Chapter 2
Particulate Matter in the Urban Atmosphere: Concentration, Distribution, Reduction – Results of Studies in the Berlin Metropolitan Area

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2.1 Introduction: Air Quality and Urban Ecology

Urban agglomerations are places of increased emissions of anthropogenic pollutants into the atmosphere. Since most of these pollutants are harmful to humans, reduction of their ambient concentrations is a major issue of environmental policy on international, national, and local levels. According to Wiederkehr and Yoon (1998), air pollutants can be grouped into major and trace or hazardous air pollutants. Major air pollutants comprise six classical pollutants: sulphur dioxide (SO_2), airborne particles, nitrogen dioxide (NO_2), carbon monoxide (CO), lead (Pb), and ozone (O_3). Hazardous air pollutants can be found in much smaller concentrations than major air pollutants and include different chemical, physical, and biological agents, like volatile organic compounds (VOCs), radio-nuclides, and micro-organisms.

Concentrations of air pollutants are not only governed by various physical, chemical, or biological processes within the atmosphere, but also by anthropogenic actions. Levels of most air pollutants in the urban atmosphere are linked to economic activity. They are typically increasing with income per capita until a turning point is reached. Beyond this point, concentration levels decrease with increasing income (Fenger 1999). Therefore, if urban ecology is defined as a multidisciplinary approach, including natural and social sciences, to improve conditions of life for humans in cities (Endlicher et al. 2007), air quality and related pollution problems are a major issue of urban ecology where natural and anthropogenic aspects are intimately interwoven.

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   The dependency of air quality on economic development is also reflected in a typical temporal sequence of air pollutant levels (Fenger 1999). Concentrations tend to raise during the start of industrial development and initial emission controls. After peaking during a period of stabilization, an improvement in air quality is reached and the subsequent application of high technology leads to concentrations levels which meet WHO guidelines or national standards. In many industrialised countries, an improvement in air quality could be observed since the 1980s. Especially, sulphur dioxide concentrations could be lowered by the use of flue gas desulphurization technology and sulphur-free fuels. In Germany, the deindustrialization in many areas of its former socialistic part accelerated this process, since the burning of sulphur-containing lignite was reduced markedly. Together with sulphur dioxide, the major pollutants carbon monoxide and lead as well as the hazardous air pollutant benzene do not exceed effective limit values in Germany at present (UBA 2010). However, concentrations of airborne particles, nitrogen dioxide, and ozone at least occasionally exceed limit or target values. Therefore, air pollution control in Germany focuses on these pollutants. Whereas annual means of particles and nitrogen dioxide are typically higher in urban areas compared to rural background stations, ozone shows highest mean concentrations outside of urban agglomerations. The elevated ozone levels at rural stations result from complex reactions and transport processes in the atmosphere, which lead both to the formation as well as to the subsequent destruction of ozone in ambient air. From an urban perspective, particles and nitrogen dioxide are thus of special interest for air pollution control.

   Within this chapter, the focus is laid on ambient particle concentrations. After an introduction including possible measures of particles, their sources and current limit values, results from two studies are shown, which were conducted in the Berlin metropolitan area. The first study used data from the air quality monitoring network to analyse particle concentrations and to investigate their dependency on weather types. During the second study, particle concentrations on an urban brownfield were measured and the influence of vegetation on particle distribution was determined.

2.2 Particles in the Urban Atmosphere

Unlike gases, which have a defined chemical structure and therefore show certain physical properties, particles in the urban atmosphere vary in size, shape, and chemical composition. Their concentrations in ambient air may therefore be quantified by different measures like number, surface, or mass concentrations, and these measures can also be given for different particle sizes.

   Since particle sizes cover a range from a few nanometres up to more than 100 μm, according to their size, several classes of particles are distinguished in literature. The most important fractions are commonly referred to as ultrafine (particle diameter D_P smaller than 100 nm), fine (D_P smaller than 2.5 μm) and

coarse particles (D_P between 2.5 and 10 μm). Sometimes, ultrafine particles are called nanoparticles, though occasionally, nanoparticles are defined as fractions including particles with diameters greater than 100 nm (e.g. Lin et al. 2005). At times, the term nanoparticles is only used in the context of artificially produced particles in the nanometer range. Particles greater than 10 μm are seldom investigated in air quality studies and therefore no common term exists for these particles. In the following, they are referred to as gigantic particles. This terminology is based on Junge (1963), who classified particles with D_P larger than 2 μm as giant particles.

2.2.1 Legal Definitions

According to the preamble of Directive 2008/50/EC on ambient air quality and cleaner air for Europe of the European Union, the main goal of air quality control is "to reduce pollution to levels which minimise harmful effects on human health, paying particular attention to sensitive populations, and the environment as a whole". Therefore, air quality research focuses on particles which can cause damage to human health. From that point of view it is of particular interest, how deep particles can penetrate into the human lung. Only particles smaller than 10 μm are able to pass the nose and penetrate into the pharynx or throat (Herbarth, 1998). Furthermore, particles in the range of 2–3 μm can reach the lower part of the respiratory airways, and finally, the remaining particles can penetrate into the terminal bronchioles and alveoli.

To measure and control inhalable particles in ambient air, air quality legislation uses mainly mass concentrations of particles belonging to different size fractions. These are abbreviated as PM_x, with PM meaning "particulate matter" and x referring to the respective aerodynamic diameter. For example, Directive 2008/50/EC defines PM_{10} in Article 2 as "particulate matter which passes through a size-selective inlet as defined in the reference method for the sampling and measurement of PM_{10}, EN 12341, with a 50% efficiency cut-off at 10 μm aerodynamic diameter". Simply speaking, PM_{10} may be considered as particles with diameter less than or equal to 10 μm. Since most particles are irregularly shaped, the determination of their diameter is based on the concept of the so-called aerodynamic diameter. The aerodynamic diameter means the diameter of a sphere of density 1 g cm^{-3} which shows the same terminal settling velocity in calm or laminar flowing air as the measured particle. Beside PM_{10}, which is also called the inhalable fraction, $PM_{2.5}$ or respirable PM and PM_1 are commonly used particle fractions.

Many industrialised countries have set up limit values for ambient concentrations of PM_{10} and $PM_{2.5}$. Within the European Union, limit values for PM_{10} became effective for the first time on 1 January 2005, based on Directive 1999/30/EC. There exist both a long-term and a short-term limit value. During a first stage, the long-term limit value refers to an annual mean of 40 μg m^{-3}, the short-term value

demands a 24-h-average of 50 µg m^{-3} not to be exceeded more than 35 times a calendar year. It was intended to tighten up these values in a second stage in 2010 in the light of further information on health and environmental effects and technical feasibility. Since many member states had difficulties in keeping the limit values for PM$_{10}$, although they implemented reduction measures, and because studies showed that PM$_{2.5}$ may be more relevant to human health, the new Directive 2008/50/EC did not change numerical values of PM$_{10}$ limits of the first stage. However, PM$_{10}$ regulations have been qualitatively relaxed, since allowable subtraction of PM from natural sources has been increased (Brunekreef and Maynard 2008). Limit exceedances are determined after this subtraction. Simultaneously, a limit value for PM$_{2.5}$ of 25 µg m^{-3} as an annual mean was set up to become effective on 1 January 2015. In comparison, the US National Ambient Air Quality Standard sets a limit value of 15 µg m^{-3} for PM$_{2.5}$.

2.2.2   Health Effects of Particles

These limit values are intended for the protection of human health, since many adverse effects of ambient particles on human health have been reported, both related to short-term and long-term exposure (WHO 2004). Several epidemiological studies have shown that long-term exposure to PM for years is directly associated with mortality, mainly due to cardiovascular and respiratory diseases (Pelucchi et al. 2009). Morbidity is also affected by impacts on respiratory symptoms, lung growth, and function of the immune system (Kappos et al. 2004). Likewise, short-term elevations of PM levels on a daily basis show negative effects on the cardiovascular system as worsening heart failure or evoking cardiac arrhythmias (Franchini and Mannucci 2009) and on the respiratory system (Kappos et al. 2004). Although there are also numerous toxicological studies showing negative effects of particles on a cellular level, like cytotoxicity, mutagenicity, and DNA reactivity (de Kok et al. 2006), the mechanisms causing negative effects on human health are not fully understood. But, there is strong evidence that health effects depend on particle composition, particle surface area, and particle size with stronger effects for fine and ultrafine particles (Valavanidis et al. 2008). A threshold value for these adverse health effects could not be found (Kappos et al. 2004).

The association of more negative health effects with fine particles may be caused, on the one hand, by their capability to penetrate deeper into the human lung than coarse particles. On the other hand, fine particles in urban areas are often produced by human activities and therefore contain more metals and toxic organic compounds like PAHs. Fine particles constitute one of the two modes, which can be commonly found in the mass distribution of ambient particles (Seinfeld and Pandis 2006). These so-called accumulation-mode (ranging from ~0.1 to ~2 µm) particles are produced by various mechanisms, including primary emissions, condensation of gaseous precursors, and coagulation of smaller particles. Particles belonging to the

coarse mode are mostly produced by mechanical processes like erosion of soil particles or abrasion. The coarse mode also contains sea-salt particles, particles from volcanic eruptions, pollen, and to a less extent, secondary sulphates and nitrates.

2.2.3 Emission of Particles

In urban agglomerations of industrialised countries, a variety of point and line sources of particles can be found. Whereas industrial activities, including power generation, and domestic heating with coal or oil are point sources, emission of particles from motorised traffic occurs mainly along roads and hence constitutes a line source. Emissions by motorised vehicles do not only include exhaust particles, but also abrasion products from tyres, brakes, clutches, and the road's surface. Furthermore, particles are emitted by re-suspension of previously deposited particles by vehicle-induced turbulence. Besides the local emissions, particle concentrations in cities are also influenced by advection due to particle transport from rural surroundings or long-range, often trans-boundary transport.

Due to this variety of different emission sources and particle formation processes within the atmosphere, particles in the urban atmosphere comprise a mixture of different sizes and diverse composition. Since most of the urban particles are of anthropogenic origin, the maximum of their number distribution is formed by particles smaller than 100 nm; the surface area distribution is dominated by particles in the 100–500 nm range. The mass distribution, on which measures for legal monitoring purposes are based, usually shows two distinct modes, one in the sub-micrometer regime and the other in the coarse particle regime (Seinfeld and Pandis 2006). The chemical composition shows clear differences between various cities. The main components of PM_{10} particles in European cities are non-sea-salt sulphates, sea salt, nitrate and ammonium, soil-derived compounds, elemental carbon, organic matter, and various trace elements (Table 2.1).

Table 2.1 Range of mean percentage contribution of main chemical compounds to $PM_{2.5}$ and $PM_{2.5-10}$ in six European cities (Sillanpaa et al. 2006)

	$PM_{2.5}$	$PM_{2.5-10}$		$PM_{2.5}$	$PM_{2.5-10}$
Non-sea-salt sulphates	14–31	0.8–6.8	Soil-derived compounds, water-soluble	1.3–3.3	9.1–22
Sea salt	1.1–10	3.5–34	Elemental carbon	5.4–9.0	1.0 –5.5
Ammonium	7.0–9.3	0.1–2.7	Particulate organic mater	21–54	9.4–27
Nitrate	1.1–18	3.7–14	Other elements	0.3–1.2	0.4–1.8
Soil-derived compounds, insoluble	1.1–4.2	13–43	Unidentified matter	−6.4 to 21	4.2–23

2.2.4 Dispersion and Distribution of Particles

The spatial distribution of particles in cities is mainly a product of the spatial configuration of emission sources and dispersion processes. On a micro-scale, an important dispersion process takes place in street canyons, where emissions are released into the atmosphere by traffic on the ground of the canyon. Dependent on atmospheric stability, wind velocity, and wind direction, vortexes may be formed within the canyon causing a complex horizontal and vertical distribution of particles (Xie et al. 2009). The dynamics of the shear layer above the canyon, which is caused by the forcing of the external flow, drives exchange processes between the street canyon and the overlying atmosphere (Salizzoni et al. 2009).

Once passed into the overlying atmosphere, particles are diluted within the so-called mixing layer. The height of the mixing layer confines the volume, which is available for particle dilution. Therefore, the height of the mixing layer is negatively correlated with particle concentrations in urban areas. In rural areas with no strong surface sources of particles and in the absence of particle advection, particle concentrations are mainly governed by formation and dissolution processes. Since particle formation itself tends to show a positive correlation with the height of the mixing layer, the correlation between mixing layer height and particle concentrations is weaker in rural areas (Schäfer et al. 2006).

The height of the mixing layer is governed by meteorology. Under anti-cylonic weather conditions, which are characterised by stable atmospheric air layers, the mixing layer height is depressed. Hence, particle concentrations tend to increase due to an accumulation of particles within the mixing layer. In contrast, lower pollution levels may be caused by increased vertical air movement and wet deposition processes during cyclonic weather conditions. Dilution processes within the mixing layer are also influenced by the topography of an urban area. On the one hand, cities located in basins or on valley bottoms, like the German cities Stuttgart or Dresden, have unfavourable air exchange conditions compared to cities on open plains, like Berlin or Hannover. On the other hand, local down-slope winds may carry fresh and cool air on clear, calm nights into the city.

The spatial distribution of emission sources and dispersion processes create a certain spatial pattern of particle concentrations. For legal monitoring purposes, it is common to distinguish between urban, suburban, and rural stations and with respect to dominant emission sources between traffic, industrial, and background stations. In urban agglomerations, three types of stations are of special interest. According to the Commission Decision 2001/752/EC, which amends the Annexes to Council Decision 97/101/EC and establishes reciprocal exchange of ambient air pollution information and data within the European Union, these three types are characterised as follows:

– Traffic stations
  Stations located such that their pollution level is influenced mainly by emissions from a nearby road or street

– Urban background stations
Stations within continuously built-up areas with a pollution level which is not mainly influenced by any single source or street but rather by the integrated contribution by traffic, combustion sources, etc. upwind of the station
– Suburban stations
Station located within a continuous settlement of detached buildings mixed with non-urbanised areas (small lakes, woods, agricultural)

2.2.5 Deposition of Particles

After emission into the atmosphere, dispersion, and chemical or physical transformations in the atmosphere, particles are removed by deposition from the atmosphere to the Earth's surface. Deposition caused by precipitation is called wet deposition; deposition processes which are not influenced by precipitation are summarized as dry deposition. A third kind of deposition, through which water droplets are deposited by interception of fog, mist, or clouds, is referred to as occult deposition. Since this process plays only a significant role in areas with frequent orographic cloud cover or advective fog (Dollard et al. 1983), it can be neglected in most urban areas, as in Berlin, too.

Wet deposition comprises processes in which particles are cooperated into droplets and subsequently transferred to the Earth's surface. Particles might either serve as condensation nuclei for atmospheric water and be incorporated into the formed droplet or collide with an existing droplet. If these processes occur within a cloud, they are called in-cloud scavenging or rainout. If they take place below the cloud, they are named below-cloud scavenging or washout (Seinfeld and Pandis 2006). Wet deposition is a very effective way to remove particles from the atmosphere. Due to numerical studies, weak precipitation with an intensity less than 0.1 mm h^{-1} is able to remove 50–80% of the below-cloud aerosol, both in terms of number and mass, during a 4-h period (Zhang et al. 2004). Urban areas are modifying precipitation patterns by a variety of processes (Kuttler 2008) and, therefore, they are thought to have an influence on wet deposition of particles. But, up to now, little is known about the order of magnitude by which cities alter wet deposition compared to rural surroundings.

The term dry deposition encompasses several mechanisms like turbulent diffusion, sedimentation, Brownian diffusion, interception, inertial forces, electrical migration, thermophoresis, and diffusiophoresis (Zufall and Davidson 1998). Resulting deposition rates are governed by many factors, including meteorological variables such as wind velocity or relative humidity; properties of the particles, such as particle size and shape; and variables of the surface, on which the particles are deposited (Sehmel 1980). If particles are deposited on technical surfaces, which show often sharp edges, friction velocity, micro-scale roughness, and surface temperature are important parameters with influence on the deposition process (Jonsson et al. 2008). When considering particle dry deposition in urban areas, a

special focus is often laid on urban vegetation, since it provides a distinct larger surface area compared to the ground on which it stands. If particle capture by trees is calculated on a citywide scale, notable amounts of PM_{10} are removed from the atmosphere (e.g. Nowak 1994; McDonald et al. 2007). An optimised planting of vegetation along roads can also reduce re-suspension of particles by motorised traffic.

2.2.6   Abatement Strategies

Considering emission, dispersion, and deposition of particles, it becomes clear that strategies to lower ambient particle concentrations may either decrease emission or enhance deposition. On a micro-scale, improving dispersion may also be a suitable activity at traffic sites. Measures to accelerate deposition, like planting vegetation in urban areas, tend to show effects on the long run, and still, their quantitative effects regarding an improvement in particle pollution remain uncertain. On the other hand, the responsible authorities are mostly committed to lower concentrations of particles on the short run if limit values cannot be kept. In Article 23, Directive 2008/50/EC requests air quality plans in the case of exceedances of the limit values, "so that the exceedance period can be kept as short as possible". Therefore, the focus is laid on activities to lower emissions of particles. Such abatement measures include installation of new filter technology at industrial sites or optimized street cleaning and improvement in traffic flow in street canyons (Bruckmann et al. 2007). To reduce emissions by re-suspension of particles from the road's surface, application of calcium magnesium acetate (Norman and Johansson 2006) or magnesium chloride (Aldrin et al. 2008) showed significant reduction of ambient particle concentrations. Since these applications require an intensive use of resources, they may only be suitable to reduce peak levels. To lower particle concentrations extensively, at present in Germany, the most important measure is the exclusion of vehicles with high particle emissions from inner parts of the city. This is achieved by establishment of low-emission zones, which are called "*Umweltzonen*" in German, literally meaning "environmental zones". The aim of these low-emission zones is not only to lower particle concentrations, but also to reduce concentrations of nitrogen dioxide. To control the observance of driving restrictions, red, yellow, or green stickers on windscreens are used. The colour of the sticker depends on the emission level of the vehicle. For example, gasoline cars with a closed-looped catalytic converter and diesel cars which meet the Euro 4 emission standard are allowed to use a green sticker. Diesel cars, which meet only the Euro 2 emission standard, have to use a red sticker. Gasoline cars without a closed-looped catalytic converter and diesel cars meeting only the Euro 1 emission standard are not allowed to use any of the mentioned stickers. Diesel cars can be upgraded to a higher emission standard by retrofitting a particle filter. The low-emission zones themselves are implemented in several stages. During the first stage, vehicles with red, yellow, or green stickers are allowed to drive into the low-emission zone. In the last

stage, only vehicles with green stickers have this permission. Hence, emission of diesel soot is particularly reduced by the implementation of the low-emission zones. As of 1 January 2008, Berlin, Hannover and Cologne were the first German cities implementing first stage low-emission zones. As of 1 January 2010, the last stage of the low-emission zone was introduced in Berlin and Hannover.

2.3 Analysis of Data from the Berlin Air Quality Monitoring Network

The study presented in this section used data from the Berlin Air Quality Monitoring Network to characterise particle pollution in Berlin and to analyse the influence of different weather types on pollution levels. Main aspects of this study have been previously published by Wolf-Benning et al. (2005). Before starting with the description of the study, a general overview of the temporal course of particle emission in Berlin is given in Table 2.2. The total amount of emitted PM_{10} particles decreased drastically from 17,580 tons in 1989 to 3,769 tons in 2005. Particles emitted by plants with need for a special permission to operate due to national air quality regulations and domestic heating showed a particular decline in the early 1990s. This was mainly caused by the closure of many heavy industry sites and the renewal of former coal-based heating systems in the eastern part of Berlin after the German reunification. Whereas the percentage of PM_{10} particles emitted by economic activities decreased from 56% in 1989 to 14% in 2005, the relative importance of traffic as a source for particle emission increased. While exhaust emissions decreased due to the modernisation of the vehicle fleet, emissions from abrasion and re-suspension remained more or less at the same level. As a consequence, the percentage of PM_{10} emitted by traffic increased from 18% in 1989 to 39% in 2005.

Table 2.2 Temporal course of PM_{10} emissions from different sources in Berlin in tons per year (online available at http://www.stadtentwicklung.berlin.de/umwelt/umweltatlas/d312_01.htm#Tab1, 25.10.2010)

	1989	1994	2000	2002	2005	Trend 2015
Sum	17,580	8,804	4,729	4,199	3,769	3,494
Plants with need for a special permission to operate due to national air quality regulations	9,563	3,161	960	650	384	376
Domestic heating	2,693	1,148	131	132	96	77
Small business enterprises	250	220	160	153	149	134
Traffic (exhaust from cars)	1,736	1,135	667	394	355	166
Abrasion and re-suspension caused by cars	1,200	1,150	997	1,050	1,024	1,100
Other traffic	238	190	124	130	105	73
Other sources	1,900	1,800	1,690	1,690	1,656	1,568

2.3.1 PM and Weather Data

To investigate the spatial and temporal variations of ambient PM_{10} concentrations, data from the Berlin Air Quality Monitoring Network (BLUME) provided by the Senate Department for Health, Environment and Consumer Protection (SenGUV) were analysed.

Within the Berlin Air Quality Monitoring Network, particle concentrations are measured as 30-min averages using a β-absorption technique (FH62-I, Friesecke& Höpfner). Particles coarser than the PM_{10} fraction are separated by impaction in an inlet before they are captured by a glass-fibre band. This glass-fibre band is continuously penetrated by β-radiation. The band itself and the deposited particles attenuate the radiation, which is measured behind the band. The increasing attenuation is a measure of the mass of captured particles.

PM_{10} and total suspended particulate (TSP) data from eight monitoring stations of the Berlin Air Quality Monitoring Network were the basis for the analyses. The locations of the selected sites are displayed in Fig. 2.1. Three of these stations were
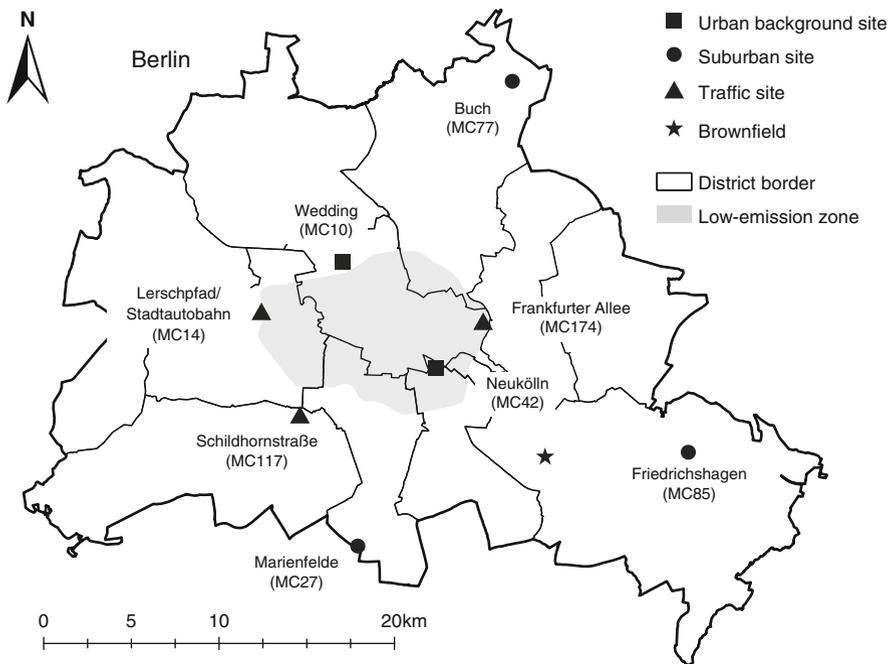
Fig. 2.1 Selected sites of the Berlin Air Quality Monitoring Network (BLUME), the measuring site on the urban brownfield and the low-emission zone in Berlin. The monitoring network is operated by the Senate Department for Health, Environment and Consumer Protection. The map shows also the borders of Berlin's administration districts. A complete PM_{10} dataset covering the years 2000–2004 is only available for five measuring sites (MC14, MC42, MC77, MC117, and MC174) (Data basis: ATKIS-Data © Geobasis-DE/SenStadt III, 2005)

traffic-related sites (MC14, MC117, and MC174), three were suburban stations (MC27, MC77, and MC85) and two stations monitored urban background concentrations (MC10 and MC42). Three further measuring stations were selected to represent the regional background PM_{10} concentrations. One measuring station is part of the air quality monitoring network of the Federal Environmental Agency and is located in the Schorfheide forest, approximately 40 km northeast of Berlin. The other two stations are part of the air quality monitoring network of the federal state of Mecklenburg-Western Pomerania. They are situated in Löcknitz (approximately 110 km northeast of Berlin) and in Göhlen (approximately 150 km northwest of Berlin).

The availability of PM_{10} data was limited due to the step-by-step progression from measuring TSP to measuring PM_{10} and the fact that PM_{10} measurements were not taken at all measuring sites. Hence, the number of monitoring stations used for the analyses varied over time. For example, a complete PM_{10} dataset covering the years 2000–2004 is only available for five of the mentioned Berlin measuring sites.

To study the impact of weather conditions on ambient particle pollution, data provided by Germany's National Meteorological Service (DWD) were analysed and correlated with PM_{10} data. To generalize weather patterns, weather types (Großwetterlagen) according to the classification of Heß and Brezowsky (Gerstengarbe and Werner 1999; Heß and Brezowsky 1977) were used. This classification is based on typical patterns of sea-level pressure and of the 500-hPa pressure surface. 29 weather types representing typical synoptic patterns for Central Europe were distinguished in this study. The number of 29 distinct synoptic patterns can be further reduced to eight general types (n – north, s – south, sw – southwest, w – west, nw – northwest, e – continental east, hm – high air pressure over Central Europe, tm – low air pressure over Central Europe). For the correlation with weather types, 24-h averages calculated from the 30-min PM_{10} data were used.

2.3.2   Temporal and Spatial Aspects of PM_{10} Concentrations

The long-term observation of PM_{10} annual averages in Fig. 2.2 exhibits a decreasing trend from 1991 to 2000. This reflects the decrease in PM_{10} emissions in Berlin (see Table 2.2).

From 2001 to 2009, PM_{10} concentrations at rural, suburban, and urban background sites stayed more or less at the same level with remarkable inter-annual variations. Only at the traffic stations, a slight decreasing trend can be observed during that period.

Highest PM_{10} concentrations were measured at sites at the kerbside of roads. At the three traffic-related sites, PM_{10} concentrations exceeded the annual mean value of 40 μg m^{-3} in 2003 (the long-term limit in the European Union, which became effective in 2005). At the urban and suburban background stations, PM_{10} annual averages were significantly below the averages measured at traffic sites. At the rural monitoring stations, lowest PM_{10} annual averages were observed. Hence, it can be
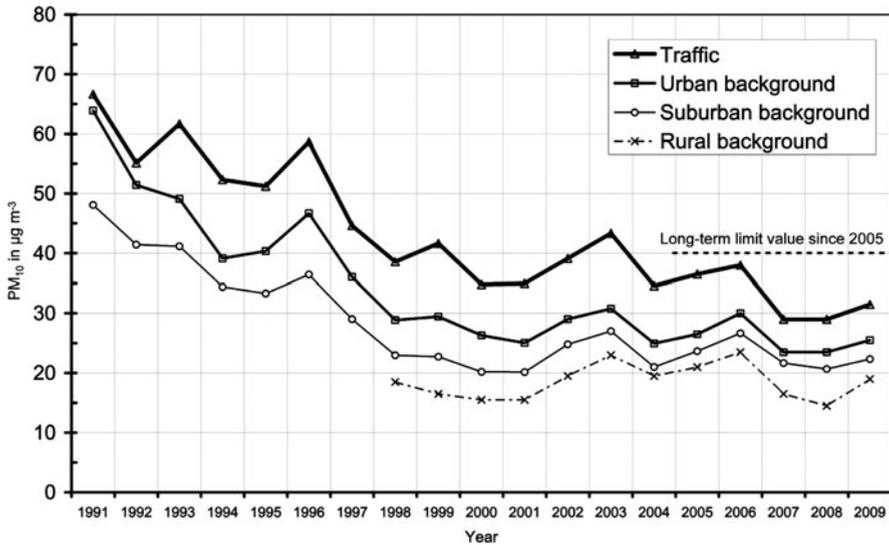
Fig. 2.2 Long-term change of ambient PM_{10} concentrations in Berlin from 1991 to 2009. Annual TSP values were converted to PM_{10} by a multiplication factor of 0.8. Selected measuring sites: MC14, MC117, MC174 (traffic); MC10, MC42 (urban background); MC27, MC77, MC85 (suburban background); concentration data were not available for all sites during the whole period. Rural background averages are based on two sites (Löcknitz and Göhlen) of the air quality monitoring network of Mecklenburg-Western Pomerania

assumed that the annual limit value of the EU will not be exceeded at background locations in the Berlin area in the future.

But, the annual limit value is only one side of the story. Keeping the short-term limit value is much more challenging for many German cities. An overview of the frequency of >50 μg m^{-3} PM_{10} daily averages (the EU short-term PM_{10} limit) between 2000 and 2009 for four measuring sites in Berlin is given in Fig. 2.3. At the two traffic-related sites, a daily average of PM_{10} exceeding 50 μg m^{-3} was observed more than 35 times per year in the 2000–2006 period. While the short-term limit could be kept at all sites in 2007 and 2008, there was an exceedance in 2009 at one of the analysed site. Exceedances at the background stations could only be observed in certain years. During these years, weather conditions may have played an important role in adhering to the short-term limit, because they influence both PM_{10} concentrations at traffic and background sites. Nevertheless, traffic-related sites are particularly vulnerable to violate the short-term limit.
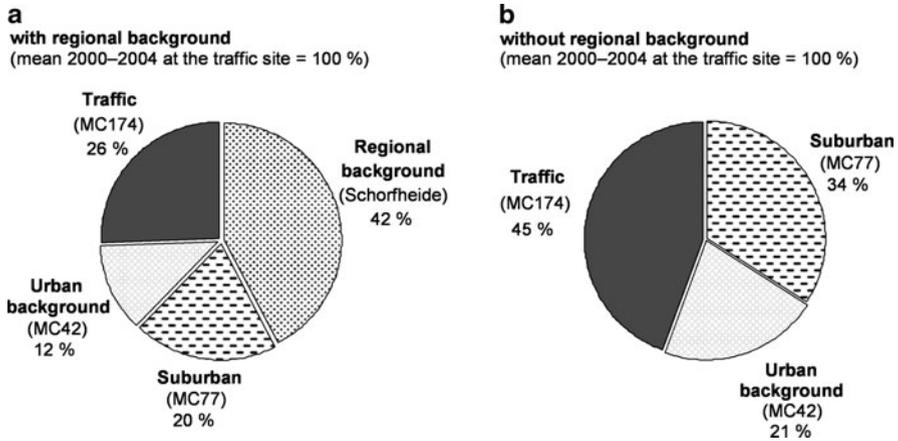
Of special interest for air quality authorities is the effect of the implementation of the low-emission zone on exceedances of the limit values. As depicted in Fig. 2.2, mass concentrations at traffic sites stagnated after introduction of the low-emission zone and showed a slight increase in 2009. Due to the overall increase in ambient PM_{10} concentrations in 2009, traffic site MC174, which is located inside the low-emission zone and where the highest frequency of daily
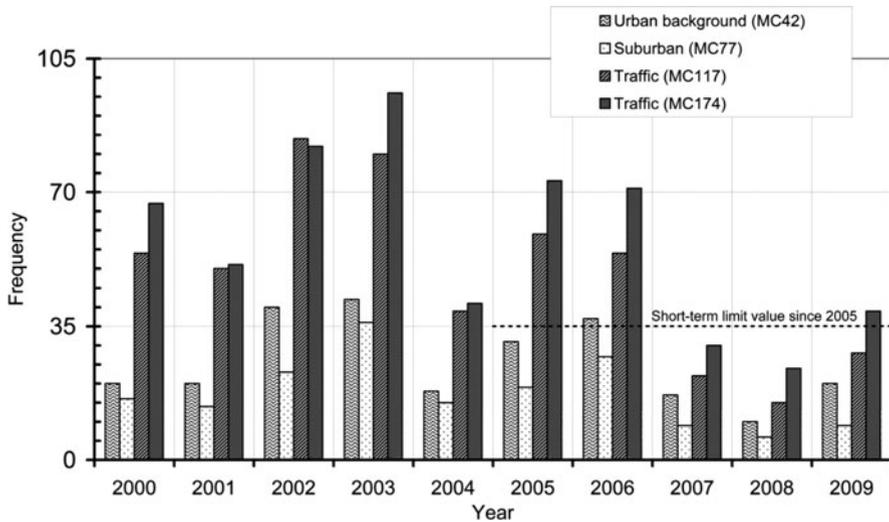
Fig. 2.3 Frequency of daily average PM_{10} concentration exceeding 50 μg m^{-3} at four selected monitoring sites in Berlin for the years 2000 to 2009. Calculation is based on data provided by the Berlin Air Quality Monitoring Network

exceedances of 50 μg m^{-3} occurred since 2006 (Fig. 2.3), showed an exceedance of the short-term limit value in the second year after implementation of the low-emission zone. Due to the dependence of PM_{10} concentrations on several factors, not only local emission but also local weather conditions and long-range particle transport, an assessment of the low-emission zone solely based on measured local PM_{10} concentrations is not appropriate. In fact, a detailed analysis revealed that PM_{10} concentrations would have been 3% higher at traffic sites without a low-emission zone in 2008 (Lutz and Rauterberg-Wulff 2009). This study also showed that the number of days with average PM_{10} values above 50 μg m^{-3} has been decreased by 4% and traffic-related ambient soot concentrations declined by 14–16%. This is of special interest, since it has been argued that mass concentrations may not be suitable to assess the effect of low-emission zones on human health. As especially emission of particles from diesel cars are lowered, which show disproportional adverse health effects compared to most other particles, positive health effects of low emission zones may be higher than expressed by variation of PM_{10} mass concentrations (Wichmann 2008).

The different PM_{10} concentrations for the four spatial categories of measuring sites (traffic, urban background, suburban, regional background) are shown in Fig. 2.4. If it is assumed that the concentration at a traffic hot spot within the agglomeration is the sum of the local traffic's share, urban background's share, suburban background's share, and regional background's share, then about 42% of the PM_{10} concentration observed at a traffic hot spot in Berlin represents the regional background (Fig. 2.4a). The relatively low settling velocity of PM_{10} particles promotes their long-distance transport, and hence, the differences between

a
with regional background
(mean 2000–2004 at the traffic site = 100 %)

Traffic
(MC174)
26 %

Regional
background
(Schorfheide)
42 %

Urban
background
(MC42)
12 %

Suburban
(MC77)
20 %

b
without regional background
(mean 2000–2004 at the traffic site = 100 %)

Suburban
(MC77)
34 %

Traffic
(MC174)
45 %

Urban
background
(MC42)
21 %

Fig. 2.4 Spatial distribution of PM_{10} in Berlin (**a**) with and (**b**) without regional background for the period 2000–2004. Average PM_{10} concentration at a traffic-related site is set to 100% [adopted from Wolf-Benning et al. (2005:109), slightly modified]

areas are not as high as observed for other pollutants, such as NO_2. Within the metropolis, nearly half of PM_{10} is attributed to the local traffic (Fig. 2.4b). Lenschow et al. (2001) mentioned that exhaust emissions and tyre abrasion contribute 55% and re-suspended soil material 45% to traffic-related PM_{10}. However, the emission data in Table 2.2 indicate that the portion of exhaust emission decreased during the last years and that mechanically generated particle emission will become more and more important.

Besides this characteristic spatial distribution within the Berlin metropolitan area, there are also pronounced intra-annual variations. An analysis of seasonal concentration patterns showed that PM_{10} concentrations during winter are typically higher than during summer (Wolf-Benning et al. 2009).

2.3.3 Influence of Weather Conditions on PM_{10} Concentrations

Weather conditions associated with high air stability and low wind velocities increase pollution levels, because of the reduced air exchange, and hence a decreased height of the mixing layer. They can also favour long-range atmospheric transport of airborne particles. Therefore, an above-average annual frequency of such weather conditions can lead to PM_{10} exceeding the EU limit more frequently.

Mean PM_{10} concentrations in Berlin associated with different weather conditions are given in Fig. 2.5 for the 2000–2009 period. Noticeably high PM_{10} concentrations in Berlin are related to three weather types in particular: "high pressure above Central Europe", "continental east", and "south", the latter two being predominantly characterised by anti-cyclonic conditions. Low PM_{10}
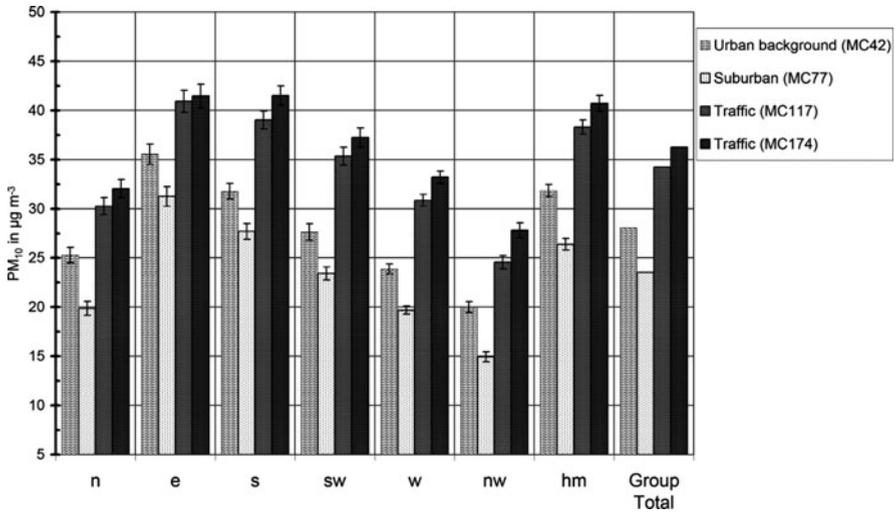
Fig. 2.5 PM$_{10}$ concentrations in association with weather types (Großwettertypen according to Heß & Brezowsky) at four selected measuring sites in Berlin (period 2000–2009). Weather types: *n* north, *s* south, *sw* southwest, *w* west, *nw* northwest, *e* continental east, *hm* high pressure above central Europe. Data are means and standard errors. Data were provided by the Berlin Air Quality Monitoring Network and Germany's National Meteorological Service (DWD)
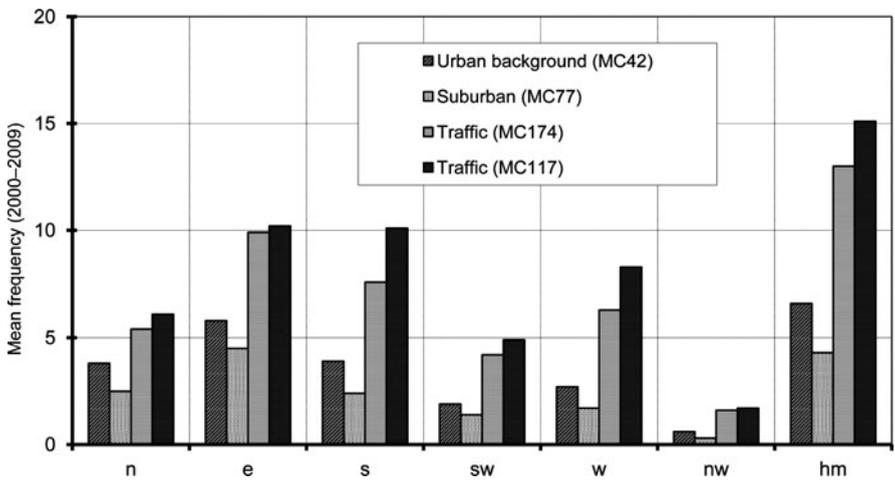
Fig. 2.6 Mean frequency (in days) of daily PM$_{10}$ averages >50 μg m^{-3} associated with weather types at four sites in Berlin (period 2000–2009). Weather types: *n* north, *s* south, *sw* southwest, *w* west, *nw* northwest, *e* continental east, *hm* high pressure above central Europe. Calculations are based on data provided by the Berlin Air Quality Monitoring Network

concentrations are associated with both the "north" and the "northwest" type of weather characterised by cyclonic conditions. The mean frequency of daily PM$_{10}$ averages exceeding 50 μg m^{-3}, correlated with different weather types, shows

nearly the same picture (Fig. 2.6). Again, most exceedances were observed on traffic-related sites and especially during the autochthonous "high pressure above central Europe" type of weather. However, exceedances at the background station in the suburban area are more associated with the continental east type of weather.

This underlines the important role of weather conditions for PM_{10} levels. The weather types "continental east", "south", and "high pressure over central Europe" are linked to above-average PM_{10} concentrations. This association is probably caused by the high dust load of continental air masses, the long-range transport of PM_{10} released from industrial combustion or even by Saharan dust episodes (Henning et al. 2003), and the reduced exchange of air under anti-cyclonic conditions.

2.4 Particle Distribution on an Urban Brownfield

Urban brownfields are a collective term of various sites in cities. They include completely abandoned areas, sites with a decreased intensity of use, and locations that will be used in a new way (Fritsche et al. 2007). Especially in shrinking cities, they present interesting areas for various studies in urban ecology.

Within the study presented in this section, particle distribution on such an urban brownfield was investigated with special focus on effects of vegetation on ambient particle concentrations. Not only PM_{10} concentrations were measured, but also gigantic particles. Although PM_{10} is of special interest from a human health point of view, it seems worthwhile to look at particles greater than PM_{10} in the urban atmosphere. Certain pollen, which are mainly larger than PM_{10} particles and therefore are not able to reach the trachea and the lungs, yet cause discomfort to allergic persons. Therefore, particles greater than 10 μm in diameter, even if their particle number concentrations are very small, may have a negative impact on human health. Another point to consider is a scavenging effect to fine particles caused by larger ones (Friedlander et al. 1991). Therefore, gigantic particles may enhance deposition of PM_{10} particles.

2.4.1 Site Description and Methods

Particle concentrations were measured on an urban brownfield formerly used as a marshalling yard during 2006 and the beginning of 2007. The brownfield is located in the Southeast of Berlin (Fig. 2.1). It has an average width of 300 m and a length of roughly 3 km. It is bordered on its northeast side by a busy six-lane road with a traffic flow of about 50,000 vehicles per day. To the southwest, small industries and various types of unused brownfields are dominating. During the measurements, many construction sites with sand-heaps were located in this area. A large fraction was also covered by bare soil. Active railway lines – three on the northeast side and

one on the southwest side – used by regional passenger trains run on both longitudinal sides of the site.

Particles were sampled both with active and passive systems. Sigma-2 samplers, as described in VDI-Guideline 2119/4, were used as passive samplers at a height of 2.5 m above ground. These sampling devices consist of a cylindrical tube with a height of about 27 cm made of antistatic plastic, which is topped by a cap. This cap has three rectangular windows at its side providing a passive entrance of particles at the top of the tube. Once entered the tube, particles settle down within the tube due to gravitation and are collected on transparent adhesive slides.

After exposition of the slides, particles were detected on the slides by the use of a light optical microscope using an 'automated optical image analysis' system. The geometric diameter of the deposited particles was determined and particles were grouped into size classes ranging from 3 to 6 µm, 6 to 12 µm, 12 to 24 µm, 24 to 48 µm, and 48 to 96 µm. The analysis system was also able to distinguish between transparent and opaque particles. The obtained deposition rates were converted to ambient concentrations using settling velocities calculated according to Stokes' law. Details of this procedure can be found in Dietze et al. (2006). The particle analyses were conducted at Germany's National Meteorological Service (DWD).

To gather information about different PM fractions, PM_{10}, $PM_{2.5}$, and PM_1 were measured with two optical particle counters (Grimm Environmental Dust Monitors #107). These particle monitors operate with an active airflow of $1.2 \ 1 \ min^{-1}$. Particles are detected and counted by the scattering of a laser beam with a wavelength of 655 nm.

During a first sampling campaign, seven Sigma-2 samplers (G-R-1, G-R-2, G-R-3, G-R-4, G-R-5, G-R-6, and G-R-7, see Fig. 2.7) were placed along the road adjoining the brownfield to assess variation of concentrations along the road. This sampling campaign lasted from 4 January to 8 March 2006; the slides were exposed in 7-day intervals. The main sampling campaign was conducted from 8 March 2006 to 28 February 2007, also with a sampling interval of one week. While one Sigma-2 sampler (G-R-1) was left at its position at the road, six samplers were placed on the brownfield (G-B-1, G-B-2, G-B-3, G-B-4, G-B-5, and G-B-6). The samplers were arranged along two transects perpendicular to the road on the sampling site to determine the influence of different vegetation structures on spatial particle distribution. The first transect is dominated by annual herbaceous plants (G-B-1, G-B-2, and G-B-3, see Fig. 2.7) whereas trees and shrubs grow along the second transect (G-B-4, G-B-5, and G-B-6). The most frequent tree species are the Norway maple (*Acer platanoides*) and the black locust (*Robinia pseudoacacia*). Transect two is also characterised by some small buildings.

Measurements with optical particle counters were performed during several campaigns from 1 June 2006 to 18 April 2007. Concentration data were recorded as 10-min averages; the particle inlet was placed 2 m above ground. The four measurements sites (P-B-1, P-B-2, P-B-3, and P-B-4 in Fig. 2.7) were restricted to locations with access to permanent power supply. Occasionally, defects of the monitors resulted in gaps in the concentration data.
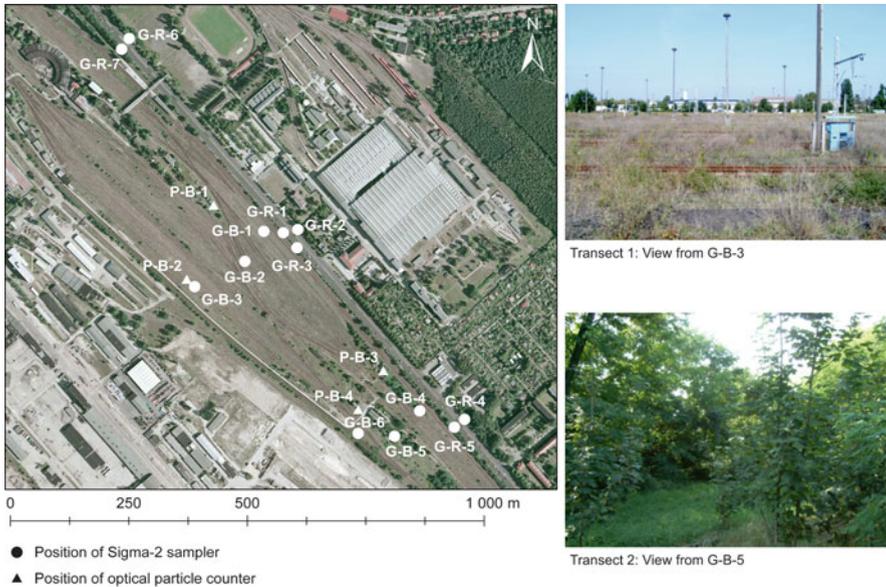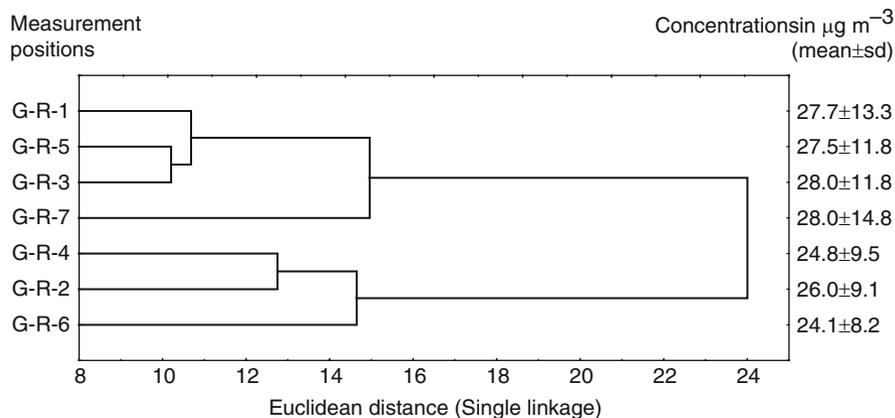
Fig. 2.7 Sampling site on an aerial photograph (original orthophoto scale 1:10,000, photo was taken in August 2004) and different types of vegetation in transect 1 and transect 2 (Data basis of the orthophoto © Geobasis-DE/SenStadt III, 2004)

The meteorological parameters wind velocity, wind direction, temperature, and relative humidity were determined at two sampling points (P-B-2 and P-B-3, Davis Weather Monitor II®). Measurements were made 3 m above ground and data were stored as 10-min averages.

2.4.2 Distribution of Gigantic Particles

During the first campaign in which particles were measured along the road, the Sigma-2 samplers were fixed to street lamps. Mean particle concentrations ranged from 24.1 to 28.0 μg m^{-3}. A cluster analysis showed two main clusters: one with positions G-R-1, G-R-3, G-R-5, and G-R-7; another with positions G-R-2, G-R-4, and G-R-6 (Fig. 2.8). The configuration of the positions depicted in Fig. 2.7 reveals that each cluster belongs to one side of the road. The formation of these clusters is mainly caused by different distances of the street lamps to the edge of the road. On the southwest side of the road this distance amounts to 0.85 m. On the northeast side, a bicycle lane separates the road from the lamps and thus they have a distance of about 3 m from the edge of the road. Within this short distance perpendicular to the road, mean particle concentrations decline from 27.8 to 25.0 μg m^{-3}, which is a reduction of about 11%. This concentrations gradient illustrates that the road is a strong line source for particles.

Measurement
positions

Concentrations in $\mu g\ m^{-3}$
(mean±sd)

|	
G-R-1	27.7±13.3
G-R-5	27.5±11.8
G-R-3	28.0±11.8
G-R-7	28.0±14.8
G-R-4	24.8±9.5
G-R-2	26.0±9.1
G-R-6	24.1±8.2

Euclidean distance (Single linkage)

Fig. 2.8 Results of the cluster analysis of measurements during the first sampling campaign from 4 January to 8 March 2006 with a sampling interval of 7 days; concentrations of all gigantic particles from 3 to 96 µm were analysed

Since traffic is not disrupted by traffic lights at this section of the road, there is a steady traffic flow which results in uniform emissions of particles by traffic. Hence, particle concentrations show only minor differences at a fixed distance from the edge of the road. Therefore, it was concluded from the first measurement campaign, that concentrations at G-R-1 are representative for particle concentrations at the section of the road which borders the brownfield.

During the measurements with Sigma-2 samplers on the brownfield, the lowest concentration of particles from 3 to 96 µm on a one-week basis was 2.9 µg m^{-3} (measured at G-B-5 from 27 December 2006 to 3 January 2007), the highest concentration amounted to 48.7 µg m^{-3} (measured at G-R-6 from 10 May to 17 May 2006). Within the brownfield, mean concentrations ranged from 12.8 µg m^{-3} at G-B-5 to 15.5 at G-B-4. At the roadside position G-R-1, concentrations were usually higher compared to the positions on the brownfield and ranged from 4.1 to 73.4 µg m^{-3} with an average of 26.6 µg m^{-3}.

The steep gradient of particle concentrations nearby the road, detected during the first campaign, could be verified by measurements on the brownfield. At G-B-1 already, approximately 20 m from the edge of the road, concentrations are in the range of those measured at the other sites within the brownfield.

Particle concentrations showed a high variation with time with only minor spatial variation (see Fig. 2.9). High correlations could be found between the various sites. The strongest coefficient of determination was calculated between G-B-2 and G-B-3 ($R^2 = 0.96$), the weakest between G-R-1 and G-B-6 ($R^2 = 0.77$). Figure 2.9 also shows that concentrations tend to be higher during the vegetation period. This might be caused by re-suspension of soil material from adjoining brownfields often covered with bare soils and increased construction activity during dry summer periods.
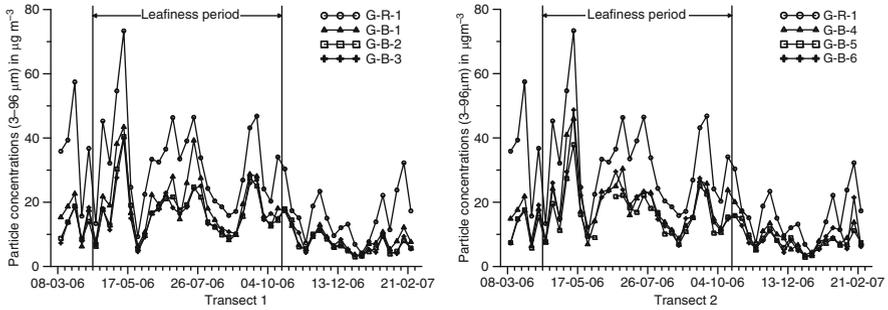
Fig. 2.9 Time course of ambient particle concentrations (3–96 μm) measured at indicated positions arranged in two transects on the urban brownfield and at position G-R-1 at the edge of the road; particles were measured with Sigma-2 samplers with a sampling interval of 7 days from 8 March 2006 to 28 February 2007
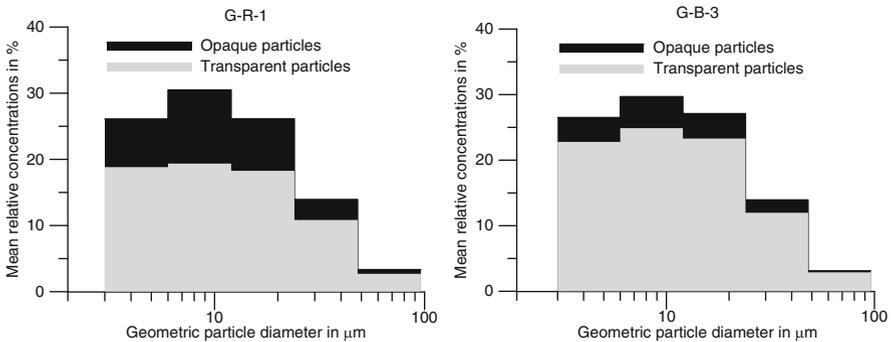
Fig. 2.10 Mean relative size distribution of opaque and transparent particles at the roadside position G-R-1 and at a measurement position within the brownfield (G-B-3). Distributions were calculated based on one-year sampling data from 8 March 2006 to 28 February 2007 obtained with a sampling interval of 7 days

There are not only quantitative differences between the roadside sampling site G-R-1 and the sampling sites on the brownfield; they differ also with respect to their qualitative composition. In Fig. 2.10, mean relative size distributions of opaque and transparent particles are shown. At G-R-1, 30% of sampled particles between 3 and 96 μm are opaque particles. On the brownfield, the fraction of these particles is much lower. At G-B-3, only 14% of all particles belong to the opaque fraction. This suggests that the road acts as a strong source of opaque particles. They consist of soot agglomerates and tyre wear, the latter being particularly enriched in the size range between 10 and 50 μm (Schultz, 1993). Although these differences between the brownfield and the roadside sampling site, the overall size distributions differ only slightly. Particles in the range between 6 and 12 μm dominate the size

distribution and amount to about 30% of mass concentration. Between 48 and 96 µm, only 3% of particle mass could be found.

To have a closer look on the distribution of various particle classes, a principal component factor analysis using varimax rotation was conducted on the particle concentration data. These method is commonly used for source apportionment studies based on elemental concentrations (e.g. Yatkin and Bayram 2007). It extracts various factors from a multivariate particle dataset, which might be attributed to distinct emission sources. Since there were no concentration data available for single elements, concentrations of opaque and transparent particles in the measured size classes were used. The results of the factor analysis in Table 2.3 revealed that the first factor F1 is always dominated by opaque particles in the range between 3 and 24 µm, both at positions within the brownfield and at the road. Therefore, this particle fraction was regarded as an indicator for traffic-generated particles. The interpretation of the second factor F2 was less clear. Within the brownfield, it always contains transparent particles in the 24–96 µm range, whereas at the roadside position G-R-1, F2 is formed by transparent particles between 3 and 12 µm.

From a health point of view, behaviour and especially removal of traffic-related particles is of special interest. Thus, influence of vegetation on dispersion of opaque particle in the range between 3 and 24 µm was investigated. Because of high temporal fluctuations of these particles, they were normalised to the concentrations at G-R-1 prior to further analysis. Furthermore, concentration data were split into concentrations during the vegetation or leafiness period, which was defined by the foliation of taller vegetation on the brownfield, and concentrations during a leaf-less period. The leafiness period lasted from 12 April to 18 October 2006 on the brownfield. Regarding the opaque particles, only minor differences could be detected on the two transects by comparing leafless with leafiness period. According

Table 2.3 Results of factor analysis after varimax rotation for particle concentrations obtained with Sigma-2 samplers based on one-year sampling data from 8 March 2006 to 28 February 2007 obtained with a sampling interval of 7 days. Only factor loadings >0.7 for the two factors F1 and F2 are given

Position	G-R-1		G-B-1		G-B-2		G-B-3		G-B-4		G-B-5		G-B-6	
	F1	F2	F1	F2	F1	F2	F1	F2	F1	F2	F1	F2	F1	F2
Opaque particles														
48–96 µm														
24–48 µm	0.82													
12–24 µm	0.91		0.77		0.83		0.77		0.82		0.83		0.79	
6–12 µm	0.91		0.93		0.87		0.86		0.91		0.87		0.85	
3–6 µm	0.73		0.93		0.85		0.84		0.89		0.88		0.79	
Transparent particles														
48–96 µm				0.72		0.78		0.70		0.87		0.76		0.85
24–48 µm				0.85		0.88		0.89		0.90		0.88		0.92
12–24 µm						0.70		0.70	0.74					
6–12 µm		0.83	0.75			0.75			0.77					
3–6 µm		0.87	0.71						0.75				0.73	
Explained variance in %														
	39	31	46	24	35	34	35	28	46	27	36	27	38	29

to the U-test of Mann-Whitney, this particle fraction showed significantly decreased concentrations during the period of foliation at the positions G-B-3 and G-B-5 (Fig. 2.11).

A decrease of particle concentrations within a vegetation structure may be either the result of particle capture by leaf surfaces or caused by a modification of air flow. Vegetation acts as an obstacle to air flow and, therefore, it is able to redirect particle-laden air resulting in lower concentrations within the vegetation. Especially in street canyons, trees can have an important influence on particle dispersion by the modification of air flow, which has been proved by wind tunnel studies (Gromke and Ruck 2009) and numerical simulations (Ries and Eichhorn, 2001). To test the hypothesis, that reduction of opaque particles ranging from 3 to 24 µm may be caused by a modified air flow, distribution of transparent particles in the same size class was also analysed. Surprisingly, no reduction during the vegetation period could be found, rather a significant increase was detected at most of the measuring positions. This might be explained by the above mentioned increased re-suspension of soil-derived material. Therefore it is concluded that a modification of air flow had only a minor effect on particle dispersion and, thus, on spatial particle distribution on the brownfield. The reduction of opaque particle is caused mainly by particle capture which seems to depend strongly on particle properties.
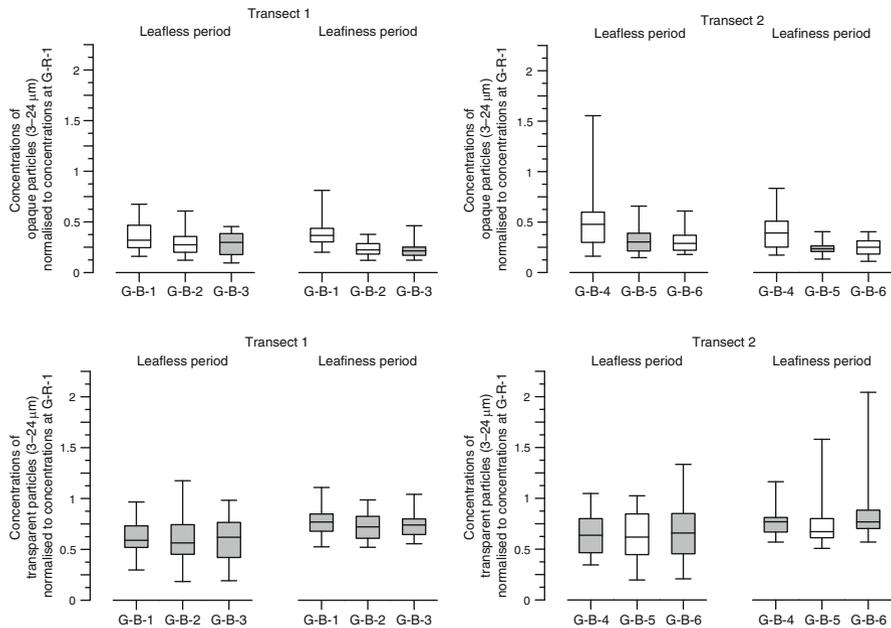
Fig. 2.11 Distribution of concentrations of transparent and opaque particles in the range from 3 to 24 µm within the two transects, divided into a leafless and leafiness period. All concentrations are normalised to concentrations at G-R-1. Concentrations with significant differences between leafless and leafiness period at one measurement position ($p = 0.05$, U-Test of Mann-Whitney) are shaded *grey*

2.4.3   Distribution of PM Particles and Examples of Short-Time Concentration Peaks

Particle measurements with Sigma-2 samplers proved to be a very robust method to get concentration data. Nevertheless, only 7-day averages could be obtained and no information about particles smaller than 3 μm is available from these measurements. To get short-time data of PM_x concentrations, additional measurements with optical particle counters were conducted.

Unfortunately, repeated parallel measurements during a period of 34 days with both counters at P-B-2 revealed remarkable differences between the two devices. Whereas mean PM_{10}, $PM_{2.5}$ and PM_1 concentrations obtain from one counter were 24.1 μg m^{-3}, 17.5 μg m^{-3} and 15.2 μg m^{-3}, respectively, the second counter gave mean concentrations of 18.1 μg m^{-3}, 15.0 μg m^{-3} and 13.5 μg m^{-3}. These problems may be caused by a faulty environmental sampling housing of one of the two samplers. Therefore, parallel measurements needed a correction prior to further analysis. This correction was performed using one counter (A) as a reference device. Concentrations of the second counter (B) were corrected using a multiple regression approach, according to

$$\frac{PM_{X,A}}{PM_{X,B}} = \beta_X + \beta_{X,10} \ln PM_{10,B} + \beta_{X,2.5} \ln PM_{2.5,B} + \beta_{X,1} \ln PM_{1,B}$$

with $X = 1, 2.5, 10$,

which proved to give the best fitting results.

The results obtained after this adjustment procedure are given in Table 2.4. For all PM_x fractions, only minor differences were found. Since the correction procedure itself has to be regarded as a source of error, the corrected data seemed not to meet the requirements for further sophisticated statistical analysis. Nevertheless, the data suggest that PM_x particles are homogeneously distributed on the brownfield.

Table 2.4 PM_x concentrations in μg m^{-3} (mean and standard deviation) at four positions within the brownfield obtained from paired measurements with two optical particle counters

Positions	Sampling period	PM_{10}, mean ± sd	$PM_{2.5}$, mean ± sd	PM_1, mean ± sd
P-B-2	34 days[a]	24.1 ± 18.5	17.5 ± 13.7	15.2 ± 13.1
P-B-2	34 days[a]	25.5 ± 20.2	18.1 ± 14.7	15.5 ± 13.7
P-B-1	40 days[b]	19.4 ± 15.5	11.8 ± 7.0	9.7 ± 6.5
P-B-2	40 days[b]	19.2 ± 12.6	11.9 ± 7.1	9.8 ± 6.5
P-B-3	37 days[c]	23.2 ± 12.2	12.2 ± 6.0	9.7 ± 5.6
P-B-4	37 days[c]	23.7 ± 18.3	11.6 ± 5.8	9.2 ± 5.4
P-B-2	128 days[d]	18.0 ± 13.9	15.4 ± 11.1	13.6 ± 10.6
P-B-3	128 days[d]	16.8 ± 16.6	13.9 ± 11.0	12.7 ± 10.4

[a]1 Jun–7 Jun 2006; 6 Sep–7 Sep 2006; 13 Sep–20 Sep 2006; 2 Apr–18 Apr 2007
[b]7 Jun–21 Jun 2006; 5 Jul–19 Jul 2006; 2 Aug–11 Aug 2006
[c]21 Jun–5 Jul 2006; 19 Jul–2 Aug 2006; 24 Aug–30 Aug 2006
[d]30 Aug–6 Sep 2006; 20 Sep–24 Sep 2006; 8 Nov 2006–22 Feb 2007; 18 Apr–25 Apr 2007

Such homogeneous distributions have been found also on another urban green (Langner and Meurer 2004) and indicate a rather low filtration of PM_x particles by vegetation. This is also supported by models of particle deposition on leaf surfaces predicting a minimum deposition velocity for particles in the size range between 0.1 and 1 μm (e.g. Slinn 1982).

Another interesting aspect of short-term measurements of particle concentrations is the detection of events with high particle loads in the ambient atmosphere. Examples of three events with short-term peaks are shown in Fig. 2.12, each of them caused by different circumstances. The first two events consist of a short and sharp increase of particle concentrations. The first short-term peak was caused by a local thunderstorm. The concentration peak was accompanied by heavy rain fall and an increase of wind speed with strong wind gusts. The latter were the reason for increased re-suspension, but the high particle concentrations may also be partly an artefact of bursting rain droplets, which are detected as particles by the optical counters. During the second event, the meteorological parameters showed no peculiarity during the concentration peak. This was clearly caused by fireworks on New Year's Eve. Although the measurements were made at P-B-2 and hence there have been no fireworks in direct vicinity to the particle counter, concentrations began to rise 10 min after midnight.

The third concentration peak differs from the other peaks in various respects. Increased concentrations lasted over several hours with a maximum PM_{10} concentration of 150 μg m^{-3} which resulted in a 24-h mean PM_{10} concentration of 56.8 μg m^{-3} on 24 March. There is also a clear difference regarding particle size distribution. Whereas the fraction of the coarse mode ($PM_{2.5}$–PM_{10}) amounted to 30% and 11% of PM_{10} during the first and second event, respectively, 54% of PM_{10}
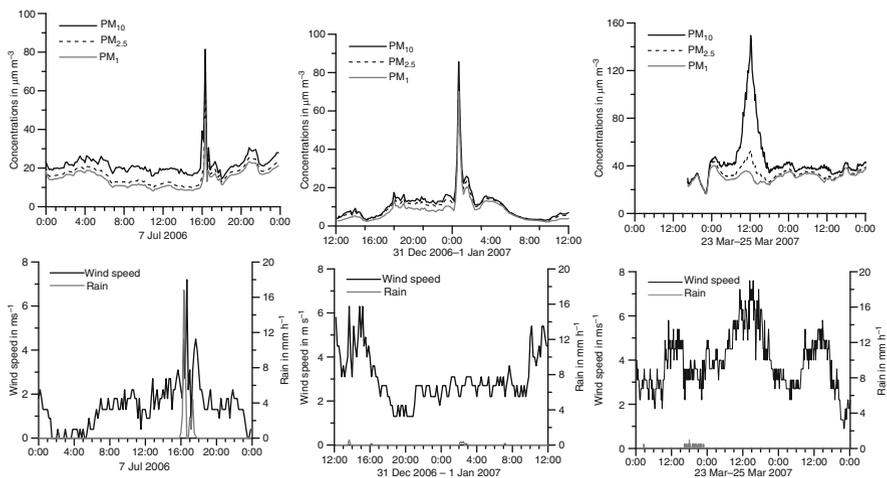
Fig. 2.12 Three examples of events with short-term peaks of PM_x concentrations and related meteorological conditions (wind speed and rain), measured with an optical particle counter on the urban brownfield. At the beginning of 23 March 2007, the particle counter was out of use

belonged to the coarse mode during the third event. Using a backward trajectory model, it could be confirmed that long-range transport of Saharan dust caused this relatively long lasting concentration peak in March 2007.

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