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Review Article

On the reduction of urban particle concentration by vegetation – a review

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Abstract

In order to assess the filtration performance of plants with respect to atmospheric dust, deposition on vegetation has been investigated by a number of different methods (field studies, numerical and physical modelling) over the past few years. The intention of this review is to assess the extent to which a reduction in particle concentration (especially PM_{10}) can be accomplished by existing vegetation or targeted planting on the basis of international publications. The range of this assessment however is limited to the quantitative filtration potential of urban vegetation. Deposition velocity was taken as a measure of filtration performance as a particle deposited on a plant is in effect taken from the atmosphere. As regarding published deposition velocities, there are differences of an order of magnitude between measured values and the results of model calculations. The average published value ($\sim$1 cm s^{-1}) corresponds to a reduction in pollutant concentration (PM_{10}) of about 1 % in urban areas. In addition, analyses carried out for a busy arterial road show that very large vegetation areas (in excess of 10,000 m^2) would be needed to compensate for local emissions of particles (PM_{10}) by vehicles at a deposition velocity of 1 cm s^{-1}. However, current *in-situ* measurements indicate deposition velocities considerably higher than 1 cm s^{-1} and, for PM_1, velocities above 10 cm s^{-1}. If these results were confirmed by further measurements, local planting campaigns covering small areas could also be beneficial for a reduction of particle concentrations.

Zusammenfassung

Um die Filterwirksamkeit von Pflanzen gegenüber Partikeln zu bestimmen, wurde in den vergangenen Jahren mit einer Vielzahl von Methoden (Feldstudien, Modellierung, Windkanal) die Deposition auf Vegetationsoberflächen untersucht. Die Intention dieses Reviews ist es auf der Basis von internationaler Literatur zu bewerten, inwieweit eine Reduktion der Immissionskonzentration von Partikeln (insbesondere von PM_{10}) durch vorhandene Vegetation bzw. durch gezielte Pflanzmaßnahmen möglich ist. Diese Einschätzung beschränkt sich jedoch vom Umfang her auf das quantitative Filterungspotential städtischer Vegetation. Als Maß für die Filterleistung wurde die Depositionsgeschwindigkeit herangezogen, da mit der Deposition auf den Pflanzen auch eine Entnahme aus der Luft einhergeht. Die veröffentlichten Depositionsgeschwindigkeiten zeigen Unterschiede von bis zu einer Größenordnung zwischen den gemessenen Werten und den Ergebnissen aus Modellrechnungen. Berechnet man mit einem Mittelwert der veröffentlichten Werte ($\sim$1 cm s^{-1}) das Reduktionspotential für den städtischen Raum, erhält man eine Verringerung der Immissionskonzentration (PM_{10}) von ca. 1 %. Analysen für eine belastete Ausfallstraße zeigen außerdem, dass bei einer Depositionsgeschwindigkeit von 1 cm s^{-1} beispielsweise eine Kompensation der lokalen Kfz-Emissionen von PM_{10} erst bei sehr großen Vegetationsflächen (> 10 000 m^2) möglich ist. Aktuelle *in-situ* Messungen weisen jedoch auf Depositionsgeschwindigkeiten hin, die deutlich größer als 1 cm s^{-1} sind und für PM_1 wurden sogar Geschwindigkeiten > 10 cm s^{-1} gemessen. Sollten diese Ergebnisse in weiteren Messungen bestätigt werden, wären auch kleinräumige, lokale Begrünungsmaßnahmen sinnvoll für eine Reduktion der Partikel-Konzentrationen.

1 Introduction

Since the introduction of a short term standard for particles (PM_{10}) in air on 1 January 2005 (National implementation of European Council Directive 1999/30/EC, e.g. 22. BImSchV in Germany), attention has once again focussed on the filtration of atmospheric particles by vegetation. The reduction of particle concentration by deposition on urban vegetation (such as street trees, vertical gardens and roof gardens) could prove to be an effective long-term alternative to disputed measures such as the wet cleaning of streets or the exclusion of vehicles for the improvement of air quality. Therefore this review focuses on the assessment of the quantitative filtration potential of urban vegetation. A review on particle deposition with a wider approach concerning all surfaces has been given by PRYOR et al. (2008), MCPHERSON (2007) discussed vegetation-atmosphere interactions as a whole, including e.g. effects on radiation, wind and energy fluxes.

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It was already known at the beginning of the 20th century that dust is deposited on plant surfaces. However, the possibility of using plants to filter the air was not discussed at that time because dust concentrations were so high that plant growth was reduced or even inhibited (e.g. WIELER, 1911). KRATZER (1956) deals with urban aerosol pollution in a prominent position in his summarizing chapter on urban air and refers to the filtration effect of parks and gardens, which is, for example, clearly evident in an early dust concentration map of Leipzig (LOEBNER, 1935). HENNEBO (1955) devotes 60 pages to the topic in his work on the filtration of dust by parks and gardens. He also discusses the importance of urban parks and gardens for the filtration of dust-laden air and makes proposals for urban planning measures to reduce particulate concentration on the basis of existing vegetation structures.

Whereas Hennebo focuses on the micro-scale effect of small wooded areas, GUDERIAN (1975) for example transfers this approach to larger areas. He proposes that planted regeneration areas should be created in the surroundings of pollution source areas with a view not only to reducing pollutant concentrations in the atmosphere but also to preventing the convergence of source areas. However, as a result of technical improvements and the introduction of more stringent legislation (BImSchG – Federal Pollution Control Act, 1974), average annual concentrations of SO_2, NO_2 and suspended particles sank in Germany (FRICKE et al., 2001) in the 1970s. For example, at the Deuselbach (Hunsrück) station of the German Federal Environmental Agency, average total suspended particle (TSP) concentration fell below 40 μg m^{-3} for the first time in 1977 and has remained below this level with the exception of 1982. Following 1999 particles have been measured as PM_{10}, which accounts for approximately 76–79 % of TSP at urban background stations and 82–85 % at traffic stations (LENSCHOW et al., 2001). In other countries, such as the United Kingdom (CLEAN AIR ACT, 1956) and the USA (CLEAN AIR ACT, 1963), similar laws came into effect, in some cases earlier than in Germany, resulting in a large-scale reduction in atmospheric pollutant concentrations. However, AULICIEMS and BURTON (1972) demonstrated that the reduced pollution was chiefly due to technical improvements in industry and socio-economic factors such as the changeover from coal to oil in residential and commercial heating.

Decreasing pollutant concentrations meant that scientific attention was no longer focussed on the impairment of human health and therefore on dust filtration by vegetation. However, the other side of the coin, i.e. the damage caused to plants by filtering polluted air, was investigated comprehensively from the beginning of the 20th century onwards as the negative impact of polluted air was evident through permanent exposure and the clear damage caused to plants. In 1916, WISLICENUS et al. already conducted controlled field tests to investigate damage to plants. These also included exposure to diesel particles (WISLICENUS et al., 1916). In their study, they confirmed the assumption that it was not particles but adsorbents (such as heavy metals and polycyclic aromatic hydrocarbons) that were chiefly responsible for the reduced vitality of plants.

However, this review will only deal with damage to plants to the extent that sustained vitality is an essential prerequisite for the successful, sustained use of vegetation to remove particles from the atmosphere. For this reason, the detrimental effects of dust on vegetation need to be taken into consideration, especially with regard to the selection of resistant species. A comprehensive overview of the various detrimental effects is given, for example, by FARMER (1993) and GRANTZ et al. (2003).

More recently, scientific attention was once again focussed on the capacity of plants to filter dust-laden air by the introduction of a short term standard for particulate matter (PM_{10}). In qualitative terms, the filtration potential of plants is not disputed; the main factors influencing the process of deposition on plant surfaces are discussed in Section 3 below.

Nevertheless, it is questionable whether the filtration capacity of plants will be adequate in quantitative terms to reduce current pollution concentrations as the particulate emissions of plants themselves and the reduction in near-surface air exchange must be set off against filtration performance (Section 4).

Taking an arterial road with heavy traffic as an example, an attempt is therefore made in Section 5 to assess the reduction potential of roadside vegetation as a function of the assumed velocity of deposition. However, before proceeding to this stage, it is necessary to explain the use of deposition velocity as a measure of filtration performance.

2 Deposition velocity as a measure of filtration performance

In the following sections, deposition velocity (v_d in m s^{-1}) is used as a measure of filtration performance. The deposition velocity is the quotient of the mass particle flow rate (F_p in μg s^{-1} m^{-2}) towards the leaf surface and the atmospheric particle concentration (C_p in μg s^{-1} m^{-3}). The resulting velocity is normally given in cm s^{-1} (Eq. 2.1).

$$v_d = \frac{F_p}{C_p} \tag{2.1}$$

Even if resuspension is taken into consideration, particles are taken from the atmosphere and deposition on a leaf surface can therefore be equated with air filtration.

3 Parameters influencing the deposition of particles

The deposition of particles on plant surfaces is influenced by a variety of factors. Not only the diameter and shape of the particles but also meteorological parameters such as humidity, wind speed and turbulence are of decisive importance and have considerable impact on deposition velocity and the filtration performance of plants. In addition, the plant species and planting configuration also affect deposition as the special structure of the vegetation and the shape of the leaf surface are key factors in the deposition and resuspension of particles.

Although particle deposition on plant surfaces corresponds to particle removal from the air and therefore the reduction of pollutant concentration, it must also be noted that plants themselves may be sources of particles and represent an obstacle to air flow which can reduce air exchange compared with non-vegetated areas (RIES and EICHHORN, 2001; GROMKE and RUCK, 2007). In this context, the term "air exchange" is used to mean the frequency with which a given volume of air is exchanged per time unit. In the case of particle emitters located within a vegetated area (as, for example, a road with roadside vegetation), a reduction in air exchange would result in an accumulation of dust emissions. The reduction in pollutant concentration through deposition must therefore be set off against this contrary effect, which tends to increase pollutant concentration. The following paragraphs give an overview of the current state of knowledge regarding the key factors that influence deposition velocity.

3.1 Effects of particle diameter

Theoretical deposition velocities as a function of particle diameter were modelled for example by SLINN (1982) and are generally recognized. Fig. 1 shows the deposition velocities calculated for two different friction velocities (u^*). For particles with a diameter of $d_p > 10\ \mu$m, *sedimentation* is the key deposition process. For particles with a diameter of $d_p < 10\ \mu$m, deposition is generally reduced, reaching a minimum at $d_p = 0.3\ \mu$m. Sedimentation is only significant up to a diameter of $d_p > 1\ \mu$m as the mass of the particle and therefore also the acceleration due to gravity are reduced with decreasing diameter. Between 1 μm and 0.1 μm diameters, *impaction* and *interception* are the main processes acting on particles in the air. These processes involve air flowing around obstacles, where the flow path of the particle is either too near to the obstacle (interception) or is curved in such a way that the particles collide with the obstacle (impaction) as a result of their inertia (LEE and RAMAMURTHI, 1993).

For particles with diameter $d_p < 0.1\ \mu$m, interception and impaction are less significant. As particle sizes fall below this value, inertia and deposition are also reduced. In this size class, only diffusion is effective, but still results in high deposition rates. Near to surfaces, particles are continuously deposited by Brownian motion, resulting in a concentration gradient which induces a mass flow towards the surface.

FOWLER et al. (2004) and GALLAGHER et al. (1997) compared deposition velocities determined theoretically by SLINN (1982) with measurement results from 12 field studies (Fig. 1) and identified marked deviations from the theoretical curve for particle sizes of 0.1 μm < d_p < 1 μm. FOWLER et al. (2004) explain these deviations of as much as an order of magnitude by phoretic processes along an electric potential gradient (electrophoresis) or a thermal gradient (thermophoresis) which were not taken into consideration in the model used by SLINN (1982).

To summarize, it can be stated that the considerable differences between modelled and measured deposition velocities indicate the need for further research in this area.

3.2 Effects of air humidity

The fundamental effect of air humidity on deposition is due to the fact that particles are mainly hygroscopic and that their size varies as a result of the absorption or discharge of water (WINKLER, 1988). In turn, this leads to a change in their deposition properties as a function of diameter (cf. previous section). The size increase with reference to dry air is an exponential function of relative humidity (r.h.). Size increases by an average factor of 1.1 at 60 % r.h., 1.2 at 80 % r.h. and 1.7 near to saturation point. There is a slight variation in size increase as a function of the hygroscopicity of the measured aerosol, e.g. rural or urban aerosol (WINKLER, 1988).

HÄNEL (1982) calculated the sedimentation mass flows of three aerosol types with reference to dry conditions (Tab. 1). The exponential character of the relationship is also clearly reflected in deposition velocities; especially in fog (99.9 % r.h.) the deposition of wet particles is increased by a factor of 16.6 to 24.8 compared with dry aerosol.

3.3 Effects of wind speed

The effect of wind speed illustrates a basic dilemma of the potential filtration of particles using vegetation. As particle concentrations are at their highest in the direct vicinity of emitters, it can be found, by converting Eq. 2.1, that the mass flow to the surface at constant deposition velocity is also at its highest close to the emitters (Eq. 3.1)

$$F_p = v_d \cdot C_p \tag{3.1}$$

Investigations carried out by SPITSYNA and SKRIPAL'SHCHIKOVA (1992) even show an exponential decrease in the mass of dust deposited per unit surface area

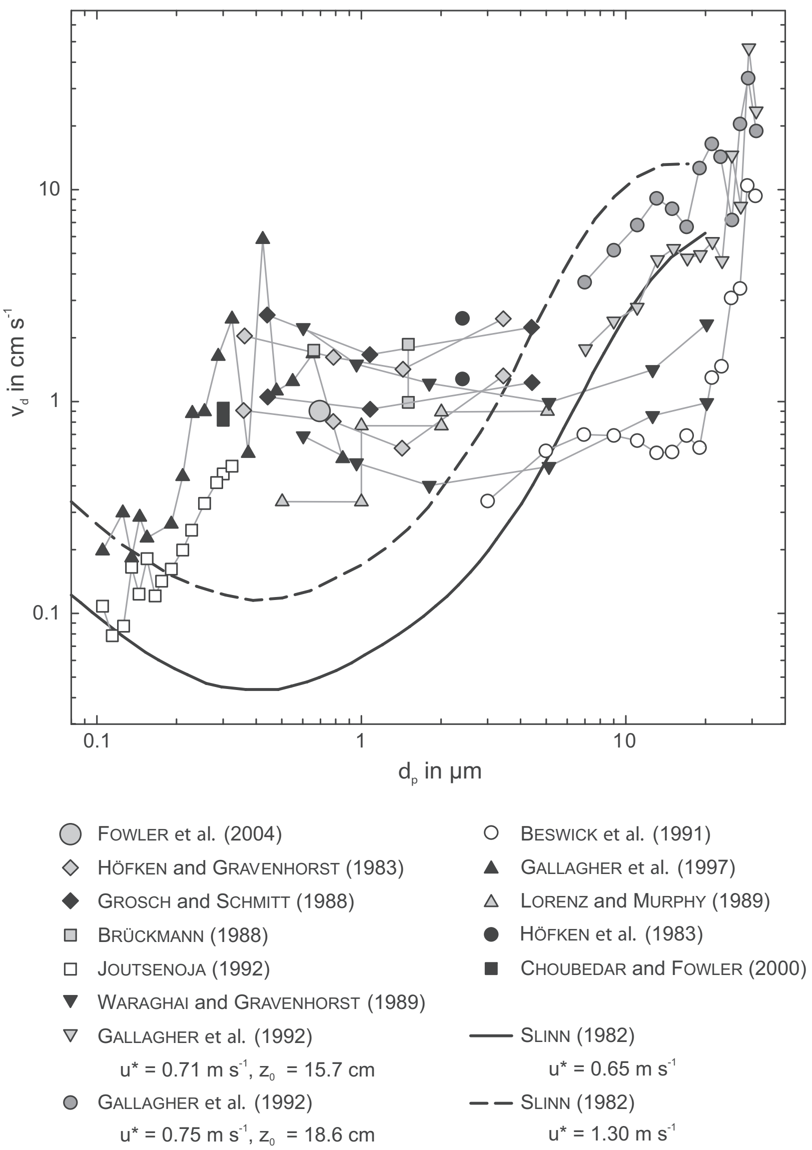


Figure 1: Comparison of modeled deposition velocity (v_d) of aerosols with aerosol size (d_p) for woodland (SLINN, 1982) with field measurements by a wide range of methods (after FOWLER et al., 2004; GALLAGHER et al., 1997). Roughness length (z_0) and friction velocity (u*) are given for GALLAGHER (1992) and SLINN (1982).

with increasing distance from the emission source. In order to maximize the efficiency of filtration, the mass flow to the surface must also be maximized; in other words, vegetation is required as near as possible to the emission source. However, the special structure of vegetation reduces near-surface air exchange, leading to an increase in atmospheric particle concentration and thus deterioration in the pollution situation near to emission sources.

Ideally, a planting concept (or the existing vegetation) should take this condition into consideration by providing as great a plant surface as possible near to the emission source without significantly reducing air exchange. Conceivably, solutions could include vertical gardens or loose ground-level planting with adequate spacing between the plants to minimize the effects on air flow. However, this only applies to autochthonous emissions (from within, such as road vehicle emissions) with particles created on site and accumulating between the plants. In the case of allochthonous emissions (from the outside, such as industrial emissions) an obstacle to air flow could help in keeping local particle pollution low, as the concentration within the plant canopy would be lower than in the air coming from the outside.

In order to investigate the effects of reduced near-surface air exchange in quantitative terms, RIES and EICHHORN (2001) modelled a vegetation scenario for an urban street canyon (Fig. 2). In this two-dimensional model (width 30 m, height 14 m), rows of trees (dedicious, LAI approx. 5 with leaf area density maximum in 9–10 m height) were positioned on both sides of the

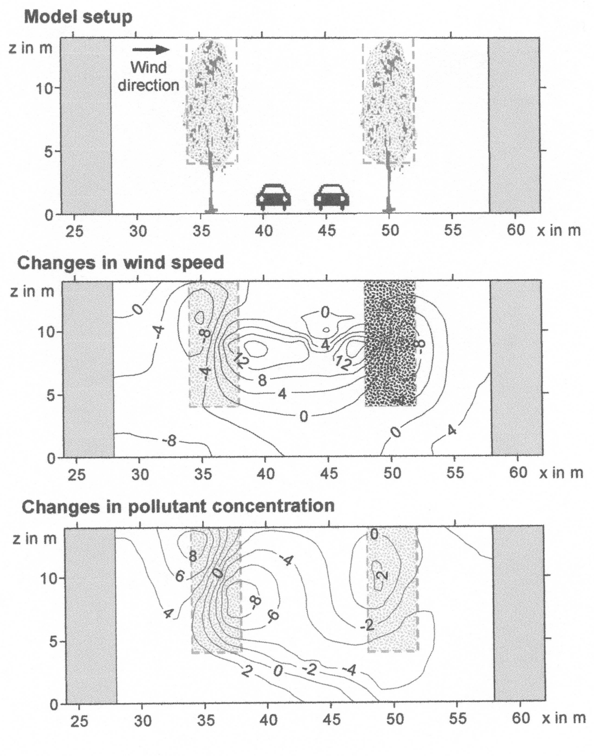


Figure 2: Calculated changes (in percent) in windspeed and pollutant concentration with model height (*z*) and length (x) after insertion of vegetation in the street canyon (RIES and EICHHORN, 2001; modified). Maximum absolute reduction in wind speed is 4 cm s^{-1}, maximum absolute increase in pollutant concentration is 1.2 μg m^{-3}.

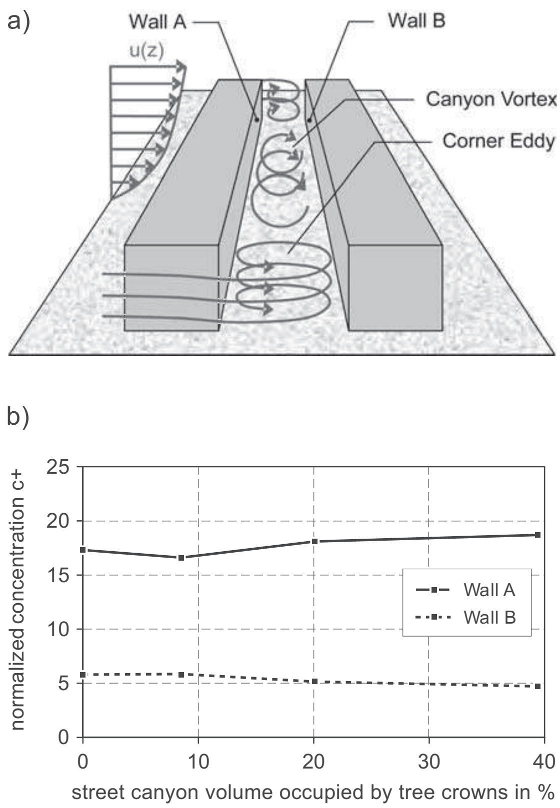


Figure 3: a) Sketch of the wind tunnel setup and wind profile u(z), the trees are located in the center of the canyon, b) dimensionless pollutant concentration values c+ for each wall depending on the street canyon volume occupied by tree crowns. Measured concentrations (c) have been normalized with wind velocity at rooftop (u_{ref}), building height (L_{ref}) and tracer gas emission rate of the line source (Q_T/l), c+ = (c u_{ref} L_{ref})/(Q_T/l). (GROMKE and RUCK, 2007; modified).

road and exposed to wind of 5 m s^{-1} (10 m above ground level) perpendicular to the canyon. A tracer was emitted as a line in the centre of the urban canyon (autochthonous emission type). The tracer was chemically inert, meaning that the calculated concentration changes could also be applied to particles. In comparison with the unplanted situation on the windward side of the inner wall of the urban canyon, wind speed near to ground level was reduced by 8 %, while wind speed was increased by 4 % on the other side. In the situation without vegetation, a rotor circulation mechanism would develop in the urban canyon during a perpendicular wind, with air flowing down the leeward side into the canyon, then across the floor of the canyon in a direction opposite to the general direction of flow to the inside windward wall, where it would leave the street (cf. Fig. 3). Vegetation reduces the cross section for downward flow on the leeward inside wall, resulting in a partial increase in the wind speed. In addition, vegetation reduces near-surface air flow in the urban canyon, leading to lower wind speeds behind the rows of plants. In the model situation, this reduction in air exchange on the windward inside wall leads to an increase of 2–4 % in tracer concentration compared with the scenario without vegetation. However, the results only apply to comparable urban canyons with a building height/road width ratio of about 0.5. In the case of narrower streets with a ratio of 1, the concentration increase near to ground level may be as high as 20 %.

GROMKE and RUCK (2007) investigated a steet canyon with perpendicular flow in a wind tunnel and also found an increase in the tracer concentration on the windward inside wall (Wall A in Fig. 3) and a reduction on the leeward side (Wall B in Fig. 3) . Fig. 3 also shows a sketch of the model configuration and the concentrations on the walls of the urban canyon as a function of different vegetation volumes (the volume taken up by the vegetation in the urban canyon). The conclusion reached was that there was no increase in tracer concentration with a vegetation volume of about 10 %. In contrast to the modelled situation, flow effects on the edges of the building blocks were also taken into consideration in the wind tunnel tests. Depending on the geometric shape of the urban canyon, the reduction in air

Table 1: Ratios of wet to dry deposition by sedimentation for different air humidities and types of aerosol (after HÄNEL, 1982).

Relative humidity (r.h.) in %	Deposition of wet aerosol (r.h.> 0) / Deposition of dry aerosol (r.h. = 0)		
	Maritime aerosol	Urban aerosol	Background aerosol
40	1,0	1,1	1,0
90	2,0	1,6	1,7
99	6,0	4,8	4,4
99,9	24,8	16,5	16,6

exchange may lead to an increase of up to 20 % in particle concentration which runs counter to the filtration effect of particle deposition.

In addition to reducing near-surface air exchange, vegetation also reduces the wind speed. Investigations carried out by BECKETT et al. (2000) in a wind tunnel with NaCl particles indicate a clear relationship between wind speed (u) and deposition velocity, as greater u gives larger particle inertia and, hence, more effective impaction. The role of vegetation as an obstacle to air flow therefore has two negative effects on the pollutant concentration situation. Firstly, the reduced air exchange means that the particle-laden air is replaced less frequently. Secondly, the deposition mass flow is lower as a result of the reduced wind speed.

3.4 Effects of plant species and surface properties

The effects of plants species on deposition depend on the size scale considered. Regarding the plant as a whole, deposition is mainly affected by the shape of the plant and the structure of the leaves or needles. Considering individual leaves, deposition may be increased or reduced by different surface structures.

The spatial structure of branches and twigs and the shape of leaves and needles play a key role in the filtration properties of a plant. Wind tunnel tests (BECKETT et al., 2000) have shown that the deposition of NaCl particles is significantly higher on selected conifers (pine and cypress) as a result of their more complex spatial structure than on deciduous trees (maple, poplar). A further disadvantage of deciduous trees is the lack of foliage outside the vegetation period which, considered in absolute terms, reduces their filtration performance. However, technical problems such as the need to ensure adequate drive-through heights, reduced solar access for inhabitants in winter and poor resistance to pollutants restrict the possible applications of conifers in urban areas. For this reason, of the 154 tree species which have been tested as street trees in Germany, only one is a conifer (Dawn Redwood, *Metasequoia glyptostroboides*), which is not a domestic species in Germany and a deciduous one at that (GALK, 2006).

BURKHARDT et al. (1995) demonstrated by electron microscope examination with a fluorescent marker that deposition was increased at certain cuticular areas of the leaf surface characterized by an especially complex three-dimensional structure. Increased micro-roughness has been discussed as a possible reason for higher deposition in these areas. Increased deposition can also be observed for comparable leaf appendages and structural features such as hairs and veins. For this reason, THÖNNESSEN (2002) distinguishes between plants with very rough leaf surfaces and higher deposition (accumulators) and plants with surfaces favouring the removal of dust deposits by precipitation (self-cleaners) of the type which has been observed for the lotus plant. However, it is not possible to make an assessment of long-term filtering performance on the basis of this distinction because no long-term studies have been performed on the leaves of accumulators. A "saturation effect", which would lead to reduced deposition in the case of large dust deposits, can therefore not be excluded.

In addition to their negative impact on near-surface exchange conditions (cf. previous section) plants may also be sources of particulate emissions under certain conditions. According to CHAMBERLAIN (1975), wind-borne pollen may reach concentrations of 14 μm^{-3} under extreme conditions (1000 pollen m^{-3}), i.e. in the vicinity of plants in bloom. However, most pollen grains have an average diameter in excess of 10 μm; they therefore do not fall into the PM_{10} size category and are less problematical considering health effects as they can scarcely reach the human bronchi but are filtered out by the upper airways. An aspect that must be viewed more critically is the creation of particles by the emission and subsequent condensation of biogenic volatile organic compounds (BVOCs) by vegetation. In studies in the Fichtelgebirge mountains in North-East Bavaria (Germany), HELD et al. (2004) registered an increase in particle concentrations through nucleation events in the case of wind flow from a large coniferous forest (Fig. 4). Other parameters favouring these effects were found to

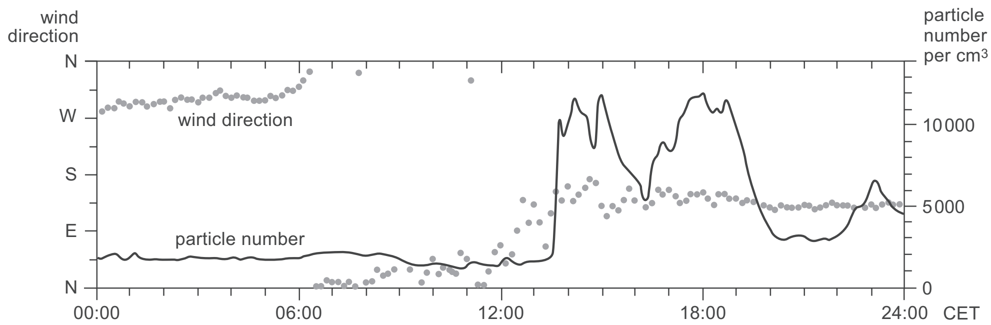


Figure 4: Wind direction (grey dots) and particle number concentration at 22 m agl within a forest stand (black line) during nucleation event on 27 July 2002 in the "Fichtelgebirge", a mountain range in NE Bavaria, Germany (modified after HELD et al., 2004).

be dry atmospheric conditions and high short-wave radiation density. Nucleation events were observed on 20 % of days in the measured period and must therefore be regarded as a relatively frequent phenomenon.

The size of the particles observed was between 0.8 μm and 0.003 μm, a size range that is subject to high deposition velocities. It can therefore be assumed that a large part of these particles were deposited again immediately following emission. However, as the mass of the particles measured increases in the course of a nucleation event as a result of condensation processes, it is difficult to draw up a mass balance. It cannot be clearly determined whether the forest is a net source or a net sink of particles with respect to their mass. Nevertheless, HELD et al. (2004) state that wooded areas are a clear sink with reference to the number of particles.

3.5 Resuspension

One of the main reasons for the variability of deposition velocities is the resuspension of particles following deposition. As resuspension decreases rapidly with time after deposition, it is normally already included in the deposition velocities measured and deposition velocities therefore do not need to be corrected for resuspension. In modelling and calculations, it is also normal practice to integrate resuspension into the deposition velocity. This also applies to the deposition velocities mentioned in previous sections. It is especially with reference to the factors of wind and surface structure that resuspension has a significant impact on the resulting deposition velocity.

Using radioactive tracers, WITHERSPOON and TAYLOR (1969) were able to assess resuspension after one hour at 91 % in the case of an oak and at 10 % for a pine-tree. The higher deposition velocities for conifers are therefore probably due at least in part to the lower resuspension rate. Comprehensive overviews of resuspension in general are given by NICHOLSON (1988) and SEHMEL (1980).

4 Mesoscale calculations and modelling

In the preceding section, the parameters affecting the deposition of particles were treated separately. However, in order to assess the possible filtration performance of vegetation, it is necessary to consider interaction between all these factors. In addition, particulate emissions by plants and increased pollutant concentrations as a result of reduced near-surface air exchange also need to be taken into consideration.

A microscale urban climate model considering air flow in and around vegetation and the deposition of particles is suitable for obtaining a view of the individual parameters taken together. For example, BRUSE and FLEER (1999) introduced the three-dimensional non-hydrostatic model ENVI-met to investigate surface-plant-air interactions in urban environments. For the purpose of this review ENVI-met has been used to model the concentration of particles above a busy road in the central business district of Duesseldorf (North Rhine-Westphalia, Fig. 5). Only local traffic on the four streets in Fig. 5 was taken into consideration as line emission sources. In order to assess the filtration performance of vegetation, the vegetation situations modelled were compared with a situation without vegetation. The vegetation scenario is based on the actual situation encountered in Duesseldorf. In the scenario with vegetation the modelled particle concentration on the road with the heaviest traffic (B in Fig. 5) was up to 5 μg m^{-3} higher than without vegetation. The results of this modelling suggest that the filtration effect of all types of vegetation is negligible in comparison with the effect of reduced air exchange, although the model includes gravitational settling of the particles and removal from air on impact with obstacles (no resuspension). However, the results are of limited applicability as the technical possibilities of the model only allowed a spatial resolution of 5 m. The difference between deposition velocities determined by modelling and measurement, which are very significant in some cases, was already mentioned in Section 3.1.

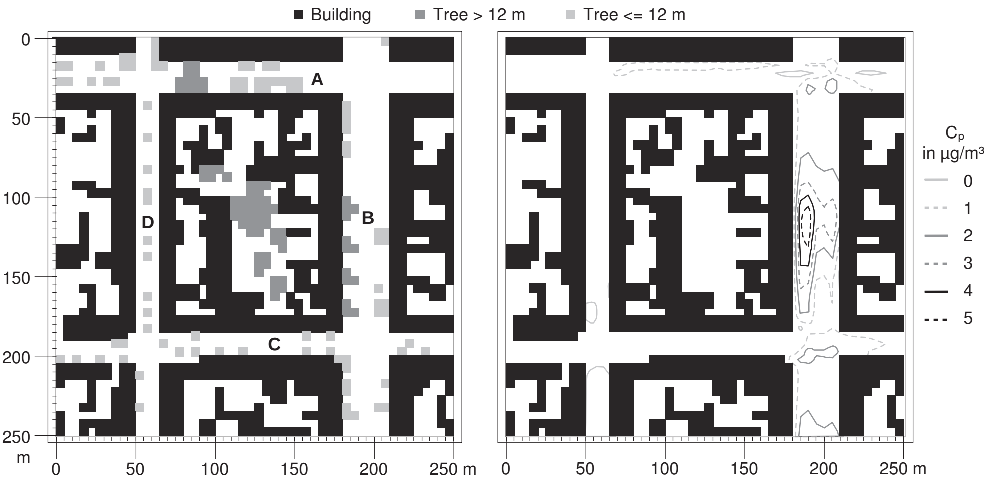


Figure 5: The left side shows the model setup of buildings and vegetation (base size of the grid 5x5 m), average building height is 17 m agl. Wind setting was 3 m s^{-1} in 10 m agl coming from the left. The particle emission (PM_{10}) of the line sources was set to 19.4 (A), 97.0 (B), 38.8 (C) and 5.9 μg m^{-1} s^{-1} (D) respectively. The right side displays isolines of the difference in particle concentration (ΔC_p) with and without vegetation ($\Delta C_p = C_{Vegetation} - C_{No\,Vegetation}$) in 2 m agl.

Despite the negative impact on local particle concentration, particles were nevertheless deposited on the vegetation surfaces and therefore contributed slightly to the improvement of general air quality. However, the specific air quality near to a road is dominated by powerful sources of emissions, which provide a continuous supply of particles and can therefore scarcely be improved by vegetation.

In order to assess the role of vegetation in improving general air quality, filtration performance has been extrapolated for a number of cities. For example, NOWAK (1994) calculated total annual deposition of 212 t PM_{10} for Chicago, IL (90 % deciduous trees, vegetated area: 11 % of city area). In the vegetation period, this would correspond to an average reduction of 0.4 % in particle concentration in the mixing layer above the city; for some densely vegetated areas of the city, even a reduction in particle concentration as high as 2.1 % was calculated. Nowak used the vegetation records of the city to calculate vegetation surfaces in 117 city districts on the basis of the leaf area index. For particle concentrations in the districts, values from the nearest of the 14 measurement stations installed were used. Deposition velocities were calculated from published values of canopy-resistance (R_c) and weather-dependant aerodynamic (R_a) und quasi-laminar boundary layer resistance (R_b) using Eq. (4.1).

$$v_d = \frac{1}{R_a + R_b + R_c} \tag{4.1}$$

Deposition flux densities were calculated for the city districts from particle concentrations and deposition velocities using Eq. (3.1). These were then multiplied by plant surfaces (A) to give deposition mass flow rates (m_d in μg s^{-1}) (Eq. 4.2).

$$m_d = v_d \cdot C_p \cdot A \tag{4.2}$$

To obtain an indication of the improvement in particle concentrations, this deposition was totalized for one hour and related to the total mass of particles in the mixing layer height (h) of the city (estimated from the values of the nearest measurement station, located in Peoria, IL). For this purpose, the total mass of particles (m_{total}) was estimated by taking the product of the mixing height (h), city area (A_{City}) and particle concentration (C_p) (Eq. 4.3).

$$m_{total} = C_p \cdot h \cdot A_{City} \tag{4.3}$$

NOWAK and CRANE (2000) adopted a similar approach for New York City (tree cover: 16.6 %). In this case, total annual deposition was 470 t PM_{10} and the reduction in particle concentration during the vegetation period 0.5 %. The data of urban vegetation (cover, species, leaf area index etc.) were determined by field investigations and to a certain extent by evaluating aerial photographs. Deposition was calculated using a method based on NOWAK (1994), and a constant deposition velocity of 0.64 cm s^{-1} was assumed for the vegetation period (0.14 cm s^{-1} outside the vegetation period).

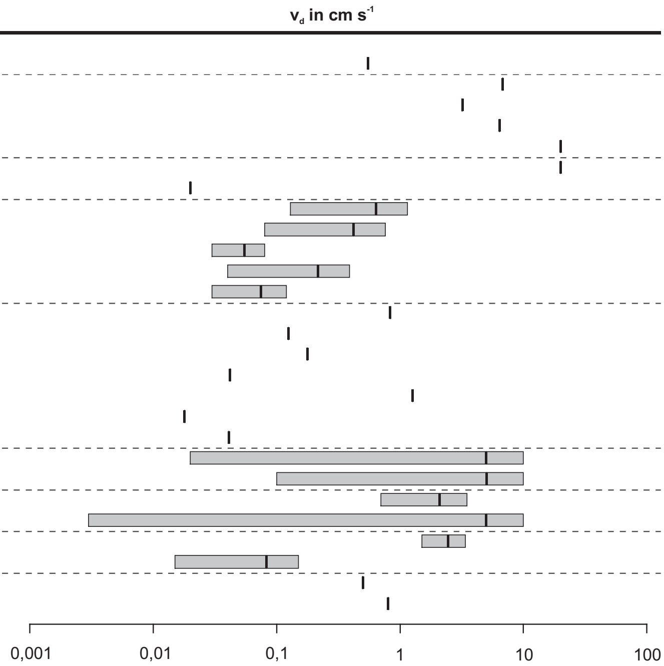
The improvement in air quality as a result of the deposition of dust on plants, calculated using the deposition rates assumed by NOWAK (1994) and NOWAK and CRANE (2000), is therefore very slight and could scarcely be confirmed by concentration measurements.

5 Estimate for an arterial road with heavy traffic

The calculations made in the preceding section show that effective filtration is largely dependant on the deposition velocity used.

Table 2: Published deposition velocities (v_d), depending on plant species, particle size (d_p) und wind speed (u) (from FREER-SMITH et al., 2005; SEHMEL, 1980).

Author	Year	Species	d_p in µm	u in m s^{-1}
BUNZL *et al.*	1989	*Picea abies*		
WHITE a. TURNER	1970	*Fraxinus excelsior*	0,1-20	2
		Quercus petrea	0,1-20	2
		Betula pubenscens	0,1-20	2
		Corylus avellana	0,1-20	2
HORI	1953	*Picea glehnii*	20	1,4
PETERS a. EIDEN	1992	*Picea abies*	1	0,5
BECKETT *et al.*	2000	*Pinus nigra*	1,28	1-3
		Cupressocyparis leylandii	1,28	1-3
		Acer campestre	1,28	1-3
		Populus deltoides	1,28	1-3
		Sorbus intermedia	1,28	1-3
FREER-SMITH *et al.*	2004	*Quercus petrea*	0,8	3
		Alnus glutinos	0,8	3
		Fraxinus excelsior	0,8	3
		Acer pseudo-platanus	0,8	3
		Pseudotsuga menziesii	0,8	3
		Ficus nitida	0,8	3
		Eucalyptus globulus	0,8	3
QUARG	1996	Grass	0,1-12	
GALLAGHER *et al.*	1997	*Pseudotsuga menziesii*	0,01-10	
CHAMBERLAIN	1966	Grass	32	1-7,4
BELOT a. GAULTHIER	1975	Pinus and Quercus shoots	2	1-10
NICKOLA a. CLARK	1976	*Artemisia tridentata*	5	
HORBERT *et al.*	1976	Grass	1-3	
SIMPSON	1961	*Artemisia tridentata*	2,5	
CLOUGH	1973	Grass	1-10	6



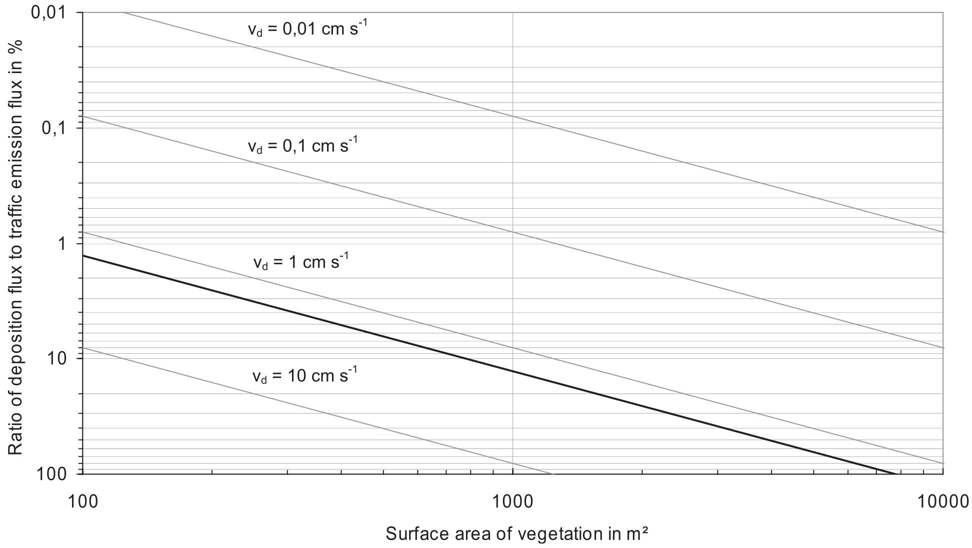


Figure 6: Ratio of deposition flux to traffic emission flux depending on surface area of vegetation and deposition velocity (v_d) in a street section of 100 m length. The black line shows the deposition velocity for 8.000 m^2 of vegetation surface area and a ratio of deposition flux to traffic emission flux ratio of 100 % ($\sim$1.6 cm s^{-1}, see text for explanations).

Deposition velocities previously published cover a range of more than four orders of magnitude (Tab. 2). The question therefore arises as to how high the deposition velocity must be in order to result in a measurable improvement in the particle concentration situation and whether previously published deposition velocities can reach this order of magnitude at all.

In a sample calculation, the filtration (deposition effect) of targeted planting on an arterial road with heavy traffic was therefore considered as a function of deposition velocity and the plant surface added to the system. In order to make filtration performance clearer, deposition mass flow to vegetation surfaces is plotted against vehicle particulate (PM_{10}) emissions in % in Fig. 6. A

Table 3: Deposition velocities of PM_{10}, PM_2 und PM_1 on the foliage of *Pinus sylvestris, X Cupressoparis leylandii, Acer campestre, Populus deltoides X trichocarpa* 'Beaupré' and *Sorbus aria* (modified after FREER-SMITH et al., 2005).

Particle size fraction	**Deposition velocity in cm s^{-1}**				
	Pine	Cypress	Maple	Poplar	Whitebeam
Sussex field site					
PM_{10}	2,8	3,4	3,6	0,6	5,4
PM_2	1,8	4,6	9,2	0,8	11,0
PM_1	36,2	33,7	31,7	25,4	27,2
Withdean Park					
PM_{10}	4,7	6,2	1,8	0,4	3,3
PM_2	6,1	3,7	2,5	0,8	4,5
PM_1	29,9	19,5	11,6	12,3	16,9

value of 100 % would mean that the filtration performance of the plants fully compensated for the additional pollutant load caused by road vehicles.

For a road section with a length of 100 m, average daily traffic of 40,000 vehicles and an assumed emission factor of 100 mg km^{-1} vehicle^{-1} PM_{10} (rounded value, e.g. DÜRING et al., 2004) the average emission mass flow is 4.63 mg s^{-1}.

At 37 μg m^{-3}, the average annual PM_{10} concentration (C_p) corresponds to that of an arterial road with heavy traffic. If 40 deciduous trees were planted on both sides of the road (5 m crown diameter, leaf area per unit ground surface covered LAI 5), a vegetation surface of about 3,000 m^2 would be created; with a façade height of 12 m and a section length of 100 m, façade vegetation on both sides of the road could provide a maximum vegetation surface of 2,400 m^2 (4,800 m^2 with an LAI of 2). As a result, the maximum possible vegetation surface area (A_{total}) for the road section would be about 8,000 m^2. The deposition mass flow can be calculated from these values using Eq. (4.2) as a function of the vegetation surface and the assumed deposition rate.

For the maximum possible vegetation area (8,000 m^2) 100 % compensation for road vehicle emissions is only reached at a deposition velocity in excess of 1.6 cm s^{-1}. Tab. 2 shows that deposition velocities of this order of magnitude have already been measured. However, it should be noted that some of these measurements were based on particles with a diameter of up to 20 μm which reach high deposition velocities by sedimentation (cf. Tab. 2). In other studies, high filtration performance was only reached at high wind speeds which are not representative of urban conditions. The figures given by GALLAGHER et al. (1997) were measured by the eddy covariance method in a Douglas fir forest with an area of 3000 ha and are therefore only of limited relevance to small areas of urban vegetation. It can therefore be stated that it would be just within the bounds of possibility to compensate for local road vehicle emissions using vegetation with the (optimistic) estimate of 8,000 m^2 of vegetation surface and the published values for deposition velocity. The negative effect of vegetation on near-surface air exchange has not been taken into consideration here as a result of the complex interactions involved. However, in order to compensate for local road vehicle emissions, very extensive vegetation would be needed and it would be necessary to consider the possibility of planting conifers by roadsides.

Not only as regarding the deposition velocity assumed but also as regarding the method used, the investigations of FREER-SMITH et al. (2005) are an exception. According to the information provided by the authors, the deposition rates used were measured in situ for the first time. For this purpose, atmospheric particle concentrations at two locations (urban area and surrounding countryside) were measured continuously by optical particle counters during a summer measurement period (7 days in August 1998). At the end of this measurement period, the particle mass deposited on the leaves of five different tree species was determined gravimetrically and a deposition mass flux density was calculated using the area of the leaf samples. In combination with the results of the concentration measurements, it was then possible to calculate deposition velocities using Eq. (2.1). The resulting deposition velocities are remarkable (Tab. 3). With the exception of the poplar, the deposition velocities determined for the two locations were significantly above 1 cm s^{-1} for all particle size fractions. The deposition velocities for PM_1 (including the poplar) were in excess of 10 cm s^{-1}, not only about two orders of magnitude above the valued determined by modelling but also higher than the deposition velocities measured by other methods (wind tunnel, eddy-covariance, cf. Fig. 1). Applying these values to the example of the arterial road with heavy traffic mentioned above, only a few trees would be adequate to compensate for road vehicle emissions in the size class PM_1 which is especially significant in view of health aspects. However, these results have not yet been confirmed by any other studies based on the same methods.

6 Conclusions

The decisive value for the assessment of dust filtration by plants is the deposition velocity. In previous publications, we find relatively large differences between deposition velocities determined by measurement and by modelling. Average deposition velocities for vegetation are normally of the order of v_d < 1 cm s^{-1}. In the case of high wind speeds, particle sizes up to 20 μm or measurements in extensive forests with an area of several thousand hectares, the values are higher. However, when considering targeted planning in urban areas with a view to reducing dust concentration, appropriate values are in the range of v_d < 1 cm s^{-1}. At these deposition velocities, the improvement in air quality as a result of filtration by plants is about 1 % (Sec. 4); in order to compensate for local particulate emissions by road traffic, very large areas of vegetation would be needed (Sec. 5). However, the validity of the deposition velocities mentioned above is called into question by the new measurement method of FREER-SMITH et al. (2005). With filtration performance in some cases an order of magnitude higher than the values published to date, improvements in air quality could be achieved with considerably less effort. However, the *in-situ* measurements using the method proposed by FREER-SMITH et al. (2005) will still need to be confirmed by further measurements. If it is in fact confirmed that plants can reach these levels of filtration performance, it should be possible to measure differences between particle concentrations in comparisons between vegetated and non-vegetated situations. However, it will be necessary to carry out further research in this area.

The wide variety of human-biometerological effects of plants in urban environments should also be mentioned at this point. Thermal stress is a key aspect of urban climatology (MAYER et al., 2008; KUTTLER, 2004). In this connection, BONGARDT (2006) was able to demonstrate the reduction in thermal stress for city-dwellers taking an urban park as an example. However, it must still be stated that urban parks and gardens may have an effect on the general particulate concentration situation (PM_{10}) but that measurable improvements in air quality can only be achieved by planting large areas of trees with a high filtration effect (e.g. conifers). According to the studies of FREER-SMITH et al. (2005), filtration effectiveness with respect to smaller particles (PM_1) could be about an order of magnitude higher. In that case, smaller, localized areas of vegetation could also be beneficial. Even if they did not bring about any improvement with respect to the statutory concentration standards for PM_{10}, targeted planting campaigns should still be considered in this case, especially in view of the greater significance of PM_1 for human health.

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