

Urban particulate pollution reduction by four species of green roof vegetation in a UK city

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HIGHLIGHTS

- ▶ Green roofs act as passive filters of airborne particulate matter.
- ▶ Species differences in particle capture efficiency were observed.
- ▶ Morphological reasons for differences in particle capture efficiency were posited.
- ▶ Spatial differences in leaf SIRM were observed in relation to PM₁₀ sources.
- ▶ 0.24 tonnes of PM₁₀ a year could be removed from Manchester city centre.

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ABSTRACT

Urban particulate pollution in the UK remains at levels which have the potential to cause negative impacts on human health. There is a need, therefore, for mitigation strategies within cities, especially with regards to vehicular sources. The use of vegetation as a passive filter of urban air has been previously investigated, however green roof vegetation has not been specifically considered. The present study aims to quantify the effectiveness of four green roof species – creeping bentgrass (*Agrostis stolonifera*), red fescue (*Festuca rubra*), ribwort plantain (*Plantago lanceolata*) and sedum (*Sedum album*) – at capturing particulate matter smaller than 10 µm (PM₁₀). Plants were grown in a location away from major road sources of PM₁₀ and transplanted onto two roofs in Manchester city centre. One roof is adjacent to a major traffic source and one roof is characterised more by urban background inputs. Significant differences in metal containing PM₁₀ capture were found between sites and between species. Site differences were explained by proximity to major sources. Species differences arise from differences in macro and micro morphology of the above surface biomass. The study finds that the grasses, *A. stolonifera* and *F. rubra*, are more effective than *P. lanceolata* and *S. album* at PM₁₀ capture. Quantification of the annual PM₁₀ removal potential was calculated under a maximum sedum green roof installation scenario for an area of the city centre, which totals 325 ha. Remediation of 2.3% (±0.1%) of 9.18 tonnes PM₁₀ inputs for this area could be achieved under this scenario.

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1. Introduction

The adverse effects on health due to atmospheric particulate pollution have been the subject of a growing number of studies in recent years (Daniels et al., 2004; Tchepel and Dias, 2011). Urban residents are at particular risk from certain anthropogenic sources such as traffic, and a rapidly increasing urban population worldwide gives these studies additional importance because higher numbers of people will be exposed to urban particulate pollution.

Particles smaller than 10 µm in diameter (PM₁₀) can penetrate deep into the pulmonary passages where any transition metals present, such as iron and copper, can release free radicals in lung fluid and cause cellular inflammation (Birmili and Hoffmann, 2006).

Modern anthropogenic sources in urban areas include vehicular traffic fuel combustion, particularly diesel, and vehicular component wear (Petroff et al., 2008; AQEG, 2005; Bukowiecki et al., 2010). The environmental burden from component wear such as brakes, and resuspended road dust particles, is high because the particles are more likely to contain metals, polyaromatic hydrocarbons (PAHs) and sulphides which are toxic or carcinogenic. UK mitigation techniques have led to a decline in industrially sourced emissions but vehicle usage is continually increasing (DFT, 2009),

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thus offsetting the impact of vehicular emission controls (AQEG, 2005). Air Quality Management Areas (AQMA) are implemented in areas where air quality objectives are not likely to be met, wherein mitigation procedures are required (AQEG, 2005).

Persistent pollution episodes in cities are more likely to develop in winter in the Northern hemisphere because of a rise in space heating and the frequency of atmospheric inversions (Birmili and Hoffmann, 2006). Climate change could also cause an increase of episodes in summer, as one of the predicted weather changes is an increase in the likelihood of stationary air masses (AQEG, 2007).

Cessation of vehicular sources is impractical so cities in developed countries will have an ongoing need to mitigate PM₁₀. Transitional nations, with increasing traffic amounts, will need to plan for the future health of their citizens. Using plants as passive filters is being seen as a practical remediation method (Beckett et al., 2000; Freer-Smith et al., 2005; Litschke and Kuttler, 2008; Peachey et al., 2009). Trees and woodland have been found to be significant sinks for gaseous, aerosol, particulate and rain-borne pollutants (Fowler et al., 1989). Four processes are responsible for deposition onto the large surface area provided by leaves – sedimentation under gravity, diffusion and turbulent transfer giving rise to impaction and interception (Petrouff et al., 2008). The large leaf area and turbulent air movement caused by their structure makes trees particularly effective for particle removal (Fowler et al., 1989). However, space in cities is at a premium and there are limited opportunities to implement urban greening programmes with trees. Greening roofs is a viable and attractive alternative solution as roofs can form up to 35% of the urban land area (Macmillan, 2004). While a large amount of studies have focussed on trees, this study will look at the potential contribution of green roofs for PM₁₀ mitigation which is a preferable strategy to the exclusion of vehicles from cities (Litschke and Kuttler, 2008).

There is already some evidence for the potential of green roof vegetation for air pollution removal. Yang et al. (2008) found that 1675 kg of air pollutants, such as NO₂, SO₂ and PM₁₀, were removed by 19.8 ha of green roofs in one year, with PM₁₀ accounting for 14% of the total. A study in Toronto found that 58 metric tonnes of air pollutants could be removed if all the roofs in the city were converted to green roofs, with intensive green roofs having a higher impact than extensive green roofs (Currie and Bass, 2008). Intensive roofs have a deeper soil substrate than extensive roofs, which allows for a larger above-ground biomass and a wider variety of plants to be grown. While these studies offer promising results, they are based on modelling alone and, to the authors' knowledge, no empirical investigation of green roof removal of air pollution has been published.

Mitchell and Maher (2009) recently confirmed the magnetic characteristics of tree leaves to be a suitable proxy for ambient air pollution. This method can be used to increase spatial and temporal pollution monitoring resolution and has been used a number of times in magnetic biomonitoring studies involving trees (Matzka and Maher, 1999; Maher et al., 2008; Hansard et al., 2011). To date the technique has not been applied to quantify PM₁₀ removal by green roof vegetation.

The objectives of the study are twofold. Firstly to investigate spatial differences in particulate load to two roofs that differ in their proximity to pollution sources and secondly to elucidate any species differences in particulate capture rates between several common types of green roof vegetation used in the UK. With the choice of study sites, it is intended to ascertain whether differences in particle capture exist when a significant traffic source is located in close proximity to a roof, elevating local concentrations, compared to a roof subject solely to urban background concentration levels. The results will further be used to quantify the potential removal of metal-containing PM₁₀ by green roofs in an area of the

city centre of Manchester. Green roofs are increasingly being recognised as investments that can help address many of the challenges facing urban residents and, in recognition of this, Manchester City Council is developing a green roof policy guidance document (MCC, 2009). In addition, this work will support a burgeoning literature on the numerous benefits of green roofs such as reduction of Urban Heat Islands (Spolek, 2008) and their role as Sustainable Urban Drainage Systems (SUDS) (Scholz-Barth, 2001).

2. Methodology

2.1. Site description

Manchester is a large city situated in north-west England. The Manchester city district, which includes the centre of the Greater Manchester conurbation, has a population of 498,000 (MCC, 2010). It has seen a gradual improvement in its air quality since its industrial past; however PM₁₀ levels remain high near roadsides and in urban centres (HFAS, 2011). Indeed, the whole of Manchester city centre and the Oxford Road corridor (Fig. 1) are within an AQMA (DEFRA, 2011). Oxford Road is a key transport route within an area of high economic activity and has an annual average weekday traffic flow of 11,529 motor vehicles a day (GMTU, 2010). In 2007, the Manchester district produced 257 tonnes of PM₁₀, with roads being the major source (HFAS, 2011).

Two roofs situated on the Oxford Road corridor were chosen as locations for this study. Roof 1, Manchester Technology Centre (MTC), is a 3 storey office block with a conventional bare roof, situated adjacent north to the point where a busy inner city motorway, the Mancunian Way, crosses Oxford Road (Fig. 1). Traffic flows on the motorway reach over 60,000 vehicles a day (GMTU, 2010). Tyre and brake wear emissions account for ~23% of total road transport emissions and, along with resuspended road dust, appear to be more important for heavy duty vehicles than cars (AQEG, 2005). Urban canyon effects can strongly influence urban wind patterns and the Oxford Road corridor might channel these winds northwards. The location of Roof 1, combined with prevailing south-westerly winds, means this site is expected to receive a high load of road derived pollution and thus represents a roof with a strong local source.

Roof 2, the precinct bridge, crosses Oxford Road and consists of an extensive paved area over the road. This 2 storey roof was chosen for its proximity to the busy Oxford Road and represents a roof with an urban background pollution source. A control site at the Firs Botanical Gardens is situated 3.5 km south of the roof monitoring sites (see Fig. 1). This peri-urban site is in a location 200 m away from major road-sourced PM₁₀ in an area of high tree-planting. This control site is used to make comparisons between the other sites by utilising enrichment ratios.

2.2. Species selection and sampling

Four perennial species were chosen for this study – *Sedum album*, *Festuca rubra*, *Agrostis stolonifera* and *Plantago lanceolata*. The *Sedum* species is a common choice of plant for commercial extensive green roofs. *F. rubra* and *A. stolonifera* are both common British grasses, which can grow on turf roofs, and *P. lanceolata* is a common invasive species on green roofs (Dunnett et al., 2008). Trays of the four species were grown from seed, or washed cuttings (*S. album*), in a 'magnetically clean' greenhouse at the control site. John Innes number 3 compost was used. Trays of mature plants were then placed on the two study roofs on 03/07/2011. Additional trays were left in the greenhouse at the control site. Trays containing just compost were also placed on Roof 2 and in the control greenhouse to investigate soil metal concentration changes when

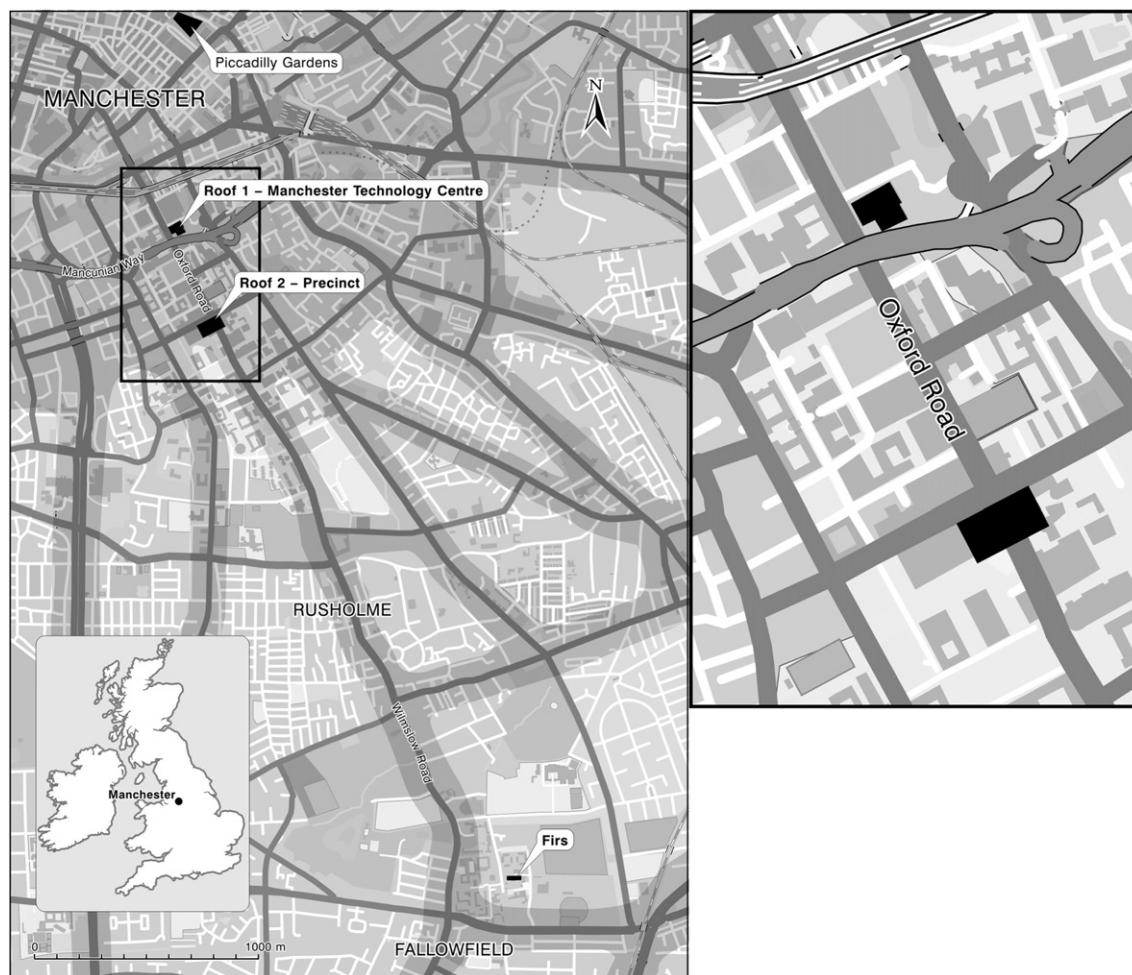


Fig. 1. Map showing locations of the study sites with Firs control site to the south of the city centre and the locations of Roofs 1 and 2 next to Oxford Road. The shaded overlay, highlighting the city centre and major roads, indicates the current extent of the Manchester AQMA (DEFRA, 2011).

plants are absent. In dry periods, plant trays were watered with pre-collected rainwater direct to the soil layer to avoid artificial rain effects of washing particles from the leaf surfaces.

Leaf samples were collected from mature plants. The sampling frequency was three times a week and all samples were collected in the mornings. Sampling began on 5/07/11 and the final samples were collected on 03/08/11. Enough leaves were collected to provide material for two samples per species, with each sample sufficient to fill a 10 ml sample pot. Very young leaves were not collected to ensure exposure times covered the entire study period. The samples were transported to the laboratory in plastic bags where they were dried in ovens for two days at 40 °C (three days for *S. album*), weighed, and stored in plastic film for subsequent magnetic measurement.

Samples were also taken from vegetation growing on or near the roofs, to investigate particulate capture by plants with prolonged exposure at the site locations. In the case of Roof 1, this was from the roof-height tree canopy of *Platanus acerifolia* and, for Roof 2, samples were taken from established plants of *Symphotrichum novae-angliae* and the evergreen *Hypericum inodorum*, growing in an adjacent green roof.

Surface areas of leaves, for area-normalising of results, were obtained by pixel counting images of scanned leaves and calculating regression equations from the relationship between area and dry weight. Subsequent surface areas could be calculated from

sample weight. Estimates of above surface biomass in the trays, useful for up-scaling pollutant removal efficiencies, were made by visual calculation of number of samples per tray.

Ambient PM₁₀ concentrations were recorded for half an hour at most sampling events at all three locations using a TSI Sidepak AM510 personal aerosol monitor. To determine the relative magnetic PM₁₀ levels of the three locations, pumped air samples of 100 l were collected from all three locations at a rate of 1.4 l min⁻¹ using the monitor in conjunction with an IOM sampler head fitted with glass filters (1 μm pore size Whatmann). The filters were folded and placed in plastic pots ready for magnetic measurements (see Section 2.3). Ambient PM₁₀ concentration data were also acquired from DEFRA's Automatic Urban and Rural Network (AURN) located in Piccadilly Gardens in Manchester City Centre (Fig. 1). These data can be compared with ambient PM₁₀ levels recorded by the personal monitor to investigate any temporal trends in PM₁₀ occurring throughout the study and to assess site differences. Rolling averages were calculated from the inter-sampling periods prior to each sampling event to see if the atmospheric PM₁₀ levels preceding sample collection influence leaf levels.

A Kestrel device was situated on Roof 1. This records wind speed and direction, useful for determining prevailing wind patterns and hence identifying upwind sources. Meteorological data were also available from the Whitworth Observatory (UOM, 2012), located

adjacent to Roof 2. Rainfall was recorded because rainfall can influence trends in leaf pollutant load as particulate matter can be partially washed off.

2.3. Magnetic measurements

Dried leaf samples were placed in 10 ml cylindrical styrene sample holders and immobilized with packing film. Saturated Isothermal Remanent Magnetisation (SIRM) was imparted on each sample in an applied field of 1 T, using a Molspin Instruments' high-field 'Pulse Magnetizer' (Walden et al., 1999). SIRMs were then measured with a Molspin Instruments' 'Minispin' fluxgate magnetometer. SIRM provides a measurement of the total concentration of magnetic particles within a sample (Muxworthy et al., 2003), once remanence values are normalised for leaf surface area (Matzka and Maher, 1999). Recent works have shown that leaf SIRM can be used as a proxy for ambient PM₁₀ levels (Mitchell and Maher, 2009; Kardel et al., 2011).

2.4. Elemental analysis

Following the magnetic measurements, subsets of the leaf samples from the start and end of the study were prepared for elemental analysis via ICP-MS (Agilent 7500cx). Leaves were digested in high purity nitric acid (15.6 M) in closed vessels using a microwave apparatus (MARS Xpress, CEM) according to US EPA method 3051A. Increases in metal concentrations over the study period may signify capture of metal-containing particles on the leaf surfaces, with Fe being of particular interest due to the ferrimagnetic properties of Fe containing minerals and hence influence on leaf SIRM. Vehicle derived PM₁₀ contains high levels of Fe due to conversion of Fe impurities in fossil fuels on combustion (Muxworthy et al., 2003).

Soil samples were also prepared for elemental analysis by ICP-OES to investigate potential metal concentration changes as particulates are washed into the substrate by rains. Samples were taken at the end of the study from one tray of each species at all three locations, and from trays of bare soil, with samples taken from the surface soil layer and deeper in the trays. Soil samples were also taken at the start of the study before the seeds were sown. The soil was dried at 40 °C and sieved to 63 µm to remove stones/twigs/leaves and homogenised in a ball grinder (Fritsch Spartan pulverisette 0) prior to acid digestion (as described above).

2.5. Scanning Electron Microscopy (SEM)

Single leaves of the four main species in the study were collected at the end of the study from Roof 2. These were transferred in plastic sampling bags to prevent contamination, and subsequently mounted on carbon stubs for analysis via SEM (FEI XL30 Environmental Scanning Electron Microscope). This was carried out on the same day to prevent desiccation and subsequent alteration of leaf surface micro morphology (Stabentheiner et al., 2010). Element mapping was undertaken via energy dispersive X-ray analysis (EDX) on clusters of particle grains to investigate any significant elemental distributions.

2.6. Statistical analyses

Correlations were carried out between average ambient PM₁₀ concentrations at the sites and with average PM₁₀ recorded at the same times from the AURN station in Piccadilly gardens. Rolling averages of ambient PM₁₀ over the pre-sampling periods at the AURN station were calculated and correlated against leaf SIRM values. Data were grouped by species or site, which allows for inter-

site and inter-species comparisons to be made respectively. Repeated Measures ANOVA (RMANOVA), with the Greenhouse-Geisser correction, was used to test significance with site or species as a between-subject factor. Linear Mixed Models (LMM), with site and species as random effects, were fitted to any trends seen from the start of the study to the end. Statistical analysis was carried out using SPSS software (SPSS 16).

2.7. PM₁₀ removal quantification

Particle numbers were counted in SEM images of the same leaf samples described in Section 2.5, sampled at random, and averaged per unit leaf area. The counts (N) were used to calculate particle volume by:

$$V = \frac{N}{6} \pi D^3.$$

where D is average particle diameter and particles are assumed to be spherical

Particle mass is then given by: $M = \rho V$.

where ρ is the particle density, assumed to be 1.3 g cm⁻³ (Held et al., 2006).

The mass per unit leaf area can then be used to quantify PM₁₀ removal by estimating average leaf area per unit area of roof based on measurements of above surface biomass of the experimental trays. To gain an annual removal figure, it was assumed that while the study species are evergreen, growth is compromised in the winter months, so a rate of 50% of the summer rate was chosen. Sensitivity to this rate was assessed by calculating removal at ±5%. Roof 2 was considered suitable for the quantification work as it represents an 'average' roof, subject to urban background PM₁₀ levels.

3. Results

3.1. Site comparisons

Average ambient PM₁₀ concentration at different days of sampling were significantly correlated between the sites (control with Roof 1, $r = 0.87$; control with Roof 2, $r = 0.59$; Roof 1 with Roof 2, $r = 0.85$) indicating that while the magnitude of PM₁₀ differs between the sites, the overall trends are broadly similar (Fig. 2). Fig. 1 shows their locations, with the maximum distance being between the control site and Roof 1 (3.5 km).

Correlations between average ambient PM₁₀ with average PM₁₀ from the AURN station (control, $r = 0.59$; Roof 1, $r = 0.71$; Roof 2, $r = 0.64$) showed good agreement, with the lower value for the control site due to the increased distance from the city centre. The filter SIRM values (Fig. 3) were lower for air sampled at the control site than on the study roofs.

Grouping the data by species (Fig. 4) shows SIRM was influenced by time, with an increase seen from start of study to end. This effect of time was significant (RMANOVA, $F = 12.914$, $p = 0.01$). There was a general trend noticeable whereby the highest SIRM values are recorded on Roof 1, followed by Roof 2. The lowest SIRM values were seen at the relatively unpolluted control site. This influence of site location on SIRM was also significant as evidenced by between-subjects effects of site ($F = 103.27$, $p = 0.01$).

The increases in SIRM over the study period were found to be significantly different between sites (LMM, $F = 55.76$, $p = 0.01$), with Roof 1 producing the strongest effect on the SIRM increase. Fig. 5 shows the SIRM enrichment ratios, the ratio of each roof over

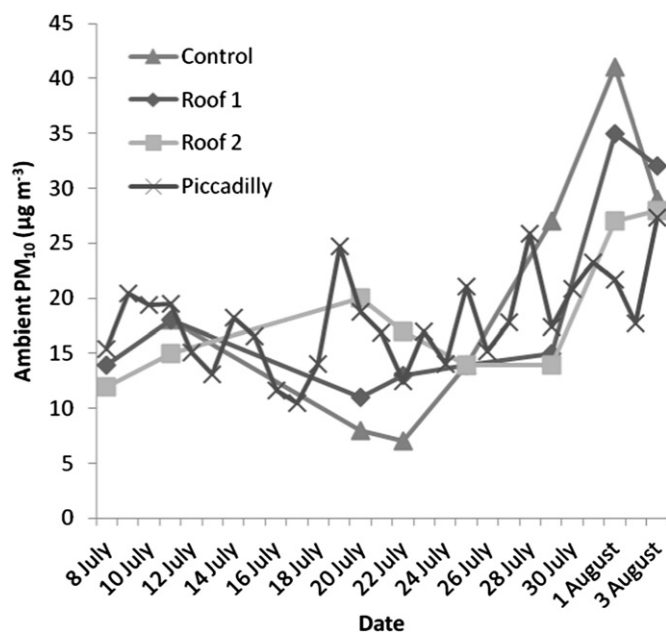


Fig. 2. Average ambient PM₁₀ levels between 11 am and 1 pm at each of the three sites on selected days throughout the study and daily average at the Piccadilly monitoring station.

the control site, at each sampling event. The plants on Roof 1 were clearly capturing more PM₁₀ than those on Roof 2. The two roofs displayed a similar trend for the latter half of the study.

Wind rose analysis of data from Roof 1 showed that the prevailing wind direction throughout the study was predominantly from the south west, especially in the drier final two weeks. These winds would potentially be carrying large amounts of road derived pollution from the proximal, southerly located Mancunian Way, which would explain the higher SIRM values at this site.

3.2. Species comparisons

Grouping the data by site (Fig. 6) shows a trend in the ability of the different species to capture magnetic PM₁₀ could be seen to follow the order *A. stolonifera* > *F. rubra* > *P. lanceolata* > *S. album*.

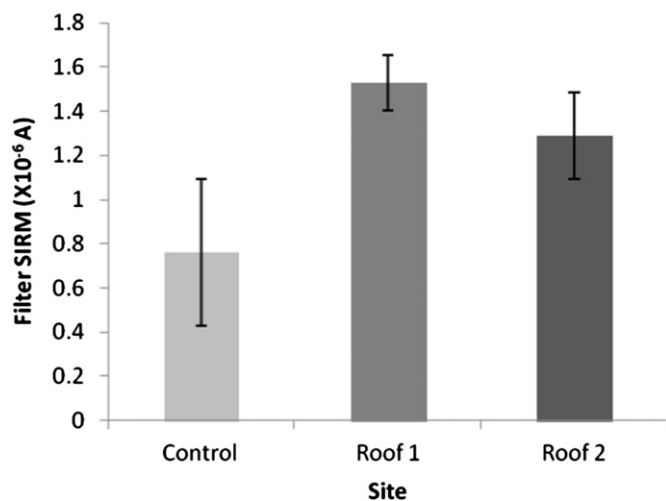


Fig. 3. Filter SIRM from 1 h pumped air samples taken on 27 July 2011, showing the lower levels of magnetisable particulate matter at the Firs control site.

Between-subjects effects showed significant effects of species on the leaf SIRM levels (RMANOVA, $F = 65.93$, $p = 0.01$), so some species were clearly collecting particles faster than others. Significant effects of species (LMM, $F = 34.99$, $p = 0.01$) were noted, with *A. stolonifera* producing the best estimates, as well as higher SIRM values.

The SEM images in Fig. 7 show the different leaf surface micromorphologies of the four green roof species and the predisposition of the particles to accumulate within parallel grooves, especially for the grasses, *A. stolonifera* and *F. rubra*. *P. lanceolata* and *S. album* also have grooved surfaces but the grooves are randomly distributed at the boundaries of tessellating plate-like structures. The grooves are more pronounced with *P. lanceolata* than *S. album*, and the latter also has a waxy surface, which may explain the higher particle capture efficiency of *P. lanceolata*. Small barbs on the surface of *A. stolonifera* leaves could also enhance particle capture.

The grains present on the leaves were of varying shapes and sizes. However, a number of grains were found to be the spherule shape characteristic of vehicle exhaust produced PM₁₀, formed from cooled droplets (Maher et al., 2008). Spot EDX analysis of these spheroidal particles yielded high Fe concentrations. Elemental mapping of a cluster of particles also showed a high occurrence of Fe – rich particles in the 2–5 µm range with larger particles being rich in silica (Fig. 8), indicating a geogenic source for these larger particles (De Berardis et al., 2007).

The results of the ICP-OES analysis of the upper and lower soil horizons from the plant trays showed no significant spatial or temporal differences in levels of Fe, Al, Ni, Pb or Zn.

Leaf Fe concentrations did not increase over the duration of the study despite a fairly high correlation between leaf SIRM values and Fe concentrations ($r = 0.67$, $p = 0.01$). Table 1 shows this relationship, with clear SIRM increases for all species and sites, but a rather erratic pattern for leaf Fe concentrations. The extent of these increases from start of study to finish manifests with the grasses showing higher average SIRM increases than *P. lanceolata* and *S. album*. Fe concentration increases were mostly apparent on the roofs, with the control site showing decreases or a minimal increase in the case of *F. rubra*.

The correlations in Table 2 show varying inter-species relationships, with *S. album* generally having the lowest correlation with other species (average $r = 0.31$), and the two permanently located species on Roof 2, *S. novae-angliae* and *H. inodorum*, having the highest correlations. Their correlation with each other was very high ($r = 0.94$). Their permanence at the site could mean that they are well-equilibrated with atmospheric PM₁₀.

The best correlations between rolling averages of ambient PM₁₀ against leaf SIRM values (Table 3) were seen with the Roof 1 samples. This is possibly due to their location closer to Piccadilly gardens. Good correlations were also seen with the permanent shrubs, *S. novae-angliae* and *H. inodorum*, on Roof 2, again indicating a potential equilibration with atmospheric PM₁₀.

The increase of the SIRM values of the permanently located green roof species and roof height tree canopy in the drier second half of the study period (Fig. 9) suggested increasing PM₁₀ levels in dry conditions, as would be expected. However, the SIRM decrease after rainfall events exceeding 2 mm, noted in Mitchell et al. (2010), did not exhibit a particular pattern. Although the results suggest a non uniform response to rainfall events by species type, it can be seen that the larger rainfall events within the period 16–22/07/2011 produce a decrease in the SIRM values for most species at most locations (Fig. 4). The reverse was also apparent, i.e. a lack of rainfall allowed the SIRM values to increase, as seen in the drier period towards the end of the study. This dry period also produced a rise in ambient PM₁₀ as seen in Fig. 2.

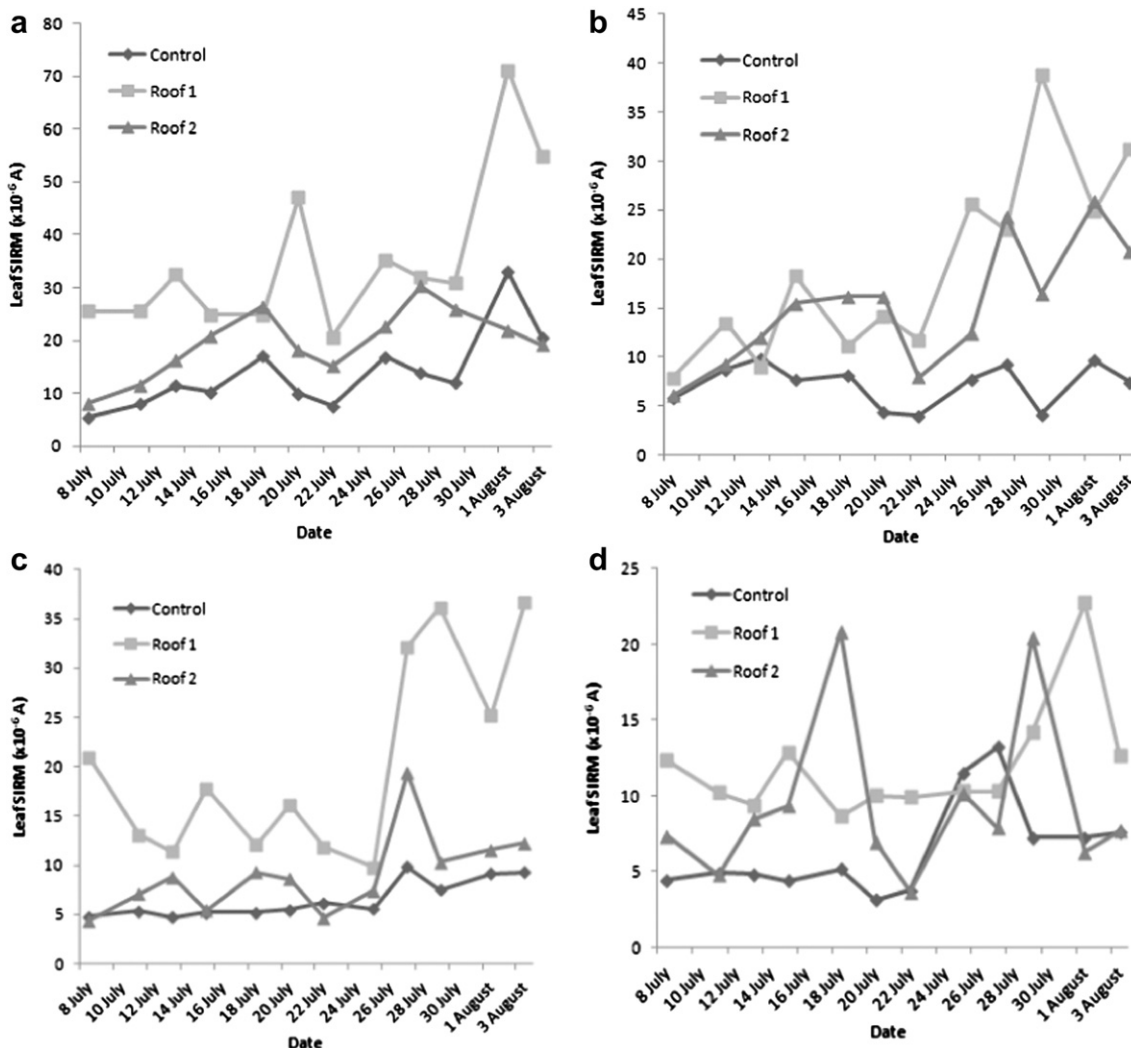


Fig. 4. Leaf area-normalised SIRM values of the study green roof species; (a) *A. stolonifera*, (b) *F. rubra*, (c) *P. lanceolata*, and (d) *S. album* at the three study sites (note the differing y-axis scales). Upward trends over the study period are noticeable, with Roof 1 capturing more PM_{10} than Roof 2.

4. Discussion

4.1. Spatial differences in PM_{10} capture

The site specific trends were found to be significant with Roof 1 being the site with the highest particle capture rates, due to its location downwind of a major source. Roof 1 is slightly different from Roof 2 in that it is adjacent to a raised busy inner city motorway which is only a few metres lower, and prevailing winds cross this motorway before reaching the roof. This could imply that Roof 1 represents a receiving environment for a local source of PM_{10} whereas Roof 2 is more dominated by urban background PM_{10} deposition. The high Fe contents of the particles found by EDX on samples from Roof 2 might indicate the vehicular source of the matter in this location, which confirms the suggestion for a considerable input from transport related PM_{10} at the sites. The agglomerated clumps of particles found were quite common and are potentially due to magnetic interactions between particles (Mitchell and Maher, 2009).

Spatial patterns of PM_{10} exist in the data with the three urban centre locations, Roof 1, Roof 2 and Piccadilly, displaying similar trends in ambient PM_{10} and Roof 1 plants having the highest correlation with the rolling average Piccadilly ambient PM_{10} data. These spatial patterns support previous work that suggest the

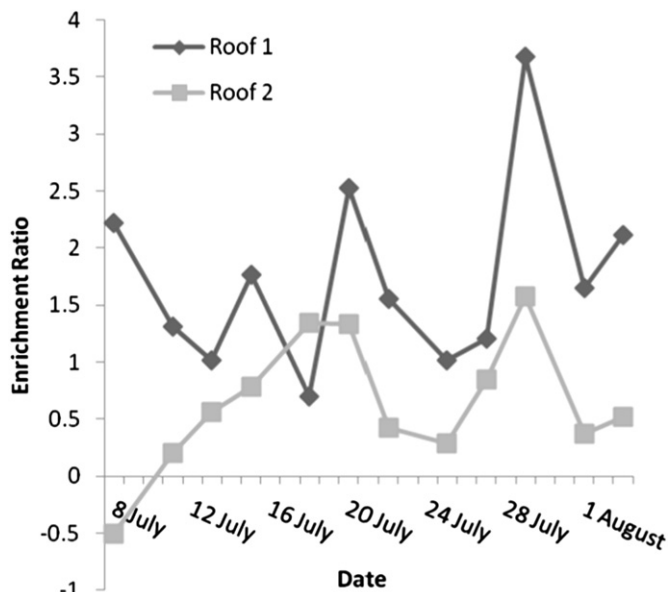


Fig. 5. SIRM enrichment ratio of the two roadside sites to the less polluted control site showing the higher enrichment on Roof 1 than Roof 2.

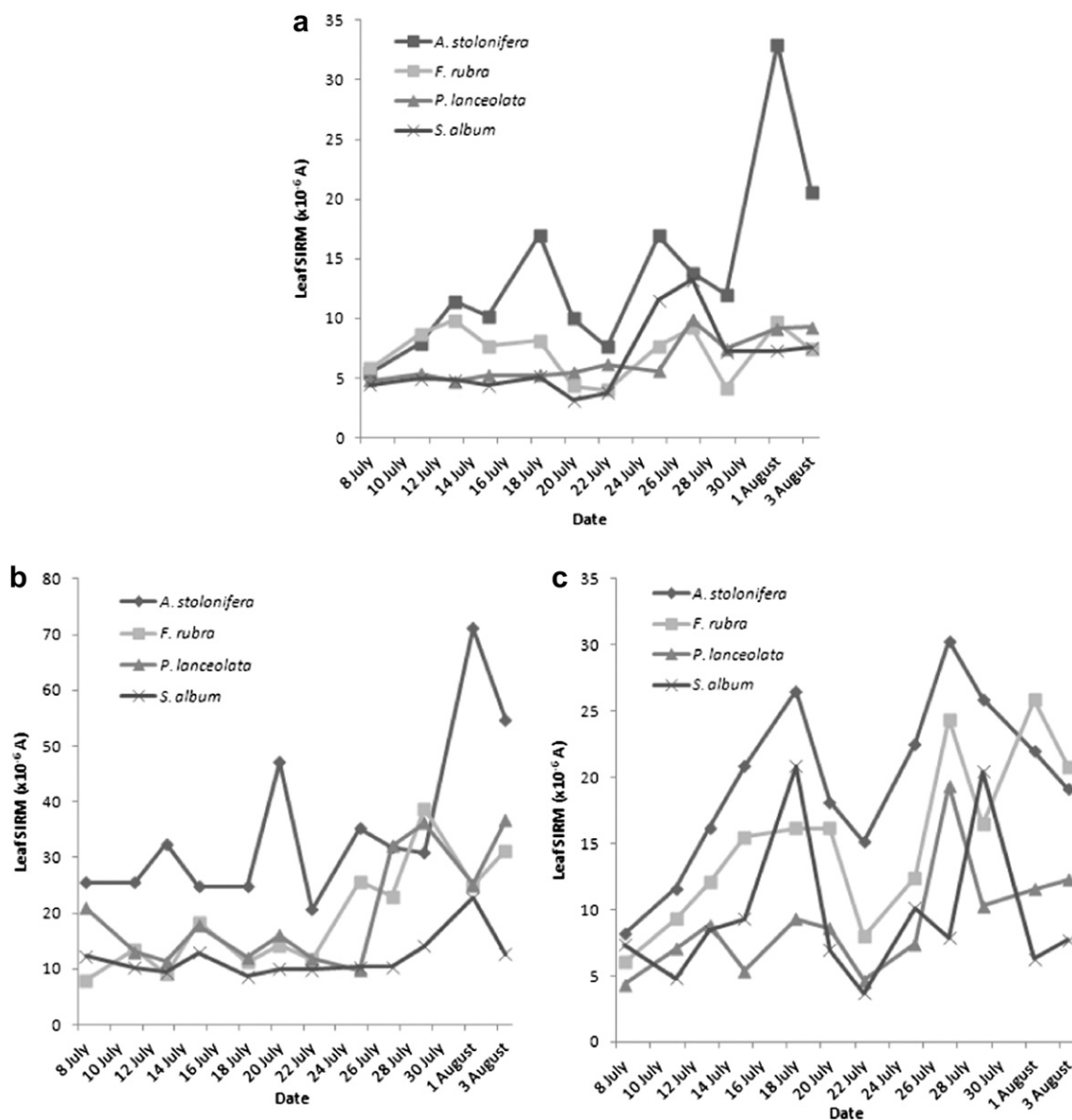


Fig. 6. Leaf area-normalised SIRM values of the study species at each of the three sites; (a) Control, (b) Roof 1, and (c) Roof 2 (note the differing y-axis scales). Upward trends over the study period are noticeable with *A. stolonifera* displaying the highest particle capture.

utility of biomagnetic monitoring for capturing fine-scale spatial variations in PM_{10} concentrations (Mitchell and Maher, 2009; Hansard et al., 2011). However the green roof species in this study, while useful due to their evergreen nature, would need further study into the time taken to equilibrate, as the results suggest equilibrium is yet to be reached.

This delay to equilibration is evidenced by the moderate leaf SIRM correlations between the species, and the fact that the highest correlation of $r = 0.95$ was seen between the two permanent species on Roof 2, indicating that they have had enough time to equilibrate. Underlying species differences in the time taken to equilibrate could also be influencing the correlations. Kardel et al. (2011) also found a progressive increase in leaf SIRM of deciduous trees over a whole growing season, even when relative change in ambient PM_{10} was negative. Equilibration times as short as 6 days have been found for roadside trees (Mitchell et al., 2010), however Lehndorff et al. (2006), investigating needle SIRM for the evergreen *Pinus nigra*, observed 26 months taken for equilibrium to be

reached, with an increase in SIRM for the first 20 months. This has implications for any conclusions made on PM_{10} removal quantities based on the results of this study. Net particle to leaf deposition will be expected to remain positive after the end of the study so calculations of PM_{10} removal might be underestimates of the true capture efficiency.

The lack of equilibration means the increase in SIRM values over the limited study time period can be viewed, for Roofs 1 and 2, as species and site specific temporal trends triggered by moving from a low pollution environment to a high one. For the control site, incremental SIRM values are due to the fact that while the site is relatively unpolluted and remote from major road sources, there is still a peri-urban background PM_{10} signal. Ambient PM_{10} levels inside the greenhouse at the control were very similar to those at the study roofs (Fig. 2), however, a larger fraction of this will be non-magnetic, biogenic PM_{10} such as pollen and condensed Biogenic Volatile Organic Compounds (BVOCs) (Litschke and Kuttler, 2008). The filter SIRM values in Fig. 3 confirm this, with

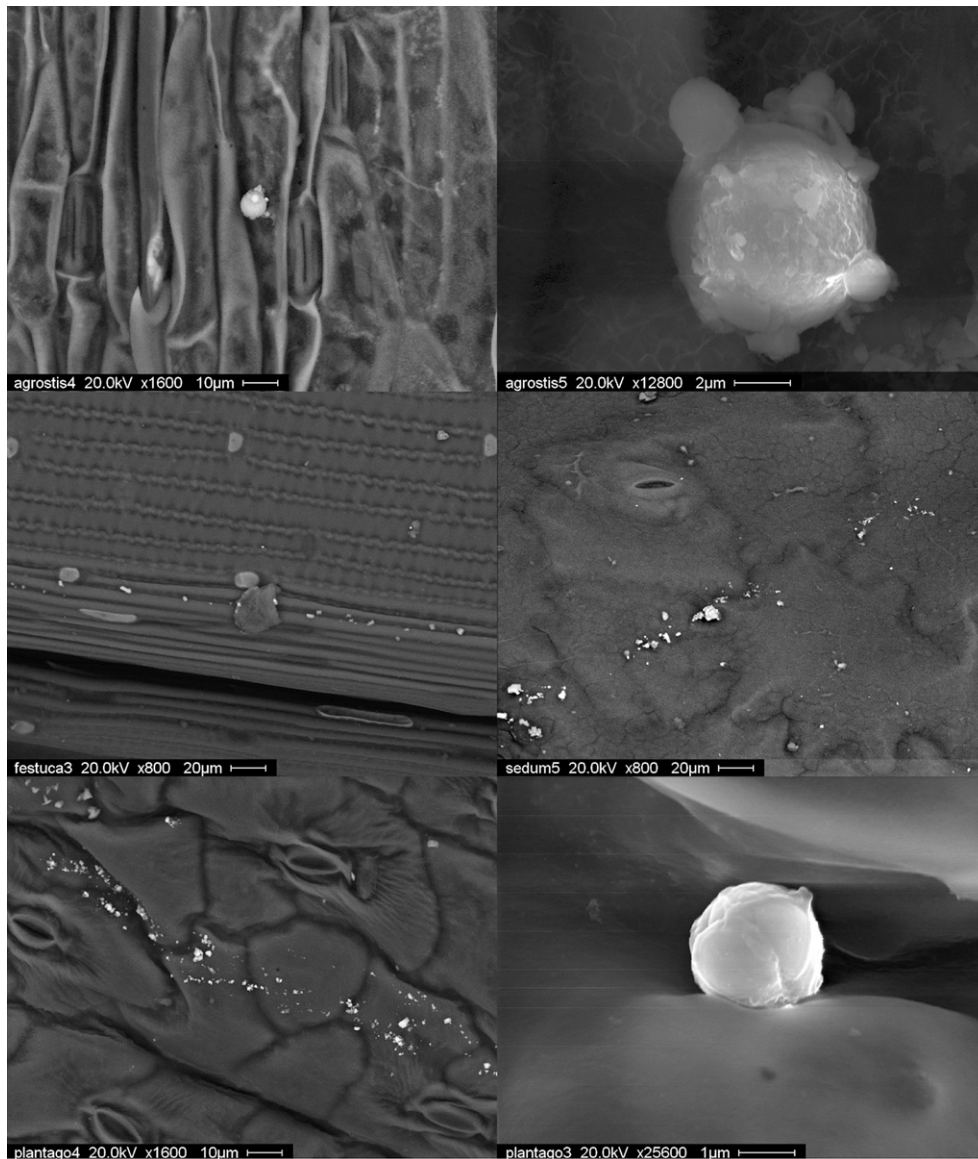


Fig. 7. SEM images of leaf surfaces of the four study species (X800–1600 mag.) with higher magnification images of selected spherules (X12,800–25,600 mag.).

a lower magnetic signal identified from air sampled at the control site.

Rooftops, both by their distance from, and obliqueness to, urban streets, represent an environment slightly removed from the vehicular source of urban PM_{10} . Roadside PM_{10} concentrations have

been found to be 100% higher at 0.3 m than at 1.5 m height (Mitchell and Maher, 2009) indicating a decrease with height. This means that PM_{10} deposition to rooftop surfaces is dependent on aerial transport mechanisms via winds within the Urban Canopy Layer (UCL) and gravitational deposition of particles from within

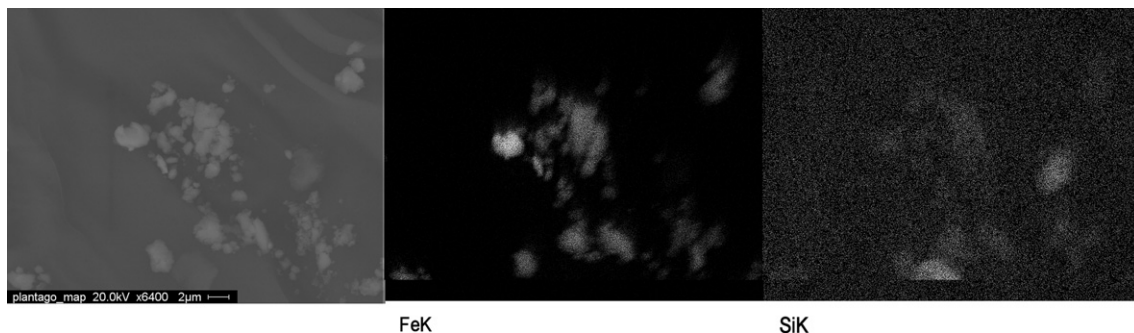


Fig. 8. EDX for *P. lanceolata* abaxial leaf surface indicating distribution of iron and silica within a particle cluster.

Table 1

Percentage increase in SIRM values and iron concentration over the duration of the study, showing clear increases in SIRM but variable patterns for iron concentration.

	% Increase SIRM				% Increase iron concentration			
	Control	Roof 1	Roof 2	Average	Control	Roof 1	Roof 2	Average
<i>A. stolonifera</i>	278.76	114.36	133.90	175.67	−24.79	35.04	−11.81	−0.52
<i>F. rubra</i>	28.02	295.69	243.27	188.99	.76	139.66	90.93	77.12
<i>P. lanceolata</i>	93.84	74.78	180.65	116.42	−60.88	86.28	124.36	49.92
<i>S. album</i>	70.62	14.79	4.82	30.08	−41.20	−32.80	7.41	−22.2

Table 2Correlation matrix for the SIRM values of the study species (A = *A. stolonifera*, F = *F. rubra*, P = *P. lanceolata*, S = *S. album*, PI = *P. acerifolia*, Sy = *S. novae-angliae*, H = *H. inodorum*) at each site (C = Control, 1 = Roof 1, 2 = Roof 2) showing wide variability. Bold values indicate significance at the 0.05 level.

	CA	1A	2A	CF	1F	2F	CP	1P	2P	CS	1S	2S	1PI	2Sy
1A	0.833													
2A	0.462	0.166												
CF	0.478	0.253	0.200											
1F	0.468	0.426	0.569	−0.092										
2F	0.774	0.701	0.740	0.398	0.579									
CP	0.622	0.594	0.539	0.200	0.682	0.823								
1P	0.307	0.401	0.378	−0.081	0.747	0.611	0.804							
2P	0.462	0.400	0.713	0.414	0.478	0.825	0.815	0.627						
CS	0.388	0.192	0.637	0.384	0.569	0.521	0.626	0.404	0.719					
1S	0.749	0.741	0.156	0.193	0.484	0.576	0.538	0.440	0.185	0.114				
2S	0.085	−0.204	0.571	−0.116	0.341	0.143	−0.067	0.185	0.126	0.099	−0.077			
1PI	0.303	0.471	0.045	−0.194	0.554	0.382	0.801	0.741	0.441	0.327	0.462	−0.310		
2Sy	0.680	0.653	0.614	0.284	0.799	0.874	0.918	0.815	0.794	0.606	0.546	0.101	0.648	
2H	0.609	0.711	0.490	0.222	0.712	0.808	0.816	0.787	0.773	0.448	0.458	0.110	0.623	0.935

this mixed layer. Litschke and Kuttler (2008) stated that to be an effective filter of urban air masses vegetation must be located as near as possible to the emission source.

4.2. Species differences in PM₁₀ capture

The SEM images (Fig. 7) show differences in the micro morphology of the leaf surfaces which may go some way in explaining the species differences in PM₁₀ capture. The parallel grooves on the leaves of the grasses can trap particles and prevent their resuspension. Hairy *Tilia* leaves have been found to be more efficient at particle capture than smooth leaved *Tilia* species (Kardel et al., 2011). Mitchell et al. (2010) observed plant species with hairy or ridged surfaces have a significantly higher leaf SIRM than those with a waxy or smooth surface, such as *P. lanceolata* and *S. album*.

The macro morphological influences on particle capture must also be considered, with the grasses having dense arrangements of long thin blade-like leaves which may affect the near-surface air flow and enhance deposition (Litschke and Kuttler, 2008). *P. lanceolata*, with its radial arrangements of lanceolate leaves, and *S. album* with its tendency to form low-lying mats and cylindrical leaves, may increase the near-surface roughness length to a lesser degree than the grasses.

The weak correlation between SIRM and Fe concentrations is potentially due to the nature of the SIRM measurement technique which is an indication of total concentration of magnetic particles (Muxworthy et al., 2003). Different Fe-containing minerals such as haematite and magnetite, have differences of mass specific SIRM at least an order of magnitude apart at the 1–10 μm size range

(Walden et al., 1999). This unaccounted for variation has the potential to weaken the relationship between SIRM and presence of Fe minerals. A possible solution to this would be to use incremental ARM acquisitions to characterise the proportions of Fe containing minerals present (Walden et al., 1999). The decoupling between SIRM and Fe may be due to species dependant effects from cycling of plant intracellular Fe, via soil uptake (Marschner et al., 1986) or surface absorption through the cuticle (Peachey et al., 2009). Further work could explore whether stronger correlations with SIRM may be achieved with Fe concentrations in leaf surface deposits, obtained through leaf-washing (Freer-Smith et al., 2005).

The soil metal concentrations did not change during the course of the study. This is potentially due to the brevity of the study being unable to capture the long-term nature of what is presumed to be a small flux of metal-containing particulates to soil.

4.3. Implications of wider green roof implementation

The species differences in particulate capture have implications for any roof-greening projects that have the specific aim of pollution remediation, with a necessity for increasing the near-surface roughness apparent. Intensive green roofs, which have a deeper substrate able to support the larger above-surface biomass, would be preferable, however higher installation and maintenance costs are associated with these (Clark et al., 2008). Green roof impacts may be mostly operating via reduction of urban background levels within the UCL, but certain locations adjacent to strong sources, as with Roof 1, indicate that strategic planting informed by location may be a key consideration.

Table 3Correlations of area-normalised leaf SIRM values with rolling average ambient PM₁₀ concentration (A = *A. stolonifera*, F = *F. rubra*, P = *P. lanceolata*, S = *S. album*, PI = *P. acerifolia*, Sy = *S. novae-angliae*, H = *H. inodorum*).

Site	Control														Roof 1														Roof 2																															
	A				F				P				S				A				F				P				S				PI				A				F				P				S				Sy				H			
	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S	A	F	P	S																								
r	0.39	−0.16	0.55	0.07	0.65	0.69	0.62	0.45	0.65	0.02	0.42	0.24	−0.14	0.64	0.70																																													

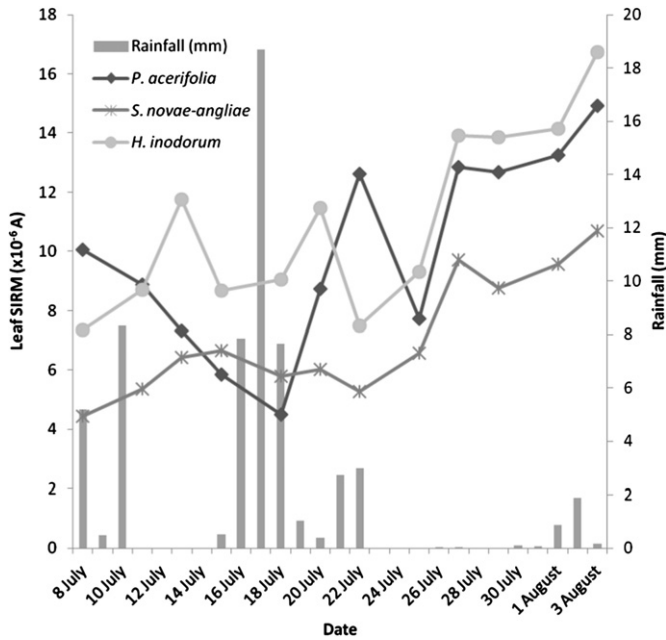


Fig. 9. Daily rainfall totals and leaf area-normalised SIRM values of the 'permanent' green roof species and the roof-height tree canopy which show a clear increase in the drier second half of the study period.

Quantifying particulate removal by green roof vegetation at the city scale can give an indication of the potential benefits of green roof installation. The removal rates achieved by grasses in Table 4 are higher than the $1.12 \text{ g m}^{-2} \text{ year}^{-1}$ for short grass and $1.52 \text{ g m}^{-2} \text{ year}^{-1}$ for tall herbaceous plants reported by Yang et al. (2008). Figures for trees in US cities are in the range $0.4\text{--}11.2 \text{ g m}^{-2} \text{ year}^{-1}$ with an average of $3.8 \text{ g m}^{-2} \text{ year}^{-1}$ (Nowak et al., 2006). In comparison to trees, green roof species do not perform as well, however the grasses, especially *F. rubra*, come close.

The green roofing potential of Manchester city centre and the Oxford Road corridor, 326 ha in area, was quantified using aerial photography in ArcGIS software. Polygons were constructed for all potential green roofs, based wholly on whether the roof was flat, and no consideration given to structural considerations. The total area of potential green roof coverage was found to be approximately 50 ha (15.3% of the selected area). A 'maximum extensive green roof scenario' was postulated, where every possible flat roof in this selected area of city centre Manchester has an extensive sedum roof. Emissions of PM_{10} for this area are estimated to be 9.18 tonnes per year, based on an emissions inventory (HFAS, 2011), and the area is mostly classed as an AQMA (DEFRA, 2011).

The potential removal of particulate matter under this scenario is 0.21 metric tonnes a year (Table 4). This equates as 2.3% ($\pm 0.1\%$) of the PM_{10} emitted by the scenario area, which is a considerable removal amount. Larger quantities could be removed with grass roofs evidenced by the 9.8–17.5% in the scenario, for *A. stolonifera*

Table 4
Estimates of quantities of PM_{10} captured by the study species with $\pm 5\%$ sensitivity to winter reduced rate. Higher removal rates were achieved by the grasses *A. Stolonifera* and *F. rubra*.

Species	$\text{g m}^{-2} \text{ year}^{-1}$	T year^{-1} under max green roof scenario
<i>A. stolonifera</i>	1.81 ± 0.06	0.9 ± 0.03
<i>F. rubra</i>	3.21 ± 0.1	1.61 ± 0.05
<i>P. lanceolata</i>	0.49 ± 0.02	0.25 ± 0.01
<i>S. album</i>	0.42 ± 0.01	0.21 ± 0.01

and *F. rubra* respectively. Consequently, intensive greening can be beneficially employed in critical locations adjacent to PM_{10} sources, and extensive green roofs installed in background locations. Sequestration of the metal-containing particles within the substrate layer of the green roof is expected (Berndtsson et al., 2009). Further research is needed, however, into the long term fate of these particles and whether green roof substrates would reach pollutant saturation.

Despite the distance from sources, the results presented here suggest that green roof vegetation has a sizeable impact on particulate matter levels in the mixed layer. The benefit of reduced particulate air pollution in cities is further evidence for the multi-beneficial nature of green roofs.

5. Conclusion

- The grasses *A. stolonifera* and *F. rubra* were found to have higher particle capture efficiency than the invasive weed *P. lanceolata* and the commercial extensive green roof species *S. album*. This should be considered if green roofs are being constructed with the aim of air pollution reduction.
- Both macro and micro morphological reasons for differences in capture efficiency were posited.
- Spatial differences in leaf magnetic SIRM of the green roof species were associated with proximity to PM_{10} sources being a main factor.
- Green roofs act as passive filters of airbourne particulate matter. While not as effective as street trees, due to lower surface roughness lengths and increased distance from sources, they can be considered for remediation of urban air pollution because their construction does not require major upheaval of the urban built environment, as tree-planting schemes often do.
- 0.21 tonnes of PM_{10} a year were removed from Manchester city centre in a scenario involving all flat roofs within a chosen area being installed with an extensive green roof. This is the equivalent of 2.3% of the PM_{10} inputs of this area. Larger quantities can be removed with grass roofs.

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