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An integrated tool to assess the role of new planting in PM_{10} capture and the human health benefits: A case study in London

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A combination of models can be used to estimate particulate matter concentrations before and after greenspace establishment and the resulting benefits to human health.

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ABSTRACT

The role of vegetation in mitigating the effects of PM_{10} pollution has been highlighted as one potential benefit of urban greenspace. An integrated modelling approach is presented which utilises air dispersion (ADMS-Urban) and particulate interception (UFORE) to predict the PM_{10} concentrations both before and after greenspace establishment, using a 10×10 km area of East London Green Grid (ELGG) as a case study. The corresponding health benefits, in terms of premature mortality and respiratory hospital admissions, as a result of the reduced exposure of the local population are also modelled. PM_{10} capture from the scenario comprising 75% grassland, 20% sycamore maple (*Acer pseudoplatanus* L.) and 5% Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) was estimated to be 90.41 t yr^{-1}, equating to 0.009 t ha^{-1} yr^{-1} over the whole study area. The human health modelling estimated that 2 deaths and 2 hospital admissions would be averted per year.

1. Introduction

Sources of PM_{10} (particles with a diameter of less than 10×10^{-6} m) within urban areas of the UK include road traffic, industry and power production (Dore et al., 2005). Results from numerous longitudinal investigations of human respiratory and other diseases show consistent statistical associations between human exposure to outdoor levels of PM_{10} and adverse health impacts. Health effects range from alveolar inflammation and respiratory-tract infection (specifically pneumonia) (Pope et al., 1995; Holgate, 1996; QUARG, 1996; Defra, 2007a) to acute cardiovascular disorders (Pope et al., 1995; Klemm et al., 2000; USEPA, 2004). These often lead to substantially increased morbidity and mortality, in particular among elderly individuals (Zelikoff et al., 2003). The adverse health effects of high ambient PM_{10} concentrations have resulted in the introduction of air quality standards which are designed to be protective of human health. When considered in an economic context, the health costs incurred by PM_{10} pollution in the UK have been estimated to range between £9.1 and 21.4 billion per annum (Defra, 2007a).

Although PM_{10} emissions in the UK have reduced in the last 30 years (Dore et al., 2005; Defra, 2007a), this trend is flattening or reversing in some major urban areas and along roads (Defra, 2007a). A range of measures have been introduced in an attempt to further reduce PM_{10} emissions, for example tightening of vehicle emissions standards and road pricing initiatives. However, a range of cost-effective abatement measures must be initiated if improvements in air quality are to be any more substantial (Defra, 2007a). Tree establishment in urban areas has been proposed as one measure to reduce ambient PM_{10} concentrations (Bealey et al., 2007; Nowak et al., 2006; McDonald et al., 2007). PM_{10} deposition to vegetation has been the subject of a number of recent investigations (Beckett et al., 1998; Gupta et al., 2004; Dammgen et al., 2005; Tiwary et al., 2006). However, the complexities involved in understanding the removal mechanisms for PM_{10} on different vegetation types, species, planting design and age class has resulted in a large degree of uncertainty regarding the level of reduction that could practically be achieved and how this would relate to human health. This uncertainty is exacerbated by the inherent assumptions and uncertainties in deposition models, where the

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interception mechanism is influenced by particle size, foliage density, terrain, and meteorological conditions (Ruijgrok et al., 1995; He et al., 2002).

Trees can serve as effective sinks for particulates at the canopy level, both via dry, wet and occult deposition mechanisms. For example, work on forest canopies (Peters and Eiden, 1992; Erisman et al., 1997; Freer-Smith et al., 1997; Decker et al., 2000; Urbat et al., 2004) found them to have high capturing efficiencies for airborne particles. The structure of trees and the rough surfaces that they provide increase the incidence of particle impaction and interception by disrupting the flow of air (Beckett et al., 1998), mainly at canopy height (Erisman et al., 1997). It has been suggested that the layered canopy structure of large trees provides a surface area for particulate deposition of between 2 and 12 times that of the area of land they cover (Broadmeadow and Freer-Smith, 1996). Fowler et al. (2004) found that woodlands in the West Midlands, England, collected three times more PM_{10} than grassland. The differences between tree species play an important role in estimating PM_{10} capture; leaves with complex shapes, large circumference-to-area ratios, waxy cuticles or fine hairs on their surfaces collect particles more efficiently. Conifers, which are also in leaf all year round, may be more effective than deciduous species (Freer-Smith et al., 2005).

Deposition models such as Urban Forest Effects model (UFORE) (Nowak, 1994) and FRAMES (Bealey et al., 2007) are available to assess the potential for particulate matter interception by trees. In UFORE, generic deposition values are assigned to trees due to a lack of empirical deposition data for specific species and for different wind speeds. However, recent reports have shown that tree species and wind speed account for large variations in deposition velocity (Beckett et al., 2000a,b; Freer-Smith et al., 2005). These variations suggest that the use of generic deposition velocities may produce imprecise estimates of PM_{10} flux if they are used to predict deposition where the species composition is different from that for which they have been derived. Recently published deposition velocities measured for specific tree species and wind speeds (Freer-Smith et al., 2004, 2005) allow more accurate PM_{10} flux estimations to be produced.

The potential use of trees to improve local air quality has been recognised by the UK Government (e.g. Scottish Executive, 2006; Defra, 2007b; Royal Commission on Environmental Pollution, 2007). There is, however, a need for greenspace to be planned and implemented strategically at the landscape level in order to fulfil the potential benefits it can bring to urban environments. These benefits include those to air quality, climate amelioration, sustainable urban drainage, health and well-being. This study aims to estimate the potential for a greenspace initiative to reduce PM_{10} levels in an area of East London and the corresponding human health impacts on mortality and morbidity. It also aims to demonstrate how this type of integrated modelling approach, comprising environmental and health models, can be used as a tool by practitioners wishing to target greenspace development to areas where air quality is of concern.

2. Materials and methods

2.1. Study area

The East London Green Grid (ELGG; Fig. 1) (Greater London Authority, 2008) is the delivery mechanism of the 'Greening the Gateway' initiative in London. It is a proposed 'network of interlinked, multi-purpose and high quality open spaces that connect areas where people live and work with town centres, public transport, the countryside in the urban fringe and the River Thames' that will be created from both new and existing greenspace (Greater London Authority, 2006). The drivers behind the ELGG development are multi-faceted and, whilst PM_{10} reduction is not a primary driver the improvement of local air quality is seen as a potential benefit of the scheme (Greater London Authority, 2006).

This study used a 10 × 10 km region (Fig. 2; National Grid Reference TQ 401801, 51°30′N, 0°01′E) of the ELGG covering the London Boroughs of Newham and Greenwich. The ELGG within this area occupies 547 ha (5.5% of the total study area).

2.2. Overview of the integrated approach

The study area falls within a heavily urbanised region of the ELGG, characterised by heavy traffic, industrial activity and London City airport. Sources of PM_{10} from the whole of Greater London were modelled using ADMS-Urban (version 2.2, Cambridge Environmental Research Consultants, UK; CERC, 2006) to calculate hourly PM_{10} concentrations at 1.5 m height (human receptor level) for the 10,000 ha study area; a map of average PM_{10} concentrations was then produced. This process used emissions data from the London Atmospheric Emissions Inventory (GLA, 2006) and meteorological data for 2004 from Heathrow Airport, UK (Meteorological Office, 2006). ADMS-Urban allows a maximum of 10,000 output points in the calculation of spatial concentrations; these can be specified using a mix of regular output grid points and additional receptor points. For the presented study the grid resolution for the 10 km × 10 km study area was chosen as a mix of points on a 40 × 40 grid (0.25 × 0.25 m) and 18 specified receptor locations. The latter were used to sample the input upstream concentrations in order to calculate the potential flux from vegetation intervention.

A canopy PM_{10}-uptake model based on UFORE (Nowak, 1994) was then used to estimate the PM_{10} interception by the proposed ELGG within the study area. However, the ELGG does not yet have specific information available on the composition of the greenspaces, e.g. species choice, percentage tree cover or planting design. Therefore, a range of possible planting options for these greenspaces was modelled. The 'most realistic scenario' of PM_{10} interception by the ELGG within the study area was then used to reproduce the PM_{10} concentration map for the area for use in the human health modelling. This scenario was thought to be most realistic based on social considerations for urban greenspace design, where broadleaves, a range of habitats and areas of open space tend to be preferred by local communities (Lee, 2001). The interception of PM_{10} by the other scenarios is presented in order to demonstrate the importance of species selection if air quality improvement is an objective of greenspace design and the beneficial role of tree cover versus grassland.

The PM_{10} concentrations post-implementation of the ELGG were estimated in ADMS-Urban using two modifications to the pre-implementation scenario. Firstly, the source strength of each grid cell was adjusted by accounting for the modelled flux to vegetation using the GIS information on the presence of the corresponding vegetation in each grid cell. Secondly, the surface roughness (in metres) was altered to take account of changes to this parameter following greenspace establishment. ADMS-Urban parameterises the boundary layer structure based on the Monin-Obukhov length and the boundary layer height. Using the hourly sequential meteorological data a pre-processor code makes accurate estimation of the boundary layer height for each hour, based on the previous history.

The impact of the ELGG on human health from PM_{10} exposure was compared with a situation of no greenspace establishment. Two models were used to estimate the premature mortality and respiratory hospital admissions, as a result of PM_{10} exposure, of the populations within the London Boroughs of Newham and Greenwich.

2.3. The potential impact of the ELGG on PM_{10} concentration

Five scenarios were used to estimate the potential for PM_{10} interception by the ELGG. These were based on the premise that trees have a greater capacity for PM_{10} reduction than grassland and that conifers have a greater capacity than broadleaves. Data for sycamore maple (*Acer pseudoplatanus* L.) and Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) were selected to provide the 'best' and 'worst' case scenarios for PM_{10} interception by tree cover. *A. pseudoplatanus* produces very low deposition velocities due to its low particle capture efficiency and *P. menziesii* exhibits very high deposition velocities (Freer-Smith et al., 2004, 2005). The five scenarios used, based on a total land area of 547 ha, in the study were:

1. 100% grassland;
2. 50% grassland, 50% *A. pseudoplatanus*;
3. 100% *A. pseudoplatanus*;
4. 75% grassland, 20% *A. pseudoplatanus*, 5% *P. menziesii*;
5. 100% *P. menziesii*.

The PM_{10} flux (F; in g m $^{-2}$ s^{-1}) to each greenspace scenario is calculated as the product of the deposition velocity (V_d; in m s^{-1}) and the pollutant concentration (C; in g m^{-3}) according to the methodology outlined in Nowak (1994):

$$F = V_d C \tag{1}$$

Deposition velocity is calculated as the inverse of the sum of the aerodynamic (R_a), quasi-laminar boundary layer (R_b) and canopy (R_c) resistances (Baldocchi et al., 1987):

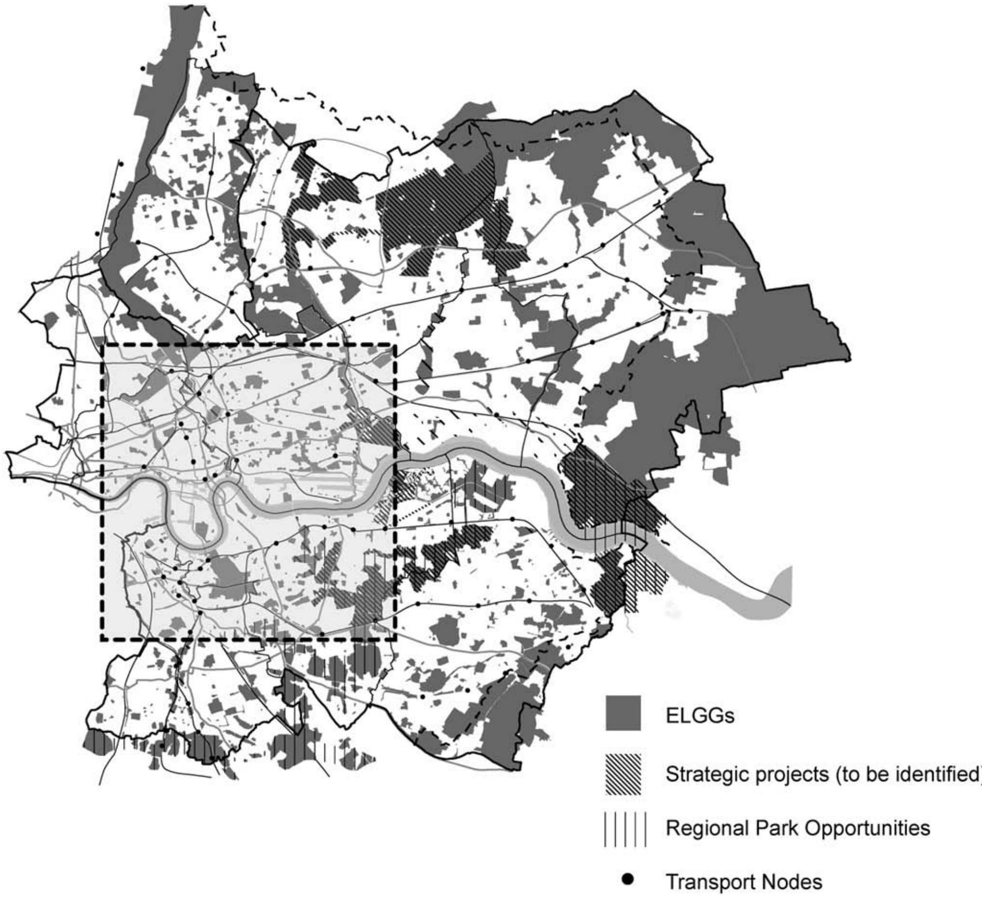

Fig. 1. Location of the study area within the wider East London Green Grid (the study area is contained within the dotted square) (Greater London Authority, 2006).

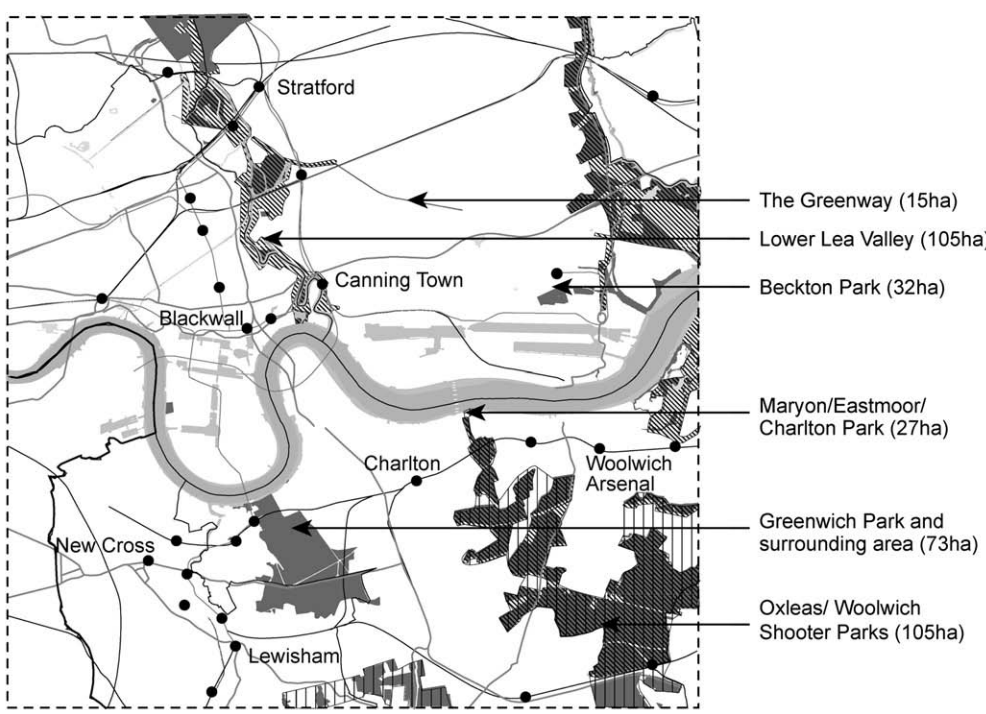

Fig. 2. Locations of greenspace within the East London Green Grid included in the assessment of vegetation intervention (Greater London Authority, 2006).

$$V_d = (R_a + R_b + R_c)^{-1} \quad (2)$$

Hourly meteorological data were used to estimate R_a and R_b. The aerodynamic resistance is calculated as (Killus et al., 1984):

$$R_a = u(z)u_*^{-2} \quad (3)$$

where $u(z)$ is the mean wind speed at height z (m s^{-1}) and u_* is the friction velocity (m s^{-1}).

$$u_* = (ku(z-d))\left[\ln\left((z-d)z_o^{-1}\right) - \psi_m\left((z-d)L^{-1}\right) + \psi_m\left(z_o L^{-1}\right)\right]^{-1} \quad (4)$$

where k = von Karman constant, d = displacement height (m), z_o = roughness length (m), ψ_M = stability function for momentum, and L = Monin-Obukhov stability length. L was estimated by classifying hourly local meteorological data into stability classes using Pasquill's (1961) stability classification scheme and then estimating $1/L$ as a function of stability class and z_o (Golder, 1972). When $L < 0$ (unstable) (Van Ulden and Holtslag, 1985):

$$\psi_m = 2\ln[(1+X)/2] + \ln\left[\left(1+X^2\right)/2\right] - 2\tan^{-1}(X) + \pi/2 \quad (5)$$

where $X = (1 - 28\,z\,L^{-1})^{0.25}$ (Dyer and Bradley, 1982). When $L > 0$ (stable conditions) (Van Ulden and Holtslag, 1985):

$$\psi_m = -17(1 - \exp(-0.29(z-d)/L)) \quad (6)$$

The quasi-laminar boundary-layer resistance was estimated as:

$$R_b = B^{-1}u_*^{-1} \quad (7)$$

where $B^{-1} = 2(2u_*)^{-1/3}$ (Killus et al., 1984).

Hourly canopy resistance (R_c) values were derived from yearly-averaged R_a and R_b and deposition velocity values for each tree species ($V_{g(s)}$; in m s^{-1}):

$$R_c = \frac{1}{V_{g(s)}} - (\overline{R}_a + \overline{R}_b) \quad (8)$$

$V_{g(s)}$ (m s^{-1}) values were species-specific (either $V_{g(A.\ pseudoplatanus)}$ or $V_{g(P.\ menziesii)}$) and calculated using known relationships between wind speed and $V_{g(s)}$ (using the data from Freer-Smith et al., 2004):

$$V_{g(A.pseudoplatanus)} = 0.00119(1.164^u) \quad \text{(number of observations} = 9; p < 0.001; R^2 = 0.68) \quad (9)$$

$$V_{g(P.menziesii)} = 0.00297(1.404^u) \quad \text{(number of observations} = 9; p = 0.007; R^2 = 0.45) \quad (10)$$

where $V_{g(A.\ pseudoplatanus)}$ is the deposition velocity for *A. pseudoplatanus*, $V_{g(P.\ menziesii)}$ is the deposition velocity for *P. menziesii* and u is wind speed.

Deposition velocities for grassland were calculated using measured relationships between deposition velocities for long grass and Pasquill's atmospheric stability classes (Vong et al., 2004).

The in leaf period for *A. pseudoplatanus* was assumed to be 15th May to 1st November. The canopy height used for both species was 10 m and the leaf area indices (LAI) were assumed to be constant throughout the in leaf period at 9.0 m^2 m^{-2} for *P. menziesii* and 7.0 m^2 m^{-2} for *A. pseudoplatanus*. The PM_{10} flux (in g m^{-2} s^{-1}) from Equation (1) was used together with the area of each greenspace (in ha) and the LAI of the grass, *A. pseudoplatanus* and *P. menziesii* (multiplied by 2 to account for deposition to both sides of the leaf) to calculate the total annual PM_{10} flux to greenspace (in t ha^{-1} yr^{-1}). Flux data were then used to modify the ADMS-Urban outputs, taking into account the orientation of the greenspace relative to wind direction, in order to estimate the PM_{10} concentrations across the study area following implementation of the ELGG.

2.4. Assessment of health benefits to the local population

The estimates of the health benefits of the reduction in PM_{10} concentration are based on exposure–response relationships obtained from time-series epidemiological analyses of daily mortality and respiratory hospital admissions with daily mean PM_{10} concentration. The exposure–response relationships quantify the short-term (i.e. acute) health effects of exposure due to changes in PM_{10} concentrations. Epidemiological studies have shown that premature mortality and respiratory hospital admissions risks are positively and linearly associated with exposure to PM_{10} (COMEAP, 1998; Atkinson et al., 1999; Brunekreef and Holgate, 2002; Medina et al., 2004). The linear coefficients of the inferred regression lines are used to quantify the changes in the risks of health events associated with changes in the pollutant concentration.

The two equations below were used to quantify the annual reduction in mortality (ΔM) and hospital admissions (ΔH) due to reduction in PM_{10} concentration in each of the affected wards in East London:

$$\Delta M_i = \alpha \times \Delta C_i \times P_i \times M_0 \quad (11)$$

$$\Delta H_i = \beta \times \Delta C_i \times P_i \times H_0 \quad (12)$$

where α and β are respectively the regression coefficients of the PM_{10} exposure-mortality and PM_{10} exposure-hospital admission relationships ($\alpha = 0.00075$ and $\beta = 0.00080$), ΔC_i is the modelled reduction in PM_{10} concentration in ward i (μg m^{-3}),P_i is the size of the population of ward i, M_0 and H_0 are respectively the annual baseline mortality (720 per 100,000 of the population) and hospital admission rates (651 per 100,000 of the population) for the overall affected area. The total health benefits are obtained by summing the health benefits over all the wards

$$\Delta\widehat{M} = \sum_{i=1}^{n} \Delta M_i \quad (13)$$

$$\Delta\widehat{H} = \sum_{i=1}^{n} \Delta H_i \quad (14)$$

where n is the total number of affected wards, $\Delta\widehat{M}$and $\Delta\widehat{H}$ are respectively the total number of deaths and hospital admissions averted per year across all wards as a result of the intervention.

The PM_{10}-mortality coefficient was taken from COMEAP (1998) and the PM_{10}-hospital admission coefficient was taken from Atkinson et al. (1999). Population data were obtained from the 2001 Census, annual baseline mortality data from the UK Office of National Statistics and annual baseline respiratory hospital admissions from the UK Hospital Episodes Statistics.

3. Results

3.1. Potential impact of the ELGG on PM_{10} concentration

The results of the PM_{10} interception modelling are shown in Fig. 3. *P. menziesii*, due to its greater V_g and LAI values, has a significantly greater capacity to intercept particulates from the atmosphere than *A. pseudoplatanus*; *A. pseudoplatanus* appears only slightly more effective than grass (12.45 t yr^{-1} removed compared to 3.75 t yr^{-1} at the Lower Lea Valley as opposed to 258.75 t yr^{-1} using *P. menziesii*). This represents a four-fold increase when trees are included in urban greenspace design; equating to a removal rate of 0.12 t ha^{-1} yr^{-1}. The amount of PM_{10} interception is directly proportional to the area of the greenspace; therefore the differences in reductions between greenspaces shown in Fig. 3 are a factor of the areas of the greenspaces. The PM_{10} reductions for the whole ELGG within the study area are 17.99 t yr^{-1} (0.03 t ha^{-1} yr^{-1}) under 100% grassland, 60.49 t yr^{-1} (0.11 t ha^{-1} yr^{-1}) under 100% *A. pseudoplatanus*, 1277.13 t yr^{-1} (2.33 t ha^{-1} yr^{-1}) under 100% *P. menziesii*.

When the more realistic planting scenario of 75% grassland, 20% *A. pseudoplatanus* and 5% *P. menziesii* (scenario 4) is used the PM_{10} removal is 90.41 t yr^{-1} (0.17 t ha yr^{-1}; Fig. 4). Fig. 5 shows the spatial distribution of PM_{10} concentrations both before and after implementation of this scenario within the ELGG study area.

3.2. Results from health modelling

The health modelling estimated that 2 premature deaths and 2 respiratory hospital admissions are averted per year due to the implementation of scenario 4 within the ELGG study area.

4. Discussion

4.1. Potential impact of the ELGG on PM_{10} concentration

This study suggests that the contribution greenspace makes to improving local air quality is dependant on the percentage cover of trees and their species. The rates of PM_{10} removal for scenario 4 are greater than those found by Nowak (1994) for Chicago who calculated that trees within the city could reduce PM_{10} concentrations by 0.004 t ha^{-1} yr^{-1}. The Nowak (1994) study had a lower

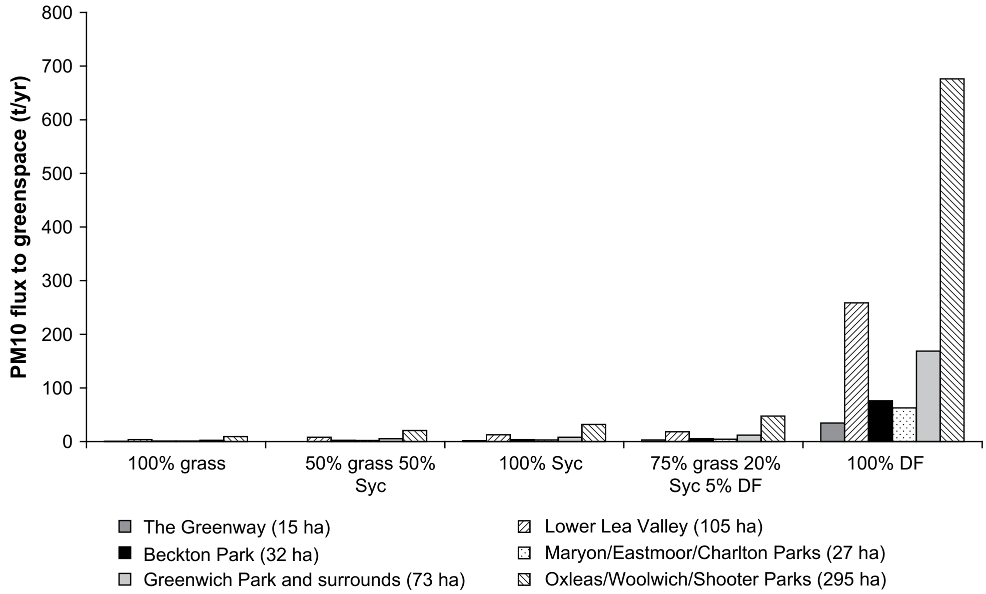

Fig. 3. Potential PM_{10} flux to different scenarios of planting composition within the study area of the East London Green Grid (Syc = *A. pseudoplatanus*; DF = *P. menziesii*).

tree cover at 11% which would, in part, explain the lower rate of PM_{10} removal. The current study also included the deposition to grassland, which was not taken into consideration in the Chicago study. In addition, the deposition velocities used by Nowak (1994) were smaller than those used for the ELGG as a result of the different species composition; 90% broadleaves and only 10% conifers, and because the re-suspension of particles from trees was assumed to be 50%. Particles deposited onto tree surfaces may remain there, be re-suspended to the atmosphere by wind or be washed off during precipitation events (Nowak, 1994). However, re-suspension of silica particles from an experimental spruce canopy, at a comparable wind speed to that in London, was found to be extremely small at approximately 1% per day of the deposited material (Ould-Dada and Baghini, 2001), for this reason re-suspension was not considered in the current study. Yang et al. (2005), using the UFORE model, found that trees in Beijing removed approximately 0.025 t ha^{-1} yr^{-1}; this greater value can probably be explained by the differences in climate, tree cover (16.4%) and initial PM_{10} concentrations in Beijing compared to Chicago.

It is unsurprising that such studies have predicted relatively similar rates of PM_{10} removal since they all use similar methodologies, models and literature data so the inherent assumptions and therefore uncertainties will be broadly equivalent. However, Broadmeadow et al. (1998) calculated the annual deposition to a *Quercus* spp. woodland located alongside the M6 (West Midlands, England) from leaf collections and measurements of LAI and found that at the end of the growing season the PM_{10} deposition rate was 0.009 t ha^{-1} yr^{-1}, suggesting that the models used in these studies are reasonably reliable.

This study assumes a canopy height of 10 m; this is likely to represent *A. pseudoplatanus* and *P. menziesii* trees of 13–15 and 14–16 years respectively (Edwards and Christie, 1981). The ability of trees to intercept PM_{10} will vary during the lifetime of the tree; larger trees are capable of removing more PM_{10} (Nowak, 1994), although younger, smaller trees are surprisingly effective due to their greater foliage densities (Beckett et al., 2000c). Street trees and the edge effect of woodland blocks were also not considered in the study, both of which have been reported to play a significant role in the removal of particles from the air (Hasselrot and Grennfelt, 1987; Neal et al., 1994; Nowak, 1994).

The UFORE model has a number of assumptions, some of which could be addressed by the use of a more complex deposition model. This would, however, probably result in a model which was more dependant on site- or regional-specific data which would be more complicated to run and interpret. The assumptions can be summarised in the following paragraphs.

The UFORE model only considers dry deposition to greenspace; it does not take into account occult or wet deposition and is therefore likely to underestimate the total deposition (Graustein and Turekian, 1989). However, in this study it is unlikely to make a significant contribution, as the site is not frequently immersed in cloud (Graustein and Turekian, 1989; Broadmeadow and Freer-Smith, 1996) and wet deposition is not affected by the surface roughness of the system so is identical between vegetation types (Fowler et al., 2004).

UFORE uses canopy resistance values (R_c) calculated from the average deposition values measured in a field in Chicago minus the R_a and R_b values (Nowak, 1994). These were unlikely to represent the situation in London, therefore the present used species- and wind speed-specific V_g values calculated from relationships developed from the raw data from Freer-Smith et al. (2004) for *A. pseudoplatanus* and *P. menziesii*. This study measured deposition velocity in wind tunnel experiments and so V_g values may not be representative of field conditions, although a range of wind speeds were used.

Deposition to *A. pseudoplatanus* was assumed to only occur during the in leaf period, but there will be some uptake onto woody surfaces during the winter. Nowak (1994) calculated that removal ranged from 0.007 to 0.06 t ha^{-1} yr^{-1} between out of leaf and in leaf seasons presumably due to the impact of leaf senescence. LAI was assumed to be constant throughout the year; this will however vary within the growing season (Broadmeadow and Freer-Smith, 1996).

4.2. Health impacts of reductions in PM_{10} concentration

In the present study, the estimated health benefits are small, with between 2 premature deaths and 2 respiratory hospital admissions being averted per year. Powe and Willis (2004) estimated that the existing forest cover within the UK would result in a reduction of 5–7 deaths and 4–6 respiratory hospital admissions

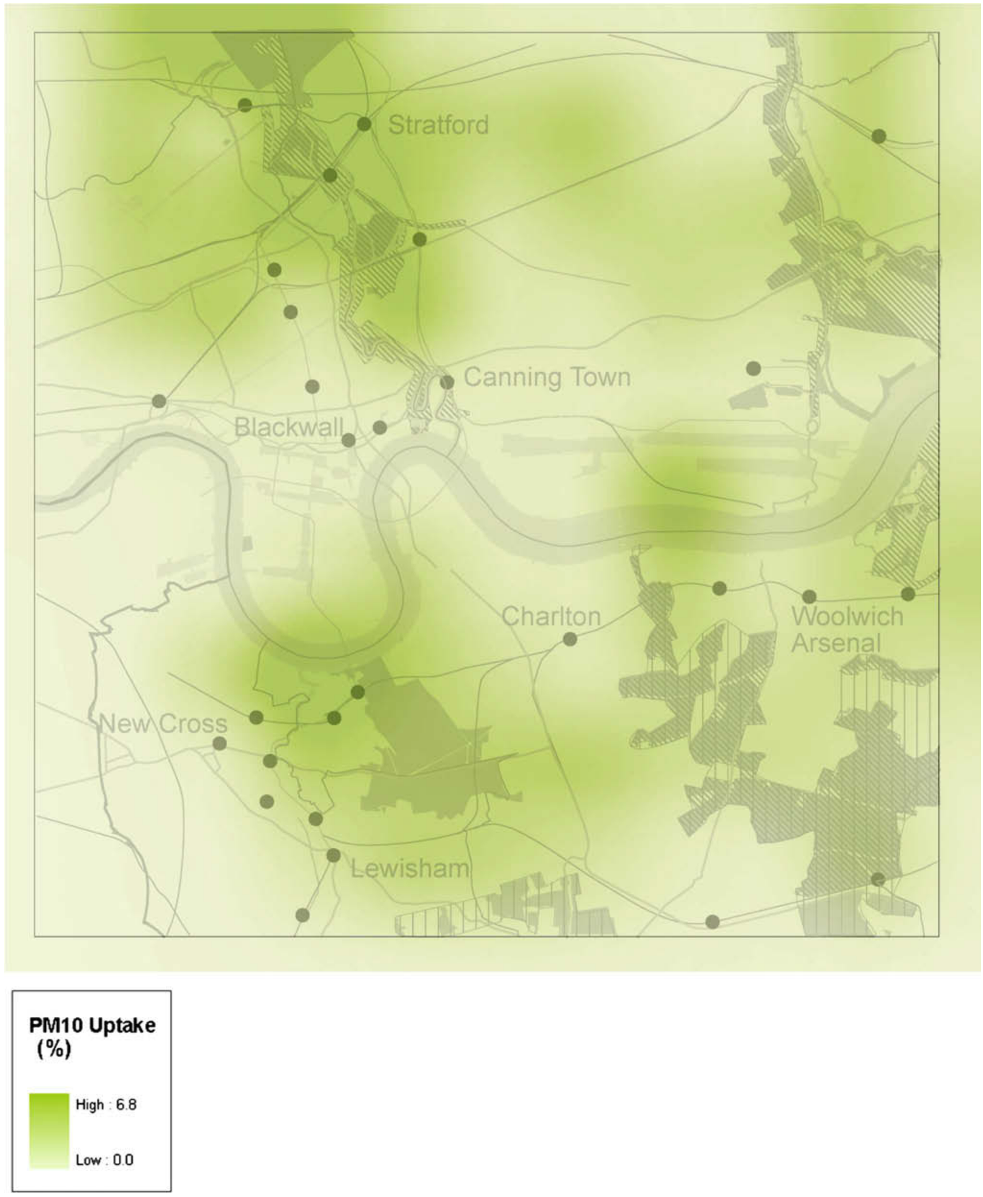

Fig. 4. Spatial distribution of the potential reduction in PM_{10} concentrations following implementation of the 75% grassland, 20% *A. pseudoplatanus* and 5% *P. menziesii* scenario in the East London Green Grid within the study area.

per year due to reductions in PM_{10} and SO_2 pollution. To put the above health benefits in perspective; a recent study on urban air quality management in the UK predicted that by reducing PM_{10} levels in Westminster (Central London) from 1996 to 1998 roadside levels to achieve an annual mean PM_{10} (gravimetric) target of 20 $\mu g\ m^{-3}$, an estimated 8–20 premature deaths would be averted in that area due to reduced short-term exposure and up to 100 deaths from long-term exposure (Mindell and Joffe, 2004).

There are also a number of uncertainties associated with the health impacts modelling. The relative risk coefficients used in the mortality calculations are subject to uncertainty. The model did not estimate the difference in respiratory symptoms between pre- and post-intervention. These symptoms would not require hospital admissions but would need medical attention. Changes in respiratory symptoms that need medical attention could be associated with changes in general practice (GP) consultations. There is epidemiological evidence to support the assumption that changes in air pollution impact on GP consultations (e.g. Wong et al., 2002). A recent study, which looked at asthma prevalence in 4–5-year-old children in New York found that the presence of street trees was associated with a 29% reduction in early childhood asthma (Lovasi et al., 2008). Although, the authors stated that 'this study does not permit inference that trees are causally related to asthma at the individual level'. The model also did not consider secondary health impacts which would be relevant if it is assumed that the implementation of the ELGG is likely to have other indirect health effects, for example though recreation, sports, increased use of pedestrian and cycle transport and increase well-being (O'Brien, 2005). There are epidemiological studies which could be used to quantify the health benefits from additional exercise (e.g. Tully et al., 2007).

4.3. Practical considerations for greenspace establishment

The aim of this study was to demonstrate how an integrated tool used be used to predict the impact of greenspace initiative in

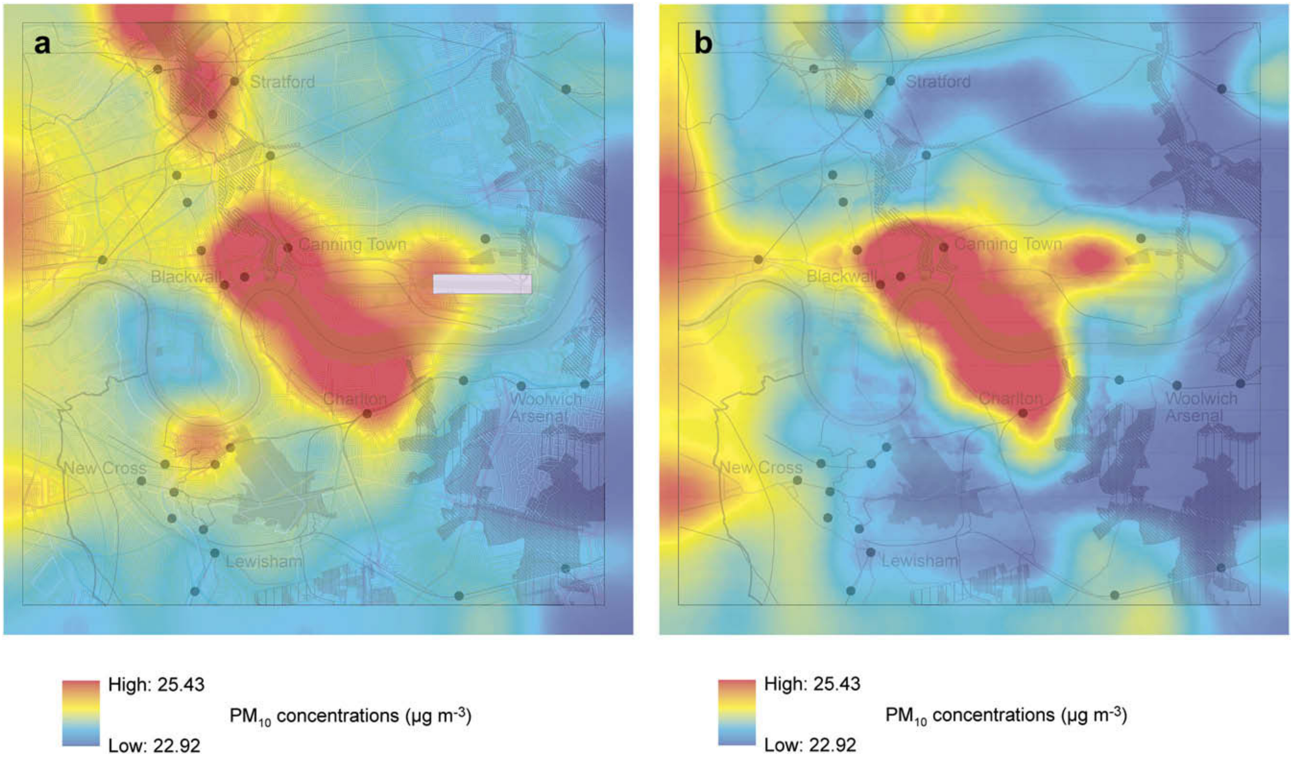

Fig. 5. PM_{10} concentrations from ADMS-Urban within the study area a) prior to and b) post-implementation of the 75% grassland, 20% *A. pseudoplatanus* and 5% *P. menziesii* scenario in the East London Green Grid. [note: the location of the London City Airport runway is shown with a rectangular patch in the upper panel.]

terms of PM_{10} concentrations. Modern greenspace aims to be multifunctional and as such must be designed to meet a number of objectives. Considering the wide range of drivers for the ELGG development, of which air quality improvements are only a small part, the relevant proportion of the greenspace taken up by trees is likely to be relatively low. The planting scenario selected for the health impact work demonstrates the value in planting a relatively small proportion of conifer species, which could also be targeted around 'hot-spots' of PM_{10} pollution in order to realise the maximum benefit.

Apart from their ability to mitigate PM_{10}, there are many other benefits to tree establishment that have not been considered in this study. These include additional improvements in air quality, for example through the uptake of O_3, SO_2 and NO_X (Broadmeadow and Freer-Smith, 1996). There are also many environmental and social benefits to greenspace creation in general, including their contribution to sustainable urban drainage, soil stabilisation, flood mitigation, shade provision, biodiversity, education, community cohesion and health and well-being. The benefits of greenspace must be considered in tandem with the other, potentially detrimental aspects of greenspace and tree establishment, including VOC emissions, which are implicated in the formation of O_3, pollen production, damage to property and maintenance costs.

People's behaviour will also have a significant impact on how the reductions in PM_{10} concentrations affect health. The most significant reductions in PM_{10} concentrations were estimated to be within the greenspaces themselves, suggesting that, in order for their full effects to be realised, the local residents would need to use the greenspaces. The most significant impacts of tree establishment are likely to be during peak traffic densities when vehicular emissions are greatest. These are also likely to be the time periods of greatest exposure to air pollution, for example when people are out of their houses or places of work and travelling to work or school. Encouraging people to walk or cycle through greenspace rather than walking along the side of roads may result in even greater benefits in terms of human exposure, although this will depend on a number of other factors including the perception of crime, ease of access and the attractiveness of the site. Alternatively, street trees could be used to provide localised improvements in air quality along busy roads or pathways.

5. Conclusions

This study demonstrates that tree planting schemes in urban areas such as the ELGG can make a positive contribution to air quality bringing additional benefits to human health. Furthermore, urban greenspace creation has received attention in recent years through the recognition of the social, environmental and economic benefits that it can bring to communities. The integrated modelling approach presented here provides a tool, which in combination with other models (e.g. to quantify climate amelioration, health and well-being), could be used to assess the potential benefit of such initiatives and provide the evidence base for their continuing role within urban environments.

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