

## 1. Introduction

Air pollution is a major environmental health problem affecting developed and developing countries around the world. Increasing amounts of potentially harmful gases and particles are being emitted into the atmosphere on a global scale, resulting in damage to human health and the environment. It is damaging the resources needed for the long-term sustainable development of the planet.

The sources of air pollution resulting from human activities are of three broad types.

- Stationary sources. These can be subdivided into:
  - Rural area sources such as agricultural production, mining and quarrying.
  - Industrial point and area sources such as manufacturing of chemicals, non-metallic mineral products, basic metal industries, power generation.
  - Community sources, e.g. heating of homes and buildings, municipal waste and sewage sludge incinerators, fireplaces, cooking facilities, laundry services and cleaning plants.
- Mobile sources. These comprise of any form of combustion-engine vehicles, e.g. light duty gasoline-powered cars, light and heavy-duty diesel-powered vehicles, motorcycles, aircraft, and including line sources such as fugitive dusts from vehicle traffic.
- Indoor sources. These include: tobacco smoking, biological sources (such as pollen, mites, moulds, insects, micro-organisms, pet allergens etc.), combustion emissions, emissions from indoor materials or substances such as volatile organic compounds, lead, radon, asbestos, various synthetic chemicals and others.

In addition, there are also natural sources of pollution, e.g. eroded areas, volcanoes, certain plants that release great amounts of pollen, sources of bacteria, spores and viruses, etc. These natural physical and biological sources of pollution are not discussed in this publication.

In recent years it has become clear that indoor air pollution from the use of open fires for cooking and heating may be a serious problem in developing countries. It has been estimated that about 2 500 000 people die annually from exposure to high concentrations of suspended particulate matter in the indoor air environment; and the excess mortality due to suspended particulate matter and sulphur dioxide in the ambient air amounts to about 450 000 people annually (Murray and Lopez 1996; Schwela 1996a; WHO 1997a). Although the indoor air database is weak due to the scarcity of monitoring results, these estimates indicate that a serious indoor air problem may exist in developing countries.

Air pollutants are usually classified into suspended particulate matter (dusts, fumes, mists, smokes), gaseous pollutants (gases and vapours) and odours.

*Suspended particulate matter (SPM)* Particulate matter suspended in air includes total suspended particles (TSP), PM<sub>10</sub>, (SPM with median aerodynamic diameter less than 10 µm), PM<sub>2.5</sub> (SPM with median aerodynamic diameter less than 2.5 µm), fine and ultrafine particles, diesel exhaust, coal fly-ash, mineral dusts (e.g. coal, asbestos, limestone, cement), metal dusts and fumes (e.g. zinc, copper, iron, lead), acid mists (e.g. sulphuric acid), fluoride particles, paint pigments, pesticide mists, carbon black, oil smoke and many others. Suspended particulate pollutants provoke respiratory diseases, and can cause cancers, corrosion, destruction to plant life, etc. They can also constitute a nuisance (e.g. accumulation of dirt), interfere with sunlight (e.g. light scattering from smog and haze) and also act as catalytic surfaces for reaction of adsorbed chemicals.

*Gaseous pollutants:* Gaseous pollutants include sulphur compounds (e.g. sulphur dioxide (SO<sub>2</sub>) and sulphur trioxide (SO<sub>3</sub>)), carbon monoxide (CO), nitrogen compounds [e.g. nitric oxide (NO), nitrogen dioxide (NO<sub>2</sub>), ammonia (NH<sub>3</sub>)], organic compounds [e.g. hydrocarbons (HC), volatile organic compounds (VOC), polycyclic aromatic hydrocarbons (PAH) and halogen derivatives, aldehydes, etc.], halogen compounds (HF and HCl) and odourous substances.

Secondary pollutants may be formed by thermal, chemical or photochemical reactions. For example, by thermal action SO<sub>2</sub> can be oxidised to SO<sub>3</sub> which, dissolved in water, gives rise to the formation of sulphuric acid mist (catalysed by manganese and iron oxides). Photochemical reactions between NO<sub>x</sub> and reactive hydrocarbons can produce ozone (O<sub>3</sub>), formaldehyde (HCHO) and peroxyacetyl nitrate (PAN); reactions between HCl and HCHO can form bis-chloromethyl ether.

*Odours:* While some odours are known to be caused by specific chemical agents such as hydrogen sulphide (H<sub>2</sub>S), carbon disulphide (CS<sub>2</sub>) and mercaptans (R-SH, R<sub>1</sub> S R<sub>2</sub>), others are difficult to define chemically.

An air pollutant concentrations inventory summarizes the results of monitoring ambient air pollutants. The data are expressed in terms of annual means, percentiles and trends of the parameters measured. In most developed countries compounds measured for such an inventory include SO<sub>2</sub>, nitrogen oxides (NO<sub>x</sub>), SPM, CO, O<sub>3</sub>, heavy metals, PAH, and VOC. In developing countries the “classical” compounds SO<sub>2</sub>, NO<sub>x</sub>, SPM, CO, O<sub>3</sub> and lead are commonly monitored.

Trends in air pollution exposure are usually shown as annual arithmetic or geometric means and as statistical measures of short-term exposure such as high percentiles, or maximal or second highest values of a sample. The general picture for the “classical” compounds considered in this publication is that SO<sub>2</sub> and SPM concentrations are decreasing in developed countries while NO<sub>x</sub> and O<sub>3</sub> concentrations are either constant or increasing (UNEP/WHO 1992). In many countries in transition and in developing countries, SO<sub>2</sub> and SPM concentrations are increasing as a consequence of increasing combustion, as are NO<sub>x</sub> and O<sub>3</sub> due to increasing traffic exhaust and emissions of VOC by industrial sources as precursors of O<sub>3</sub>.

WHO's air quality guidelines were first published as *Air Quality Guidelines for Europe* in 1987 (WHO 1987). Since 1993 the *Air Quality Guidelines for Europe* has been revised and updated after a review of the literature published since 1987 (WHO 1992a; WHO 1994a; WHO 1995a; WHO 1995b; WHO 1995c; WHO 1996a; WHO 1998a; WHO 1999a). Also, the following additional compounds were considered in the review procedure: 1,3 butadiene, environmental tobacco smoke (ETS), fluoride, man-made-vitreous fibres (MMVF) and platinum. Parallel to the review of the air quality guidelines for Europe, the Environmental Health Criteria series of the International Programme on Chemical Safety has continued and the health risks of more than 120 chemical compound and mixtures were assessed between 1987 and 1998.

Trends of ambient air pollution were assessed in the WHO/UNEP Global Environmental Monitoring System/Air Pollution (GEMS/Air) which operated from 1973 to 1995 (UNEP/WHO 1993). The GEMS/Air programme has been replaced by a new programme under the umbrella of WHO's Healthy Cities Programme: Air Management Information System (AMIS). AMIS is intended as an information turntable which collects information on all issues of air quality management from its participants and distributes this information among them via the information centre at WHO. Several databases have already been developed (WHO 1997b; WHO 1998b). The AMIS core database of ambient air pollutant concentrations contains summary data, including annual means, percentiles and the number of days on which WHO Air Quality Guidelines are exceeded, from more than 100 cities in the world. A database on air quality guidelines and air quality standards contains data from about 60 countries. A database on air pollution management capabilities contains data from 70 cities. A database of the AMIS focal points helps AMIS participants in different countries to communicate with each other. A database on indoor air pollutant concentrations and a noise database have been developed and will be available in the near future.

The WHO Air Quality Guidelines for Europe (WHO 1987) were based on evidence from the epidemiological and toxicological literature published in Europe and North America. They did not consider exposure to ambient air concentrations in developing countries and the different conditions in these countries. However, these guidelines were used intensively throughout the world. In view of the

different conditions in developing countries, the literal application of the WHO Air Quality Guidelines for Europe could be misleading. Factors such as high and low temperature, humidity, altitude, background concentrations and nutritional status could influence the health outcome after exposure of the population to air pollution. To make the WHO *Air Quality Guidelines for Europe* globally applicable, a task force group meeting was convened at WHO Headquarters from 2-5 December 1997. The outcome of this meeting is this publication of the globally applicable *Guidelines for Air Quality*.

The objective of WHO's *Guidelines for Air Quality* is to help countries derive their own national air quality standards, to help protect human health from air pollution. The guidelines are technologically feasible and consider socio-economic and cultural constraints. They provide a basis for protecting public health from the adverse effects of air pollution, and for eliminating or reducing to minimum, air pollutants likely to be hazardous to human health. Consequently, the instruments of air quality management are also addressed in this publication.

## **2. Air Quality and Health**

### **2.1 Basic facts**

Pure air comprises oxygen (21%) and nitrogen (78%) and a number of rarer gases, of which argon is the most plentiful. Carbon dioxide (CO<sub>2</sub>) is present at a lower percentage concentration (0.03%) than argon (0.93%).

Water vapour, up to 4% by volume, is also present. Oxygen is produced by plants as a by-product of photosynthesis and the earth's atmosphere is now described as oxidant, or oxidising, in comparison with the hydrogen-rich reducing atmosphere that was present before life began. The increase in oxygen has led to the development of anti-oxidant defences in many living organisms.

The atmosphere contains a number of gases which, at higher than usual concentrations, are poisonous to humans and animals and damaging to plants. These include O<sub>3</sub>, SO<sub>2</sub>, NO<sub>2</sub>, CO and a wide range of VOC. Some of the latter are carcinogenic, for example benzene and butadiene. All these potentially toxic gases are referred to as air pollutants.

As well as gases, the atmosphere contains a wide variety of particulate matter, both solid and liquid, ranging in size from a few nanometres to about 0.5 mm. Small particles (<2.5 µm) persist in the air for long periods, forming a more or less stable aerosol. Larger particles are more quickly lost as their mass leads to rapid sedimentation.

This publication is focused on gases and particulate materials that have been accepted as posing a threat to health. The relative importance of the different pollutant gases and particles varies with their concentrations over both time and distance. This implies that the extent of the effects of air pollutants on health may vary from country-to-country. Careful monitoring of the concentrations of polluting gases and the particle size distribution, concentration, and composition is thus needed before an acceptable estimate of effects can be produced. The picture is further complicated as some combinations of pollutants act in an additive manner and some perhaps synergistically.

#### **2.1.1 Physico-chemical aspects of air pollution and units used to describe concentrations of air pollutants**

A consistent system of units is necessary if concentrations of air pollutants in different countries are to be compared. For both gases and particles WHO has adopted a mass per unit volume system, with concentrations generally expressed as µg/m<sup>3</sup>. The volume of a mass of air varies with ambient temperature and atmospheric pressure and thus these conditions should be specified. In considering pollutants on a global scale this is clearly important.

The alternative system, the volume mixing ratio, is applicable only to gases. In this system the concentration of gas is expressed as parts per billion, for example, and assuming ideal gas behaviour, does not depend upon the conditions of sampling because these will affect the air containing the pollutant and the pollutant itself to the same extent. A gas present at one part per million thus occupies 1 cm<sup>3</sup> per m<sup>3</sup> of polluted air; is present as 1 molecule per 1 x 10<sup>6</sup> molecules and exerts a partial pressure of 1 x 10<sup>-6</sup> atmospheres.

The two systems are interconvertible as under ideal conditions, 1 Mole of gas occupies 22.4 litres at 273K and 13mb pressure, dry air Standard Temperature and Pressure Dry (STPD). The interconversion formula is:

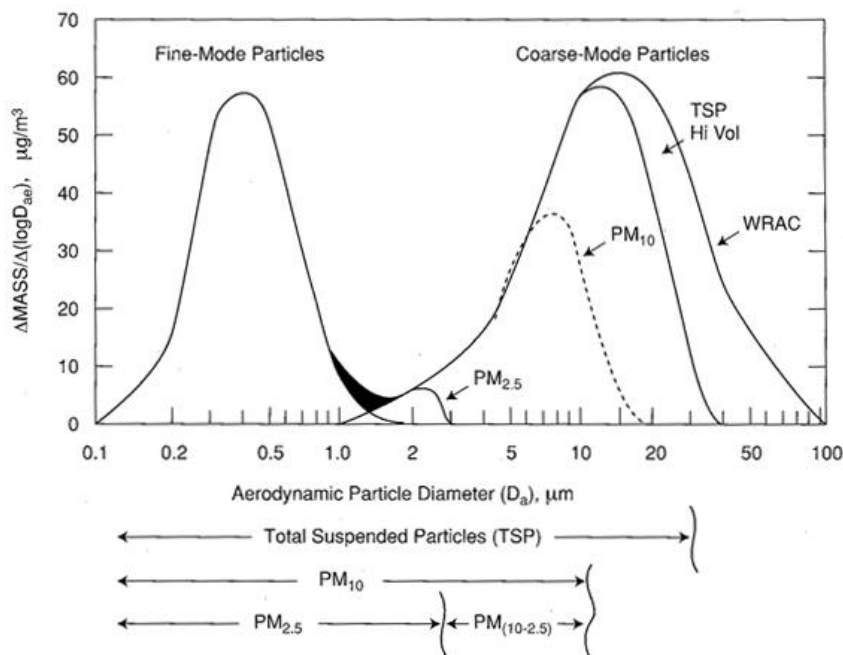
$$\text{mg/m}^3 = \text{ppm} \times (\text{molecular weight/molar volume})$$

$$\text{molar volume} = 22.4 \times T \times 1013/273 \times P$$

T = absolute temperature (K)

P = atmospheric pressure (mb)

For particles, the mass per unit volume approach is used. Particle deposition in the respiratory tract depends upon the dimensions of the particles (WHO 1979a). Thus, in describing the particle loading of the air, information on the distribution of particle size should be given in addition to the mass concentration. A representative size distribution of urban particulate matter is provided in figure 2.1. It may also be important to specify the number of particles present in each of several specific size ranges per unit volume of air.



Source: USEPA, 1996

Figure 2. 1. Representative example of a mass distribution of ambient PM as function of aerodynamic particle diameter. A wide-ranging aerosol collector (WRAC) provides an estimate of the full coarse mode distribution. Inlet restrictions of the high volume sampler for TSP, the PM<sub>10</sub> sampler, and the PM<sub>2.5</sub> sampler reduce the total mass reaching the sampling filter.

The distribution of sizes of particles arising from each source of aerosols has been shown to follow a log-normal distribution: thus the geometric mean (or median) diameter and the geometric standard deviation are often quoted and specify the distribution. In defining the median diameter of the particles of an aerosol it should be specified whether this value reflects the mid-point of the distribution of the mass or number of particles present. Thus the Mass or Count Median Diameter (MMD or CMD) should be specified. An additional refinement involves adjusting for the aerodynamic properties of the particles and so the Mass or

Count Aerodynamic Median Diameters should be quoted. In naturally occurring aerosols the geometric standard deviation tends to vary from about 2 to 4  $\mu\text{m}$  with 84% of the distribution being of size less than that specified by the median diameter multiplied by the geometric standard deviation. It is incorrect to refer to the median diameter of a single particle: the term refers to the distribution of sizes present in an aerosol cloud.

Current techniques used to measure the mass concentration of particles in air make use of size-specific sampling devices and thus the mass of particles of less than 10  $\mu\text{m}$  diameter may be determined ( $\text{PM}_{10}$ ) as an index of the mass concentration of particles that can penetrate into the human thorax. Sampling devices allow a fairly definite separation of particles of greater or less than the specified size. To be precise, the percentage of particle mass in the specified size range accepted by the sampling system, should be specified (e.g. 10  $\mu\text{m}$  in the above example). A  $\text{PM}_{10}$  sampling head accepts 50% of particles of aerodynamic diameter exactly 10  $\mu\text{m}$ , the acceptance fraction rising rapidly for particles of smaller diameter and declining rapidly for particles of greater diameter.

The mass concentration of particles of less than 2.5  $\mu\text{m}$  diameter ( $\text{PM}_{2.5}$ ) is a means of measuring the total gravimetric concentration of several chemically distinctive classes of particles that are emitted into or formed within the ambient air as very small particles. In the former category (emitted) are carbonaceous particles in wood smoke and diesel engine exhaust. In the latter category (formed) are carbonaceous particles formed during the photochemical reaction sequence that also leads to  $\text{O}_3$  formation, as well as sulphate and nitrate particles resulting from the oxidation of  $\text{SO}_2$  and nitrogen oxide released during fuel combustion and their reaction products.

The coarse particle fraction, i.e., those particles with aerodynamic diameters larger than about 2.5  $\mu\text{m}$ , are largely composed of soil and mineral ash that are mechanically dispersed into the air. Both the fine and coarse fractions are complex mixtures in a chemical sense. To the extent that they are in equilibrium in the ambient air, it is a dynamic equilibrium in which they enter the air at about the same rate as they are removed. In dry weather, the concentrations of coarse particles are balanced between dispersion into the air, mixing with air masses, and gravitational fallout, while the concentrations of fine particles are determined by rates of formation, rates of chemical transformation, and meteorological factors. Concentrations of both fine and coarse particulate matter are effectively depleted through in-cloud and below-cloud scavenging by precipitation. Further elaboration of these distinctions is provided in Table 2.1.

**Table 2.1. Comparisons of ambient fine and coarse mode particles**

	<b>Fine Mode</b>	<b>Coarse Mode</b>
Formed from:	Gases	Large solids/droplets
Formed by:	Chemical reaction; nucleation; condensation; coagulation; evaporation of fog and cloud droplets in which gases have dissolved and reacted.	Mechanical disruption (e.g. crushing, grinding, abrasion of surfaces); evaporation of sprays; suspension of dusts.
Composed of:	Sulphate, $SO_4^{2-}$ ; nitrate $NO_3^-$ ; ammonium, $NH_4^+$ ; hydrogen ion, $H^+$ ; elemental carbon; organic compounds (e.g., PAHs); metals (e.g. Pb, Cd, V, Ni, Cu, Zn, Mn, Fe); particle-bound water.	Resuspended dusts (e.g., soil dusts, street dust); coal and oil fly ash, metal oxides of crustal elements (Si, Al, Ti, Fe); $CaCO_3$ , NaCl, sea salt; pollen, mould spores; plant/animal fragments; tire wear debris
Solubility	Largely soluble, hygroscopic and deliquescent	Largely insoluble and non-hygroscopic
Sources	Combustion of coal, oil, gasoline, diesel, wood; atmospheric transformation products of $NO_x$ , $SO_2$ and organic compounds including biogenic species (e.g. terpenes) high temperature processes, smelters, steel mills, etc.	Resuspension of industrial dust and soil tracked onto roads; suspension from disturbed soil (e.g. farming, mining, unpaved roads); biological sources; construction and demolition; coal and oil combustion; ocean spray
Lifetimes	Days to weeks	Minutes to hours
Travel Distance	100s to 1000s of kilometres	< 1 to 10s of kilometres

Source: USEPA (1996a, b)

As indicated in Table 2.1, fine and coarse particles generally have distinct sources and formation mechanisms, although there may be some overlap. Primary fine particles are formed from condensation of high temperature vapours during combustion. Secondary fine particles are usually formed from gases in three ways:

1. Nucleation (i.e., gas molecules coming together to form a new particle).
2. Condensation of gases onto existing particles.
3. By reaction of absorbed gases in liquid droplets.

Particles formed from nucleation also coagulate to form relatively larger aggregate particles or droplets with diameters between 0.1 - 1.0  $\mu m$ , and such particles normally do not grow into the coarse mode. Particles form as a result of chemical reaction of gases in the atmosphere that lead to products that either have a low enough vapour pressure to form a particle, or react further to form a low vapour pressure substance. Some examples include:

1. The conversion of  $\text{SO}_2$  to sulphuric acid droplets ( $\text{H}_2\text{SO}_4$ ).
2. Reactions of  $\text{H}_2\text{SO}_4$  with  $\text{NH}_3$  to form ammonium bisulphate ( $\text{NH}_4\text{HSO}_4$ ) and ammonium sulphate ( $(\text{NH}_4)_2\text{SO}_4$ ).
3. The conversion of  $\text{NO}_2$  to nitric acid vapour ( $\text{HNO}_3$ ), which reacts further with  $\text{NH}_3$  to form particulate ammonium nitrate ( $\text{NH}_4\text{NO}_3$ ).

Although some directly emitted particles are found in the fine fraction, secondary particles formed from gases dominate the fine fraction mass. By contrast, most of the coarse fraction particles are formed directly as particles, and result from mechanical disruption such as crushing, grinding, evaporation of sprays, or suspension of dust from construction and agricultural operations. Basically, most coarse particles are formed by breaking up bigger masses into smaller ones. Energy considerations normally limit coarse particle sizes to greater than  $1.0\ \mu\text{m}$  in diameter. Some combustion-generated mineral particles, such as fly ash, are also found in the coarse fraction. Biological material such as bacteria, pollen, and spores may also be found in the coarse mode.

In general, fine and coarse particles exhibit different degrees of solubility and acidity. With the exception of carbon and some organic compounds, fine particles are largely soluble in water and hygroscopic (i.e., fine particles readily take up and retain water). Except under fog conditions, the fine particle mode also contains almost all of the strong acid. By contrast, coarse mineral particles are mostly insoluble, non-hygroscopic, and generally basic.

Fine and coarse particles typically exhibit different behaviour in the atmosphere. These differences affect several exposure considerations including the representativity of central-site monitored values and the behaviour of particles that were formed outdoors after they penetrate into homes and buildings where people spend most of their time.

Fine accumulation mode particles typically have longer atmospheric life times (i.e. days to weeks) than coarse particles, and tend to be more uniformly dispersed across an urban area or large geographic region. Atmospheric transformations can take place locally, during atmospheric stagnation, or during transport over long distances. For example, the formation of sulphates from  $\text{SO}_2$  emitted by power plants with tall stacks can occur over distances exceeding 300 kilometres and 12 hours of transport time; therefore, the resulting particles are well mixed in the air shed. Once formed, the very low dry deposition velocities of fine particles contribute to their persistence and uniformity throughout an air mass.

Larger particles generally deposit more rapidly than small particles; as a result, total coarse particle mass is less uniform in concentration across a region than that of fine particles. The larger coarse particles ( $> 10\ \mu\text{m}$ ) tend to fall out of the air rapidly and have atmospheric lifetimes of only minutes to hours depending on their size, wind velocity, and other factors. Their spatial impact is typically limited by a tendency to fall out in the nearby downwind area. The atmospheric behaviour of the small particles within the "coarse fraction" ( $\text{PM}_{10-2.5}$ ) is intermediate between that of the larger coarse particles and fine particles. Thus, some of the smaller coarse fraction particles may have lifetimes on the order of days and travel distances of up to 100 km or more. In some locations, source distribution and meteorology affects the relative homogeneity of fine and coarse particles, and in some cases, the greater measurement error in estimating coarse fraction mass precludes clear conclusions about relative homogeneity.

The composition of airborne particles is seldom routinely determined though this can vary significantly from site to site. This is important in interpreting the results of epidemiological studies of the effects of particles on health. Extrapolation from data collected in one country to conditions in another may be unwise unless some comparability of particle composition has been established.

### 2.1.2 Sources of air pollutants

The sources of air pollutants may be divided into anthropogenic and natural. However, as human activity disturbs natural systems, the distinction may become blurred. Natural sources include dust storms, volcanic action, forest fires and the formation of radioactive particles from gases such as radon. Incursions from the stratosphere increase ground level (tropospheric) concentrations of O<sub>3</sub>. For some pollutants, e.g. SO<sub>2</sub>, natural sources exceed anthropogenic sources on a global scale. However, when considering the effects of air pollutants on health, especially in urban areas where population densities are high, anthropogenic sources are very important and are those to which attention is usually directed with a view to control.

Most anthropogenic sources of fine particles, i.e. those less than 2.5 µm in aerodynamic diameter, involve combustion of some sort. Materials of biological origin (e.g. wood, coal and oil) burn in air by virtue of their carbon content. If a substance containing only hydrocarbon compounds burns with complete efficiency, only water and CO<sub>2</sub> are produced. Such combustion demands a stoichiometric ratio of oxygen to fuel and, in practice, is never attained. Unburned fragments of combustible material, semi-volatile organic compounds, which vaporise and subsequently recondense as tarry droplets and incombustible matter are usually emitted as components of smoke during and following the combustion process. Improving the mix of air and fuel and pre-removal of volatile compounds may reduce smoke production. Smokeless fuel is prepared and burnt in this way. If the supply of oxygen is inadequate, large increases in CO production occur.

In most countries, motor vehicles, industrial activity and the generation of electricity account for a large percentage of the anthropogenic production of the oxides of nitrogen and sulphur. These, in addition to CO, particles and VOC are described as primary pollutants in that they are produced directly by the combustion process. Reactions taking place in the troposphere generate secondary pollutants: O<sub>3</sub> is a classic example. NO<sub>2</sub> breaks down photochemically under the action of ultra-violet light to generate NO and atomic oxygen. The latter combines with molecular oxygen to produce O<sub>3</sub>. The presence of peroxy radicals derived from atmospheric reactions of HC and other organic compounds ensures that NO is oxidized back to NO<sub>2</sub> without loss of O<sub>3</sub>. Thus an O<sub>3</sub>-generating series of reactions is established. The formation of O<sub>3</sub> typically occurs as polluted air drifts away from sites of production including urban areas; O<sub>3</sub> may thus occur at large distances from sources of NO<sub>2</sub> and HC.

NO<sub>2</sub> is both a primary and a secondary pollutant. Motor vehicles emit both NO and NO<sub>2</sub>. In the atmosphere, NO is oxidized to the dioxide, slowly by oxygen but rapidly by O<sub>3</sub>. This explains the low concentrations of O<sub>3</sub> generally found close to sources of oxides of nitrogen.

In addition to the above, sulphur in fuel also gives rise to both primary and secondary pollutants. SO<sub>2</sub> is formed by oxidation during combustion. Further oxidation of SO<sub>2</sub> leads to SO<sub>3</sub>, which rapidly undergoes hydration to form sulphuric acid and this, in turn, is neutralized by NH<sub>3</sub> to ammonium bisulphate and ammonium sulphate. These compounds make an important contribution to the ambient fine particle aerosol.

The combustion of oil and petrol in internal combustion engines leads to the release of organic compounds, which condense in the air to produce small particles of the order of 1 µm in diameter. These and the freshly formed sulphuric acid droplets of similar size are described as nucleation mode particles. Such particles have a short lifetime (< 1 hour) and aggregate or agglomerate to produce particles in the 0.2-2.0 µm diameter range which are defined as accumulation mode particles. These particles are stable and long-lived, and may be transported many hundreds of kilometres before being eventually lost from the air, generally as a result of below-cloud scavenging by precipitation.

### **2.1.3 Factors affecting the concentrations of air pollutants**

Local concentrations of air pollutants depend upon the strength of their sources and the efficiency of their dispersion. Day to day variations in concentrations are more affected by meteorological conditions than by changes in source strengths. Under some conditions both factors may play a part: in cold, still weather, dispersion is reduced whilst production is increased by the increased use of domestic space heating.

Wind is of key importance in dispersing air pollutants: concentrations being inversely related to wind speed for ground level sources. Turbulence is also important: a "rough" terrain, as produced for example by buildings, tends to lead to increased turbulence and better dispersion of pollutants.

Temperature inversions are of great importance in controlling the depth of the layer of air adjacent to the ground in which pollutants are well mixed (the mixing depth). As a mass of air rises it is exposed to decreasing atmospheric pressure and expands accordingly. This causes the temperature of the air mass to fall. The rate at which temperature falls with height is described as the adiabatic lapse rate: for dry air the rate of decline of temperature is about 1°C for each 100 m of height. Air saturated with water vapour loses heat more slowly than dry air, since the heat capacity of water vapour is twice that of dry air. As temperature falls and the saturated vapour pressure also falls, water condenses out as droplets and latent heat is released. As air containing water vapour, but not saturated, cools on rising it will reach saturation and thereafter the adiabatic lapse rate will be reduced.

As a mass of air rises it cools but as long as its temperature remains greater than that of the surrounding air it will retain buoyancy and continue to rise. Conversely if the actual temperature falls more slowly than that of the mass of air, or even increases, the cooling air will rapidly become heavier than the surrounding air and it will fail to rise. Consequently, a temperature inversion occurs when the air temperature rises with height above the ground.

At night, with low wind speeds and clear skies, rapid cooling of the ground and the adjacent air causes air to be coldest close to the ground and thus air cannot rise. Polluted air will not rise in the layer in which the usual temperature gradient is reversed and thus the concentration of pollutants in this layer will increase, sometimes leading to a thick layer of polluted air close to the ground.

Temperature inversions occur in summer as well as in winter. With strong sunlight and high traffic density, temperature inversions contributed to the high incidence of photochemical smog first described in the early 1950s in Los Angeles and now seen commonly in other large cities surrounded by mountains, such as Mexico City, Sao Paulo, and Caracas.

### **2.1.4 Exposure to air pollutants**

The total daily exposure of an individual to air pollution is the sum of the separate contacts to air pollution experienced by that individual as he/she passes through a series of environments during the course of the day (also called micro-environments, e.g. at home, while commuting, in the streets, etc.). Exposures in each of these environments can be estimated as the product of the concentration of the pollutant in question and the time spent in the environment. In this model, the concentration of pollutants is assumed to be approximately constant during the time a person spends time in it. Exposure should not be confused with dose: i.e., the amount of pollutant absorbed. As the number of micro-environments studied is increased, a better estimate of total daily exposure is produced. The daily average concentration of a pollutant recorded at a single, fixed-site outdoor monitoring point provides only a very approximate guide to actual exposure.

One obvious and important micro-environment is the indoor environment where the types and concentrations of pollutants may be very different from those outdoors. For example, O<sub>3</sub> concentrations are generally much lower indoors in the absence of indoor sources, and O<sub>3</sub> penetrating from outdoors is destroyed by reaction

with interior surfaces. By contrast, indoor concentrations of a chemically non-reactive fine particle such as sulphate may reach 90% of those outdoors. For some pollutants, indoor concentrations usually exceed outdoor concentrations.

In some cool climate countries people living in urban areas spend as much as 90% of their time indoors; this should be considered in interpreting the results of epidemiological studies relating outdoor concentrations of pollutants to effects on health. In other countries where climates are warm and many occupational activities are conducted outdoors, the percentage of the day spent indoors may be very much less. In some developing countries, indoor air pollution may be much higher than outdoor air pollution due to use of biomass fuels in open stoves (Section 4.2).

Besides varying temporally, outdoor concentrations of air pollutants vary from place-to-place. For example, concentrations of primary pollutants generated by motor vehicles decline rapidly as one moves away from busy roads. However, concentrations of pollutants generated by motor vehicles may be significantly higher inside motor vehicles than indicated by single site monitors and thus the motor car may itself be a significant micro-environment. Some pollutants are comparatively evenly distributed across large areas: O<sub>3</sub> and fine particles are examples. For such pollutants, monitoring at a limited number of sites may provide an adequate indication of concentrations over wide regions.

Personal monitoring devices have been developed for some pollutants. At their simplest these provide an integrated assessment of personal exposure over a given period. An overview of some aspects of the technology of monitoring devices is provided in Chapter 5.

### **2.1.5 Health significance of air pollution**

Exposure to air pollution is probably as old as human exposure to fire. There is a large amount of archaeological evidence that indoor air pollution must have been troublesome to early humans, who used fire in confined spaces (Brimblecombe 1987). The classical writers record the oppressive fumes of Rome. Attention to effects of air pollution on health was focused during the early and mid 20th Century by a series of air pollution episodes, which produced dramatic effects on health. The Meuse Valley in Belgium (1930), Donora in the USA (1948) and London, England (1952) all experienced air pollution episodes which were investigated in some detail. In the 1952 London air pollution episode it was estimated that 4000 extra deaths occurred as a result of a smog largely consisting of high concentrations of SO<sub>2</sub> and particulate matter (Brimblecombe 1987), and in Donora some 43% of the population were affected by symptoms including headache, eye irritation, dyspnoea and vomiting. Analysis of the London episode showed that the elderly, especially those suffering from pre-existing cardio-respiratory disorders and the very young were at greatest risk. Later studies demonstrated a decline in urban levels of chronic bronchitis as concentrations of air pollutants fell (Chin et al 1981).

Emphasis on severe episodes of pollution may have distracted attention from the effects of long term exposure to pollution. Studies in London in the 1950s and 60s showed that the self-reported state of health of a panel of patients suffering from chronic bronchitis varied with day-to-day levels of air pollution (Waller 1971). It was noted, using simple methods of analysis, that symptoms did not increase unless the concentrations of smoke (measured as Black, or British Smoke) and SO<sub>2</sub> exceeded 250 and 500 µg/m<sup>3</sup>, respectively. It is likely that, had more searching methods of analysis been applied, effects would have been seen at lower concentrations.

Since the 1950s a great body of evidence has accumulated showing that air pollutants have a damaging effect on health. Some of the key studies are reviewed in Chapter 3 of this publication. Two especially important groups of studies will be dealt with briefly here as they have played an important role in the formulation of these guidelines.

When the WHO Air Quality Guidelines for Europe were developed in 1987 (WHO 1987) emphasis was placed on the results of studies of volunteers exposed to air pollutants under controlled conditions. Where such studies demonstrated a Lowest Observed Effect, or Adverse Effect Level this was used as a starting point for deriving the relevant air quality guideline. Epidemiological studies that demonstrated a threshold of effect were used in the same way.

A new database of epidemiological studies emerged in the late 1980s and 1990s. This database of time-series studies was developed first in the United States and later in Europe and other areas (Schwartz *et al* 1996). In essence the time series approach takes the day as the unit of analysis and relates the daily occurrence of events, such as deaths or admissions to hospital, to daily average concentrations of pollutants whilst taking careful account of confounding factors such as season, temperature and day of the week. Powerful statistical techniques have been applied and coefficients relating daily average concentrations of pollutants to effects have been produced. The results of these studies have been remarkably consistent and have withstood critical examination well (Samet *et al.* 1995). Such methods cannot, of course, be expected to prove the possible or probable causal nature of the associations demonstrated, but detailed examination of the data and application of the usual tests for likelihood of causality have convinced many that it would be unwise to disregard the findings.

Associations have been demonstrated between daily average concentrations of particles, O<sub>3</sub>, SO<sub>2</sub>, airborne acidity, NO<sub>2</sub>, and CO. The associations for each of these pollutants were not significant in all studies though, taking the body of evidence as a whole, the consistency is striking. More remarkable than the consistency of the results was the demonstration of associations at levels of pollution hitherto expected to be quite safe: indeed, below the levels recommended in the 1987 WHO Air Quality Guidelines for Europe.

For particles and O<sub>3</sub> it has been accepted by many that the studies provide no indication of any threshold of effect. This was reflected in the tables relating small differences in daily concentrations of particles and O<sub>3</sub> to effects on health (Chapter 3).

In time - series studies, daily counts of events are related to the daily average concentration of pollutants measured, usually at a single, fixed, monitoring site or predicted from such measurements. In any city it is likely that there will be a distribution of personal exposure across the population. Thus, on days when the measured or predicted level of pollution is low, some individuals may be exposed to greater than the reported concentration. If such exposure exceeded some threshold then effects would be recorded and attributed to occurring as a result of exposure to the recorded or predicted concentration. It might then be asked whether time series studies are capable of discerning a threshold of effect, especially if the threshold is low. This problem is by no means limited to particulate matter and O<sub>3</sub>: similar difficulties in identifying a threshold of effect at a population level apply to lead. This is an important point with regard to defining an air quality guideline based on such data: it is unlikely that a single guideline value can be derived from such a database and thus the "guideline" should be accepted to be a relationship relating events to airborne concentrations. This is a significant departure from the concept of a guideline value as a level of exposure at which the great majority of people, even in sensitive groups, would be unlikely to experience any adverse effects. Translation of this new form of guideline into an air quality standard is likely to be difficult. Junker and Schwela further discussed this issue in some detail (Schwela and Junker 1978; Junker and Schwela 1998).

Time-series studies relate the concentrations of air pollutants to their effects on health: in fact they provide the slope of a regression line relating concentrations to health events. There are no grounds for simple extrapolation of the concentration-exposure relationship to high levels of pollution. Several studies have shown that the slope of regression line is reduced when the annual average concentration of pollution is high (Schwartz and Marcus 1990).

Elevations in daily rates for various adverse health outcomes are sometimes referred to as the acute effects of air pollutants. For example, an increase in pollutant concentration might cause an increase in asthma attacks. It is assumed that without an increase in pollution, neither would asthma attacks increase. It is also

likely that long-term exposure to air pollution produces chronic effects on health. For example, lifelong exposure to air pollution in England amongst those born in the late 19th Century is likely to have increased their chances of developing chronic bronchitis and dying earlier than expected as a result of the illness. (Chinn et al. 1981). In the United States, cohort studies in a range of towns have demonstrated associations between long term average concentrations of fine particles (PM<sub>2.5</sub> and sulphates) and the Standardized Mortality Ratios of communities (Dockery *et al* 1993; Pope *et al* 1995). Attempts to estimate the public health impact of air pollution have been made on the basis of both the cohort studies and the time-series studies. On the basis of one of the cohort studies (Pope et al. 1995), Brunekreef (1997) has reported that exposure to current levels of air pollution in the Netherlands may lead to a average reduction in longevity of 1 year. Work reported from the United States reports slightly larger effects: perhaps 2 years are lost in polluted communities compared with unpolluted areas of the United States. Loss of life expectancy may be distributed statistically across the affected population. This is the case amongst cigarette smokers where the average loss of life expectancy is of the order of 3-5 years, though some smoking-related deaths occur among people in their forties.

## **2.2 Air pollutant concentrations and factors affecting susceptibility**

The concentrations of emitted pollutants and population exposures to air pollution vary substantially from country to country. In addition, human responses to air pollutant exposure also vary. Outdoor and indoor concentrations of air pollutants, and a number of examples of factors affecting responses to pollutants, are considered in this section.

### **2.2.1 Concentrations of classical pollutants in ambient air**

There are far more data available on the ambient outdoor concentrations of certain classic air pollutants in many countries around the world than for any of the other pollutants, as monitoring records on black smoke (BS) and SO<sub>2</sub> in particular go back for five decades or more. There are, however, relatively few locations where all of the classical air pollutants have been measured simultaneously, or over extended periods. Additionally, historical data are often of limited value for retrospective or cross-sectoral analyses of air quality and health. Only recently, and only for a limited number of sites, have the specificity of analyses, the validity of calibrations, the identification of site representativity for the specific sampling purpose, the consistency of averaging times and/or sampling intervals, and the frequency and data management procedures been standardized to appropriate quality assurance procedures (see Chapter 5).

Available air pollutant concentration data were reviewed by the Task Group and selected data summaries are presented here to give the reader some general perspectives on recent pollutant levels and trends in the various WHO regions. The presentation is organized into three categories for each of the classical air pollutants.

The first category contains data on air quality in the European region on the basis that these summary data provided key input to the *Air Quality Guidelines for Europe* (WHO 1999a), which in turn provided the basis for the WHO *Guidelines for Air Quality* summarized in this publication. These data underwent a limited peer review by the WHO/EURO Working Group that judged them to be sufficiently representative and reliable for inclusion in the *Air Quality Guidelines for Europe*.

The second category contains data on ambient outdoor air quality in other WHO regions that were collected from countries by representatives from those regions on the WHO Global Air Quality Guidelines Task Group. In most cases, the Task Group was not able to assess data quality. Thus, it was not possible to endorse these data in terms of their accuracy and representativity. Although some data may be of high quality, some of the data were based on intermittent sampling programmes and cannot be reliably used for determining longer-term average concentrations.

Despite these limitations, the Task Group considered that presentation of some of the available summary data would provide a valuable frame of reference for the readers of this report. Accordingly, for each WHO Region other than Europe, a restricted set of data was selected for this report. Wherever possible, they represent: (a) at the one extreme, data for point source monitoring in regions as being representative of high-end human exposures; (b) non-typical levels, selected from data for urban sites not greatly affected by industrial point sources; and (c) at the other extreme, non-urban site data selected as being representative of the low end of concentrations for the country.

Each region was represented by data from a limited number of countries in that region, that differed in size and extent of industrial development, to demonstrate the extent of potential exposure of people in that region. Wherever possible, data that are summarized include available information on the source of the data, averaging times, and the quality assurance procedures followed in producing the data.

The third category of data consists of summaries from the WHO Air Management Information System (AMIS) programme (WHO 1997b, WHO 1998b). Since AMIS collects data from collaborating centres in all WHO regions, there is some overlap in coverage with the data summarized in the regional reports within the second category. The primary justification for including the AMIS data as a separate category is that the procedures used to generate and report these data are more uniform and were subjected to more validation, providing an independent source of data of assured quality.

An examination of the data summaries that follow clearly shows that air quality in large cities in many developing countries is remarkably poor, and that very large numbers of people in those countries are exposed to ambient concentrations of air pollutants well above the WHO *Guidelines for Air Quality*.

### ***AIR QUALITY DATA IN DEVELOPING COUNTRIES***

The main source of information on air pollution in developing countries is the Air Management Information System AMIS (WHO 1997b) set up by the WHO as a continuation of GEMS/Air (UNEP/WHO 1993). AMIS is based on voluntary reporting of data by municipalities of the WHO member states. The AMIS core data base collects information on annual (arithmetic) mean and high (95-, 98-) percentiles of daily mean concentrations of SO<sub>2</sub>, NO<sub>2</sub>, O<sub>3</sub>, CO, SPM, lead and other potentially monitored compounds. In principle, data from three types of monitoring stations are stored: "industrial," reflecting levels in areas affected by emissions from industry; "city center / commercial," which will be mostly affected by traffic; and "residential," which should reflect the best basic level of population exposure. Until now the coverage of the system has been limited to 100 cities, but the intention is to acquire current information from some 300 cities by the end of 2000. The analysis of the data and its limitation is discussed by Krzyzanowski and Schwela (1999).

### ***AIR POLLUTION LEVELS AND TRENDS***

#### **Sulphur dioxide**

In most analysed cities, the annual mean concentrations of SO<sub>2</sub> in residential areas have not exceeded 50 µg/m<sup>3</sup>. Notable exceptions are several cities in China, with the SO<sub>2</sub> concentration of 330 µg/m<sup>3</sup> in Chongqing and 100 µg/m<sup>3</sup> in Beijing in 1994. In some Chinese cities, the levels reported from "residential" locations exceed those from "commercial" regions of the city and are comparable with the levels in industrial zones. This may reflect the impact of combustion of sulphur-containing coal for domestic heating and cooking.

High levels of SO<sub>2</sub> may also be seen in other developing countries, especially in those with cold winters, as illustrated by the report from Nepal (Sharma 1997). Daily mean concentration of SO<sub>2</sub> was in the range 273 - 350 µg/m<sup>3</sup> in residential areas of Kathmandu during September - December 1993. In monitoring sites close to main roads, the reported range is 310-875 µg/m<sup>3</sup>, reflecting the influence of emissions from

traffic. More than half of the vehicles registered in the city are equipped with two-stroke engines and many are old and ill maintained.

In most of the cities with data allowing trend assessment a decline in mean annual SO<sub>2</sub> concentration was seen over the 1990s. The most dramatic reduction of air pollution with SO<sub>2</sub> was reported from Mexico City, where the concentration in various residential areas dropped from 100-140 µg/m<sup>3</sup> in 1990-1991 to 32-37 µg/m<sup>3</sup> in 1995-1996. In the most polluted Chinese cities an annual means declined between 1% and 10%.

### **Suspended particulate matter**

The most commonly monitored and reported indicator of this type of air pollution is the mass concentration of TSP. In most of the cities, the TSP annual mean concentration exceeds 100 µg/m<sup>3</sup>, with the levels exceeding 300 µg/m<sup>3</sup> in several cities of China and India. There is no evidence of any overall systematic and significant change in TSP levels: the data from the 1990s show increasing as well as decreasing trends in a similar number of cities. The most visible relative decrease of TSP concentrations is shown by the data from Bangkok, but the progress is not steady there either. More consistent, though with a smaller relative rate, is the decrease in TSP concentration in Mexico City. The opposite tendency can be seen in some Chinese cities, with the most rapid increase of TSP concentration in Guangzhou (from less than 150 µg/m<sup>3</sup> in 1990-1992 to more than 300 µg/m<sup>3</sup> in more recent years).

In a limited number of cities reporting data to AMIS levels of PM<sub>10</sub> are also measured. The most commonly registered annual average PM<sub>10</sub> levels ranged from 50 - 100 µg/m<sup>3</sup> in the years 1995-1996. The highest concentrations, exceeding 250 µg/m<sup>3</sup>, were observed in Calcutta and New Delhi. In most towns with high PM<sub>10</sub> average in the last year, an increase in the pollutant concentration was seen over the 1990s. In most cases, this increase has occurred even when the decrease in TSP was reported. An opposite trend and a decrease in PM<sub>10</sub> level were seen in the Central and Southern America cities. In Mexico City, the relative decrease in PM<sub>10</sub> was faster than that of TSP.

This limited information on the size-specific particulate pollution allows a comparison of the mass concentration of TSP and PM<sub>10</sub>. For most sites and years with data on both indicators, the PM<sub>10</sub> to TSP ratio was in the range between 0.4-0.8. However, in a few cases, the ratio exceeded 1.0. This fact indicates that the measurements reported to AMIS might not have been done at the same locations and/or periods. In a southeastern part of Mexico City, the ratio remained between 0.25 and 0.32 in all years 1991-1996, while in the southwestern part of the city it was consistently between 0.44 and 0.55. More specific studies of the size distribution of airborne particles, conducted in the northern cities of China in the mid-1980s, indicate that some 70% of the mass concentration of TSP are due to the PM<sub>10</sub> (Ning et al 1996). During the heating season, particles with diameter less than 2 µm were found to make some 30-50% of TSP. The elemental analysis of the particles confirmed that human activities are the main source of the fine fraction of particulate matter. Similar results were reported from Jakarta, where particles with diameter less than 7.2 µm contributed more than 80% of TSP (Zou et al 1997). Traffic-related compounds contributed significantly to the overall pollution mass, and especially to the fine particle fraction.

### **Nitrogen dioxide**

In most of the cities reporting to AMIS the annual mean concentrations of NO<sub>2</sub> remain moderate or low, not exceeding 40 µg/m<sup>3</sup>. However, in Mexico City and in Cape Town, the annual average of 70 µg/m<sup>3</sup> has been exceeded regularly in the 1990s. A paper based on data from centrally located monitors in Sao Paulo indicates annual mean of 240 µg/m<sup>3</sup> in 1990/91 (Saldiva et al. 1995). The trends vary between the cities, but a 5-10% annual increase in concentration of this pollutant was more common than a decrease.

The observed pattern is consistent with the volume of car traffic in each city. The highest pollution levels, and the increasing trends, are observed in the cities with high and increasing car traffic. In Southern Asia or in Latin America, this high NO<sub>2</sub> concentration combined with the intense UV radiation results in photochemical smog with high oxidant concentrations. It is illustrated by the analysis of temporal and spatial patterns of tropospheric O<sub>3</sub> in New Delhi (Singh et al 1997). The build-up of O<sub>3</sub> over the day is faster than scavenging of O<sub>3</sub> by the NO<sub>2</sub>. In Mexico City, the mixture of high NO<sub>2</sub> emissions from gasoline combustion and intense UV radiation is the cause of the notorious photochemical smog in that city. According to the data reported to AMIS, the O<sub>3</sub> concentration exceeded a concentration of 120 µg/m<sup>3</sup> in over 300 days a year in 1994-96, and the 95<sup>th</sup> percentile of maximum daily 1-hour average O<sub>3</sub> concentration was around 500 µg/m<sup>3</sup>. Some decrease was seen, however, in the annual mean O<sub>3</sub> concentration, indicating slow improvement of air quality in non-extreme days.

### **2.2.2 Factors affecting susceptibility to indoor air pollution**

Indoor air pollutants usually differ in type and concentration from outdoor air pollutants. Indoor pollutants include environmental tobacco smoke, biological particles (such as pollen, mites, moulds, insects, microorganisms, pet allergens etc.), non-biological particles (such as smoke), VOC, NO<sub>x</sub>, lead, radon, CO, asbestos, various synthetic chemicals and others. Degradation of indoor air quality has been associated with a range of health effects, including discomfort, irritation, chronic pathologies and various cancers.

With growing public concern about indoor air quality, action has been taken in many developed countries to characterize levels of indoor air pollutants, to improve ventilation and fuel emissions, and to reduce exposure to environmental tobacco smoke, biological contamination and radon among other actions. Even though there is considerable evidence that indoor air quality is a serious and widespread problem in many developing countries, the information and resources to control indoor air quality are often lacking (Ferrari et al. 1995). Management of indoor quality is discussed in section 6.2.

Perhaps the most important factor that causes qualitatively and quantitatively different exposures to air pollutants across different countries is that of indoor heating and cooking by solid fuel burning. This topic deserves special attention. The emissions, concentrations, exposures and health effects are discussed in detail in Chapter 4. On a global scale, biomass fuels (wood, crop residues, dung and grass) are used daily in about half the world's households as energy for cooking and/or heating. In China, for example, it has been estimated that coal burning results in particle concentrations up to 5000 µg/m<sup>3</sup> in indoor living areas, whereas smoky houses in Nepal and Papua New Guinea have peak levels of 10 000 µg/m<sup>3</sup> or more (Smith 1996). An unknown, but significant, proportion of this activity takes place in conditions where much of the airborne effluent is released into the living area. Therefore, some of the highest concentrations of particulate matter other pollutants occur in rural, indoor environments in developing countries.

Biomass smoke contains significant amounts of several important pollutants: CO, particles, HC and to a lesser extent, NO<sub>x</sub>. However, biomass smoke also contains many organic compounds, including PAH, that are thought to be toxic, carcinogenic, mutagenic or otherwise of concern. Coal smoke contains all of these as well as additional pollutants, e.g. sulphur oxides and heavy metals such as lead. In many parts of the world these pollutants are released from stoves in poorly ventilated homes or in enclosed courtyards. Due to the high concentrations and the large populations involved, the total human exposure to many important air pollutants can be much higher in the homes of the poor in developing countries than in the outdoor air of cities in the developed world.

### **2.2.3 Meteorological factors**

At increased altitude the partial pressure of oxygen falls and inhalation increases in compensation. For gaseous pollutants no increase in effects over those experienced at sea level would be expected as a result of

the increased inhalation, as the partial pressures of the pollutant gases will fall in line with that of oxygen. For particles, on the other hand, increased inhalation volumes will lead to increased intake of airborne particles and perhaps changes in patterns of deposition. Differences in effects between those who have always lived at high altitude and those who have recently relocated there might be expected.

Temperature has a very significant effect on health and has been shown to be an important confounding factor when examining the effects of air pollutants. The relationship between ambient temperature and ill health is "U"- or "V"-shaped with excess daily deaths increasing in both cold and hot conditions. Local populations tend to be acclimatised to local conditions and cope better with changes in temperature than do immigrants from other countries. The effect of low temperatures in winter is more marked in countries with temperate climates, than in much colder countries. Inhaled volumes increase under hot conditions, and thus the intake of pollutants also increases. In addition, warm days encourage people to spend more time out-of-doors and so personal exposure patterns to pollutants may change. Of course, living in well-ventilated houses in warm weather, when doors and windows may be open, may decrease exposure to pollutants from indoor sources and increase exposure to outdoor sources.

Humidity is unlikely to have a significant effect on the toxicity of gaseous pollutants, and it may reduce the effects of some particles by permitting hygroscopic growth in particle size prior to inhalation, thus changing the patterns of deposition from smaller to larger airways in the lung.

#### **2.2.4 Demographic factors**

The age structure of populations differs markedly from country to country. Old people tend to show increased susceptibility to air pollution as a result of reduced functioning of physiological defence mechanisms, reduced physiological reserves and the increased prevalence of disease. Very young children may also be at increased risk due to incompletely developed defence mechanisms, higher ventilation rates per unit body mass and a tendency to spend more hours out of doors than adults.

#### **2.2.5 Socio-economic factors**

People with a poor standard of living suffer from nutritional deficiencies, from infectious diseases due to poor sanitation and overcrowding, and tend to be provided with a poor standard of medical care. Each of these factors may render individuals more susceptible to the effects of air pollution. A dietary lack of anti-oxidant factors may decrease defence mechanisms against oxidant pollutants such as O<sub>3</sub> and NO<sub>2</sub>. Delayed clearance of particles in airways already damaged by infection is likely. In developing countries, poor air quality may be closely associated with the incidence of infectious diseases.

#### **2.2.6 Effects of differing levels of disease in the population**

Diseases which produce narrowing of the airways, a reduction in the area of the gas-exchange surface of the lung and an increased alteration of inhalation-perfusion ratios are likely to make the subject more susceptible to the effects of a range of air pollutants. Epidemiological studies have shown that patients suffering from asthma or chronic obstructive pulmonary disease suffer an increase in symptoms when levels of pollutants are raised (see Chapter 3). It should be noted that asthma is less common in developing than in developed countries. However, the prevalence of infectious disease in developing countries, including tuberculosis, may militate against the development of the IgE antibody response, which is characteristic of asthma.

#### **2.2.7 Specific differences in prevalence levels of air pollutants**

Concentrations of air pollutants vary greatly from country to country. In countries where indoor air pollution is common, due to cooking over open fires with poor ventilation, indoor exposure may be an important cause of damage to health especially among women. In other countries, including those of the Middle East, particle

concentrations in outdoor air are high due to wind-blown dust. In desert areas this dust contains a high proportion of silica, and silicotic nodules have been described in the lungs of residents. However, high concentrations of volcanic ash do not seem to be associated with acute effects on health. Specific examples of the levels of ambient urban air quality, and indoor air quality in various countries around the world are provided by the AMIS (WHO 1997b; WHO 1998b).

Countries burning brown coal (or lignite) for domestic heating are likely to experience high concentrations of smoke and SO<sub>2</sub>. To these may be added the pollutants produced by motor vehicles. Leaded motor vehicle fuel is in use in many parts of the world and in these areas airborne lead particles make an important contribution to total lead intake both by inhalation and by ingestion.

### **2.3 Exposure to air pollutants**

An ideal characterization of the distribution of human exposures would be based on direct measurements of each pollutant concentration in the breathing zone of each member of a representative cross section of the population of interest. At present, however, such a programme is technically impossible and probably impractical as well. Instead, ambient air quality measurements at central, fixed, air monitoring sites are widely used surrogates for population exposures, and are generally the only widely available quantitative resource that can be related to exposures. Personal monitors for exposure estimates could overcome some of the shortcomings of ambient air monitors, but they can be applied only in a small sample of the population.

There are many factors that can account for the substantial differences between the concentrations of pollutants measured at central sites and those in the breathing zone of residents of the community. Air pollutants emitted into outdoor air can be attenuated during infiltration into indoor air. This attenuation can be expected to be minimal for all pollutants of outdoor origin when barriers such as windows and doors are open or absent. In contrast, attenuation can be very large for tightly sealed buildings during times of maximal heating or cooling needs.

The attenuation of indoor air pollutant concentrations by removal to indoor surfaces is highly dependent on the physico-chemical characteristics of the pollutant. At one extreme is a chemically stable fine particle component such as sulphate ion, where indoor concentrations are typically 90% or more of outdoor concentrations. At the other extreme, indoor concentrations can be low for larger particles deposited by sedimentation in the relatively still air.

For a relatively non-reactive gas, such as CO, the indoor-outdoor concentration ratio is usually near unity in a home without indoor CO sources. However, indoor concentrations can be much higher than outdoor concentrations when there are sources such as burning cigarettes and open flames used for cooking or space heating. By contrast, chemically reactive gases, such as O<sub>3</sub> and SO<sub>2</sub> fairly rapidly diffuse to, and react with, interior surfaces. As a result, indoor-outdoor concentration ratios are typically much lower than unity.

Lead is the only classic air pollutant that can gain access to humans through indirect transport routes. Where leaded motor vehicle fuels are used, fine particle emission from vehicle exhausts can be inhaled. In addition, the particles that deposit on terrestrial surfaces can be ingested, either directly from soil in play yards, or after being carried indoors as a component of house dust. Furthermore, particulate lead deposited on plants or agricultural fields can be retained in food products and add to body burdens. Similar pathway considerations also apply to toxic air pollutants other than lead.

Humans engage in a variety of daily activities, and the concentrations of air contaminants in their breathing zone can vary substantially as they move through various microenvironments, each of which may be affected by different attenuation factors or increments from indoor sources. Furthermore, even a complete knowledge of the concentrations of all relevant pollutants in each microenvironment would

not provide an adequate basis for predicting physiological and pathological responses to their exposures. Pollutant uptake could also be greatly affected by ventilation rate and pattern, entry of air via the nose or mouth, airway sizes (which exhibit great individual variability), past and current history of exposure to other toxicants (such as cigarette smoke), and prior disease histories and genetic predispositions. Many of these factors can be modelled and such models have been used for estimating dose distributions associated with ambient air concentrations.

### **Sulphur dioxide**

SO<sub>2</sub> is a colourless pungent, irritating, water-soluble and reactive gas. Concentrations in ambient air in cities of developed countries have mostly decreased in the last two or three decades due to tighter emissions control, increased use of low sulphur fuels and industrial restructuring. Consequently, high ambient concentrations in earlier decades have been replaced by annual mean concentrations of about 20-40 µg/m<sup>3</sup> in most cities in developed countries and daily means rarely exceed 125 µg/m<sup>3</sup>.

However the situation is more complex in developing countries. In cities, the annual mean concentrations of SO<sub>2</sub> in ambient air may range from very low levels up to 300 µg/m<sup>3</sup> (WHO 1998b). Peak concentrations measured as ten-minute averages may exceed 2000 µg/m<sup>3</sup> under conditions of poor atmospheric dispersion such as inversions (Section 2.2.2), or when emissions from a major source are brought to ground levels by certain atmospheric conditions. SO<sub>2</sub> can also reach high concentrations in air in some indoor environments through the use of sulphur containing fuels such as coal for heating and cooking (Section 4.2).

As it is highly reactive, SO<sub>2</sub> has a highly non-uniform dose distribution along the conductive airways of the respiratory tract. For low to moderate tidal volumes and nasal breathing, the penetration into the lungs is negligible. For larger tidal volumes and oral inhalation, doses of interest may extend into segmental bronchi. SO<sub>2</sub> can only reach the gas-exchange region of the lungs after sorption onto fine particles; and the available particle surface is limited except when very large mass concentrations of fine particles are present (WHO 1987; WHO 1994a).

Another special consideration for SO<sub>2</sub> is that there is a great variation in susceptibility to a bronchoconstrictive response. Persons having asthma or atopy can be about 10 times more responsive than healthy subjects.

### **Nitrogen dioxide**

Ambient concentrations of NO<sub>2</sub> in air are variable. Natural background concentrations in ambient air can be less than 1 µg/m<sup>3</sup> to more than 9 µg/m<sup>3</sup>. In ambient air in cities annual mean concentrations can range from 20-90 µg/m<sup>3</sup>, with hourly maximum concentrations from 75-1000 µg/m<sup>3</sup> (WHO 1994a). Concentrations of NO<sub>2</sub> in indoor air can reach average concentrations of 200 µg/m<sup>3</sup> over several days, with hourly maximum concentrations of 2000 µg/m<sup>3</sup> where there are unvented gas heating appliances and poor ventilation (WHO 1994a).

NO<sub>2</sub> is a relatively water-insoluble gas and appreciable amounts of inhaled NO<sub>2</sub> can penetrate to, and elicit biological responses in, small lung airways. As with SO<sub>2</sub>, there is much greater susceptibility to a bronchoconstrictive response in individuals with asthma.

### **Carbon monoxide**

Natural ambient concentrations of CO range between 0.01-0.23 mg/m<sup>3</sup> (WHO 1994a). In urban environments, mean concentrations over eight hours are usually less than 20 mg/m<sup>3</sup>, and one-hour peak levels are usually less than 60 mg/m<sup>3</sup>. Highest concentrations are usually measured near major roads, as

vehicles are the major source of CO. Concentrations of CO can be high in vehicles, underground car parks, road tunnels and in other indoor environments where combustion engines operate with inadequate ventilation. In these circumstances, mean concentrations of CO can reach up to  $115 \text{ mg/m}^3$  for several hours. In houses with unflued combustion heaters, peak CO concentrations can reach up to  $60 \text{ mg/m}^3$  (WHO 1994a).

CO exerts its toxic effects after binding with hemoglobin in the capillaries of the lungs. It is not removed in larger airways.

### **Ozone**

Background concentrations of  $\text{O}_3$  in remote and relatively unpolluted parts of the world are often in the range of  $40$  to  $70 \text{ }\mu\text{g/m}^3$  as a one-hour average. In cities and areas downwind of cities, maximum mean hourly concentrations can be as high as  $300$  to  $400 \text{ }\mu\text{g/m}^3$ . High  $\text{O}_3$  concentrations can persist for 8 to 12 hours per day for several days, when atmospheric conditions favour  $\text{O}_3$  formation and poor dispersion conditions exist (Section 2.2.2).  $\text{O}_3$  is normally at higher concentrations in ambient air outdoors than in indoor air.

$\text{O}_3$  is a relatively water-insoluble gas. It reacts and produces toxic effects on small airway surfaces. The dose-delivery is greatest in terminal and respiratory bronchioles. Unlike  $\text{NO}_2$  and  $\text{SO}_2$ , there is very little difference in lung function responsiveness between asthmatics and healthy subjects. There is, however, a great variability in individual responsiveness that is not yet understood.

### **Particulate matter (PM)**

Concentrations of particulate matter in air are highly variable. In some areas very high levels occur naturally due to wind-blown dust from arid soils. Human activities, such as fires, overgrazing, agricultural practices and mining, can increase particle concentrations in air in remote areas. In Western Europe and North America efforts to control emissions of particulate matter have generally resulted in lower levels of particles in ambient air. In many cities the annual average concentrations of  $\text{PM}_{10}$  are in the range  $20$  to  $50 \text{ }\mu\text{g/m}^3$  for ambient air (WHO 19994a). However, annual average concentrations in some cities in Eastern Europe and in some developing countries can be above  $100 \text{ }\mu\text{g/m}^3$ . Concentrations of  $\text{PM}_{2.5}$  are usually about 45 to 65% of the concentrations of  $\text{PM}_{10}$ .

Concentrations of particulate matter in indoor air can be extremely high when biomass fuels such as wood, crop residues and dung, or coal are used for cooking or heating. Indoor concentrations of up to  $2000$  to  $5000 \text{ }\mu\text{g/m}^3$  of total suspended particulate matter have been measured in some circumstances during cooking with biomass fuels in developing countries (Section 4.2).

Particle size is a critical factor in internal dose distribution. The location of initial deposition in the airways depends on particles size, with coarse particles being deposited in the upper respiratory tract and fine particles being transported to the lower respiratory tract. The rate of deposition in conductive airways also depends on particle size (see Section 2.1).

### **Lead**

Levels of lead found in air, food, water and soil/dust vary widely throughout the world and depend on the degree of industrial development, urbanization and lifestyle factors. Ambient air levels over  $10 \text{ }\mu\text{g/m}^3$  have been reported in urban areas near smelters, whereas lead levels below  $0.1 \text{ }\mu\text{g/m}^3$  have been found in cities where leaded petrol is no longer used. In cities of developing countries traffic-related lead levels range between  $0.3$  and  $1 \text{ }\mu\text{g/m}^3$  with extreme annual mean values between  $1.5$ - $2 \text{ }\mu\text{g/m}^3$ .

Lead is inhaled as fine particles and deposited in the lungs. Since lead uptake by blood is dependent on deposition pattern and solubility (which is influenced by chemical form and particle size), total lead content is only a surrogate for the biologically effective dose. Furthermore, as noted in earlier sections, airborne lead can also reach humans indirectly via deposition on soil and vegetation, and through food chains.

### **Other air pollutants**

In nearly all countries routine air quality monitoring programmes are concentrated almost exclusively on selected classic pollutants. Relatively few of the other air pollutants (considered in detail in Section 3.2) are routinely monitored, except in a few occupational environments. Data are sometimes collected on personal exposures to classic and other air pollutants, but seldom are there standardised protocols for sample collection and analysis, and data processing and storage. As a result, estimates for personal exposures are generally based on highly uncertain models and the assumptions built into them. In general, the situation with respect to ambient concentrations of other air pollutants considered in section 3.2 is characterized as described in the second columns of Tables 3.2 and 3.3 in that section.

## **2.4 Role of guidelines and standards**

The purpose of these guidelines is to provide a basis for protecting public health from the adverse effects of air pollution and for eliminating, or reducing to a minimum, those air contaminants that are known to be, or are likely to be, hazardous to human health and well being (WHO 1987).

These *Guidelines* should provide background information for nations engaged in setting air quality standards, although their use is not restricted to this. The *Guidelines* are not intended to be standards. In moving from guidelines to standards, prevailing exposure levels and environmental, social, economic and cultural conditions in a nation or region should be taken into account (see Section 2.4.4). In certain circumstances there may be valid reasons to pursue policies, which will result in pollutant concentrations above or below the guideline values (WHO 1987).

### **2.4.1 The 1987 WHO Air Quality Guidelines for Europe**

Already in 1958, WHO recognized that air pollution was a threat to the health and well-being of peoples throughout the world. As a consequence, WHO has taken its first steps to marshal the facts and to suggest procedures by which preventive and remedial action may be taken by its member countries, before serious harm is done to the health of their people (WHO 1958). In a forthcoming Technical Report, criteria for guidelines for air quality are described as tests, which permit the nature and magnitude of air pollution on man and the environment to be determined. Guidelines were defined as sets of concentrations and exposure times that are associated with specific effects of varying degrees of air pollution on man, animals, vegetation and on the environment in general (WHO 1964). In 1972, guidance as to the levels of ambient air pollutants that constitute hazards to health were first formulated for the “classic” compounds SO<sub>2</sub>, SPM, CO and photochemical oxidants (WHO 1972). These attempts culminated in 1987 in the publication of the *Air Quality Guidelines for Europe* for a much extended set of air pollutants (WHO 1987).

In the *Air Quality Guidelines for Europe* (WHO 1987), relevant information on the pollutants was carefully considered during the process of establishing guideline values. It was noted that ideally, guideline values should represent concentrations of chemical compounds in air that would not pose any hazard to the human population. However, the realistic assessment of human health hazards necessitated a distinction between absolute safety and acceptable risk. To aim at achieving absolute safety, one would need to know the complete dose-response relationships in individuals in relation to all sources of exposure. Moreover, the type of toxic effect elicited by specific pollutants or their mixtures; the existence (or not) of “thresholds” for specified toxic effects; the significance of interactions; and the variation in

sensitivity and exposure levels within the human population would all have to be known. However, such comprehensive and conclusive data on environmental contaminants are not always available. Scientific judgement and consensus, therefore, play an important role in establishing acceptable levels of population exposure.

### **Criteria for endpoints other than carcinogenicity**

For compounds reportedly without carcinogenic effects, or for which data on carcinogenicity were lacking or insufficient, the starting-point for the derivation of guideline values was to define the lowest concentration at which effects are observed in humans, animals and plants. The difference between the lowest level at which an effect is observed, and the level, at which no effect is observed, is among the factors included in judgements concerning the appropriate margin of protection. In the case of irritant and sensory effects on human, it was considered desirable where possible to determine the no-effect level.

### **Criteria for selection of a lowest-observed adverse-effect level (LOAEL)**

The distinction between adverse and non-adverse effects was stated to pose considerable difficulty. The definition of an adverse effect was given as “any effect resulting in functional impairment and/or pathological lesions that may affect the performance of the whole organism, or which contributes to a reduced ability to respond to an additional challenge”. Even with such a definition, a significant degree of subjectivity and uncertainty was found to be present. To resolve this difficulty, data were ranked in three categories: (i) Single observations, even of potential health concern, were not readily used as a basis for guideline values; (ii) A lowest-observed-effect level might result in pathological change, and therefore was considered a higher degree of health concern; (iii) A substantial change in the direction of pathological effects has had a major influence on guideline considerations.

### **Criteria for selection of uncertainty factors**

The toxicology of pollutants, including the type of metabolites formed, variability in metabolism, or response in humans suggesting hypersusceptible groups, and the likelihood that the compound or its metabolites will accumulate in the body, was taken into account by uncertainty factors. Uncertainty factors were essentially determined through scientific judgement in consensus.

### **Criteria for selection of averaging times**

As a chemical may cause acute, minor, reversible effects after brief exposure, and irreversible or incapacitating effects after prolonged exposure, expert judgement had to be applied, based on the weight of the evidence available. Generally, when short-term exposures lead to adverse effects, short-term averaging times were recommended. In other cases, exposure-response knowledge was sufficient to recommend a long-term average.

### **Criteria for consideration of sensory effects**

Some of the substances selected for evaluation have malodorous properties at concentrations far below those at which toxic effects occur. Although odour annoyance cannot be regarded as an adverse health effect in a strict sense, it affects the quality of life. Therefore, odour threshold levels (detection threshold, recognition threshold, and nuisance threshold) for such chemicals have been indicated where relevant and used as a basis for separate guideline values.

### **Criteria for Carcinogenic Endpoint**

Cancer risk assessment involves a qualitative assessment of how likely it is that an agent is a human carcinogen, and a quantitative assessment of the cancer rate the agent is likely to cause at given level and duration of exposure.

### **Quantitative assessment of carcinogenicity**

The decision to consider a substance as a carcinogen is based on the classification criteria of the International Agency for Research on Cancer:

Group 1 – Proven human carcinogens.

Group 2 – Probable human carcinogens. This category is divided into two subgroups according to higher (Group 2A) and lower (Group 2B) degrees of evidence.

Group 3 – Unclassified chemicals

It was decided that for all chemicals not categorized in Groups 1 and 2A guidelines values based on non-carcinogenic health endpoints were to be given.

### **Quantitative assessment of carcinogenic potency**

Quantitative risk assessment was found to include the extrapolation of risk from relatively high dose levels to relatively low dose levels. High dose levels are characteristic of animal experiments or occupational exposures, where cancer responses can be measured. Low dose levels are of concern in environmental protection, where such risks are too small to be measured directly, either in animal or epidemiological studies.

In the 1987 guidelines, the risk associated with lifetime exposure to a certain concentration of a carcinogen in the air has generally been estimated by linear extrapolation. The carcinogenic potency has been expressed as the incremental unit risk estimate. The incremental unit risk estimate of an air pollutant was defined as “the additional lifetime cancer risk occurring in a hypothetical population in which all individuals are exposed continuously from birth throughout their lifetimes to a concentration of 1  $\mu\text{g}/\text{m}^3$  of the agent in the air they breathe”.

Necessary assumptions for the average relative risk method were: (i) the response (measured as relative risk) is some function of cumulative dose or exposure; (ii) there is no threshold dose for carcinogens; (iii) the linear extrapolation of the dose-response curve towards zero gives an upper-bound conservative estimate of the true risk function, if the unknown (true) dose-response curve has a sigmoidal shape; (iv) there is constancy of the relative risk in the specific study situation.

Advantages and limitations of the method used in the 1987 guidelines were extensively discussed.

#### **2.4.2 The development of the guideline setting process**

During the development of the 1987 WHO *Guidelines*, emphasis was placed on specifying the guidelines in terms of a concentration and averaging time, which would define an exposure unlikely to produce adverse effects, even in the majority of those members of groups with increased sensitivity to the pollutant in question. Small changes, or so called physiological changes, for example in indices of lung function, were agreed to fall outside the definition of “adverse effects”.

For many of the classic air pollutants the guidelines were based on controlled exposure studies, or on epidemiological studies which demonstrated a threshold of effect. Uncertainty factors, or protection factors, were applied to the published data to allow for more sensitive individuals who might not have been adequately represented in the studies. The guidelines were statements of levels of exposure at which, or below which, no adverse effects can be expected. This does not imply that as soon as a guideline is exceeded

adverse effects occur, but rather that the likelihood of such effects occurring would be increased. The guidelines have sometimes been misinterpreted as Lowest-Observed-Adverse-Effect Levels (LOAEL), which they are not.

Genotoxic carcinogens were treated differently: a Unit Risk was estimated from calculating the additional risk from a lifetime exposure to a unit concentration of the carcinogen. For a few pollutants, including O<sub>3</sub>, the guideline was specified as a range of concentrations.

During the period between the publication of the 1987 *Guidelines* and their revision, a number of meetings were held to consider how the guidelines might be updated (WHO 1992a; WHO 1994a; WHO 1995a; WHO 1995b; WHO 1995c; WHO 1996a). A number of important decisions were made and these are detailed in the reports of the meetings. Among these, the desirability of providing guidance on the exposure-response relationship for as many pollutants as possible was stressed. This has been an important feature of the revised guidelines.

In the updated version of the *Air Quality Guidelines for Europe*, a similar approach was applied as in the 1987 air quality guidelines. However, total tolerable intakes were calculated for multimedia pollutants first, and then adequately partitioned among the different exposure routes. The term "protection" factor used in the 1987 guidelines was abandoned. Instead, uncertainty factors to account for the extrapolation from animal to man (alternatively, human equivalent concentrations were calculated), and to account for individual variability. Wherever information on inter- and intraspecies differences in pharmacokinetics was available, data-derived uncertainty factors were employed. Additional uncertainty factors were applied whenever necessary to account for the nature and severity of the observed effects and for the adequacy of the database. For most of the compounds considered, information on the dose/exposure-response relationship was provided, both to give policy makers clear guidelines on the possible impact of the pollutant at different exposure levels, and to permit an informed decision making process to take place. For some compounds, e.g. platinum, a guideline value was considered unnecessary as exposure through ambient air levels was considerably below the lowest level at which effects were seen. For other compounds, for example PM<sub>10</sub>, no threshold of effect could be found and therefore no guideline value could be derived. Instead, exposure-effect information highlighting the public health impact of different pollutant levels was provided.

In the updating process for carcinogens, a more flexible approach than in the 1987 *Air Quality Guidelines* was applied. Although, as a default approach, low-dose risk extrapolation was conducted for groups 1 and 2A, and an uncertainty factor approach applied in the case of agents in groups 2B and 3, the mechanism of action was the determining factor for the method of assessment. Hence, it was decided that compounds classified under 1 or 2A could be assessed using uncertainty factors, if evidence for a non-threshold mechanism of carcinogenicity existed. By way of contrast, compounds classified under 2B could be assessed by low-dose extrapolation methods, if a non-threshold mechanism of carcinogenicity in animals was proven. Flexibility was also given in terms of the choice of the extrapolation model, depending on the available data (including data for PBPK modelling). The linearized multistage model was used as a default approach. Besides providing unit-risk estimates in cases where low-dose risk extrapolation was conducted, levels associated with excess cancer risk of 1:10 000, 1:100 000 and 1:1 000 000 were calculated.

In evaluating ecotoxic effects of major air pollutants, the effects of O<sub>3</sub>, nitrogen-containing compounds and SO<sub>2</sub> on vegetation (crops, forests) were evaluated. Besides the deposition effects of nitrogen compounds, those of sulphates and total acidity were also evaluated. The principles applied were those developed by the Working Group on Effects under the Convention on Transboundary Air Pollution of the UNECE, and the evaluations were carried out jointly with that group. Critical levels and critical loads were derived. Critical levels are concentrations of pollutants in the atmosphere above which direct adverse effects on receptors such as plants, ecosystems or materials may occur. Critical loads represent

quantitative estimates of an exposure, in the form of deposition, to one or more pollutants, below which significant harmful effects on specified sensitive elements of the environment will not occur.

### **2.4.3 Exposure-response relationships**

These guidelines place some emphasis on epidemiological data. Epidemiological studies are sometimes preferable to controlled exposure studies in that they provide information on responses in populations and on the effects of real exposures to pollutants and pollutant mixtures. However, the results of epidemiological studies are less easy to use than the results of controlled exposure studies in defining guidelines.

Most epidemiological studies relate responses to concentrations of pollutants, often measured at single fixed site monitors. These data tell us little about the exposure-response relationships of individuals but, rather, tell us about the concentration-response relationship of the population studied. This relationship depends upon the pattern of exposure of the population considered and thus the relationship may vary from country-to-country. When the results of time-series studies on the effects of particles in the USA and Europe were compared, only small differences were seen (Wilson and Spengler 1996). But whether the differences were, in fact, due to differences in exposure patterns, or to differences in the toxicity of the ambient particle aerosol, or differences in the particle indices that were measured, remains unknown. Differences in response to air pollution may occur between developed and developing countries.

For both particles and O<sub>3</sub> an assumption of linearity was made when defining the exposure-response relationships included in the revised guidelines. Extrapolation beyond the available data is unwise, since there is evidence to suggest the exposure-response relationship may become less steep as ambient levels of particles rise (Schwartz and Marcus 1990; Lippmann and Ito 1995). For O<sub>3</sub>, the relationship at low concentrations may be concave upwards. These are important points to be considered if the guidelines are to be used in countries with levels of pollution different from the range covered by the guidelines.

### **2.4.4 Moving from guidelines to standards**

An air quality standard is a description of a level of air quality that is adopted by a regulatory authority as enforceable. At its simplest, an air quality standard should be defined in terms of one or more concentrations and averaging times. In addition, other data should be added, including information on the form of exposure (e.g. outdoor), on monitoring which is relevant in assessing compliance with the standard, and on methods of data analysis, quality assurance and quality control.

In some countries the standard is further qualified by defining an acceptable level of attainment or compliance. Levels of attainment may be defined in terms of the fundamental units that define the standard. For example, if the unit defined by the standard is the day, then a requirement for 99% compliance allows the standard to be exceeded by three days a year. The cost of meeting any standard is likely to depend on the degree of compliance required. Consequently, it may be sensible to consider carefully the costs and benefits of different levels of compliance when deciding on the standard.

It is important to remember that the development of air quality standards is only a part of an adequate air quality management strategy (see Chapter 6). Legislation, identification of authorities responsible for enforcement of emission standards and penalties for exceedances are all also necessary. Emission standards may play an important role in the management strategy, especially if exceedance of air quality standards is used as a trigger for abatement measures. These may be needed at both the national and the local level.

Air quality standards are also important in informing the public about air quality. Used in this way they are a double-edged weapon as the public commonly assumes that once a standard is exceeded adverse effects on health will occur. This may not be the case, as discussed in Section 2.4.2.

### **2.4.5 Factors to be considered in setting an air quality standard**

The process of setting standards is simplified when the WHO *Guidelines* provide a guideline value. In general, local review of the health effects database may be unnecessary. However, when published studies on associations between air pollutants and health effects in the local region are available, it is prudent for the authorities responsible for setting national standards to give them due consideration in their evaluation of the applicability of the WHO *Guidelines for Air Quality*. If no single value is offered but rather a Unit Risk estimate, or a concentration-response relationship is defined, then the following should be considered in setting standards:

- a. The nature of the effects indicated should be examined and decisions made as to whether they represent adverse health effects.
- b. Special populations at risk should be considered.

Sensitive populations or groups are defined here as those impaired by concurrent disease or other physiological limitations and those with specific characteristics that make the health consequences of exposure more significant (e.g. developmental phase in children). In addition, other groups may be judged to be at special risk because of their exposure patterns and because the effective dose for a given exposure may be increased, as in the case of children for example. The sensitive populations may vary between countries due to differences in the number of people with inadequate access to medical care, in the prevalence of certain endemic diseases, in the prevailing genetic factors, or in the prevalence of debilitating diseases or nutritional deficiencies. The regulator needs to decide which specific groups at risk should be protected by the standards.

These factors have been considered in the development of these guidelines and have been included when a guideline value has been offered.

The WHO Guideline for SPM was developed to address the health effects associated with exposures to particulate matter released into the ambient outdoor environment, as well as the secondary ambient particulate matter found in the atmosphere from gaseous precursors (e.g. sulphate, nitrate, and the organic products of photochemical reaction sequences). The exposures take place in the outdoor air and in indoor microenvironments following infiltration of the particles into occupied indoor spaces. The numerical effects relationships described in the *Guideline* were based on size-selective mass concentration data that were obtained from numerous, and generally consistent, study results for urban population in North and South America and Europe. However, the transfer of these relationships to other parts of the world should be conducted with caution for several reasons. These include:

1. The chemical composition of the particles may be substantially different in the nation developing the air quality standard, when compared with the regions in which the community studies were conducted and which contributed to the development of the guideline.

Mass concentration in selected particle size ranges (i.e.  $PM_{10}$  and/or  $PM_{2.5}$ ) is, at best, a surrogate index for the biologically active components in the mixture. The mixture in the communities studied in the development of the guideline was dominated by primary and secondary effluents from motor vehicles, central station power generation, and space heating by natural gas and light oil combustion. The mixtures in communities in less developed countries may be different. They may be dominated by the effluents of inefficient combustion units and wind-blown soil, with quite different toxic properties from those in the studies used by WHO.

2. The particle concentration range may be substantially different.

The WHO response-concentration relationships for particulate matter are based on a linear model of response, which is a suitable approximation within the range of particle concentrations typically found in the studies used by WHO. However, it is well established that the coefficient tends to decrease toward the upper end of the concentration range. In addition, the slope

established for the lower concentrations cannot reliably be used to predict responses at the higher mass concentration levels that may be observed in urban areas in less developed countries.

3. The responsiveness of the population may be substantially different. The WHO response-concentration relationships were based on responses of populations that were mostly well nourished and who had access to modern health services. By contrast, the populations exposed to higher concentrations of particles in less developed countries are likely to have lower quality nutrition and health care. Alternatively, they may well be a hardy survivor population with fewer people in a fragile condition of health. It is currently unclear whether the responsiveness of the populations in other parts of the world differ from those studies in North and South America and Europe.

For these reasons, the WHO response-concentration relationships should be used with caution as predictors of health impacts in less developed countries. In particular, the relationships should not be extrapolated to concentrations beyond the ranges given in Figures 3.6 to 3.8.

#### **2.4.6 Uncertainty factors**

In development of these guidelines, the size of uncertainty factors applied to published data in deriving a guideline was considered to be a matter for expert judgement, rather than prescription (WHO 1987). Where the database was strong, smaller uncertainty factors were used than where the database was weak. Database strength depends upon the availability of published studies relevant to the circumstances of a country for which the guidelines are intended. In moving from guidelines to country-specific standards, the size of the uncertainty factors may require revision.

Impact assessment or risk assessment plays an important part in setting standards. This will depend on exposure and an assessment of population exposure will be required. In considering the appropriate form of exposure assessment needed attention should be paid to the database from which the guideline was derived.

Acceptability of risk varies from country to country and is in part dependent on social conditions, priorities and on the other risks to which a population is exposed. In some countries a risk that would be unacceptable elsewhere might be considered small.

#### **2.4.7 Cost-benefit analysis and other factors**

The costs of reducing levels of air pollution should be weighed against the benefits produced. Cost-benefit analysis is one way of formally setting out this process, and it uses money as a common currency for costs and benefits.

The concept is that pollutant concentrations are reduced at least until the associated costs and benefits are balanced: more strictly, emissions are reduced until the marginal costs and benefits are equal. While the cost of abatement measures may be relatively easy to quantify, this may not be the case when non-technical measures are employed. In any case, it is likely to be more difficult to assign monetary values to the benefits obtained. Some aspects of reduced morbidity, such as a reduction in the use of hospital facilities and drugs, are comparatively easy to cost; others such as reductions in premature deaths and symptoms are not. Applying monetary values based on a "willingness to pay" basis has been suggested, and has been accepted as appropriate by many health economists. This approach has been seen as preferable to one based only on such indices as loss of production, earnings or hospital expenses. Cost-benefit analysis is discussed in detail in Section 7.9.

In practice the strict theoretical precepts of cost-benefit analysis should be supplemented by broader social and economic considerations. This process is sometimes described as "Stakeholder Input". Stakeholders are defined as those who have an interest in the outcome of a decision making process. The aim is to ensure as far as possible social equity and fairness to all involved parties. An adequate and early involvement of all concerned stakeholders will increase the transparency of the process and is likely to increase the acceptability of the outcome.

Factors other than monetary concerns also need to be evaluated when considering the setting of national air quality standards. These include the technical capacity of a country to achieve and maintain an air quality within the desired standards; the social implications of adopting certain standards to ensure an equity of costs and benefits among the population; and environmental costs and benefits.