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Review

Review on urban vegetation and particle air pollution – Deposition and dispersion

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HIGHLIGHTS

- Combining deposition and dispersion helps designing urban vegetation related to air quality.
- The dilution of emissions with clean air from aloft is crucial; limit high urban vegetation.
- High concentrations of air pollutants increase deposition; vegetation should be close to the source.
- Air floating above, and not through, vegetation barriers is not filtered; decides barrier porosity.
- Differently designed vegetation catch different particle sizes.

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ABSTRACT

Urban vegetation affects air quality through influencing pollutant deposition and dispersion. Both processes are described by many existing models and experiments, on-site and in wind tunnels, focussing e.g. on urban street canyons and crossings or vegetation barriers adjacent to traffic sources. There is an urgent need for well-structured experimental data, including detailed empirical descriptions of parameters that are not the explicit focus of the study.

This review revealed that design and choice of urban vegetation is crucial when using vegetation as an ecosystem service for air quality improvements. The reduced mixing in trafficked street canyons on adding large trees increases local air pollution levels, while low vegetation close to sources can improve air quality by increasing deposition. Filtration vegetation barriers have to be dense enough to offer large deposition surface area and porous enough to allow penetration, instead of deflection of the air stream above the barrier. The choice between tall or short and dense or sparse vegetation determines the effect on air pollution from different sources and different particle sizes.

1. Background

Urban vegetation is currently popular for the ecosystem services it can provide, such as reducing problems with flooding. Positive effects on air quality through filtration of polluted air are often mentioned, but without taking reduced dilution into account. As urban vegetation is also a way to abate the effects of climate change, e.g. rising sea level and global warming, many cities are increasingly including urban vegetation in their plans (Andersson-Sköld et al., 2015). A few reviews have been published in related areas, focussing on e.g. particle deposition on vegetation (Litschke and Kuttler, 2008); dry deposition on plant canopies (Petroff et al., 2008a); urban green space and social justice (Wolch et al., 2014); and dispersion without the complication of vegetation (Xia et al., 2014). Many studies have attempted to estimate the economic benefits of improving air quality, although the effect of vegetation on urban air quality is not yet fully understood (Tiwary et al., 2009; Escobedo et al., 2011).

The aim of this literature review was to appraise the physical effects linking vegetation to air quality from two perspectives, deposition and dispersion, and to provide input on the design of urban vegetation related to air quality. Particulate pollutants were considered in particular, as they have major health impacts and as physical processes differ for different size classes, introducing an extra complication compared with gaseous pollutants. The physical processes were reviewed at different scales, including the effects of particle properties and vegetation properties. Emissions from vegetation were excluded, as was transformation of pollutants in

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the air. Dispersion was assessed by reviewing on-site measurements, wind tunnel studies and modelling approaches, with examples from both street canyons and vegetation barriers. Unfortunately, few experimental studies combining detailed descriptions of both dispersion and deposition were available for review.

This paper commences with a description of the deposition process, followed by vegetation and then dispersion within urban areas. Finally, the effects of vegetative barriers are described and recommendations on vegetation design are provided. All parts include measured and modelled data and each part ends with a short summary of the topic.

2. Deposition

Airborne particles and gas molecules can be deposited when they pass close to a surface. Most plants have a large surface area per unit volume, increasing the probability of deposition compared with the smooth, manufactured surfaces present in urban areas. For example, 10–30 times faster deposition has been reported for sub-micrometre (<μm) particles on synthetic grass compared with glass and cement surfaces (Roupsard et al., 2013). Particle size, among other parameters, has a great effect on deposition. Ultrafine particles, below ~0.1 μm, behave more like gas molecules and are deposited by diffusion; 1–10 μm particles impact on surfaces that force the air stream to bend; and particles >10 μm in diameter also fall to the ground by sedimentation (Hinds, 1999).

Deposition on vegetation is usually described as one-dimensional vertical deposition on a homogeneous layer of vegetation in the form of a forest or field. For urban applications, the vegetation is often merely single trees or bushes, or linear stands forming avenues and barriers, and the deposition process needs to be modelled in more detail. However, most of the physics can easily be described using the situation of an airstream passing a single leaf surface instead of a whole forest.

Simplified one-dimensional deposition is divided into transport from free air to the surface; across the laminar layer adjacent to the surface; and processes relating to surface properties. The deposition velocity, v_d, is often described as the reciprocal of resistance to deposition, R_{tot} (equation (1)). R_{tot} can be divided into a sum of resistances relating to each of these transport processes, namely R_a = aerodynamic resistance, R_b = boundary resistance and R_c = surface resistance (Davidson and Wu, 1990).

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

The aerodynamic resistance is normally considered small compared with the other types and is thus set to zero, unless the study is focussing on particles with high settling velocity[1] (Hinds, 1999), i.e. with a particle diameter above 10 μm diameter (Slinn, 1982; Davidson et al., 1982). The deposition velocity is always larger than the settling velocity (Petroff et al., 2008a). In this context, the aerodynamic resistance is also related to dispersion. For aerodynamic resistance, R_a, meteorology is important and both R_a and the boundary layer resistance, R_b, depend on the reciprocal of the friction, or shear, velocity (Bruse, 2007; Petroff et al., 2008a).

Vong et al. (2010) showed that the deposition velocity measured for 0.2–0.5 μm particles depends on the atmospheric stability of the boundary layer, described by the Monin-Obukov length, L, and linearly on the particle diameter, D_p (equation (2)). Other dependencies are collected within the empirical constant A, which is 0.63 over pine forest (Vong et al., 2010), 1.35 over forests (Gallagher et al., 1997) and 0.2 over grass (Wesely et al., 1985).

$$v_d = A{*}u_*{*}D_p{*}\left(1 + \left(-\frac{300}{L}\right)^{2/3}\right) \tag{2}$$

The deposition velocity for super-micrometre particles increases with size due to increasing impaction rate and, for vertical deposition, settling velocity, while for sub-micrometre particles it decreases with size. The minimum deposition velocities reported in the literature are 0.1–0.3 μm (Slinn, 1982; Davidson et al., 1982; Litschke and Kuttler, 2008; Petroff et al., 2008a; Lin and Khlystov, 2011). Particulate matter (PM) size is often reported in large size classes, e.g. PM_{10} includes particles <10 μm in diameter which have an average diameter of either 5.0 or 0.1 μm, giving deposition velocities differing by about 100-fold (Litschke and Kuttler, 2008). Number of particles emphasises smaller particles, while particle mass emphasises larger particles.

Discrepancies can also arise depending on the complexity of the measurements. For example, Freer-Smith et al. (2005) divided particles into size fractions obtained from samples in solution and attributed all dissolved particle mass to the sub-micrometre particle size range, i.e. to airborne sub-micrometre particle mass, which thus got a huge deposition velocity. Litschke and Kuttler (2008) reported that hygroscopic particles (marine) can increase their deposition velocity by 5- to 6-fold, changing the relative humidity from 40% to 99%, and with deposition 16- to 25-fold faster in 99.9% relative humidity. Thus if humidity is not stated in the literature source, the deposition velocity for hygroscopic particles might be difficult to use. Deposition velocity data obtained from net transport of particles to surfaces indicate that sticky surfaces have greater deposition velocity than dry surfaces, e.g. as shown for 18 μm particles by Petroff et al. (2008a). Many discrepancies between published deposition velocity values are due to differences not included in the analysis (Litschke and Kuttler, 2008; Petroff et al., 2008a).

Deposition velocity, v_d, for different types of vegetation is often measured in wind tunnels, which normally force all available air to pass through the vegetation. However, this is usually not the case under ambient conditions, where the air stream can pass above or around the vegetation (see section on Barriers). In a study where the particles tested were 0.01–0.1 μm in diameter and the wind speed was 0.3–1.5 m s^{-1} , cypress (*Cupressus leylandii*) and pine (*Pinus sylvestris* L.) hedges were found to be filters with an effective filter diameter in the same range as pine needles (Lin et al., 2012). Those results confirm earlier findings that deposition velocity decreases with size for sub-micrometre particles (Petroff et al., 2008a).

$$Deposited\ amount\left(g/m^2\right) = LAI{*}v_d{*}C{*}t \tag{3}$$

The amount of material deposited per unit ground area and time is often calculated by equation (3), where LAI is Leaf Area Index, i.e. the amount of vegetation surface area per m^2 of ground area; v_d is the deposition velocity; C is the air concentration of the pollutant; and t is the time. The definition of LAI varies slightly, see below. A detailed model for transport and deposition on needles was successfully applied to three different datasets by Petroff et al. (2008b), who found slight over-prediction of capture efficiency for super-micrometre particles in light winds. The model is based on a data review (Petroff et al., 2008a) and has been further developed for broad-leaved canopies (Petroff et al., 2009). These models include all different kinds of deposition of particles (diffusion, interception, impaction, sedimentation) but exclude some processes, e.g. interactions among particles and between particles and gases,

[1] Velocity of a falling particle under zero acceleration.

thermophoresis (impact of temperature gradients) and meteorological stability. The original model has since been simplified for larger scale modelling uses and is available as an open-source model (Petroff and Zhang, 2010). Kouznetsov and Sofiev (2012) describe all processes included in their model thoroughly and suggest a description of thermophoresis. However, the model still excludes turbulent impaction, which may be important as dense vegetation in forests reduces local wind speeds, while in urban areas single vegetation elements are normally not shielded.

Deposition is transport from a point in the air to a plant surface. As the description of deposition is simplified in different ways, the experimental set-up used for measurements must be described in detail when publishing deposition velocities and modellers using these data must consider the set-up thoroughly. Thus, deposition models should be transferred between different applications with great caution. The deposition velocity has a minimum around 0.1–0.3 μm particle diameter.

3. Description of vegetation

Vegetation density affects both deposition and dispersion. For deposition, the vegetation area is either described as LAI (leaf area/ground area, dimensionless) or as Leaf Area Density (LAD; leaf area/unit volume, $m^2\ m^{-3}$ or $m^2\ m^{-1}$). For dispersion, the porosity, drag force or pressure drop is measured. Many different measures are used in the literature, reducing comparability, and either deposition and dispersion is commonly estimated from the other, introducing large uncertainty.

LAI can be measured practically by cutting all leaves within a volume and measuring the surface area directly or by laser. For example, Bouvet et al. (2007) measured the LAI of four rows of maize with both a FASTRAK three-dimensional digitiser and manually, resulting in LAI values of 3.54 and 3.52, respectively. Other studies have measured pressure loss coefficient (Gromke, 2011). Optical porosity has proven useable for large particles (80 μm) and small vegetation elements (1–10 mm), but at low optical porosity the pressure drop is lower through vegetation barriers than through solid barriers (Raupach et al., 2001). Deciduous trees commonly have porosities of around 96–97.5%, with a pressure loss coefficient of 80–200 m^{-1} (Gromke and Ruck, 2012). In a study by Lin et al. (2012), the packing density in a wind tunnel, defined as the volume of vegetation divided by the tunnel volume, was 3.7% and 5.5% for juniper (*Juniperus chinensis*) branches of different orientation and 1.7% and 4.0% for pine, relative to LAD of 109 and 197, respectively, for juniper and 94 and 138, respectively, for pine.

The porosity changes at high wind speeds, with decreased porosity for broad-leaved trees and increased for conifers (Tiwary et al., 2005). The drag forces on trees decrease with increasing wind speed (Gromke and Ruck, 2008). At 10 m s^{-1} the capture efficiency and, to an even larger extent, the deposition velocity decrease for deciduous trees compared with at lower wind speeds (Beckett et al., 2000). Hedges of different species are affected by wind speeds only above a certain threshold, e.g. 0.8, 1.2 and 1.7 m s^{-1} for hawthorn, holly and yew, respectively (Tiwary et al., 2005).

As the amount of deposited mass is directly related to air concentrations close to the surface, Weber et al. (2014) ensured that the air concentrations around different herbaceous plants based on distance to the road were similar when measuring the mass deposited. They found that hairy leaves increased deposition substantially for 3–180 μm particles. Speak et al. (2012) analysed the deposits on different vegetation species on a rooftop and found greater deposition on grass and on hairy leaves than on other herbaceous plants.

Greater deposition velocities for conifers than deciduous trees have been found in several studies e.g. by particle capture of 0.8 μm NaCl particles in a wind tunnel (Beckett et al., 2000; Freer-Smith et al., 2004). The deposition velocity increased from 0.1 to 0.3 m s^{-1} to 2.9 m s^{-1} when the wind speed was increased from 3 m s^{-1} to 9 m s^{-1} (Freer-Smith et al., 2004). The relative deposition velocity on stems compared with leaves increased for smaller stems and larger leaves, while typical semi-arid region trees had low deposition velocities (thick leaves) (Freer-Smith et al., 2004). In other studies, Przybysz et al. (2014) found greater deposition on pine than on yew, and even less deposition on ivy (*Hedera helix* L.); soot particles had greater deposition velocities on needles than on broadleaved species (Hwang et al., 2011); and juniper gave larger deposition in wind tunnel tests than loblolly pine (*Pinus taeda*), but also affected the wind field more (Lin et al., 2012).

Particles, mainly the coarser fraction, are washed off from foliage during rain (Przybysz et al., 2014). If deposited in the leaf wax, the removal of particles with wind or rain is lower (Dzierżanowski et al., 2011). Measurements on 13 plant species showed that approximately 60% of the particle deposit was washed off with water, while 40% was included in the wax layer, with a large variance between species (Popek, 2013). Three-year-old needles had more polycyclic aromatic compounds (PAH) transferred into the needle wax, i.e. impossible to wash away with water and ultrasonic bath, showing an age effect on the wax in the needle surface (Terzaghi et al., 2013). Only particles smaller than 10 μm were encapsulated into the leaf/needle cuticle, i.e. could not be washed away by water (Terzaghi et al., 2013).

Using samples from more than 40 species, Saebo et al. (2012) found a positive correlation between particle deposition and hairy leaves and the wax content of the leaves. Thick leaves showed lower deposition for all particle sizes, apart from 0.2 to 2.5 μm particles. There was a 10–20 fold difference between different species in terms of particle deposition (Saebo et al., 2012).

Vegetation density or porosity is generally measured using several different parameters. High vegetation density increases deposition of pollutants that reside close to the surface, but can also hinder the transport of pollution towards the surface. There is a need for standardised measurements of vegetation density, increasing comparability between studies.

Different vegetation species have different deposition velocities even for the same particle size range, but the available data cannot yet give a parameterised description. Establishment of specific parameters to describe vegetation is important for standardising vegetation parameterisation in experiments. Hairiness and possibly wax content have been shown to increase deposition, while a difference between thick and thin leaves relating to particle size is also probable.

4. Dispersion in urban areas

Dispersion relies on descriptions of wind systems that transport and dilute air pollutants at different scales. Regional wind fields, including vertical layering, affect air pollutants at a larger scale, while fluid dynamics often describe air flows around obstacles in street canyons and inside vegetation barriers. Surface roughness is a simplification describing the effect of surface texture on the wind field and is used at different scales in the literature. The buildings in the urban area give a surface roughness in regional models, but need to be resolved as objects within the urban area. Resolving vegetation details like twigs and leaves is also sometimes needed.

Most dispersion studies exclude temperature effects, e.g. sunlit surface versus shadow (Lindberg et al., 2008) and sinking cold air (Baik et al., 2012). The temperature-humidity system is closely linked to vegetation, as plants tends to decrease temperature differences in urban areas (Lee and Park, 2008).

A common subsystem of urban areas is the street canyon. For simplicity, the air flow in a street canyon is generally described for a constant, perpendicular rooftop wind that creates a vortex in the street canyon, forming a ground level wind that has the opposite direction from the rooftop wind (Oke, 1987) (Fig. 1). In reality, the vortex is affected both by building configuration and by vegetation (Ng and Chau, 2012, 2014). For wind directions parallel to the street canyon, the flow is channelled through the canyon, and other wind directions are considered combinations of the two. Street canyon vegetation gives different effects on dispersion in these two cases (Oke, 1987).

Salmond et al. (2013) used the difference between seasons, i.e. with and without leaves on the trees, to examine the effect of vegetation on air quality, implying a need to limit or understand other seasonal differences. They measured NO and NO_2 both at street level and one floor up, i.e. below and within the tree crowns. They found a rapid, large fluctuation in concentration in all data apart from one floor up during summer, showing the decreased mixing within the foliated tree crowns. During summer, the air pollutant concentrations differed less between leeward and windward sides of the street when wind direction was perpendicular to the street canyon, showing that the normally created street canyon vortex was reduced (Salmond et al., 2013). The NOx concentrations also differed more between the urban background and the street canyon during the foliated season (Salmond et al., 2013).

Buccolieri et al. (2011) measured and modelled the concentration of PM_{10} in a real junction in two different wind directions by FLUENT (www.ansys.com). As one of the streets passing the junction had trees and the other did not, the ratio between the wind directions parallel to each street canyon was used to compare the modelled concentration ratio (1.1) to the measured (1.5). The modelled ratio for the same situation without trees was 0.3, emphasising the importance of including vegetation in the model (Buccolieri et al., 2011).

Wind tunnels are powerful tools for studying fluid dynamics, since it is possible to scale the fluid while keeping dimensionless numbers constant (http://www.cfd-online.com). However, scaling of complex vegetation is still a challenge (Gromke, 2011). Gromke and colleagues have studied this issue in detail and summarised many of their findings in a recent paper (Gromke and Ruck, 2012). A street canyon with a height/width (H/W) ratio of 1–2 was built in the wind tunnel with length 10 times the height of the buildings. The emissions were introduced as a line of point sources of inert gas and the traffic turbulence introduced by small rectangular plates moving with the traffic flow (Gromke et al., 2008). At both building walls, the concentration of the gas and wind speed were measured. In the first studies, the trees in the street canyon looked like small-scale trees, with spherical tree crowns on thin stems (Gromke and Ruck, 2007). The crowns had different porosity and different material. To simplify the studies of different vegetation density, the trees were replaced with metal cages that were filled with different amounts of fibre filling. Test with empty cages and with filling in every second cage showed only minor effects on the flow (Gromke, 2011; Gromke and Ruck, 2012). More or larger trees increased the concentration and reduced traffic turbulence (Gromke and Ruck, 2007). The wind field was found to be disturbed by the presence of the tree at a distance of at least 5 times the crown diameter downwind (Gromke and Ruck, 2008) for the rather low tree porosity used (Gromke and Ruck, 2012). The largest effect on the wind field was from trees with high porosity (~97.5%) (Gandemer, 1981; Grant and Nickling, 1998; Frank and Ruck, 2005).

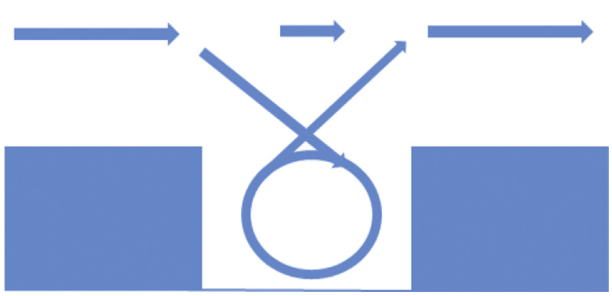

Fig. 1. The vortex of a street canyon with perpendicular wind direction.

The wind tunnel findings above were used in CFD (Computational Fluid Dynamics) modelling (Gromke et al., 2008), where two different turbulence schemes, k-ε and RMA (Yakhot et al., 1992), were tested in the FLUENT model (www.ansys.com). Both gave slightly lower dispersion compared with the wind tunnel measurements, but the RMA scheme provided the best description of the measured wind tunnel data in this case. The Schmidt number (turbulence description) was decreased from the commonly used 0.7 to 0.3, approaching the recommendation of 0.4 for urban street canyons (Di Sabatino et al., 2007). For perpendicular winds, larger tree crowns increased the difference in pollutant concentration between the sides of the street, and for wind directions parallel to the street canyon, the effect of trees was limited. The effect of vegetation was greatest (by a factor of 3 at maximum concentration) for winds at a 45-degree angle (Buccolieri et al., 2011). For larger H/W-ratios (i.e. deeper or narrower street canyons), the effect of trees increased (Buccolieri et al., 2009). A Large Eddy Simulation model for street canyons with and without trees showed slightly over-predicted concentrations far from crossings, but still a close resemblance to the wind tunnel studies (Moonen et al., 2013). CFD models including vegetation improved resemblance between modelled and measured concentrations (Amorim et al., 2013a) and also improved calculation of exposure to traffic emissions in children walking different paths to school (Amorim et al., 2013b).

Most street canyon models describe vegetation as a sink for turbulence, but without considering deposition. Two studies using ENVImet (http://www.envi-met.com/) showed higher pollutant concentration due to vegetation, both in street canyons (Wania et al., 2012) and between different buildings (Vos et al., 2013). Larger and denser trees greatly reduced the dispersion, while the impact was limited for smaller and sparser trees (Wania et al., 2012; Vos et al., 2013). Vos et al. (2013) recommends lower hedges or even walls between traffic and pedestrians, limiting polluted air transport to pavements and placing vegetation close to the source, increasing deposition (Vos et al., 2013; Pugh et al., 2012). Deep street canyons are more sensitive to larger tree coverage than shallower, and the building design can have a large effect on dispersion too (Ng and Chau, 2014).

Pugh et al. (2012) used a simplified model including the different concentrations in the street canyon and at rooftop height to calculate the deposition and demonstrated the importance of placing the vegetation close to the source. A thorough review of the state-of-the-art of environmental benefits of green roofs only dedicated a small section to air pollution (Berardi et al., 2014).

Some studies publish limited datasets to validate models, but extensive experimental datasets, including a thorough description of the vegetation inside urban areas, are needed to improve existing models.

Wind tunnel studies provide many insights into pollutant dispersion and e.g. the downscaling of vegetation adds large uncertainty to these studies. Thus, such studies are highly recommended and should be further linked to on-site measurements.

Dispersion modelling shows a negative impact on air quality if trees are introduced in trafficked street canyons, a limited effect from sparse street trees and positive effects of low barriers between traffic and exposed inhabitants. Dispersion modelling normally does not include the effect of vegetation on heat flux and buoyancy, which influence dispersion in urban areas.

5. Parks and regional deposition

The regional removal of pollutants by deposition on vegetation in urban areas has been calculated from reported deposition velocities and averaged concentrations, together with measured or estimated vegetation surface areas. Due to large spatial variability of both vegetation surface area and air pollutant concentrations, averaging problems are common.

Most studies calculate the total deposition from urban background concentrations and average LAI, and report PM_{10} reductions of a few per cent (Nowak, 1994; Nowak et al., 2006; McDonald et al., 2007; Bealey et al., 2007; Litschke and Kuttler, 2008; Baumgardner et al., 2012). These calculations often do not take account of the limited dispersion due to increased amounts of vegetation. Meteorological inversion and spatial heterogenity have been shown to have large impact on the vegetation effect (Escobedo and Nowak, 2009).

A study combining the UFORE model (Escobedo and Nowak, 2009) and GIS (Geographical Information System), forming i-Tree (http://www.itreetools.org), showed the possibility to use GIS-based systems to find areas where increased vegetation would be beneficial for air quality (Hirabayashi et al., 2012). This helped in abating ultrafine, but also coarse, particles with concentrations differing by several orders of magnitude within urban areas, while $PM_{2.5}$ concentrations vary less within urban areas (Whitlow et al., 2011; Hagemann et al., 2014).

Quantifying the total amount of deposition over larger areas needs further studies, due to the large spatial variation in most air pollutants and in vegetation cover.

6. Barriers and varying pollution

Barriers between the source of pollution and humans can be used both as a way to change the wind system and for filtering the air, i.e. through dispersion and deposition. They are easier to study due to their simple geometry, but still include many interesting complications. Barrier studies are often performed at roadsides outside urban areas in order to limit disturbances to traffic, but the theory can help understand the effects on urban air quality. In one study in which wind speed was measured around an 8 m high cypress barrier, there was no effect of the barrier 160 m downwind (Tuzet and Wilson, 2007). Solid barriers reduced pollutant concentrations downwind of the barriers, with different reduction rates for different pollutants and different barriers. The measured pollutants reported include number of particles (20 nm, 75 nm and total; Baldauf et al., 2008); NO_2, black carbon (BC), CO, particle number and mass for sub-micrometre particles (Ning et al., 2010); and ultrafine particles (Hagler et al., 2012). At 20 times the barrier height, Ning et al. (2010) found higher concentrations than without a barrier.

One of the particle number peaks measured by Hagler et al. (2012) was used with the Comprehensive Aerosol and Gas Chemistry (CAGC) model by Wang (2013), working with FLUENT as the k-ε turbulence solver. With measured LAD, the model gave slightly larger capture of number of particles below 50 nm diameter than the measured value, but on using lower LAD the capture rate for number of larger particles became too low. The deposition velocity is related to particle size, but does not seem to include particle size-differentiated wind speed effects (Steffens et al., 2012). Particles below 50 nm are common in traffic exhausts, but also difficult to model in the complex near road environment, where particle dynamics play an important role (Steffens et al., 2012). Hagler et al. (2012) found limited effect on particle concentrations from a vegetation barrier. This was attributed to low LAI (around 3 during summer) and gaps between the trees allowing transport of unfiltered air through the barrier (Hagler et al., 2012).

A maple and oak barrier with under-vegetation close to a road reduced BC concentrations by 12%, with a maximum reduction of 22%. Particles between 0.5 and 10 μm diameter showed a limited reduction in the study (Brantley et al., 2014). Twenty sites in Finland were analysed by diffusive sampling for particle deposition, NO_2 and a selection of VOCs at parallel sites with and without trees adjacent to the road (Setälä et al., 2013). The effect of vegetation was limited, even with under-vegetation; the reason might be low traffic impact on air quality as only NO_2 correlated to traffic flow at the site. During a shorter study, number of particles was reduced by one-third by vegetation compared with no vegetation, suggesting larger effects on exhaust particles (Setälä et al., 2013).

At four locations less than 2.2 m from a conifer barrier, numbers of particles of different sizes were determined: in an opening in the vegetation; close to the barrier at both sides; and inside the barrier (Al-Dabbous and Kumar, 2014). For wind directions from the road, the number of particles was slightly higher close to the road and at the barrier than in the opening. Directly after the barrier, the concentration had decreased by ~40%. All wind directions showed lower concentrations of particle numbers within and behind the vegetation than close to the road, again pointing to an effect on exhaust particles.

Many studies focus on dust and coarse particles (Raupach et al., 2001; Bouvet et al., 2007). Combined modelling and measurements around a barrier of four rows of 2 m high maize plants, with LAI of 3.5 and an optical porosity between 0.05 and 0.67, showed that a large quantity of 10–50 μm diameter glass beads passed above the barrier. Thus, deposition in the barrier was not possible for most of the glass beads. High porosity vegetation barriers are penetrated by air streams, allowing deposition of pollutants, while low porosity vegetation forces air streams to pass above it (Tiwary et al., 2005).

The effect of ~2 m high hedges on wind fields and concentrations of super-micrometre particles was attributed either to the wind field change or to the deposition in the hedge by Tiwary et al. (2005). For yew, the wind did not penetrate the hedge and most of the air passed above it, while the porosity was higher for holly and still higher for hawthorn. The collection efficiency at two-thirds the height of the barrier increased with particle size (from 0.8 to 15 μm) and decreased with porosity, with maximum collection efficiency for 15 μm particles of 3% for yew, 18% for holly and 27% for hawthorn (Tiwary et al., 2005). Tiwary et al. (2008) repeated the study for the hawthorn hedge and found similar collection efficiencies of 38%, 30% and 33%, implying statistically sound data.

Placing vegetation barriers close to a road increases the amount of deposition on the vegetation, as the concentration of dust is high when the plume impacts on the vegetation and as the full height of the plume passes through the barrier (Etyemezian et al., 2004). Tall oaks and cedars 25 m from a road halved PM_{10} and $PM_{2.5}$ concentrations, while tall prairie grass reduced the concentrations by 35% (Cowherd et al., 2006). A 100 m barrier of sparse vegetation reduced PM_{10} concentrations by less than 10% and 17–25 μm particles by 25% (Etyemezian et al., 2004). For a model of urban areas, formed by placing containers on a field, Veranth et al. (2003) found an 85% decrease in PM_{10}. This large decrease was possibly related to the high friction velocity. If the barrier is too far from the road to capture the full plume height, the collection efficiency is low, e.g. 10 m high trees 60 m from a gravel road gave no detectible effect on

PM_{10} concentrations (Mao et al., 2013). Mao et al. (2013) showed reasonable agreement between measured and modelled wind speeds, while dust concentrations differed, implying a need to improve the description of deposition in the model.

Pardyjak et al. (2008) described a simple quasi-2D Eulearian atmospheric dispersion model that accounts for dry deposition of fugitive dust onto vegetation and buildings, using measured v_d*LAD as input data calculating the total mass concentration in the dust plume. The model is easily available and helps planners to understand how vegetation design affects the plume concentration by relating the importance of each process to the relation between the turbulent diffusion time scale and the deposition time scale.

Focussing on the physical dispersion around trees, Endalew et al. (2009a, b) describe a model that resolves the tree by excluding the leaves in 3D as stem and twigs that can grow according to different parameters forming different tree types. The leaves are then added to the system as turbulence sinks that surround the stem and twigs. The drag force used in the model is calculated by leaf area density or leaf drag area in m^{-1} (Endalew et al., 2009a), drag coefficient and a sheltering factor (Endalew et al., 2009b). The sheltering factor is of greater importance in vegetation with a larger extension like parks, but also e.g. inside tree crowns. This approach has been compared to wind tunnel data on 1:10 scale trees and to modelling with the common horizontal averaging technique for wind speeds of 10 and 15 m s^{-1} with positive results (Endalew et al., 2009a, 2009b). One or two trees are measured with photography techniques and a canopy can be formed by symmetry boundary conditions. In the wind tunnel, the roughness on tree branches without leaves was important, but gave no significant effects in real world, and unfoliated trees gave a 50% reduction of the wind speed in the centre of the canopy (Endalew et al., 2009a). There is a need for further studies of leafless vegetation.

Vegetation barriers have been studied more frequently than vegetated street canyons giving important insights into the effect of vegetation on air quality. They show the great importance of designing urban vegetation, carefully relating it to the kind of air pollution targeted. Thus, if not considering reduced dilution, vegetation barriers should be placed close to the road where the concentration is high and have at least the same height as the plume from the road. The barrier should allow polluted air to pass through, allowing deposition, or to pass above, protecting areas close behind the barrier.

Barriers are efficient study objects, as they reduce the complexity of studies of vegetation and air pollution. Different vegetation types and different kinds of pollution or particle sizes should be studied.

7. Conclusions

The effect of vegetation on urban air quality depends on vegetation design and on level of air pollution in the area. This review identified the following vegetation design considerations based on air quality arguments:

1. Dilution of emissions with clean air from aloft is crucial; the vegetation should thus preferably low and/or close to surfaces.
2. Proximity to the pollution source increases concentrations of air pollutants and thus deposition; vegetation should be close to the source.
3. Air passing above, and not through, vegetation is not filtered; barriers should be high enough and porous enough to let the air through, but solid enough to allow the air to pass close to the surface.

Other interesting findings are that deposition of coarse particles is more efficient at high wind speeds, while the opposite is true for ultrafine particles; and that vegetation density often changes due to strong winds. To improve deposition, the vegetation should be hairy and have a large leaf area index, but still be possible to penetrate.

7.1. Research outlook

Dispersion and deposition related to vegetation in urban areas are both interesting and vivid research areas. This review suggests that these areas be further combined, as the environmental problem in which they interact, urban air quality, is crucial to human health and results are rapidly transferred into policy. Thus, results from one area must be modified with results from the other before action is taken in urban planning.

The effect of non-foliated vegetation during wintertime needs further studies, as they might have an impact e.g. in northern countries with air quality problems during winter and spring. In these areas, air inversion during wintertime often limits dilution, so pollution levels might be high. There is also a yearly variation, with different particle sizes being most important during different parts of the year.

The deposition process differs substantially between different particle sizes and detailed interactions with various vegetation elements require combined studies of different particle sizes together with different plant species.

Barriers are important for experimental data collection due to their simple geometry, which is a requirement for detailed deposition and dilution studies. The possibility of studying roads where a barrier is present along only a part of the road, providing surroundings and emissions that are similar for stretches with and without vegetation, is very important. Barrier studies can give great insights into differences between pollutants and between different kinds of vegetation.

The description of the vegetation is important, as recommendations can scarcely include all available species, but must group them in some way. Parameters such as hairiness, stickiness, LAI, thickness of leaves etc., but also porosity and the species in question, are described in the literature. Vegetation can interact with air pollutants in more ways than these, however, e.g. through emissions from vegetation and active uptake of water and nutrients. Therefore studies of vegetation effects need to include these other factors before vegetation implementation in urban planning can be efficient.

This review examined the deposition and dispersion of particle pollution of all size classes and showed that the effects of urban vegetation on local air quality are complex, so different disciplines must work together to identify these effects. Such work must be described in great detail, as we do not yet understand all the parameters influencing the effects of vegetation on air pollution.

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