out}} + \frac{N_{\text{cig}} S_{\text{smk}} + T_{\text{cook}} S_{\text{cook}}}{(a + k)Vt} + \frac{S_{\text{other}}}{(a + k)V}$$

where:

C_{in} = pollutant concentration indoors;

C_{out} = pollutant concentration outdoors;

P = penetration fraction (unitless);

a = air exchange rate (hour^{-1});

k = pollutant decay or deposition rate (hour^{-1});

N_{cig} = number of cigarettes smoked;

V = house volume (m^3);

T_{cook} = cooking time (hours);

t = sampling period (hours);

S_{smk} = estimated source strength for cigarette smoking (mg/cigarette);

S_{cook} = estimated source strength for cooking (mg/min); and

S_{other} = re-suspension or estimated source strength for other indoor sources (mg/hour).

Indoor or personal concentrations can also be modelled using an empirical relationship where indoor personal concentrations are expressed as a function of outdoor concentrations:

$$C_{\text{indoor or personal}} = \beta_0 + \beta_1 C_{\text{outdoor}}$$

This formulation more readily allows statistical estimation of the influence of outdoor concentrations on either the indoor or personal pollutant concentrations. Using both types of exposure modelling formulations, researchers have shown indoor-outdoor relationships for selected key pollutants: PM_{10} , $\text{PM}_{2.5}$, O_3 , NO_2 , CO and SO_2 .

For homes with no smokers, PTEAM data and physical modelling show that indoor-outdoor ratios are about 60–70% for $\text{PM}_{2.5}$ and about 50% for PM_{10} (8,10). PTEAM results also indicate that about 60% of outdoor PM_{10} is expected to contribute to personal PM_{10} . However, the composition of the indoor particles usually differs from that of outdoor particles because of contributions from indoor cooking, smoking, vacuuming and other personal activities that generate aerosols with different physical and chemical characteristics. Estimates of source-specific emission rates for particulate matter are therefore required in modelling the indoor PM_{10} or $\text{PM}_{2.5}$ concentrations. Özkaynak et al. (8) provide in-

formation on these values as well as on typical values for the deposition or decay rates for PM₁₀ ($0.65 \pm 0.28 \text{ h}^{-1}$) and PM_{2.5} ($0.39 \pm 0.16 \text{ h}^{-1}$).

Information on the air exchange rate is also needed to predict indoor pollutant concentrations using the physical mass balance model. Air exchange rates vary by season and by the type of structure. In the United States these range from 0.1 to 3 h^{-1} , with a median value of 1 h^{-1} and a geometric standard deviation of 2.

Indoor NO₂ levels are also influenced by outdoor concentrations and by emissions from indoor gas combustion appliances. Typically, ambient NO₂ infiltrates readily indoors. About $60 \pm 10\%$ of outdoor NO₂ penetrates indoors (11,12). NO₂ emissions from gas cooking appliances with no pilot lights contribute about 4–6 ppb to indoor NO₂ levels. Older types of gas stoves with continuously lit pilot lights add between 8 and 15 ppb to indoor NO₂ concentrations (12). NO₂ is a moderately reactive gas and decays quickly indoors through chemical and surface-based reactions. The experimental decay rates are given in Özkaynak et al. (13) and also in Nazaroff et al. (14).

Indoor ozone concentrations are typically much lower than those outdoors since ozone is a highly reactive gas. A study in 43 homes in southern California (15) showed that the mean indoor-outdoor ozone ratio in residences with open windows was 0.7 ± 0.2 . In homes with air conditioning, however, the mean indoor-outdoor ratio was quite small, 0.1. Indoor ozone concentrations in homes typically are 10% to 30% of outdoor values. Lee et al. (15) also estimated a mean indoor ozone decay rate of $2.8 \pm 1.3 \text{ h}^{-1}$.

Indoor VOC concentrations are typically several times higher than those outdoors (16). According to Wallace (16), indoor sources of VOC include: smoking (benzene, xylenes, ethylbenzene and styrene in breath), dry-cleaned clothes (tetrachloroethylene), air fresheners (limonene), house cleaning materials, use of chlorinated water (chloroform), deodorizers (*p*-dichlorobenzene), various types of occupational exposure and pumping petrol (benzene). Most of the outdoor VOC, however, quite efficiently penetrate indoors without much loss (17). Therefore, for some VOC, such as benzene, which has fewer indoor sources (cigarette smoking and attached garages are the primary indoor sources of benzene), indoor VOC concentrations are correlated well with out-

door levels, and indoor-outdoor ratios are closer to 1. In contrast, new buildings have VOC levels about 100 times higher than outdoor levels and only decrease to 10 times the outdoor levels after 2–3 months of occupancy. Paints and adhesives are assumed to be the primary contributors to indoor VOC levels in new buildings.

Finally, pollutants such as CO and SO₂ also penetrate quite efficiently indoors and are not reactive. Aside from possible emissions from kerosene space heaters, typically SO₂ is an outdoor pollutant. CO is also principally an outdoor pollutant. Motor vehicle and petrol combustion engines are the dominant sources of ambient CO. Gas cooking or heating appliances and smoking, however, also contribute to indoor CO concentrations. In general, indoor CO levels are slightly higher than the outdoor values for homes with gas cooking appliances (18).

Average time concerns for exposure assessment

The appropriate averaging period for monitoring and modelling and the minimum number of microenvironments have to be determined based on the specific pollutant and the biological averaging time of interest. In some cases (such as CO), technology exists to monitor minute-by-minute CO concentrations, either fixed location or personal exposure. However, for either modelling purposes or assessment of health effects, it is not clear whether such a short time resolution is always necessary. Pollutants that have been shown to produce acute health effects, such as CO, need to be studied on a near real-time basis. Since activities and microenvironments vary in time intervals greater than 1 minute, successful and practical exposure modelling suggests appropriate simulation time steps of minutes to an hour for CO. However, for VOC, particulate and metals that are typically linked with chronic health effects, the required time interval for estimating the predictions may range from 1 hour to 24 hours. Exposure models for these pollutants have typically assumed either 24-hour or 12-hour average concentrations as inputs, often because of measurement limitations. On the other hand, exposure models, such as SHAPE (for CO), THEM (for particles), NEM (for CO and ozone) and BEADS (for benzene), choose different time intervals in estimating personal or population exposures. In general, the averaging period of interest for health effects dictates the optimum resolution time required for modelling. Models such as SHAPE and NEM are time dependent in that they require real-time (minute-by-minute) knowledge of

time–activity and microenvironmental concentration data (19). In contrast, the benzene BEADS model developed by Harvard researchers (20) is time independent in that daily or 12-hour average activity and concentration profiles are used in simulating population exposure. The differences between time-dependent versus time-independent methods have important implications for input requirements, the time required to run the model, general applicability and the desired accuracy and precision in the predictions.

Alternative exposure models and their use in investigating health effects

A number of different approaches to exposure assessment have typically been used in environmental epidemiology investigations. In terms of increasing order of sophistication, these include:

- classification of individual exposure (high versus low);
- measured or modelled outdoor concentrations;
- measurement of indoor and outdoor concentrations;
- estimation of personal exposure using indoor, outdoor and other microenvironmental concentrations along with time–activity diaries;
- direct measurement of personal exposure; and
- measurement of breath and other biomarkers of exposure.

Clearly, the least sophisticated approach in classifying exposure groups using a categorical variable (such as homes with gas versus electric cooking stoves for NO₂ impact assessment) could lead to significant exposure misclassification bias (21). However, many existing environmental health studies are based on ambient or community surveillance monitoring data. Aside from the usual spatial variation in outdoor pollutant concentrations, human exposure to many pollutants also involves pollutant exposure sources and locations other than outdoor pollutants and monitored ambient environments (such as for particulate matter, NO₂ and VOC). For reactive pollutants, such as ozone, indoor pollution levels are significantly lower than the outdoor concentrations. Since people spend more time indoors, personal ozone exposure is more closely related to indoor than outdoor ozone concentrations. In general, therefore, exposure models based on ambient data only are much less accurate than the microenvironmental models that combine indoor and outdoor concentration measurements (or predictions) with time–activity

data and information on sources and household characteristics. Various investigators have demonstrated this. For example, Xue et al. (22) showed that the estimated coefficient of determination R^2 (a measure of model fit) increased from 0.28 to 0.74 when different exposure models were applied to 2-day average ambient bedroom and personal NO_2 data. These results showed that the predictive power of the NO_2 personal exposure models was quite poor ($R^2 = 0.28$) if no ambient or indoor NO_2 measurements were used except for home characterization questionnaire variables, such as the use of gas cooking appliances and air conditioners. NO_2 exposure models with either outdoor or indoor NO_2 measurements in addition to the home characterization questionnaire variables had greater predictive power ($R^2 \cong 0.6$). The full microenvironmental model weighted by time–activity data, however, had a much higher R^2 value ($R^2 = 0.74$). Ideal exposure models should clearly combine outdoor measurements with indoor concentration measurements or predictions proportional to the fraction of time spent in each of these two key microenvironments. These findings have important implications in designing monitoring studies to support investigations of health effects.

Conclusions

In conclusion, applying exposure models requires collecting and using different types of data as inputs to models (such as emission rates to various media, time–activity, dispersion and removal data) and new statistical techniques for implementing the relationships between various model parameters, time and activities and concentration distributions. Model input needs have important implications for developing monitoring strategies and programmes. A more detailed temporal and spatial profile of ambient and personal concentrations is often needed than is available. Personal exposure during commuting, for example, requires more kerbside and personal monitoring. Increasing the resolution of ambient monitors helps enhance the reliability of personal exposure models. Studying more subjects and collecting time–activity diaries and the strength of indoor pollution sources in unique locations or countries with limited baseline measurements is highly recommended. Finally, the real success of any present or future exposure models strongly depends on the results of carefully constructed field validation studies. Without proper validation, exposure models will remain as interpretive tools or as complementary information to measurement data. The challenge for

the future is to increase the accuracy, precision and the use of exposure models in environmental monitoring studies, assessment of human health effects and regulatory decision-making.

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WHO requirements for air quality indicators^a

The air quality data in a health and environment geographical information system (HEGIS) should represent the so-called classical air pollutants that are most widely distributed in the European Region and pose a risk to human health.

Whereas indicators of emissions are intended to describe long-term national trends, the WHO concentration indicators aim to describe human exposure to air pollution. Based on these indicators and available population distribution data, the number of inhabitants who live in areas with a certain air quality can be calculated for the urban and rural population within Europe. This requires incorporating data from air dispersion models and emission inventories to determine the geographical distribution of air pollution at the place of concern and the contribution from different sources.

In summary, the following principle is proposed: for a given population, use one average indicator for a specified time.

Typical ambient concentrations derived from monitoring networks do not necessarily reflect human exposure, which may vary greatly depending on occupation, location of dwelling and individual activity. Thus, the monitoring stations that can produce the values that can most accurately represent the exposure of the population being monitored will be selected based on expert judgement of the knowledge of local conditions and spatial patterns of pollution.

^a Extracts from *Health related air quality indicators and their application in health impact assessment in HEGIS: report on a WHO consultation, Sosnowiec, Poland, 21–23 November 1995 (1)*

Thus, the WHO requirements for air pollution data for population exposure assessment are listed here.

Network and station characteristics^a

(a) Country

(b) Network affiliation (complete name of the network)

(c) Geographical coverage of the network

Local industry: covers an industrial estate or around a power plant

Town or city: covers a small town or city with less than 250 000 inhabitants

Urban area or conurbation (agglomeration): covers an extensive urban area with more than 250 000 inhabitants

Administrative unit: a province, county, etc.

Region: for example, the Ruhr region

National: covers an entire country

Unknown

(d) Name of the city in which station is located (if not in rural zone)

The local name of the station and its local code

Name of the technical body responsible for the station

Complete name of the organization responsible for network management and/or supplying data and the name of the responsible person and his or her address.

Depending on the data source, the following characteristics and description of the station type and monitoring site are kept.

^a Compatible with the station classification form used by the AIRBASE system (<http://www.etcaq.rivm.nl/airbase/airbadm.html>, accessed 22 August 1999).

Station type

The station types include:

- traffic: station used for monitoring of air pollution induced by road traffic;
- industrial: station used for monitoring industrial air pollution;
- background: station used for monitoring background air pollution levels located inside (urban background) or outside (regional or background) cities; and
- unknown: station type is not known.

For traffic stations, the type of street and traffic volume must be specified.

Type of street

Wide $D/H > 1.5$ (D = distance between axis street and buildings; H = height of buildings)

Canyon $D/H < 1.5$

Highway Average speed of vehicles > 80 km/h

Unknown

Traffic volume

High $> 10\,000$ vehicles/day

Medium $2000\text{--}10\,000$ vehicles/day

Low < 2000 vehicles/day

Unknown

Type of zone

The types of zones include:

- urban: station is located in a city;
- suburban: station is located on the outskirts (periphery) of a city or in small residential areas outside the main city;
- rural: station is located outside a city; and
- unknown.

Characterization of zone

The zones include:

- residential
- commercial
- industrial
- residential and commercial
- commercial and industrial
- industrial and residential
- residential, commercial and industrial
- agricultural
- natural
- agricultural and natural.

Methods and units of measurement

Although the methods used in a specific country are based on national and international standardization, it would be preferable to follow the air quality directives of the European Union and at least to determine the differences between the different methods being used. Air pollutant concentrations should be expressed in $\mu\text{g}/\text{m}^3$. The pollutant code makes it possible to specify different measurement units for each pollutant.

Time coverage

Year: calendar year (1 January to 31 December).

Seasons: winter – from October to March inclusive; summer – from April to September inclusive.

The ratio between the number of valid data for the two seasons of the year considered cannot be greater than 2.

Criteria for validity of stations

The following criteria have been used to establish the validity of a station.

To obtain 1-hour average values from data with a smaller averaging time, at least 75% of valid data should be used.

To obtain 8-hour moving average values from hourly measures, the number of hours where valid measures have been performed must be at least 18 (75%).

To obtain 24-hour average values from data with a smaller averaging time, over 50% of 1-hour valid data should be used and not more than 25% of successive data values not accepted.

To obtain seasonal and annual average values, at least 50% of the valid data for the reported period should be used.

Indices and statistical parameters

For the stations that comply with the validity criteria, the following indices can be calculated:

- 1-hour average for CO and NO₂;
- maximum 1-hour average and a maximum 8-hour moving average in a day (24 hours) for ozone;
- daily (24-hour) average for SO₂, total suspended particulate, black smoke and PM₁₀; and
- seasonal and annual average (with valid winter period) for lead and benzo[*a*]pyrene.

Calculation of statistical parameters requires:

- for the mean (arithmetic): over 50% of data accepted; and
- for the percentile (98) and maximum: over 75% of data accepted.

To obtain an annual mean, the following criteria of completeness must be met:

- for CO and NO₂: valid winter and summer periods;
- for ozone: a valid summer period; and
- for SO₂, total suspended particulate, black smoke and PM₁₀: a valid winter period.

Population exposure profile

For each city or agglomeration with a defined population, the number of monitoring stations in operation, each station's characteristics and information about their validity will be provided. Based on a station's validity criteria, the annual and seasonal population exposure profile will be calculated according to the pollutant of concern.

A profile of the population exposure and/or a profile of subgroups of the population will be estimated based on the averaging of the corresponding data from the valid stations, which in the experts' opinion most accurately represent the exposure of the population under surveillance. Once selected, the stations selected for calculating the population exposure profile, must remain stable for 1 year and should not be replaced. If the selection of stations for calculating the population exposure profile is changed next year, this should be recorded and reported if the data are used or distributed.

The population exposure profile should be calculated based on the averaging of the daily valid data obtained from at least two thirds of the previously selected stations. If the data from less than two thirds of the stations are available on a certain day, the daily average exposure is not calculated and the data from this day do not contribute to the description of the population exposure profile.

If a profile of population exposure was estimated by daily averaging of the data from more than one monitoring station, the additional data (specified below) for the profile of lowest stations and the profile of highest stations will be required.

Calculating the population exposure profile by averaging the seasonal or annual monitoring station profile is not acceptable.

The data to be captured include the number of days (or hours for CO and NO₂) from which the annual and the seasonal averages have been calculated for each pollutant (the number of days for which the daily (or hourly) average population exposure profile was calculated is taken into consideration).

A profile of the population exposure for the classical air pollutants, together with the other relevant information allowing data identification, is recommended to be reported using forms A5.1, A5.2 and A5.3. The software tool AIRQ can also be used.

Reference

1. *Health related air quality indicators and their application in health impact assessment in HEGIS: report on a WHO consultation, Sosnowiec, Poland, 21–23 November 1995*. Copenhagen, WHO Regional Office for Europe, 1997 (document EUR/ICP/EHAZ 94 06/MT03).

Form A5.1a: Profiles for population exposure to air pollutants
Data collection form for SO₂, total suspended particulate, black
smoke and PM₁₀^a

Calendar year _____

Country _____

Agglomeration name _____

Total population size (thousands) _____

Population size for which the exposure profile is presented^b (thousands) _____

Data capture: annual (days) _____

winter (days) _____

summer (days) _____

Annual arithmetic mean of 24-hour averages	
Winter arithmetic mean of 24-hour averages	
Summer arithmetic mean of 24-hour averages	
Annual 98th percentile of 24-hour averages	
Annual maximum value of 24-hour average	
Winter maximum value of 24-hour average	
Summer maximum value of 24-hour average	

Pollutant's concentration (µg/m ³)	Total number of days
<10	
10–20	
20–30	
30–40	
40–50	
50–60	
60–70	
70–80	
80–90	
90–100	
100–110	
110–120	
120–130	
130–140	
140–150	
150–160	
160–170	
170–180	
180–190	
190–200	
200–250	
250–300	
300–350	
350–400	
≥400	

^a Check off what is inappropriate according to the specified pollutant.

^b If the exposure profile presented is for the total population of the agglomeration, the number should be the same as for the total population size.

The profile of lowest stations and the highest stations

If a profile of population exposure was estimated by daily averaging of the data from more than one monitoring station, the following additional data for the profile of lowest stations and the highest stations are required.

<i>Lowest station</i>	<i>Local name and code of the station</i>	<i>Highest station</i>
	Annual arithmetic mean of 24-hour averages	
	Winter arithmetic mean of 24-hour averages	
	Summer arithmetic mean of 24-hour averages	
	Annual 98th percentile of 24-hour averages	
	Annual maximum value of 24-hour averages	
	Winter maximum value of 24-hour averages	
	Summer maximum value of 24-hour average	

Optional information

If this information is not available, please send the measured concentrations of specified pollutants in the calendar year period on daily basis in electronic form for the stations you selected to assess the population exposure profile together with required statistics (arithmetic mean and 98th percentile).

Form A5.1b: Profiles for population exposure to air pollutants
Data collection form for SO₂ and CO^a

Calendar year _____

Country _____

Agglomeration name _____

Total population size (thousands) _____

Population size for which the exposure profile is presented^b (thousands) _____

Data capture: annual (hours) _____

winter (hours) _____

summer (hours) _____

Annual arithmetic mean of 1-hour averages	
Winter arithmetic mean of 1-hour averages	
Summer arithmetic mean of 1-hour averages	
Annual 98th percentile of 1-hour averages	

Pollutant's concentration (µg/m ³)	Total number of hours
<10	
10–20	
20–30	
30–40	
40–50	
50–60	
60–70	
70–80	
80–90	
90–100	
100–110	
110–120	
120–130	
130–140	
140–150	
150–160	
160–170	
170–180	
180–190	
190–200	
200–250	
250–300	
300–350	
350–400	
≥400	

^a Check off what is inappropriate according to the specified pollutant.

^b If the exposure profile presented is for the total population of the agglomeration, the number should be the same as for the total population size.

The profile of lowest stations and the highest stations

If a profile of population exposure was estimated by daily averaging of the data from more than one monitoring station, the following additional data for the profile of lowest stations and the highest stations are required.

<i>Lowest station</i>	<i>Local name and code of the station</i>	<i>Highest station</i>
	Annual arithmetic mean of 1-hour averages	
	Winter arithmetic mean of 1-hour averages	
	Summer arithmetic mean of 1-hour averages	
	Annual 98th percentile of 1-hour averages	

Optional information

If this information is not available, please send the measured concentrations of specified pollutants in the calendar year period on daily basis in electronic form for the stations you selected to assess the population exposure profile together with required statistics (arithmetic mean and 98th percentile).

Form A5.1c: Profiles for population exposure to air pollutants
Data collection form for ozone^a

Calendar year _____	<table border="1"> <thead> <tr> <th>Pollutant's concentration in ($\mu\text{g}/\text{m}^3$)</th> <th>Total number of days</th> </tr> </thead> <tbody> <tr><td><10</td><td></td></tr> <tr><td>10–20</td><td></td></tr> <tr><td>20–30</td><td></td></tr> <tr><td>30–40</td><td></td></tr> <tr><td>40–50</td><td></td></tr> <tr><td>50–60</td><td></td></tr> <tr><td>60–70</td><td></td></tr> <tr><td>70–80</td><td></td></tr> <tr><td>80–90</td><td></td></tr> <tr><td>90–100</td><td></td></tr> <tr><td>100–110</td><td></td></tr> <tr><td>110–120</td><td></td></tr> <tr><td>120–130</td><td></td></tr> <tr><td>130–140</td><td></td></tr> <tr><td>140–150</td><td></td></tr> <tr><td>150–160</td><td></td></tr> <tr><td>160–170</td><td></td></tr> <tr><td>170–180</td><td></td></tr> <tr><td>180–190</td><td></td></tr> <tr><td>190–200</td><td></td></tr> <tr><td>200–250</td><td></td></tr> <tr><td>250–300</td><td></td></tr> <tr><td>300–350</td><td></td></tr> <tr><td>350–400</td><td></td></tr> <tr><td>≥ 400</td><td></td></tr> </tbody> </table>	Pollutant's concentration in ($\mu\text{g}/\text{m}^3$)	Total number of days	<10		10–20		20–30		30–40		40–50		50–60		60–70		70–80		80–90		90–100		100–110		110–120		120–130		130–140		140–150		150–160		160–170		170–180		180–190		190–200		200–250		250–300		300–350		350–400		≥ 400	
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Country _____																																																					
Agglomeration name _____																																																					
Total population size (thousands) _____																																																					
Population size for which the exposure profile is presented ^b (thousands) _____																																																					
Data capture: annual (days) _____																																																					
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^a Check off what is inappropriate according to the specified pollutant.

^b If the exposure profile presented is for the total population of the agglomeration, the number should be the same as for the total population size.

The profile of lowest stations and the highest stations

If a profile of population exposure was estimated by daily averaging of the data from more than one monitoring station, the following additional data for the profile of lowest stations and the highest stations are required.

<i>Lowest station</i>	<i>Local name and code of the station</i>	<i>Highest station</i>
	Annual 98th percentile of daily maximum 1-hour averages	
	Maximum annual value of daily maximum 1-hour averages	
	Maximum winter value of daily maximum 1-hour averages	
	Maximum summer value of daily maximum 1-hour averages	
	Annual 98th percentile of daily maximum 8-hour moving averages	
	Maximum annual value of daily maximum 8-hour moving averages	
	Maximum winter value of daily maximum 8-hour moving averages	
	Maximum summer value of daily maximum 8-hour moving averages	

Optional information

If this information is not available, please send the measured concentrations of specified pollutants in the calendar year period on daily basis in electronic form for the stations you selected to assess the population exposure profile together with required statistics (arithmetic mean and 98th percentile).

Form A5.1d: Profiles for population exposure to air pollutants**Data collection form
for lead ($\mu\text{g}/\text{m}^3$)**

Calendar year _____

Country _____

Agglomeration name _____

Total population
size (thousands) _____Population size for
which the exposure
profile is presented^a
(thousands) _____Data capture:
annual (days) _____

winter (days) _____

**Data collection form
for benzo[a]pyrene (ng/m^3)**

Calendar year _____

Country _____

Agglomeration name _____

Total population
size (thousands) _____Population size for
which the exposure
profile is presented^a
(thousands) _____Data capture:
annual (days) _____

winter (days) _____

summer (days) _____

Annual average	
Winter average	
Summer average	

^a If the exposure profile presented is for the total population of the agglomeration, the number should be the same as for the total population size.

Annual average	
Winter average	
Summer average	

^a If the exposure profile presented is for the total population of the agglomeration, the number should be the same as for the total population size.

The profile of lowest stations and the highest stations

If a profile of population exposure was estimated by daily averaging of the data from more than one monitoring station, the following additional data for the profile of lowest stations and the highest stations are required.

<i>Lowest station</i>		<i>Highest station</i>
	<i>Local name and code of the station</i>	
	Annual averages	
	Winter averages	
	Summer averages	

Form A5.2: Station characteristics
(should be attached to Form 5.1 for each pollutant)

Pollutant: _____

Calendar year:	
Country	
Agglomeration name	
Agglomeration coordinates (degrees and minutes)	
Total population size (thousands)	
Population size for which the exposure profile is presented (thousands)	
A. Total number of stations in operation in the agglomeration area	
B. Total number of stations in operation in the area in which the population exposure profile has been assessed	
C. Number of stations the data of which had been used in assessing the population exposure profile ^a	
Network affiliation (name of network) of C stations	
Name of technical body responsible for C stations	

Fill in the information referred to for each station selected for the assessment of the population exposure profile (C stations)

Station local name and code	
Station type	
Type of zone	
Characterization of zone	
For traffic station type only: Type of street Volume of road traffic	
Station height above ground (metres)	
Monitoring technique (manual passive, manual active, automatic fixed, light scattering or absorption systems)	
Measurement method or device (chemoluminescent, diffusion tube etc.)	
Normalization conditions (temperature and pressure – ISO standard, European Union directive or WHO guidelines)	
Estimated magnitude of the error (% of measured value caused by uncertainty in calibration and measurement)	
Data capture (number of days with valid measurements out of 365)	

^aThe numbers in A, B and C may be the same if all stations operating in the agglomeration area are valid and their parameters were used in assessing the population exposure profile.

Form A5.3

[to be filled in and attached to Form A5.1 and Form A5.2]

Information about the data supplier

Country	
Name of the organization responsible for managing the network and supplying data	
Mailing address	Street, postal box, postal code City
Your e-mail address and telephone and fax numbers	For telephone and fax numbers use format: +xx yy zzzzz xx = international access code of the country yy = area code (city code) zz = telephone number of the organization
Last and first name of responsible person	

Air quality assessment is frequently driven by the need to determine whether a standard or guideline has been exceeded, but it should also provide the information needed to estimate population exposure to air pollution and the effects on the health of the population. Most air quality monitoring systems do not fully address population exposure to toxic air pollution. Health impact assessment combines estimates of population exposure with information on toxicity.

Given the importance of the availability of valid information on population exposure to air pollutants, the WHO European Centre for Environment and Health organized a working group to define the features of monitoring networks that allow their use in assessing the potential exposure of the population to air pollution from ambient air. This work resulted in this book. The principles outlined are intended to promote progressive modification of the networks monitoring air quality to improve their usefulness for health impact assessment.

This book is directed specifically to network managers, to those who design new networks or modify existing ones, to policy-makers and to those who influence policy.

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