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Modelling the effectiveness of urban trees and grass on $PM_{2.5}$ reduction via dispersion and deposition at a city scale

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HIGHLIGHTS

- We model the effectiveness of trees and grass on traffic $PM_{2.5}$ reduction.
- City scale CFD simulations were performed under the OpenFOAM software.
- Aerodynamics effect of tree prevails over deposition.
- Tree are beneficial for wind speeds greater than 2 m s^{-1}.
- $PM_{2.5}$ deposition on buildings is negligible with less than 0.03 %.

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ABSTRACT

Green infrastructure can reduce $PM_{2.5}$ traffic emissions on a city scale, by a combination of dispersion by trees and deposition on buildings, trees and grass. Simulations of $PM_{2.5}$ concentrations were performed using a validated CFD model. A 2 × 2 km area has been reconstructed as a 3D representation of Leicester (UK) city centre which is on a scale larger than most of the other CFD studies. Combining both the effects of tree aerodynamics and the deposition capabilities of trees and grass is also something that has not yet been modelled at this scale. During summer time in Leicester City, the results show that the aerodynamic dispersive effect of trees on $PM_{2.5}$ concentrations result in a 9.0% reduction. In contrast, a decrease of $PM_{2.5}$, by 2.8% owing to deposition on trees (11.8 t $year^{-1}$) and 0.6% owing to deposition on grass (2.5 t $year^{-1}$), was also observed. Trees and grass are shown to have greater effects locally, as smaller decreases in $PM_{2.5}$ were found when considering reduction across the whole boundary layer. Densely built areas like Leicester City centre have relatively less vegetation and subsequently have a smaller effect on $PM_{2.5}$ concentration. It was found that particle deposition on buildings was negligible with less than 0.03%. An empirical equation was derived to describe the changes in $PM_{2.5}$ based on ground surface fraction of trees and grass, and their deposition velocities.

1. Introduction

Road traffic emissions are one of the largest contributors of air pollution in urban environments, contributing to up to 66% of particulate matter with aerodynamic diameter less than 2.5 μm ($PM_{2.5}$) (Sundvor et al., 2012). An excess of inhaled particulate matters can present adverse health effects such as premature death, lung cancer, cardiovascular disease and asthma attacks among health outcomes (Kim et al., 2015). The World Health Organisation recommends that $PM_{2.5}$ concentrations should not exceed the guideline value of 10 $\mu g/m^3$ as a yearly average and 25 $\mu g/m^3$ as a daily average (WHO, 2006). 85% of the European population lives above these recommended levels of $PM_{2.5}$ (Guerreiro et al., 2013). In developing countries such as China, these thresholds are sometimes exceeded by an order of magnitude (Chan and Yao, 2008). Urban vegetation and green barriers have been shown to offer passive mitigation for air pollution (Gallagher et al., 2015; Li et al., 2016; Tong et al., 2016; Al-Dabbous and Kumar, 2014; Morakinyo and Lam, 2015). Regional scale modelling studies have shown a modest impact of trees on particle deposition with less than a few percent reduction (Tallis et al., 2011; Beckett et al., 1998; Nowak et al., 2006; Selmi et al., 2016). However, at the

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street canyon scale, modelling studies suggest that green walls can decrease pollutant concentrations as much as 40% for NO_2 and 60% for PM_{10} (Pugh et al., 2012). Vegetation barriers were also shown to reduce pedestrian exposure on ultrafine particles up by 37% under realistic wind conditions (Al-Dabbous and Kumar, 2014). There are also other benefits of urban green spaces, in that they contribute to the well being of the urban population (White et al., 2013) and road side vegetation also regulates the traffic noise level of busy streets (Kalansuriya et al., 2009). Vegetation can sometimes have adverse effects. Urban trees have been shown to increase pollutant concentrations in some street canyon configurations, as they modify the street canyon roughness properties changing the wind flow behaviour (Gromke et al., 2008; Buccolieri et al., 2011; Wania et al., 2012; Vos et al., 2013; Salmond et al., 2013; Gromke and Blocken, 2015). However, for winds parallel to street canyons, urban trees have been found to decrease road traffic emissions (Amorim et al., 2013; Abhijith and Gokhale, 2015). When looking at the global city scale, Barnes et al. (2014) have demonstrated that the urban surface has a direct impact on the dispersion of air pollution with pollutant concentrations increasing with lower surface roughness. This result has been confirmed in a modelling study around the City of Leicester where the aerodynamic effects of trees have been shown to decrease pollutant concentrations owing to an increase in turbulence production (Jeanjean et al., 2015). Recent reviews have pointed out that very little has been done attempting to integrate both the aerodynamics and deposition effects of trees on a city scale (Janhäll, 2015; Salmond et al., 2016), only a few studies focus on this aspect at the street canyon scale (Vos et al., 2013; Vranckx et al., 2015; Steffens et al., 2012), and not the city scale. The objective of this paper is to study both the aerodynamics and deposition effects of trees and grass on road traffic emitted $PM_{2.5}$. The study focuses on the City of Leicester in the United Kingdom over a total 2 × 2 km area (4 km^2) using real 3D trees, grass, roads and 3D buildings data. The simulations were performed using a Computational Fluid Dynamics (CFD) model, previously validated against available wind tunnel measurements (Jeanjean et al., 2015). The impact of each individual model components such as buildings, trees and grass on $PM_{2.5}$ reduction were individually studied. The atmospheric lifetime of $PM_{2.5}$ ranges from days to week whereas PM_{10} lasts for a few hours to days (Gugamsetty et al., 2012). This translates into a settling velocity of around 0.5 cm s^{-1} for PM_{10} which is an order of magnitude larger than a factor than the $PM_{2.5}$ settling velocity of 0.02 cm s^{-1} (Lapple, 1961).

2. Experimental

2.1. Model description

CFD models are often used to simulate wind flow in urban street canyons as they can reconstruct wind vorticity inside the canyons. The model used here is based on the OpenFOAM (Open Field Operation and Manipulation) software platform using the k-ε model. OpenFOAM is open source and is freely available. This model was previously validated for wind flow and particles dispersion (Jeanjean et al., 2015). The validation exercise shows an overall model accuracy of 30–40% on modelled pollutant concentrations. In the literature, RANS CFD models based on OpenFOAM show similar performances (Vranckx et al., 2015; Vranckx and Vos, 2013). Other CFD models, such as LES (Large Eddy Simulations) capture time dependent flow structures and are known to offer improved performance compared to RANS simulations (Di Sabatino et al., 2013). RANS simulations are especially tied to the importance of the turbulent Schmidt number (Sc_t) for pollutant dispersion compared to LES (Gousseau et al., 2011). RANS simulations are however much less computationally demanding than LES which makes them more suitable for city scale simulations, as it is the case for the present study. The computational grid was modified to take into account the grass surface (see Fig. 1). Guidelines in respect to CFD simulation of air flow in urban environments provided in the COST Action 732 (European Cooperation in the field of Scientific and Technical Research) was used to parameterise the CFD model (Franke et al., 2007).

To develop a model for the City of Leicester, a 3D LIDAR dataset of the buildings was constructed in 2007 with a resolution of 25 cm (see developed city model in, Fig. 1). This was combined with a road map from the same year provided by Leicester City Council. The road map includes major junctions and omits residential roads that have low traffic volumes. The traffic in this study was assumed to be uniform across all roads with an arbitrary $PM_{2.5}$ road emission value of 190 μg s^{-1} m^{-1}, which roughly led to an average ground concentration of 44 μg m^{-3} at a wind speed of 4.6 m s^{-1}. The National Tree Map^{TM} (NTM) Crown Polygon produced by Bluesky Ltd was used in the tree database to represent individual trees or closely grouped tree crowns (Bluesky, 2014). The areas covered by grass were obtained by downloading the *OS MasterMap Topography Layer* produced by the UK governmental mapping agency, Ordnance Survey. The grass was treated as a smooth surface with a surface roughness of 0.03 m according to the World Meteorological Organisation classification (WMO, 2008). The rest of ground surface between buildings was treated as a surface roughness of 0.10 m, which corresponds to large occasional obstacles. An idealised tree population was considered, which corresponds to the average tree profile encounters in the East Midlands region of the UK. The vertical distribution of leaf was kept constant which corresponds to the average tree LAD in Leicester, previously estimated at 1.6 m^2 m^{-3} (Jeanjean et al., 2015). In regards to tree species management, studying the impact of vertical distribution of leaf for different tree species as well different canopy shapes leaves room for future research. The trees were modelled as a porous media which results in a perturbation of the air flow and in removal of particles via deposition. To take into account an average wind profile, 12 wind directions were calculated every 30° and then aggregated into a single average of $PM_{2.5}$ concentration. The impact of the wind speed was also investigated with simulations for a turbulent wind flow of 4.6 m s^{-1} (which corresponds to the average wind speed in the UK) and for a low wind speed of 1 m s^{-1} (which corresponds to a laminar flow, without turbulence). The mean velocity flow and the turbulent dissipation were set up to follow a logarithmic law to reflect an atmospheric boundary layer profile on the bounding edges of the computational domain. Five independent scenarios were modelled looking tree aerodynamics and deposition on trees, grass and buildings. These cases were compared against a reference scenario without any tree aerodynamics and deposition (see Table 1). To investigate areas with different tree and grass cover settings, the City of Leicester was divided into smaller subsets: city centre, suburbs, road sides and suburb road sides (see Fig. 2). All reported values are at ground concentration of 1.5 m, to reflect the effect of tree at pedestrian height.

2.2. Deposition velocities

The model was enhanced with additional sink terms which take into account the deposition of $PM_{2.5}$ on trees, grass and buildings using the same implementation method as Vranckx et al. (2015). The range of dry deposition velocities in the literature are very wide, as dry deposition velocities are highly dependent on the vegetation species and particle diameters (size distribution). As a single average deposition velocity would not be representative, the simulations were bounded by the lowest and highest published deposition (Litschke and Kuttler, 2008). For trees, the deposition

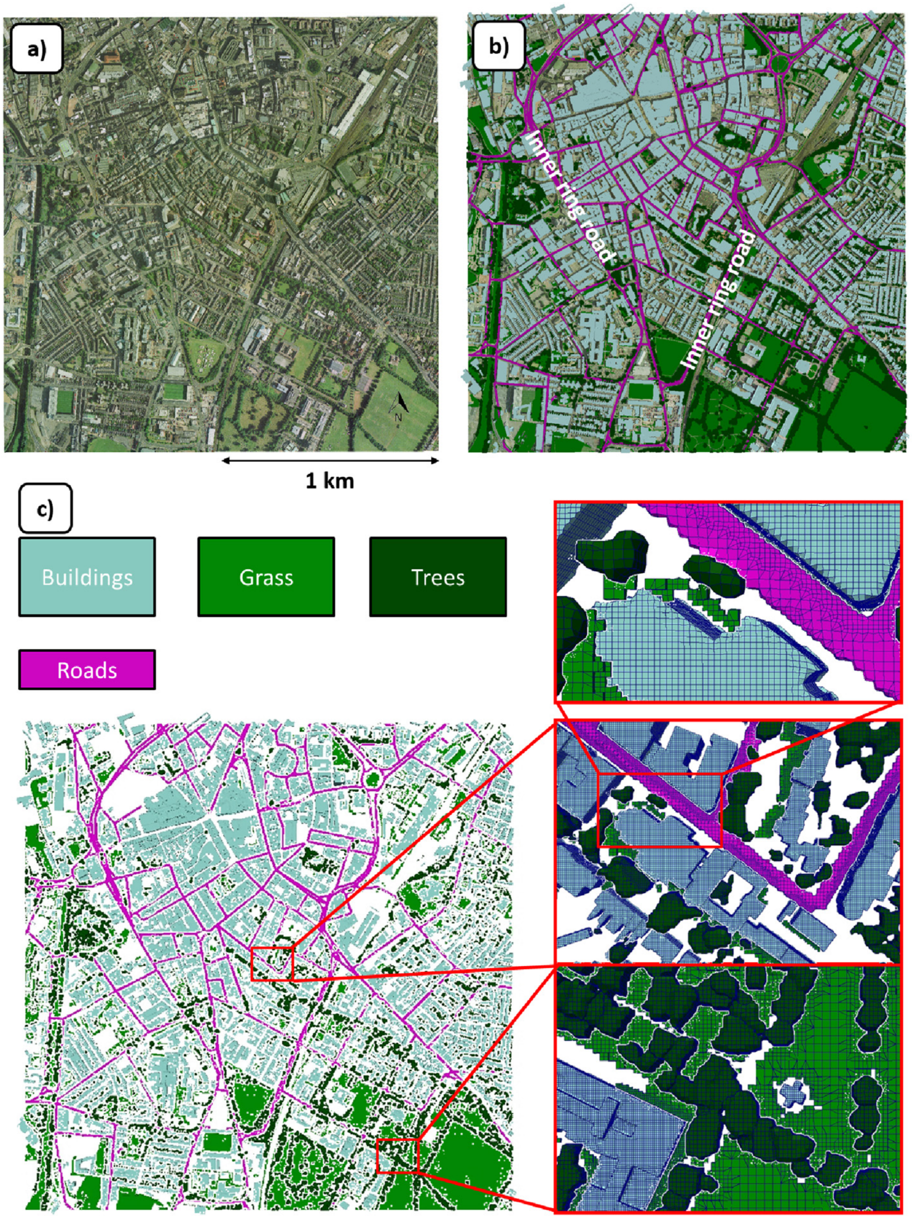


Fig. 1. Leicester City 2 × 2 km area of interest. (a) Aerial photography of Leicester City in summer 2007. Urban structures are predominant with some green spaces located at the South East of the city. (b) Aerial photography combined with the LIDAR data of the buildings, the road map, the national tree map (NTM^{TM}) from Bluesky Ltd and the grass areas from the UK mapping agency (Ordnance Survey). (c) Mesh of the Leicester City area viewed from the CFD software OpenFOAM. More than 17 million cells were used with a resolution of 1 m for each individual building, 2 m for the trees and roads, and 4 m for the grass. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

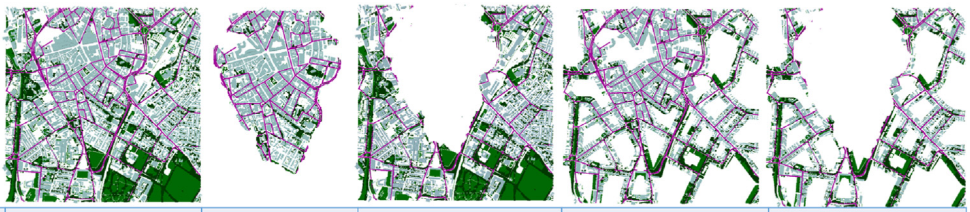
Table 1
Set of modelled scenarios and associated deposition velocities.

Scenario	Deposition velocity (cm s^{-1})		
	Low	Average	High
1. Reference scenario: building aerodynamics	–	–	–
2. Building aerodynamics and deposition	–	3.6× 10^{-3}	–
3. Tree aerodynamics	–	–	–
4. Tree aerodynamics and deposition	0.02	0.64	30
5. Grass deposition	0.01	0.64	8

velocities range from 0.02 cm s^{-1} for species such as Picea (Peters and Eiden, 1992) and Ficus (Freer-Smith et al., 2004) to 30 cm s^{-1} for the common Hazel (White and Turner, 1970). For grass, the dry deposition velocities range from 0.01 cm s^{-1} (Horbert et al., 1976) to 8 cm s^{-1} (Harrison et al., 1996). Although an average deposition velocity can be challenging to estimate, a conservative value of 0.64 cm s^{-1} chosen by Pugh et al. (2012) was used for trees and grass. Regarding building deposition, the dry deposition velocity of particles on cement of 3.6× 10^{-3} cm s^{-1} was used (Roupsard et al., 2013). The deposition inside the tree crown cells was parametrised as a sink term applied at each Eulerian step such that

$$\Delta C(t) = C(t-1) \times LAD \times Vd \tag{1}$$

where Δ C(t) is the change in particles concentration via deposition in an Eulerian forward step (g m^{-3}), C(t−1) is the particles concentration (g m^{-3}), LAD is the Leaf Area Density (m^2 m^{-3}) and Vd is



Ground surface (%)	Full scene	City centre	Suburb	Road sides	Suburb road sides
Buildings	29	42	24	28	22
Roads	9	14	7	14	12
Trees	12	6	14	10	13
Grass	19	4	25	13	18

Fig. 2. Subsets of Leicester city. These subsets were used to investigate the changes in $PM_{2.5}$ and their relation to tree and grass ground surface fraction (%).

the deposition velocity ($m\ s^{-1}$). Deposition on grass and buildings differ from trees as grass and buildings are represented as surfaces. The change in $PM_{2.5}$ concentration via deposition on grass and buildings was expressed as

$$\Delta C(t) = C(t-1) \times Vd \times \frac{S}{V} (\times LAI_{grass}) \qquad (2)$$

where C(t−1) is the particles concentration ($g\ m^{-3}$), S is the surface of grass (m^2) and V the volume of the cells where the buildings or grass are present (m^3). The leaf area index (LAI - $m^2\ m^{-2}$) is an index used here to represent the total area of grass per meter square of ground occupied by grass. Urban grass areas have been parametrised in previous models with a LAI ranging between 1 and 2 $m^2\ m^{-2}$ (Petroff and Zhang, 2010; Pugh et al., 2012). A LAI of 1 $m^2\ m^{-2}$ was used for urban grass, which corresponds to the lower end range of LAI. In this study, no changes in aerodynamics resistance were considered for the deposition sink terms used in Eq. (1) and Eq. (2) which were kept constant across the two wind speeds of 1 and 4.6 $m\ s^{-1}$.

3. Results

3.1. Reduction by trees and grass

For a wind speed of 4.6 $m\ s^{-1}$, the aerodynamic effect of trees increases traffic-sourced concentrations in street canyons but shows a decrease in open terrain (Fig. 3a). For a wind speed of 1.0 $m\ s^{-1}$ (Fig. 3b), no turbulent dispersion occurs under laminar conditions and trees are shown to significantly increase concentrations along the road sides. This effect occurs as trees are reducing the wind speed which then decreases the net dispersion. For deposition, trees decrease $PM_{2.5}$ concentrations locally (close to where they are planted) and are more efficient when placed close to road sides where particle concentrations are greatest (Fig. 3c,d). Deposition on trees is more important at a wind speed of 4.6 $m\ s^{-1}$ and almost insignificant at a wind speed of 1 $m\ s^{-1}$. Like for deposition on trees, grass decreases $PM_{2.5}$ concentrations locally and close to the road sides where particle concentrations are greatest (Fig. 3e,f). Grass deposition shows similar results in terms of loss at both strong and low wind speeds. The change of $PM_{2.5}$ by building deposition was less than 0.03% and is subsequently not detailed owing to its low impact on $PM_{2.5}$ reduction.

The model results in Fig. 4 show that the aerodynamic effects of trees prevails over the tree and grass deposition. Although the dispersive effects can appear important, similar results are found in the literature with a 12% increase in concentration for winds perpendicular to the street canyon and a 16% decrease for parallel winds (Amorim et al., 2013). Other studies measured an overall reduction in black carbon concentration by 12% downwind of trees by combining dispersion and deposition, which is comparable to the results presented in Fig. 4 (Brantley et al., 2014). In Fig. 4, the model uncertainties on estimating the trees and grass depositions are large as the choice of deposition velocities were wide (see Table 1). For a wind speed of 4.6 $m\ s^{-1}$, observed depositions was greater for trees, (with a reduction of 2.5%) than for grass, (with a reduction of 0.8%), over the full scene. The aerodynamic dispersion induced by trees reduces $PM_{2.5}$ concentrations by 11% over the full scene and up to 14% for the suburbs where the tree population is larger. This result is consistent with our previous work, where trees promote air turbulence which has a regional beneficial impact (Jeanjean et al., 2015) with the addition that trees increase the probability of particle deposition significantly more than on shorter vegetation like grass (Chen et al., 2016). For a wind speed of 1 $m\ s^{-1}$, trees were found to increase $PM_{2.5}$ concentrations by 8% over the full scene. In the model settings, a wind speed of 1 $m\ s^{-1}$ is considered laminar and therefore no turbulent dispersion occurs which explains why trees trap traffic emissions at pedestrian height as they decrease the wind flow (Jeanjean et al., 2015). Deposition on trees was negligible (less than 1%) over the full scene whereas deposition on grass was greater (1.7%). The model error was smaller for a wind speed of 1 $m\ s^{-1}$, which suggests that uncertainties in the model increase as the wind speed increases.

3.2. Generalisation between PM reduction and vegetation cover

3.2.1. Tree cover and $PM_{2.5}$ reduction

The data in Fig. 5 suggest that a linear relation can be approximated between the tree ground surface fraction (%) and the $PM_{2.5}$ reduction. By combining linear coefficients, an equation was formed to predict what the reduction in $PM_{2.5}$ would be, depending on the tree ground surface cover, such that

$$\Delta PM_{2.5}(\%) = X\left(K_{t_1} + K_{t_2}\left(V_{d_{trees}}\right)^{\alpha}\right) \qquad (3)$$

where X is the tree ground surface cover (%), K_{t1} is the aerodynamic coefficient, K_{t2} is the deposition coefficient, V_{dtrees} is the tree deposition velocity and α is a power law coefficient. Derived coefficients from this work are summarised in Table 2. It is interesting to note the nature of these linear relationships given the spatial inhomogeneity in any given sub-class of the city. It seems to suggest a level of robustness for predicting city-wide effects from

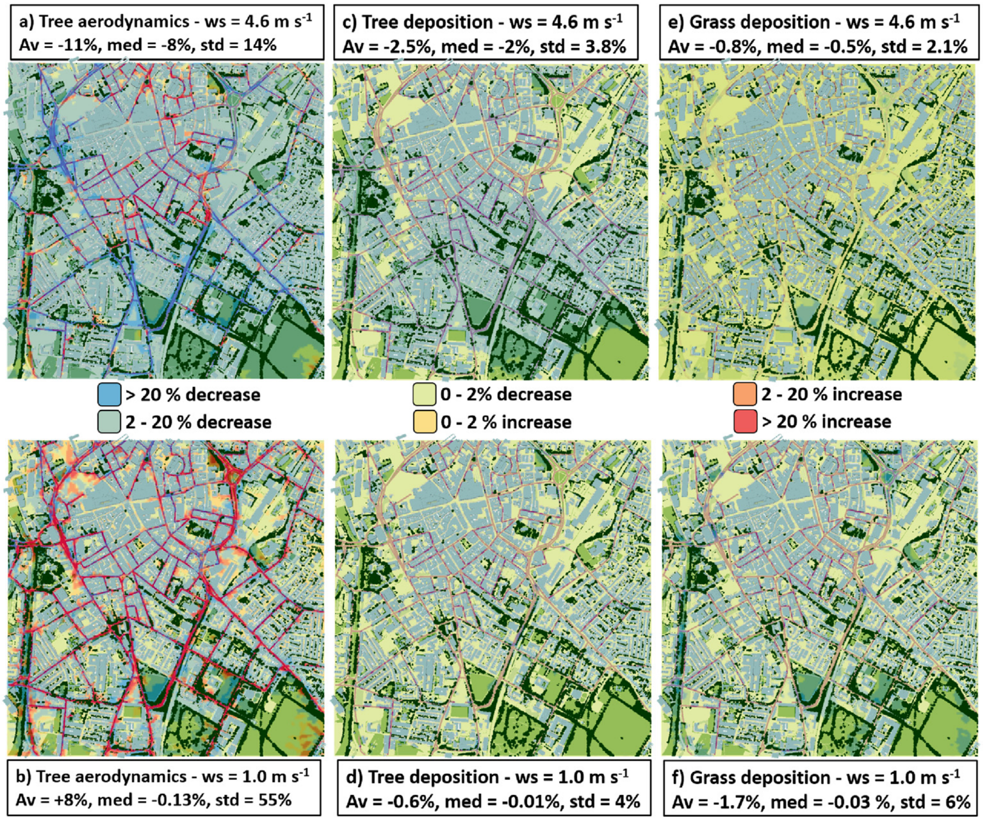


Fig. 3. Modelling output of the spatial change of $PM_{2.5}$ concentrations emitted from road sides at a height of 1.5 m. Tree aerodynamic effects for (a) a wind speed of 4.6 m s^{-1} and (b) 1.0 m s^{-1}. Tree deposition (aerodynamic effects of trees removed) calculated for an average deposition velocity of Vd = 0.64 cm s^{-1} for (c) a wind speed of 4.6 m s^{-1} and (d) 1.0 m s^{-1}. Grass deposition calculated for an average deposition velocity of Vd = 0.64 cm s^{-1} for (e) a wind speed of 4.6 m s^{-1} and (f) 1.0 m s^{-1}.

simple models. Knowing the effect of an urban tree population on $PM_{2.5}$ concentrations could be assessed using this empirical equation supposing the relation translates from Leicester City to another city, which was not investigated here.

3.2.2. Grass cover and $PM_{2.5}$ reduction

As with the case of trees, a similar linear relation can be approximated between the grass ground surface fraction (%) and the $PM_{2.5}$ reduction (see Fig. 6). By combining these linear coefficients, an equation was built to predict what the reduction in $PM_{2.5}$ would be, depending on the grass ground surface cover, such that

$$\Delta PM_{2.5}(\%) = K_g X V_{d_{grass}} \tag{4}$$

where X is the grass ground surface cover (%), K_{t1} is the aerodynamic coefficient, K_{t2} is the deposition coefficient, V_{dgrass} is the grass deposition velocity and α is a power law coefficient. Derived coefficients are summarised in Table 2. In contrast to the relationship for tree (Eq. (3)), the data in Fig. 6 suggest a greater effect of spatial inhomogeneity particular at higher deposition velocities.

3.2.3. Tree and grass relations evaluation against initial model results

Fig. 7 shows a comparison of the predicted reduction of $PM_{2.5}$ by trees or grass using the simple linear representations (Eq. (3) and Eq. (4)) against the full CFD results (see Fig. 4), plotted across a range of ground surface fractions and deposition velocities.

The linear relations (Eq. (3) and Eq. (4)) are based on the following assumptions. Each tree has been modelled as the average of the tree population of Leicester (Jeanjean et al., 2015). The dispersive effects of trees was calculated with a zero background concentration of $PM_{2.5}$ (road emissions were the only source). Eq. (3) and Eq. (4) are dependent on the wind speed and they have been derived at only two wind speeds of 4.6 and 1.0 m s^{-1}. At this stage, applying the relationship to other wind speeds would require re-running a large set of simulations. The coefficients were also calculated for ground surface fraction of vegetation, going up to 25% for grass and 15% for trees. For vegetation cover greater than these, the proposed relation has not been verified but may be a topic for further research. Exploring the applicability of these relations on other cities is also something that could be considered for future work. For example, it was shown that deposition on trees is of minor importance in Northern countries owing to the short time of the leaf season, which would change the relations coefficients (Setälä et al., 2013).

3.3. Wind speed dependence

Owing to the very high computational resources needed to run the CFD model over different wind speeds, only two wind speeds were performed in this study. Although a crude simplification, it was supposed that the change observed with wind speed is linear between the two wind speeds of 1 and 4.6 m s^{-1}. Therefore the initial coefficients from Table 2 were linearly interpolated (see Table 3). Using the previous relations (Eq. (3) and Eq. (4)), the change of wind speed on the tree and grass deposition was investigated. From these estimations, Fig. 8a) shows that tree start to be beneficial for a wind speed of 2.5 m s^{-1} when considering the tree aerodynamics and 2.0 m s^{-1} when considering both the tree

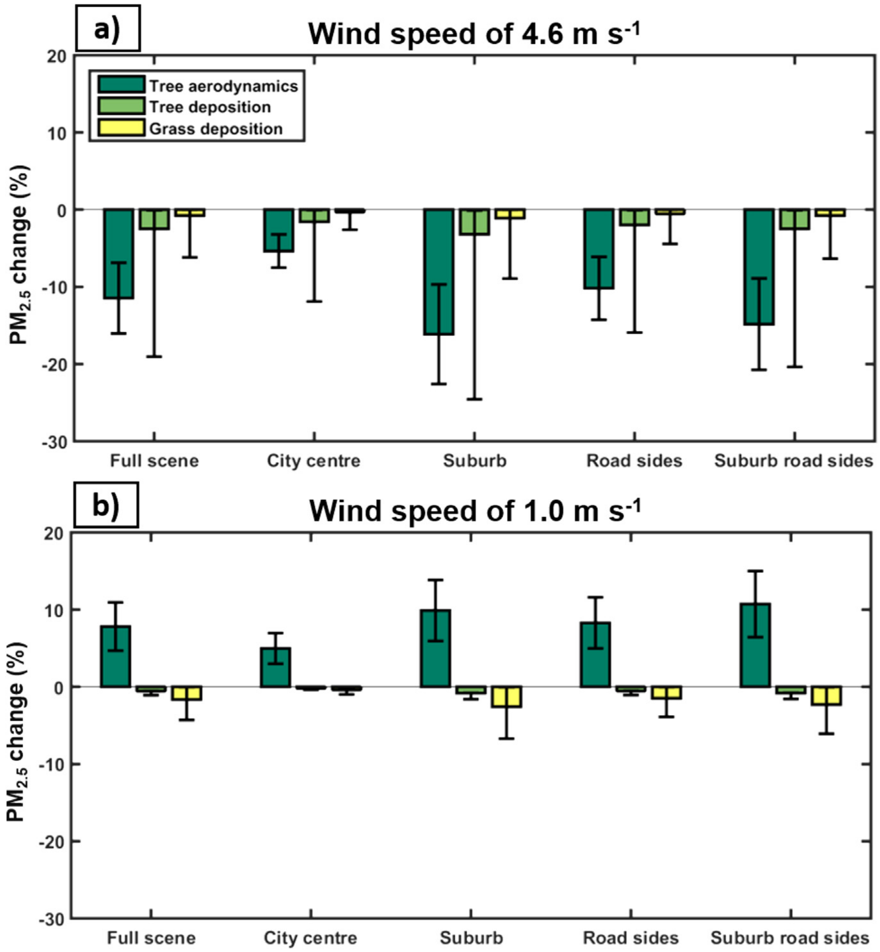


Fig. 4. Modelled change in traffic-emitted $PM_{2.5}$ concentrations induced by the tree aerodynamics, tree deposition and grass deposition for (a) a wind speed of 4.6 m s^{-1} and (b) a wind speed of 1.0 m s^{-1}.

aerodynamics and deposition. As seen previously, the $PM_{2.5}$ deposition on grass has much less impact on $PM_{2.5}$ concentrations than trees. Although it needs more CFD simulations at different wind speeds to be confirmed, it appears in Fig. 8b) that the ideal wind speed for the deposition on tree is 3 m s^{-1}. This could be understood as at low wind speed (1 m s^{-1}), not enough flux of pollutant is brought to the tree and at high wind speed the flux of pollutant passes quickly inside the tree and settle less.

3.4. *Net flux of trees and grass in the summer season over Leicester*

In this study, the seasonality of spring and autumn was not investigated, as it requires further modelling of tree profiles with growing or falling leaves. The tree profile used here corresponds to tree with fully grown leaves mainly present during the summer season in England (21^{st} June to 21^{st} September). The net flux of trees and grass in the summer season over Leicester was estimated using half hourly wind measurement from the East Midlands Airport weather station, located 30 km North of Leicester City. From these measurements, the average wind speed in Leicester City was 4.0 m s^{-1} during the summer 2014. Based on these assumptions, the average net flux of tree and grass on $PM_{2.5}$ concentration emitted by traffic was estimated to be a 9.0% reduction from the tree aerodynamics, 2.8% reduction from the deposition on trees and 0.6% reduction from the deposition on grass.

Local measurements of urban background concentrations of $PM_{2.5}$ in 2014 in Leicester City as part of the Automatic Urban and Rural Network (AURN) monitored by the UK Department for Environment, Food and Rural Affairs (DEFRA, 2014) had an average concentration of 13.3 $\mu g\ m^{-3}$. Using this average leads to an overall deposition of $PM_{2.5}$ of 11.8 t $year^{-1}$ (2.9 t $year^{-1}$ km^{-2}) on trees and 2.5 t $year^{-1}$ (0.6 t $year^{-1}$ km^{-2}) on grass.

A previous study looking at the same scene over Leicester concluding that tree were reducing road traffic emissions by 7% at a wind speed of 4.6 m s^{-1} (Jeanjean et al., 2015). In this present study, a reduction of 11.5% was found (see Fig. 8a). The main difference from the previous study mainly resides in the difference of surface roughness which was altered by the introduction of grass, treated here as a smoother surface than in the previous model set-up. This shows that results from CFD modelling studies shall be treated with caution as they are highly dependent on the boundary conditions and are limited by their modelling accuracy (being here 40%).

4. Discussion and links with previous studies

4.1. *Comparison with the i-Tree model*

To compare the decrease in particulate matter concentrations by deposition on trees, the model results were then compared with values calculated under the UFORE model (i-Tree dry deposition module) (Escobedo and Nowak, 2009). The UFORE model has been used across a wide ranges of studies to characterise the impact of

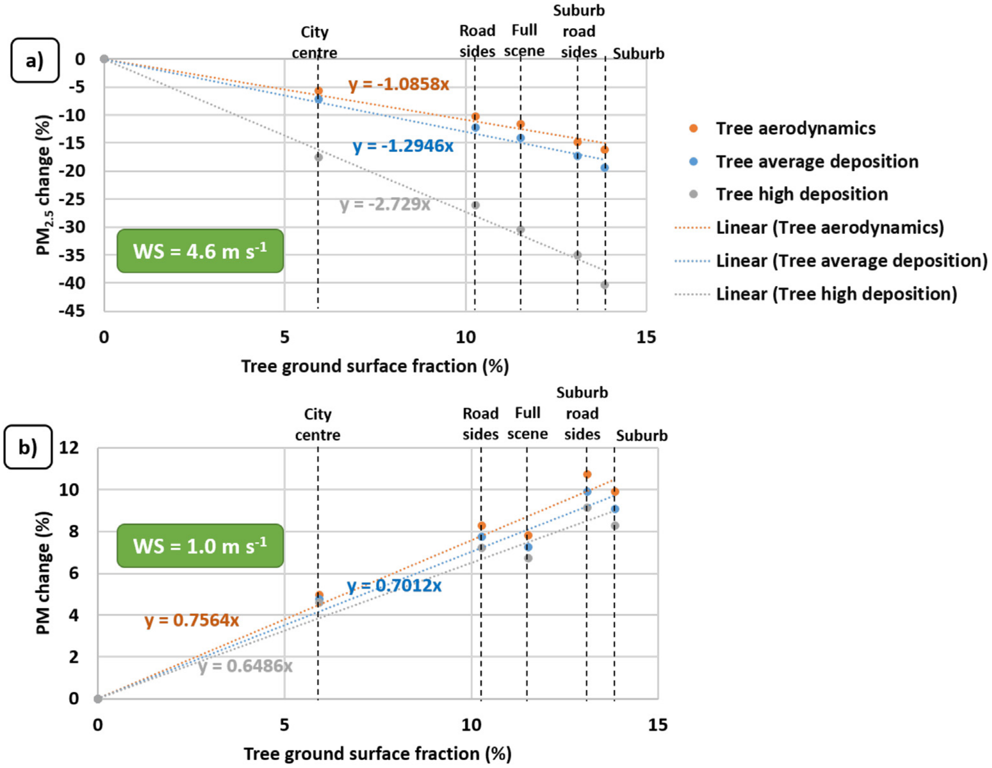


Fig. 5. Relation between the tree ground surface fraction (%) and the $PM_{2.5}$ reduction with the associated first order linear regression coefficients. The tree aerodynamics were included for the tree average (Vd = 0.64 cm s^{-1}) and high deposition (Vd = 30 cm s^{-1}) calculation. The different spatial subsets of Leicester city are detailed in Fig. 2.

Table 2
Coefficients of $PM_{2.5}$ reduction at different wind speeds expressed in Eq. (3) and Eq. (4).

Wind speed (m s^{-1})	K_{t1}	K_{t2}	α	K_g
4.6	−1.09	−3.12	0.53	−4.2
1.0	0.76	−0.13	0.16	−6.3

trees on PM_{10} deposition (e.g (Nowak et al., 2006; Tallis et al., 2011; Baumgardner et al., 2012).) and $PM_{2.5}$ deposition (Nowak et al., 2013). In the UFORE model, the reduction in particulate matter concentrations is calculated over the whole boundary layer. The results in Fig. 9 show the changes in $PM_{2.5}$ provided by the CFD model at a wind speed of 4.6 m s^{-1}. Although the Earth boundary layer height is typically 1–2 km (Seidel et al., 2010; Garratt, 1994), the maximum height of the simulation domain used here was 500 m as the main focus of the study was the dispersion and deposition of road traffic emissions. When averaging across the whole height, a reduction of 0.25% was found for the deposition on trees and 0.03% for grass averaged across the whole domain height. The estimated removal of $PM_{2.5}$ by deposition on trees is in the same order of magnitude than provided by Nowak et al. (2013), where the average improvement of air quality by urban trees was reported to range between 0.05% and 0.24% across 10 major US cities. It is worth noting that comparing to steady state CFD model, the UFORE model integrates more temporal variation such as meteorology (changes in wind speed, direction, boundary layer) and measurements of $PM_{2.5}$ concentrations over time. Nonetheless, CFD simulations have the ability to provide spatial information to see where particulate matter concentrations are decreased. In Fig. 9, it can be seen that tree and grass have the greatest effect close to the ground and that there effects decrease with height. This suggests that trees have a lot more effects locally than on a larger scale. This finding agrees with recent empirical studies that have empirically demonstrated a reduction in PM close to green spaces (e.g. Irga et al., 2015).

4.2. Urban tree management

One of the main question arising from this study is how trees can best be used for air quality improvement, combining both their aerodynamic and deposition effects. A suggestion coming from this modelling work is that in general for cities with average wind speeds greater than 2 m s^{-1}, the more trees the better for both aerodynamic dispersion and deposition of $PM_{2.5}$ in an urban environment. It is important to note that the maximum tree cover used here was 20%, findings could be altered for greater tree cover. As aerodynamic effects are the most important, trees species that are planted in urban environment shall not only be chosen based on their deposition capability. Trees species with high LAD and high deposition are the best to enhance deposition but they shall at the same time favour aerodynamic dispersion. For cities with low average wind speeds (less than 2 m s^{-1}), trees were shown to increase $PM_{2.5}$ concentrations. In this special cases tree species shall be chosen to decrease as little as possible the wind speed to avoid trapping pollution. As shown by Tiwary et al. (2009), it was found that trees have more abilities to decrease particulate matter concentrations than grass.

Another question arising is where shall trees be planted in cities. Although street canyon trees would be the most effective for particulate matter deposition as exposed to greater concentrations (Tallis et al., 2011), previous studies have shown that the aerodynamic effect of trees would end up trapping road emissions in this situation (Gromke et al., 2008; Buccolieri et al., 2011; Wania

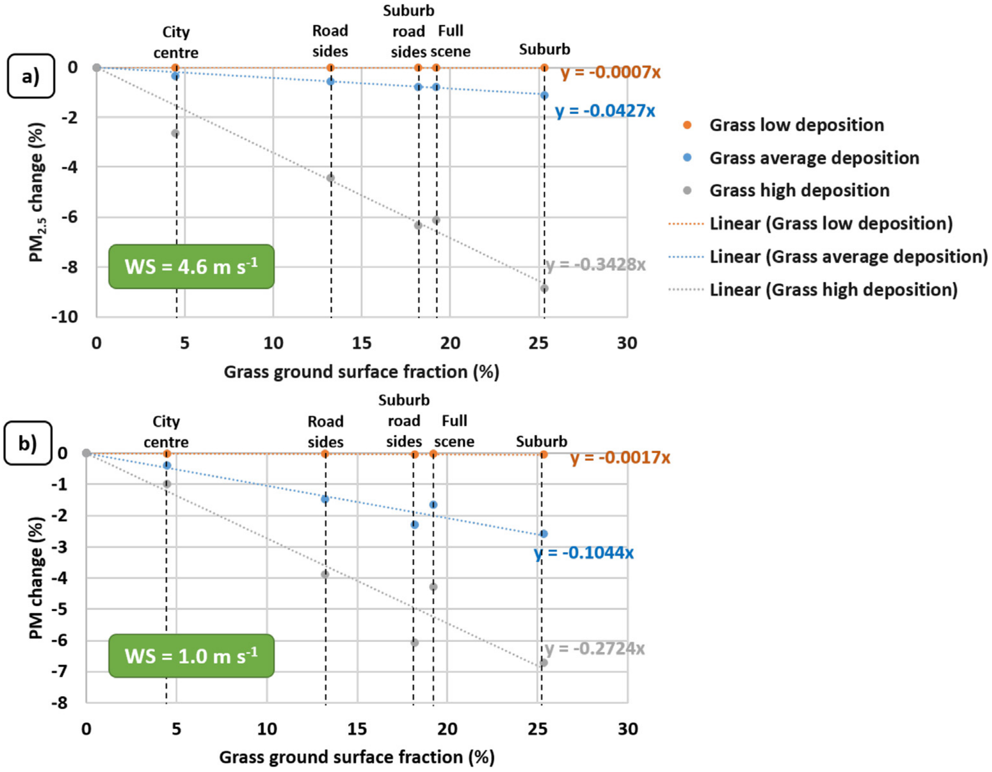


Fig. 6. Relation between the grass ground surface fraction (%) and the $PM_{2.5}$ reduction with the associated first order linear regression coefficients. The $PM_{2.5}$ deposition on grass were calculated for low (0.01 cm s^{-1}), average (0.64 cm s^{-1}) and high (8 cm s^{-1}) deposition velocities. The different spatial subsets of Leicester city are detailed in Fig. 2.

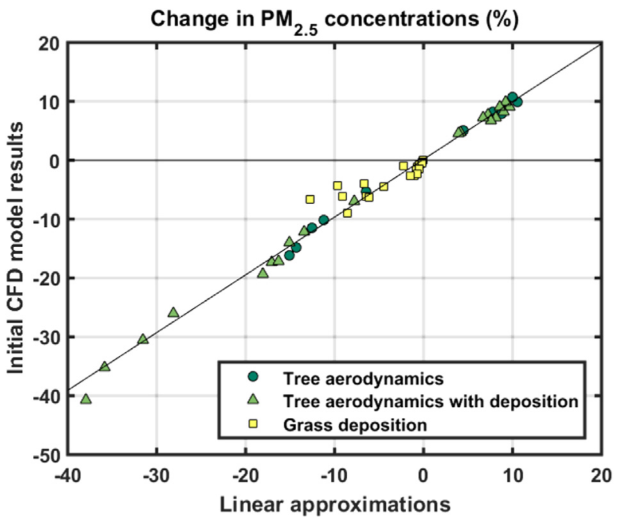


Fig. 7. Comparison of change in $PM_{2.5}$ concentrations (%) between the initial CFD model results (expressed in Fig. 4) at pedestrian height and estimated changes through two linear relations (Eq. (3) and Eq. (4)) based on trees and grass ground surface fraction (%). The aerodynamics of trees were added into the tree deposition calculation.

Table 3
Coefficients of the $PM_{2.5}$ reduction relations linearly interpolated from Table 2 between the wind speeds of 1 and 4.6 m s^{-1}.

Wind speed (m s^{-1})	1	2	3	4	5
K_{t1}	0.76	0.25	−0.27	−0.78	−1.30
K_{t2}	−0.13	−0.96	−1.8	−2.6	−3.5
α	0.16	0.27	0.37	0.47	0.57
K_g	−6.3	−5.7	−5.1	−4.6	−4.0

et al., 2012). However, most of these studies have looked at perpendicular wind directions which exacerbates tree trapping effects in street canyons. Amorim et al. (2013) found that for perpendicular wind directions trees are actually beneficial. The present study results suggest that street canyon trees could actually be beneficial, depending on the prevailing wind direction, wind speed, street canyon and surrounding building geometries.

5. Conclusion

The model results show that there is a direct relationship between changes in $PM_{2.5}$ concentration and the trees and grass ground surface cover. This suggests that there is level of geometry independence combining buildings and trees that is dominated by the aerodynamics of trees. In terms of urban planning, the linear relationship observed in this study (Eq. (3) and Eq. (4)) provides an easy tool to monitor the effectiveness of green infrastructure on the local scale, at pedestrian height. Although only computed for 2 wind speeds, the aerodynamic effects of trees show that dispersion appears to be more important than deposition. Working on removing street pollution via dispersion will prove to be as or even more efficient than deposition technologies.

In Leicester City Centre, the overall global decrease in particle concentrations when considering trees and grass deposition together, is very limited, with 2.8% reduction from the deposition on trees and 0.6% reduction from the deposition on grass. The aerodynamics effect has a much stronger effect, owing a 9.0% reduction during summer time in Leicester City. This study suggests that reducing the road emissions by 10%, equivalent to one vehicle in 10, will have the same effect as all the combined green infrastructure in Leicester City Centre. Regarding the decrease of background particles (non-road emissions, which is not studied here),

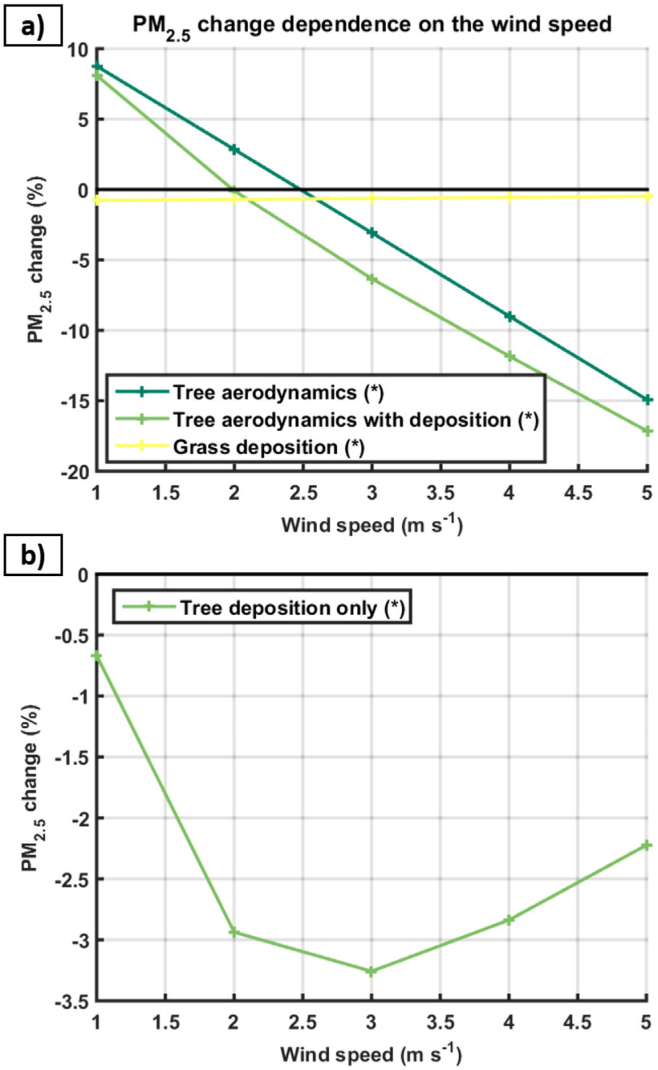


Fig. 8. $PM_{2.5}$ change dependence on the wind speed over Leicester City. (a) Tree aerodynamics, tree aerodynamics with deposition and grass deposition dependence on the wind speed (b) Tree deposition dependence on the wind speed (*) $PM_{2.5}$ estimated from linear relations (Eq. (3) and Eq. (4)) with a linear interpolation between the two reference wind speeds of 1 and 4.6 m s^{-1}.

the literature shows as well that the deposition on vegetation is limited to less than a few percent decrease (Tallis et al., 2011; Beckett et al., 1998; Nowak et al., 2006).

Green infrastructures are beneficial but they do not represent a solution to completely remove air pollution from cities. The tree and grass species of a city could lead to very different reduction in $PM_{2.5}$ from a few percent to almost 20% as suggested by the results. These reductions would only occur during the leaf-period (non-winter period), although for a temperate city like Leicester some trees and hedges are evergreen. It is clear that green infrastructure has a role to play at a city scale but only when co-ordinated with understanding of local implementation and traffic planning.

Because of the steady flow produced by the CFD model, it shall be noted that time dependent effects such as fluctuations in wind speed or direction, solar heating or chemical reactions were not reproduced by the simulations.

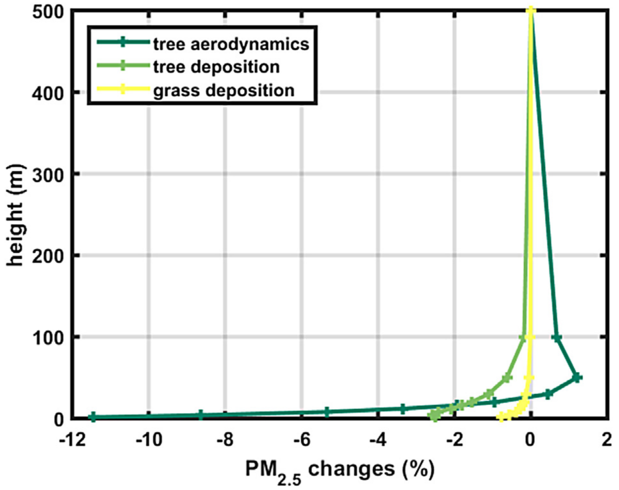


Fig. 9. CFD results of changes in $PM_{2.5}$ across the whole height of the simulation domain for a wind speed of 4.6 m s^{-1}. All relative reductions (%) where calculated using the average ground concentrations of the reference scenario (see Table 1) as denominator.

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References

Abhijith, K., Gokhale, S., 2015. Passive control potentials of trees and on-street parked cars in reduction of air pollution exposure in urban street canyons. Environ. Pollut. 204, 99–108.

Al-Dabbous, A.N., Kumar, P., 2014. The influence of roadside vegetation barriers on airborne nanoparticles and pedestrians exposure under varying wind conditions. Atmos. Environ. 90, 113–124.

Amorim, J., Rodrigues, V., Tavares, R., Valente, J., Borrego, C., 2013. CFD modelling of the aerodynamic effect of trees on urban air pollution dispersion. Sci. Total Environ. 461, 541–551.

Barnes, M., Brade, T.K., Mackenzie, A.R., Whyatt, J., Carruthers, D., Stocker, J., Cai, X., Hewitt, C., 2014. Spatially-varying surface roughness and ground-level air quality in an operational dispersion model. Environ. Pollut. 185, 44–51.

Baumgardner, D., Varela, S., Escobedo, F.J., Chacalo, A., Ochoa, C., 2012. The role of a peri-urban forest on air quality improvement in the Mexico city megalopolis. Environ. Pollut. 163, 174–183.

Beckett, K.P., Freer-Smith, P., Taylor, G., 1998. Urban woodlands: their role in reducing the effects of particulate pollution. Environ. Pollut. 99 (3), 347–360.

Bluesky, 2014. National Tree Map Specification. http://www.emapsite.com/downloads/product_guides/NTM-Specification.pdf.

Brantley, H.L., Hagler, G.S., Deshmukh, P.J., Baldauf, R.W., 2014. Field assessment of the effects of roadside vegetation on near-road black carbon and particulate matter. Sci. Total Environ. 468, 120–129.

Buccolieri, R., Salim, S.M., Leo, L.S., Di Sabatino, S., Chan, A., Ielpo, P., de Gennaro, G., Gromke, C., 2011. Analysis of local scale tree–atmosphere interaction on pollutant concentration in idealized street canyons and application to a real urban junction. Atmos. Environ. 45 (9), 1702–1713.

Chan, C.K., Yao, X., 2008. Air pollution in mega cities in China. Atmos. Environ. 42 (1), 1–42.

Chen, L., Liu, C., Zou, R., Yang, M., Zhang, Z., 2016. Experimental examination of effectiveness of vegetation as bio-filter of particulate matters in the urban environment. Environ. Pollut. 208, 198–208.

DEFRA, 2014. Automatic Urban and Rural Network, Data Files for Leicester University Monitoring Site. https://uk-air.defra.gov.uk/data/flat_files?site_id=LECU.

Di Sabatino, S., Buccolieri, R., Salizzoni, P., 2013. Recent advancements in numerical modelling of flow and dispersion in urban areas: a short review. Int. J. Environ.

Pollut. 7 (52), 172–191, 3-4.

Escobedo, F.J., Nowak, D.J., 2009. Spatial heterogeneity and air pollution removal by an urban forest. Landsc. urban Plan. 90 (3), 102–110.

Franke, J., Hellsten, A., Schlunzen, K.H., Carissimo, B., 2007. Best Practice Guideline for the CFD Simulation of Flows in the Urban Environment. COST Office Brussels.

Freer-Smith, P., El-Khatib, A., Taylor, G., 2004. Capture of particulate pollution by trees: a comparison of species typical of semi-arid areas (ficus nitida and eucalyptus globulus) with european and north american species. Water, Air, Soil Pollut. 155 (1–4), 173–187.

Gallagher, J., Baldauf, R., Fuller, C.H., Kumar, P., Gill, L.W., McNabola, A., 2015. Passive methods for improving air quality in the built environment: a review of porous and solid barriers. Atmos. Environ. 120, 61–70.

Garratt, J.R., 1994. Review: the atmospheric boundary layer. Earth-Science Rev. 37 (1–2), 89–134.

Gousseau, P., Blocken, B., Stathopoulos, T., Van Heijst, G., 2011. CFD simulation of near-field pollutant dispersion on a high-resolution grid: a case study by LES and RANS for a building group in downtown Montreal. Atmos. Environ. 45 (2), 428–438.

Gromke, C., Blocken, B., 2015. Influence of avenue-trees on air quality at the urban neighborhood scale. part ii: traffic pollutant concentrations at pedestrian level. Environ. Pollut. 196, 176–184.

Gromke, C., Buccolieri, R., Di Sabatino, S., Ruck, B., 2008. Dispersion study in a street canyon with tree planting by means of wind tunnel and numerical investigations-evaluation of CFD data with experimental data. Atmos. Environ. 42 (37), 8640–8650.

Guerreiro, C., de Leeuw, F., Foltescu, V., 2013. Air Quality in Europe: 2013 Report. European Environment Agency.

Gugamsetty, B., Wei, H., Liu, C.-N., Awasthi, A., Hsu, S.-C., Tsai, C.-J., Roam, G.-D., Wu, Y.-C., Chen, C.-F., 2012. Source characterization and apportionment of PM10, PM2.5 and PM0.1 by using positive matrix factorization. Aerosol Air Qual. Res. 12, 476–491.

Harrison, R., Brimblecombe, P., Derwent, R., Dollard, G., Eggleston, S., Hamilton, R., Hickman, A., Holman, C., Laxen, D., Moorcroft, S., 1996. Airborne Particulate Matter in the United Kingdom. Third Report of the Quality of Urban Air Review Group.

Horbert, M., Vogt, K., Angeletti, L., 1976. Studies on the Deposition of Aerosols on Vegetation and Other Surfaces. ORNL-tr-4314.

Irga, P., Burchett, M., Torpy, F., 2015. Does urban forestry have a quantitative effect on ambient air quality in an urban environment? Atmos. Environ. 120, 173–181.

Janhäll, S., 2015. Review on urban vegetation and particle air pollution–deposition and dispersion. Atmos. Environ. 105, 130–137.

Jeanjean, A., Hinchliffe, G., McMullan, W., Monks, P., Leigh, R., 2015. A CFD study on the effectiveness of trees to disperse road traffic emissions at a city scale. Atmos. Environ. 120, 1–14.

Kalansuriya, C., Pannila, A., Sonnadara, D., 2009. Effect of roadside vegetation on the reduction of traffic noise levels. In: Proceedings of the Technical Sessions of the Institute of Physics Sri Lanka, vol. 25, pp. 1–6, 2009.

Kim, K.-H., Kabir, E., Kabir, S., 2015. A review on the human health impact of airborne particulate matter. Environ. Int. 74, 136–143.

Lapple, C., 1961. Characteristics of particles and particle dispersoids. Stanf. Res. Inst. J. 5, 94.

Li, X.-B., Lu, Q.-C., Lu, S.-J., He, H.-D., Peng, Z.-R., Gao, Y., Wang, Z.-Y., 2016. The impacts of roadside vegetation barriers on the dispersion of gaseous traffic pollution in urban street canyons. Urban For. Urban Green. 17, 80–91.

Litschke, T., Kuttler, W., 2008. On the reduction of urban particle concentration by vegetation–a review. Meteorol. Z. 17 (3), 229–240.

Morakinyo, T.E., Lam, Y.F., 2015. Simulation study of dispersion and removal of particulate matter from traffic by road-side vegetation barrier. Environ. Sci. Pollut. Res. 1–14.

Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban For. urban Green. 4 (3), 115–123.

Nowak, D.J., Hirabayashi, S., Bodine, A., Hoehn, R., 2013. Modeled pm 2.5 removal by trees in ten us cities and associated health effects. Environ. Pollut. 178, 395–402.

Peters, K., Eiden, R., 1992. Modelling the dry deposition velocity of aerosol particles to a spruce forest. Atmos. Environ. 26 (14), 2555–2564.

Petroff, A., Zhang, L., 2010. Development and validation of a size-resolved particle dry deposition scheme for application in aerosol transport models. Geosci. Model Dev. 3 (2), 753–769.

Pugh, T.A., MacKenzie, A.R., Whyatt, J.D., Hewitt, C.N., 2012. Effectiveness of green infrastructure for improvement of air quality in urban street canyons. Environ. Sci. Technol. 46 (14), 7692–7699.

Roupsard, P., Amielh, M., Maro, D., Coppalle, A., Branger, H., Connan, O., Laguionie, P., Hébert, D., Talbaut, M., 2013. Measurement in a wind tunnel of dry deposition velocities of submicron aerosol with associated turbulence onto rough and smooth urban surfaces. J. Aerosol Sci. 55, 12–24.

Salmond, J., Williams, D., Laing, G., Kingham, S., Dirks, K., Longley, I., Henshaw, G., 2013. The influence of vegetation on the horizontal and vertical distribution of pollutants in a street canyon. Sci. Total Environ. 443, 287–298.

Salmond, J.A., Tadaki, M., Vardoulakis, S., Arbuthnott, K., Coutts, A., Demuzere, M., Dirks, K.N., Heaviside, C., Lim, S., Macintyre, H., et al., 2016. Health and climate related ecosystem services provided by street trees in the urban environment. Environ. Health 15 (1), 95.

Seidel, D.J., Ao, C.O., Li, K., 2010. Estimating climatological planetary boundary layer heights from radiosonde observations: comparison of methods and uncertainty analysis. J. Geophys. Res. Atmos. 115 (D16).

Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L., Nowak, D., 2016. Air pollution removal by trees in public green spaces in Strasbourg city, France. Urban For. Urban Green. 17, 192–201.

Setälä, H., Viippola, V., Rantalainen, A.-L., Pennanen, A., Yli-Pelkonen, V., 2013. Does urban vegetation mitigate air pollution in northern conditions? Environ. Pollut. 183, 104–112.

Steffens, J.T., Wang, Y.J., Zhang, K.M., 2012. Exploration of effects of a vegetation barrier on particle size distributions in a near-road environment. Atmos. Environ. 50, 120–128.

Sundvor, I., Balaguer, N., Mar Viana, X., Reche, C., Amato, F., Mellios, G., 2012. Road traffics contribution to air quality in european cities. ETC/ACM Tech. Pap. 14 (2012), 1–74.

Tallis, M., Taylor, G., Sinnett, D., Freer-Smith, P., 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of london, under current and future environments. Landsc. Urban Plan. 103 (2), 129–138.

Tiwary, A., Sinnett, D., Peachey, C., Chalabi, Z., Vardoulakis, S., Fletcher, T., Leonardi, G., Grundy, C., Azapagic, A., Hutchings, T.R., 2009. An integrated tool to assess the role of new planting in pm 10 capture and the human health benefits: a case study in london. Environ. Pollut. 157 (10), 2645–2653.

Tong, Z., Baldauf, R.W., Isakov, V., Deshmukh, P., Zhang, K.M., 2016. Roadside vegetation barrier designs to mitigate near-road air pollution impacts. Sci. Total Environ. 541, 920–927.

Vos, P.E., Maiheu, B., Vankerkom, J., Janssen, S., 2013. Improving local air quality in cities: to tree or not to tree? Environ. Pollut. 183, 113–122.

Vranckx, S., Vos, P., 2013. OpenFOAM CFD Simulation of Pollutant Dispersion in Street Canyons: Validation and Annual Impact of Trees. http://atmosys.eu/atmosys/faces/doc/ATMOSYS_Deliverable_9.2b_windtunnel%20validation.pdf.

Vranckx, S., Vos, P., Maiheu, B., Janssen, S., 2015. Impact of trees on pollutant dispersion in street canyons: a numerical study of the annual average effects in Antwerp, Belgium. Sci. Total Environ. 532, 474–483.

Wania, A., Bruse, M., Blond, N., Weber, C., 2012. Analysing the influence of different street vegetation on traffic-induced particle dispersion using microscale simulations. J. Environ. Manag. 94 (1), 91–101.

White, E., Turner, F., 1970. A method of estimating income of nutrients in a catch of airborne particles by a woodland canopy. J. Appl. Ecol. 441–461.

White, M.P., Alcock, I., Wheeler, B.W., Depledge, M.H., 2013. Would you be happier living in a greener urban area? A fixed-effects analysis of panel data. Psychol. Sci. 24, 920–928.

WHO, 2006. WHO Air Quality Guidelines for Particulate Matter, Ozone, Nitrogen Dioxide and Sulfur Dioxide: Global Update 2005: Summary of Risk Assessment. World Health Organization, Geneva.

WMO, 2008. Guide to Meteorological Instruments and Methods of Observation. Report. World Meteorological Organization.