AIR QUALITY EXPERT GROUP

Impacts of Vegetation on Urban Air Pollution

Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government; and
Department of the Environment in Northern Ireland

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This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of the Environment in Northern Ireland, on the impacts of vegetation on urban air pollution. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

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PB14503

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- Provide judgements on the quality and relevance of the evidence base;

- Suggest priority areas for future work, and advise on Defra's implementation of the air quality evidence plan (or equivalent);

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- Provide independent advice and operate in line with the Government's Principles for Scientific Advice and the Code of Practice for Scientific Advisory Committees (CoPSAC).

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Contents

1 Introduction

This report addresses the research and policy aspects of the interactions between vegetation and air pollutants in urban areas. The focus is on the ways in which existing vegetation and potential plantings influence atmospheric composition through emission of trace gases and by dispersion and deposition of air pollutants, and as a consequence, the concentrations of pollutants to which people in urban areas are exposed. The main criteria for selection of the reviewed literature for inclusion was quantification of the effects on fluxes, concentrations or mass budgets. Additional reference material to provide context and links to the wider literature on this subject have been included. ***Overall, vegetation and trees in particular are regarded as beneficial for air quality, but they are not a solution to the air quality problems at a city scale***.

Compared with emissions control at source, removing pollutants once diluted into the atmosphere is challenging because of the large volume of air into which the pollutants have been dispersed compared to the surface area to which any potential abatement technology may be applied. The report comprises two main sections, on the effect of vegetation on dispersion, and on the capture of pollutants by vegetation. The report begins with a summary of the policy implications.

2 Policy implications

The report aims to answer the following questions.

Is there definitive observational evidence of the effectiveness of urban vegetation in mitigating air pollution?

The effects of realistic planting schemes to alleviate air quality problems by enhancing deposition to the surface with vegetation in cities are small. Reductions in concentrations of PM_{10} for realistic planting schemes would be expected to be at the scale of a few percent. The work to date both from measurement and modelling shows that it is unlikely that large reductions in concentration (>20%) could be achieved using vegetation to enhance deposition over a substantial area.

For nitrogen dioxide (NO_2), vegetation is, generally speaking, of little benefit; it is not a very efficient sink. The deposition occurs in daytime, and primarily in the warmer months, when NO_2 is less of a problem. Vegetation is a very poor sink for nitric oxide (NO) and soil is a source of NO, at least partially offsetting any potential benefit of uptake by vegetation.

Locally (tens to hundreds of square metres) tree planting may enhance or reduce dispersion; this redistributes pollution but does not remove it. Where vegetation acts as a barrier close to a source, concentrations immediately behind the barrier owing to that source are reduced typically by a factor of about 2 relative to those which would occur without the barrier, whereas on the source side of the barrier concentrations are increased. Tree planting may also exacerbate the build-up of pollution within street canyons by reducing air-flow.

The use of trees to improve air quality is not without negative impacts as some tree species are important sources of biogenic volatile organic compounds (BVOCs), notably isoprene. BVOCs can enhance the formation of pollutants including PM and ozone. However, BVOC emissions could be avoided by selecting low emitting species. Similarly, the choice of plant species which are known sources of aeroallergens should be avoided.

It is important in communicating the potential benefits of vegetation in mitigating urban air pollution problems to provide quantitative estimates, supported by measurement and modelling and their uncertainties, and avoid the campaigning zeal, which is commonly associated with popular publications on the subject.

What role does vegetation and its effects on air pollution play in integrated urban planning and policy?

Recognising that the potential for improving Air Quality using vegetation is modest, an important limitation to mitigation of current Air Quality problems with vegetation is that the most polluted areas of cities are those with very limited space for planting, greatly reducing the potential for mitigation using these methods. An integrated policy which separates people spatially from major pollution sources (especially traffic) as far as possible and in which vegetation is used between the sources and the urban population maximises its beneficial effects.

Are the data and models to quantify effects of urban planting schemes on air quality in the major cities of the UK generally available?

A range of models have been applied to address the problem, all of which are available. However, the complexity of the physical and chemical environment in urban areas and knowledge of the interactions between the flow regimes and urban structures with emission sources have all been greatly simplified in the modelling to date. Thus, while the scale of the effects of vegetation on concentrations of particulate matter in urban areas is probably correct within a factor of 2, there is a great deal more to be learned from further very detailed measurement and modelling.

In the following sections, evidence has been selected from the literature where the papers directly address the quantification of effects of vegetation on dispersion and deposition of pollutants and their effects on ambient concentrations. In addition a range of recent reviews of the wider subject area are included for context.

3 Effects of vegetation on atmospheric dispersion

3.1 Introduction.

This section considers the impact of vegetation, in particular trees, on pollutant concentrations due to their effect on the dispersion of pollutants. The impacts on deposition are considered in Section B below. Janhall (2015), for example, includes a review of the impact of both processes.

It is the impact of trees on airflow and turbulence which determines any resulting impact on dispersion. This effect is first outlined for different examples, including different numbers, configuration and density of trees. This information is then extended to consider the dispersion of pollutants from a source upwind of, within and downwind of trees.

3.2 Effects of trees on airflow and turbulence.

Figure 1 illustrates the impact on the airflow and turbulence of a single tree and groups of trees of different density and layout. Considering first a single tree (Figure 1(a)) there is some deceleration of the flow upwind and around the tree due to blocking of the flow. Close to the tree there is further deceleration of the flow and also generation of turbulence as the air flows around and through the branches of the tree. Downstream in the near wake, the flow may be highly perturbed with substantial turbulence generation; in the far wake the flow slowly recovers to its upstream form with acceleration and decay of the wake turbulence. As trees change shape to some extent with wind speed, the magnitude of these effects is also dependent on wind speed. Detailed studies of the impact of a single tree on airflow are given in Gross (1987) and Green (1990).

Understanding how groups of trees affect the airflow and turbulence can be deduced from consideration of the effects of single trees in combination (Belcher *et al.* 2003, Britter and Hanna 2003, Belcher *et al.* 2012). For a sparse array of trees more than about 8 tree diameters apart (Figure 1(b)) the perturbation to the flow can broadly be considered as the sum of perturbations from individual trees – the direction of the flow is little affected by the trees but the mean flow speed is reduced and there is an increase in turbulence. As the density of the tree array increases, the flow through and around the trees is reduced and there is increased airflow over the trees, which descends again on the downwind edge of trees (see Figure 2). In this case the largest shear stresses are at the top of the canopy, so turbulence levels are increased here but reduced within the tree canopy. As the tree array density increases still further so that trees are less than 2 tree diameters apart (Figures 1(c,d)), the flow becomes increasingly blocked and for the highest array densities it is almost completely blocked, diverging over and around the group of trees much as it does around a building and providing a barrier to noise.

3.3 Effects of Trees on Dispersion

Once the effects of the trees on airflow and turbulence have been determined, it is straightforward in principle to determine their impact on dispersion and hence on pollutant concentrations. The mean airflow determines the plume trajectory and also impacts on the near field concentration which is inversely proportional to wind speed. The turbulence determines the rate of mixing with ambient air and hence the rate of dilution and spread of the plume. So, in general, an increase in wind speed or turbulence increases dispersion. Neglecting cases where the plume has high momentum or buoyancy, for ground level sources an increase in dispersion reduces ground level concentrations and vice versa. For elevated sources no such simple rule applies, but downward flows (e.g. behind obstacles) or an increase in the vertical component of turbulence brings pollutant down towards the ground tending to increase maximum surface concentrations. In these cases the sensitivity of concentration to wind speed is similar to ground level sources.

We have used these considerations in Table 1 to describe the impact on sources of pollution located upwind of trees, within the trees and downwind of trees, dependent on the proximity of trees to each other. Drawing the most general conclusions from this table, we can surmise that there is negligible impact upstream of well-spaced trees. However, there is an increasing adverse impact as tree density increases, and a slight positive impact within widely spaced trees changing to strongly adverse impact within closely packed trees. Impacts downstream are complex and depend on the location of the source and the density of the tree array.

In view of specific interest of the effect on pollutant concentrations of tree barriers and trees within street canyons we now consider these two cases separately and provide some limited quantitative estimates of their impact on pollutant concentrations.

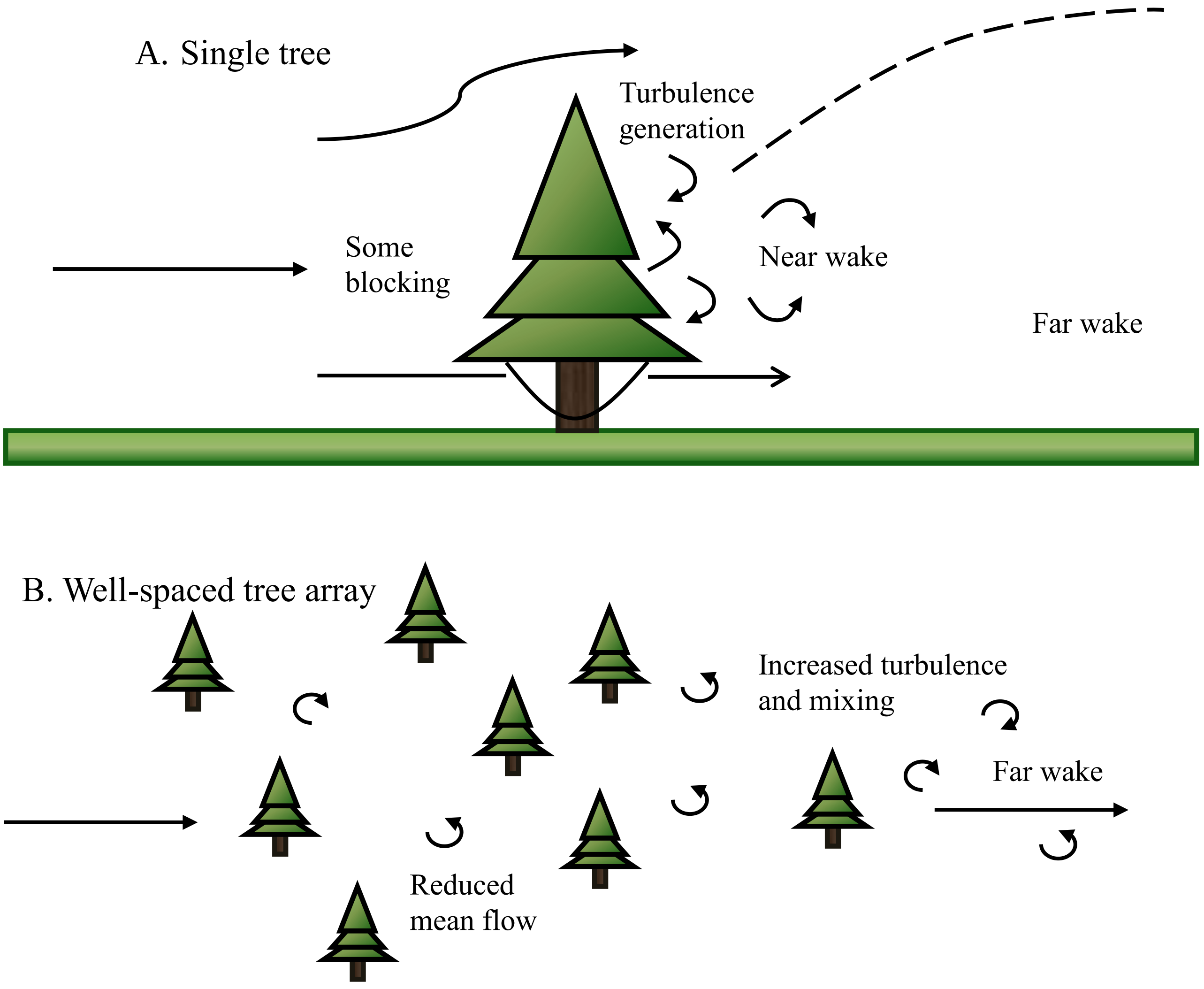
3.4 Tree Barriers

The impact of tree barriers (or lines of closely packed trees Figure 1(D)) and also of noise barriers, particularly those adjacent to roads, have been studied in some detail in the field, in wind tunnels and with CFD models (e.g. Baldauf *et al.* 2008, Baldauf *et al.* 2013, Al-Dabbous and Kumar 2014). Consistent with the broad descriptions above, when the wind blows from the road to the barrier there are reductions in concentrations on the downwind side of the barrier. These reductions decrease with distance away from the barrier and depend on the height and density of the barrier as well as other factors such as atmospheric stability and building morphology in the neighbourhood of the barrier. The measurements show a broad range in the maximum reduction in concentrations up to a factor of 5 (Baldauf *et al.* 2013), but reductions within a factor of 2 are more typical; see for example noise barrier studies of Heist *et al.* 2014 and Baldauf *et al.* 2008 and tree barrier studies of Hagler *et al.* 2012, Al-Dabbous and Kumar 2014 and Brantley *et al.* 2014. It is noted that for the studies conducted in the field, some of the concentration reduction may be attributable to deposition rather than dispersion effects. In very light winds reductions in concentration are less apparent and in some cases increases are observed. Roadside of the barrier, concentrations may show some increase close to the barrier, as in a single sided street canyon, both when the wind blows from the road to the barrier and from the barrier to the road.

3.5 Trees within street canyons.

There have been fewer studies of this case, which presents a greater challenge for experimental design. Most studies have considered impacts on deposition (Pugh *et al.* 2012), however Gromke and Ruck (2008) and Gromke *et al.* (2008) have conducted wind tunnel studies and CFD modelling of the impacts of avenues of trees within canyons. As discussed, single trees may increase turbulence levels with relatively little impact on the mean flow, however this is likely to be of little consequence in an urban street canyon where turbulence levels are already typically high. More important is the blocking effect which is enhanced in the confined space of a street canyon, leading to decreased dispersion and higher concentrations due to road sources (Woodland Trust 2012). This effect increases with the number of trees or their foliage density (Gromke *et al.* 2008) and may increase concentrations by as much as a factor of 2 when there are a sufficient density of trees to substantially reduce the air flow within the canyon.

Finally, we note that the magnitude of the impacts on concentrations discussed above are for primary pollutants. Impacts on secondary pollutants, in particular NO_2, are reduced.



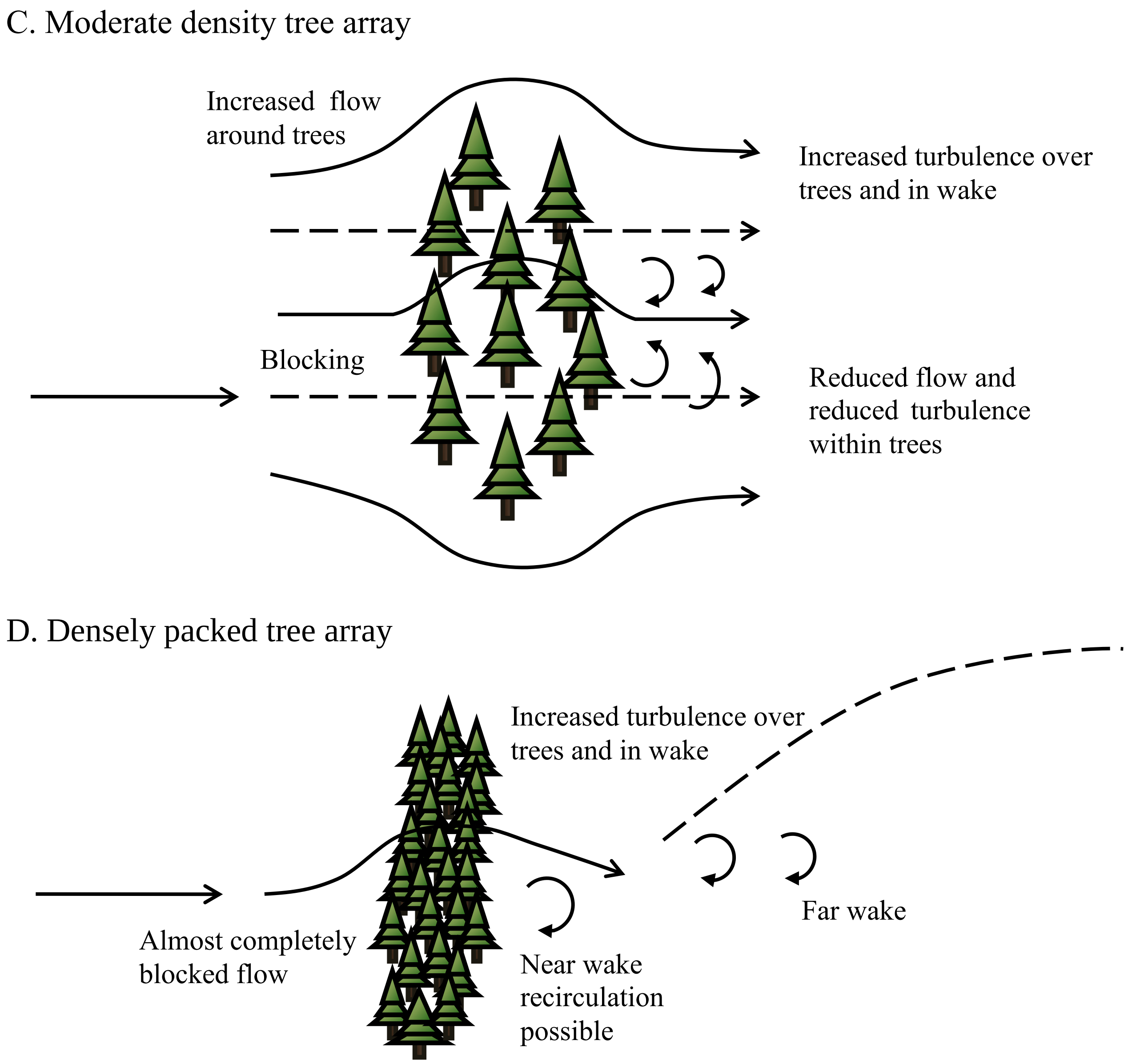


Figure 1. Schematic depiction of the impact of trees of different packing density on airflow and turbulence

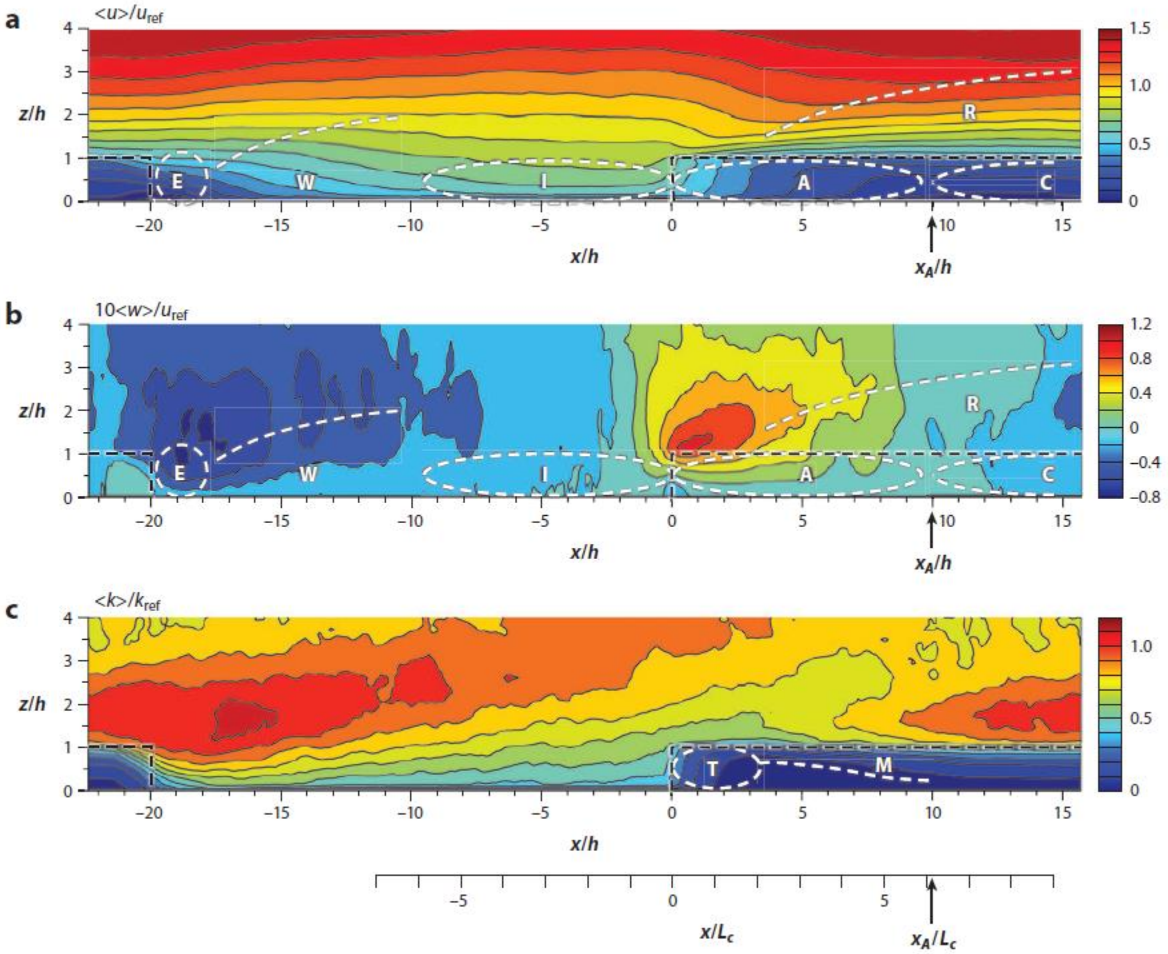


Figure 2 (From Belcher et al. 2012). Contour plots of the evolution of (a) the mean streamwise velocity, (b) mean vertical velocity, and (c) turbulent kinetic energy across a forest edge replotted from the original LES data of Dupont and Brunet (2008). The domain is periodic in the x direction. The black dashed lines mark the location of the canopy. Overlain are white dashed lines indicating a schematic of the adjustment of the flow (adapted from Belcher et al. 2003). Adjustment of the mean flow is indicated in panels a and b, and adjustment of the turbulence is indicated in panel c. Abbreviations: A, adjustment region; C, canopy flow region; E, exit region; I, impact region; M, mixing-layer region; R, roughness change region; T, turbulence impact region;W, wake region.

		Source location		
		Source upwind of trees	**Source within trees**	**Source downwind of trees/in the wake**
Density of group of trees	**Well spaced trees**	Negligible effect upwind of trees; within trees or downwind may increase dispersion	Increased dispersion due to increased turbulence	Increased dispersion due to increased turbulence
	Moderate density	Some blocking close to trees reducing dispersion upwind of trees	Decreased dispersion due to decreased flow and turbulence	For source near trees effects are complex: an increase in turbulence increases dispersion, a reduction in the mean flow reduces it; there may be down-flow behind the trees increasing ground-level impact of elevated sources. For sources further downwind an increase in turbulence increases dispersion
	Densely packed trees	Close to trees blocking reduces dispersion; main trajectory of plume over or around trees	Much decreased dispersion	For source near trees effects are complex: an increase in turbulence increases dispersion, a reduction in the mean flow and recirculating flow (near wake) reduces it; down flow behind the trees increasing ground-level impact of elevated sources. For sources further downwind an increase in turbulence increases dispersion

Table 1. Impacts of trees on dispersion. Increased/decreased dispersion results in reduced/increased pollutant concentrations for ground-level sources. For elevated sources increased/reduced dispersion may increase/decrease maximum ground level concentrations as the pollutants can be mixed more rapidly to the surface.

4 Effects of vegetation on dry deposition in urban areas

4.1 Introduction

Terrestrial surfaces are an important sink for pollutants in the boundary layer by direct deposition to the surface (dry deposition), and enhancing the deposition flux to the surface has the benefit of reducing concentrations near the ground and thus exposure of people in urban areas. As with so many aspects of urban air quality, the matter is not straightforward: the rates of deposition of gaseous and particulate pollutants on vegetation vary greatly between individual gases and for particulate matter mainly with particle size. Quite a lot is known about deposition rates in the countryside where most measurements have been made. The (micrometeorological) techniques most widely used to measure deposition of pollutants require extensive uniform areas of vegetation to quantify the vertical flux to the surface. In urban areas, vegetation is usually present in areas which are too small for micrometeorological methods of flux measurement. Urban vegetation typically in parkland or within gardens or as amenity planting along roads requires more specialised techniques for measurement, which few have attempted.

Vegetation also changes the dispersion rates of pollutants as discussed in section 3 above. This section provides a guide to current knowledge of deposition from physical and chemical principles and research to date. Where possible the focus has been to quantify the scale of reductions in concentrations in urban areas in response to planting schemes rather than provide a comprehensive review of the literature. The discussion is based on published research, and where possible for UK conurbations.

The literature on the subject of pollutant deposition is reasonably large and includes both direct measurements of rates of deposition to canopies of vegetation for a range of gaseous and particulate pollutants, many of which are described in a recent review (Fowler et al 2009). The literature also includes laboratory studies of the exchange of pollutants with plant surfaces (Freer-Smith et al 2004) and models which simulate deposition fluxes over large areas or time scales, to provide e.g. annual deposition fluxes to the UK (Smith et.al 2000).

There is a much wider literature on the benefits of vegetation for improving air quality in urban areas and many recent reviews (e.g. Janhall 2015, Salmond et al 2016, Abhijit et al 2017, Gallagher et al 2015, Berardi et al 2014). These reviews provide a useful guide to the extensive recent literature, but they are not focussed on the scale of reductions in concentrations of the major pollutants that can be achieved by policies to increase vegetation in urban areas. Furthermore, there is a general lack of the quantitative analysis required to quantify the benefits of urban vegetation for air quality, (e.g. Willis and Petrokosky 2017). It has therefore become difficult to separate the campaigning zeal for vegetation for all its acknowledged benefits from an analytical assessment of the value of vegetation to augment the dry deposition sink.

4.1.1 The deposition process

Pollutant gases and sub-micron particulate matter are transported from the atmosphere to absorbing surfaces by turbulent transfer (Figure 3), sedimentation only becomes important for particles appreciably larger than a micron in diameter. Close to surfaces, turbulence is supressed and transport of gases relies on molecular diffusion across a layer of laminar air flow to reach sites of reaction or sorption at the surface. Most of the particulate matter (by mass) is too large to diffuse efficiently through the laminar sub-layer and relies on impaction, interception and phoretic processes for transport to the surface (Garland 2001). The large flat structures of buildings are associated with substantial laminar boundary layers, so that capture of particulate matter is most efficient on those parts of the structure with the shallowest boundary layers, as is distinct on some buildings from the pattern of soiling.

One of the potential benefits of vegetation is that the finely divided structure of many leaves, especially of conifers provides both larger collecting surface per unit ground area and shallow laminar boundary layers over the leaves, especially at the edges to collect particles and reactive gases. A consideration of the leaf area available for capture of pollutants raises the question of which species are best suited to the role of pollutant deposition and whether evergreen or deciduous species are preferred. These aspects have been discussed by Grote et al (2016) and by Hewitt (2002) and includes a consideration of the emission of BVOCs, considered later in this report.

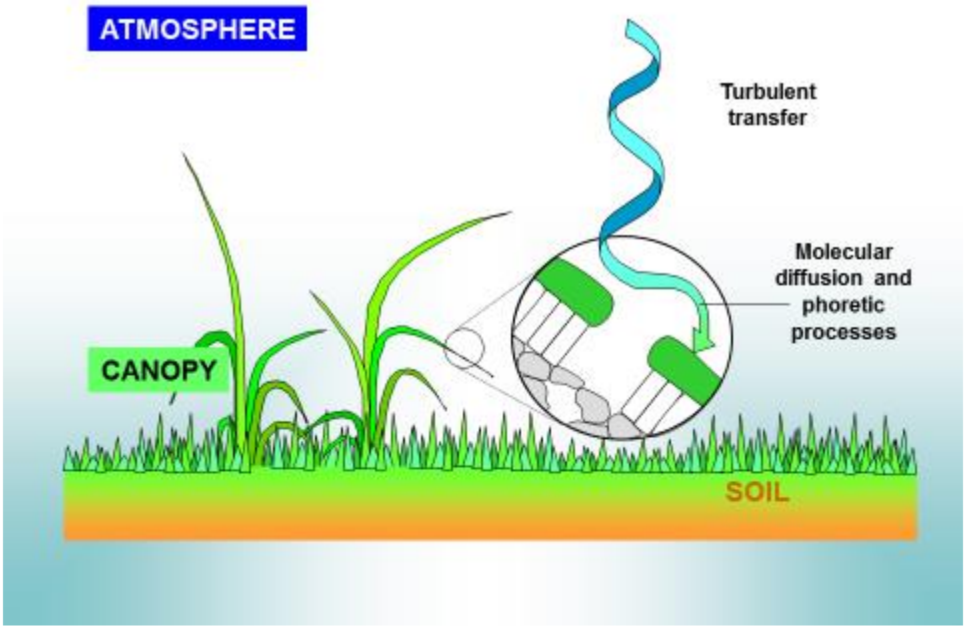


Figure 3. The deposition process for pollutant gases and particulate matter. The atmospheric transport to the surface is largely by turbulent diffusion for both gases and sub-micron particulate matter except very close to the surface where molecular diffusion transports gases to the absorbing surfaces while particulate matter relies on a combination of impaction, interception, diffusion, gravitational settling, diffusion and phoretic process, which vary greatly with particle size.

In principle, adding vegetation to an urban landscape introduces both extra surfaces for uptake and larger deposition velocities per unit area than most building surfaces. Vegetation may be added to urban areas by planting in green spaces or by covering buildings with vegetation (green walls).

4.2 Measurements of trace gas and particulate matter deposition in urban areas

The majority of published measurements of pollutant deposition to vegetation are in rural areas, where fluxes have been measured over extensive areas of uniform vegetation. A requirement of the widely applied micrometeorological methods is a uniform surface within the flux footprint (e.g. Vesala et al 2008) in which the deposition to the surface is deduced from the vertical flux measured at a height above the surface. The vertical flux needs to be constant with height and this in turn requires stationarity of the mixing ratios with time and in space along the footprint within the measurement period to avoid the effects of storage and advection contaminating the measured flux. Such conditions are hard to satisfy in urban areas, close to spatially and temporally variable sources of pollutants. In addition, in urban areas it is difficult to identify a particle metric that only undergoes deposition, these are areas in which the sources are much larger than the sinks, and most reported vertical flux measurements within urban areas to date are dominated by emissions (e.g. Dorsey et al 2002).

Large areas of uniform vegetation make it possible to deduce the effects of specific processes on the exchange processes, such as the response of changes in stomatal conductance on vertical exchange fluxes. The practical and theoretical difficulties of making and interpreting flux measurements of pollutant gases or particulate matter in urban areas prevented significant measurements in urban areas until quite recently (Nemitz et al 2008).

For the gaseous pollutants, the only measurements of urban fluxes have reported net emissions rather than deposition (Velasco et al 2009, Vaughan et al 2016). There are few mechanisms by which dry deposition rates of the major pollutant gases to vegetation would be expected to differ substantially between urban and rural areas because the main restriction to transfer is at the surface, characterised by surface resistance. The added complexity of flow regimes in urban areas will probably lead to small underestimates of deposition fluxes, especially for the very reactive gases (notably HNO_3) but for those gases relying on stomatal deposition pathways for uptake (NO_2) and for those which exhibit a cuticular resistance which is at least an order of magnitude larger than the aerodynamic resistance (e.g. for O_3) deposition velocities from the wider literature appear appropriate. In the case of HNO_3 this very reactive species appears to deposit as quickly as the diffusional processes are able to deliver molecules to the surface, and adding substantial areas of absorbing surfaces per unit land area are likely to appreciably enhance removal rates. However, HNO_3 is an exception and is a minor component of urban NO_y (NO_y = the sum of all oxidized nitrogen compounds, NO, NO_2, HONO, HNO_3 PAN…). For NO_2, deposition rates on vegetation are quite small relative to other gaseous pollutants (SO_2, O_3, NH_3), and the process is restricted to stomatal uptake, thus are smallest in winter and night periods when vegetation removes negligible amounts of NO_2 and when urban NO_2 concentrations are largest (Fowler et al 2009). Vegetation is therefore generally considered an inefficient sink for urban NO_2. This said, some studies have found that plant species differ greatly in their capacity to assimilate nitrogen from gaseous NO_2 (e.g. Morikawa et al 1998), but the study was not set up to quantify the NO_2 removal rate from the atmosphere and it cannot currently be concluded that "NO_2-phyllic" plant species provide an efficient sink, especially during the conditions of high NO_2 pollution. It nevertheless suggests that through careful selection of plant species, the benefit may be maximised. Vegetation is also a very poor sink for nitric oxide (NO), with deposition velocities substantially smaller than 1 mm s^{-1}, and soils, in particular forest soils, act as sources of NO, emission from soils at least partially offsetting any benefits of uptake by vegetation of NO_2 from the air.

Lichens and moss have been used as passive collectors of pollutants, especially for metals and to a lesser extent nitrogen compounds (Barqaqli et al 2002) and have proved useful to map the spatial pattern of metal deposition to moss and to detect hot spots. However, the method does not quantify the overall deposition to other surfaces (taller vegetation, soil, etc.), because moss accumulates pollutants at a different rate than the vegetation canopy as a whole, and it requires calibration with other methods to provide deposition estimates to the landscape.

Measurements of aerosol fluxes over urban areas by eddy covariance methods have been used to quantify aerosol fluxes (Dorsey et al 2002, Nemitz et al 2008, Zalakeviciute et al 2012, Deventer et al 2015). These studies show that urban areas are primarily a net source, rather than a sink for particulate matter.

Measurements above closed urban forest canopies are rare. Particle deposition on a 'peri-urban' forest has been measured by eddy covariance by Fares et al (2016) who showed deposition velocities up to 10 cm s^{-1} in the 3 hours centred on mid-day, but much smaller values at other times of day. The measurements were made at a site 25 km from Rome with the flux footprint being entirely within an area of vegetation dominated by holm oak. Thus these measurements are not really in an urban area, and so do not reveal the effects of vegetation within an urban environment on city scale deposition rates.

Measurements of particle deposition have also been made using radioactive tracer methods by Graunstein and Turekian (1989) in which the inventory of ^{210}Pb (half-life 22.3 years) in surface soil horizons of organic matter is used to deduce the long-term average flux to the surface. The ^{210}Pb in the atmosphere arises from the radioactive decay of Radon (^{222}Rn) emitted from soil as a gas. In the atmosphere, the ^{222}Rn decays through a series of very short lived daughters to ^{210}Pb and is transformed from a gas to a particle, which in turn attaches within a minute or so to existing particulate matter (Chamberlain, 1991). Thus the ^{210}Pb effectively tags particulate matter in the atmosphere and also in the soil. The inventory of ^{210}Pb in soil when in equilibrium with atmospheric input provides a measure of the long-term average flux. As the half-life is 22.3 years, the soil needs to be undisturbed and land use constant for 50 years or more to bring the system to equilibrium. However, the benefit of the method is that it integrates the deposition events over many decades, and provides a long-term average deposition rate for the site.

The method was applied by Fowler et al (2004) at a range of sites in the West Midlands conurbation, measuring rates of particle deposition onto grassland and small pockets of mixed woodland close to urban development in Moseley, Edgbaston and Sutton Park.

The sites were chosen to span a wide range of locations in the West Midlands urban area, but the requirement for sites with undisturbed soils for 50 years or more is quite restricting in an urban area. The results, shown in Table 2, show the enhancement in deposition of particulate matter in woodland, relative to grassland. Clearly, the large height and aerodynamically rough surfaces of woodland capture particulate matter by dry deposition at approximately three times the rate of shorter grass surfaces. Further, these measurements provide deposition velocities that can be used in models to simulate effects of additional woodland within an urban area on deposition.

Site	Sutton Park	Edgbaston	Moseley	Average
Grass				
Dry Deposition ($Bqm^{-2}a$-1)	24	21	17	
Deposition velocity mm s^{-1}	3.8	3.3	2.8	**3.3**
Woodland				
Dry Deposition ($Bqm^{-2}a$-1)	45	60	67	
Deposition velocity mm s^{-1}	7	9.4	10.7	**9**

Table 2. Dry deposition of deposition velocities of ^{210}Pb aerosols on grassland and woodland in the West Midlands conurbation (from Fowler et al 2004)

4.3 Modelling effects of trees in urban areas

The evidence so far suggests that planting more trees in an urban area will increase deposition rates of particulate matter. The next step is to quantify the scale of the reduction in ambient concentrations for a specified increase in tree cover in UK conurbations. Such exercises have been explored using dispersion models over UK cities by McDonald et al (2007), and by Bealey et al (2006) and for a more complex treatment of vegetation covering building surfaces (green walls) within urban street canyons by Pugh et al (2012) and by Jeanjean et al (2017).

The approach by McDonald et al was to use a multi-layer trajectory model to simulate the emission, transport and deposition of particulate matter in two large conurbations, the West Midlands and Glasgow. For both conurbations detailed land use information was available, and in the West Midlands a separate study identified the species composition and location of the urban tree population and the locations for new planting throughout the urban area (Donovan 2003). The deposition parameters were taken from measurements over moorland and grassland (Nemitz et al 2002) and closed forest (Gallagher et al 1997) and the scenarios simulated included removing all existing trees, and planting 25%, 50% 75% and 100% of the future planting potential throughout the area.

Modelled concentration and deposition changes due to tree planting for the West Midlands

	Concentration		Deposition	
	Average µg m^{-3} Primary PM_{10}	% change of Primary PM_{10}	Primary PM_{10} tonnes	% change
Status Quo	2.3	n/a	575	n/a
No trees	2.4	4	536	-7
FPP_{25}	2.1	-10	685	19
FPP_{50}	1.9	-17	747	30
FPP_{75}	1.8	-22	773	34
FPP_{100}	1.7	-26	774	35

Table 3: Modelled concentration and deposition changes for the West Midlands as a consequence of tree planting in 6 scenarios, (From McDonald et al 2007). (FPP refers to the Future Planting Potential and the subscript 100 means that ALL open space not already covered by hard surfaces is planted)

The results (Table 3) for the West Midlands show that current tree cover is responsible for removal of 7% of the primary particulate deposition and reduces the concentration by 4%. By planting 100% of the potential maximum, the concentrations of primary PM_{10} are reduced by 26%. It must be appreciated that planting all available space in a city is not practical or desirable, and even the least ambitious planting scheme (FPP_{25} meaning 25% of the area available for planting is used), would include planting important amenity areas (gardens, parkland), and would reduce concentrations of primary PM_{10} by 10%. The exercise does however quantify the scale of the effect of vegetation on concentration and deposition of the primary emissions within the conurbation.

Modelled concentration and deposition changes due to tree planting for Glasgow

	Concentration		Deposition	
	Average µg m^{-3} Primary PM_{10}	% change of Primary PM_{10}	Primary PM_{10} tonnes	% change
Status Quo	1.26	n/a	72	n/a
No trees	1.30	3	67	-7
FPP_{25}	1.23	-2	76	6
FPP_{50}	1.21	-4	80	11
FPP_{75}	1.19	-6	83	14
FPP_{100}	1.17	-7	85	18

Table 4: Modelled concentration and deposition changes for Glasgow (From McDonald et al 2007)

For Glasgow, the model shows that current tree cover removes 3% of the primary PM_{10} and that by planting all available areas of the city the average primary PM_{10} concentration would be reduced by 7%. (Table 4). Again taking a more realistic planting scenario (25% of the maximum) reductions in the primary PM_{10} would be 2%.

Because deposition rates were taken from woodlands, the results reflect the effect of introducing closed urban woodlands into the urban matrix, rather than small groups of trees or individual trees. More detailed modelling of the interactions within the street canyons would be required to quantify the effect of the latter, which is computationally expensive and has been applied to relatively small areas to date. Using a street canyon focussed study with or without green-walls Pugh et al (2012) showed that the vegetation could appreciably reduce local concentrations.

These modelling studies quantify the effect of tree planting on primary PM_{10}, but the nature of the modelling did not allow the larger scale sources of PM_{10} outside the city to be fully quantified. Furthermore, the sources of PM_{10} from outside the urban area, and including long range transported material from continental sources frequently contribute a large fraction of the PM_{10} in urban areas (e.g. Keuken et al 2013). The deposition processes operate in exactly the same way, but the atmospheric boundary layer, which in daytime conditions is often 1000m or more deep is not as efficiently scavenged by deposition to the surface as the primary urban emissions. Thus the values presented for the efficiency of vegetation in removing PM_{10} are overestimates. Further, more detailed modelling is required to provide the extent of the overestimation. A modelling study of the effectiveness of urban vegetation in reducing $PM_{2.5}$ concentrations by Jeanjean et al (2017), using CFD approaches over a 2km x 2km area of central Leicester showed that the dispersive effect of trees reduced $PM_{2.5}$

concentrations by 9% while dry deposition to the trees reduced concentrations by 2.8%. Thus modelling and measurement approaches provide broadly consistent reductions in concentrations of particulate matter. Consideration of local ‘micro-scale’ interventions, using planters or living walls on local exposure have not been studied in detail but the same principles apply and are unlikely to offer significant benefit except very close to the absorbing surface in calm conditions. Given the scale at which planting is practical, the scale of mitigation of exposure to PM_{10} (or $PM_{2.5}$) in urban areas is unlikely to exceed a few percent.

While the use of the naturally occurring radioactive tracer is valuable in quantifying the additional dry deposition of particulate matter by tree canopies, there is additional evidence of the effect of the trees in capturing particulate matter from additional measurements at the West Midlands sampling locations noted above. Atmospheric lead in particulate matter from vehicle emissions in the pre lead-free petrol days and a range of other metal processing industries has accumulated in organic matter in soils of most cities, and the additional filtering by trees leads to the accumulation of large concentrations in the surface soils in these areas. The particle deposition studies in the West Midlands reported above revealed concentrations of Pb in surface soil in the range 100ppm to 400 ppm (Table 5). The proximity of the largest Pb values to major roads strongly suggests traffic as the main source. The concentrations of other heavy metals were also enhanced in soils beneath woodland at the measurement sites (Table 5).

Heavy metal concentrations in soils of the West Midlands conurbation

Location	Cd	Cu	Ni	Pb	V	Zn
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Recommended upper limit	3	135	75	250		300
Sutton Park Pageant area, 0–5 cm	0.53	53	14	100	11	110
Sutton Park Pageant area, 5–15 cm	0.12	33	5	80	16	22
Sutton Park Pageant area, 10–20 cm	0.064	17	1.4	28	13	7.6
Westwood Coppice edge 4, 0–5 cm	2.1	150	28	400	25	410
Westwood Coppice edge 4, 5–10 cm	0.92	140	19	330	30	140
Westwood Coppice edge 4, 10–20 cm	0.26	33	4.3	91	14	38
Westwood Coppice middle 5, 0–5 cm	0.59	150	21	29	29	110
Westwood Coppice middle 5, 5–10 cm	1.2	88	22	14	14	180
Westwood Coppice middle 5, 10–20 cm	0.084	9.2	1.5	13	13	10
Moseley Golf Course, 0–5 cm	0.61	26	14	62	14	130
Moseley Golf Course, 5–15 cm	0.34	44	21	100	36	100
Moseley Golf Course, 10–20 cm	0.39	39	22	84	35	110
Moseley Wood, 0–5 cm	0.58	63	11	120	13	120
Moseley Wood, 5–10 cm	0.43	60	13	160	21	79
Moseley Wood, 10–20 cm	0.33	86	14	200	32	68
Edgbaston Golf Course, 0–5 cm	0.6	130	26	220	33	150
Edgbaston Golf Course, 5–10 cm	0.2	89	13	140	37	48
Edgbaston Golf Course, 10–20 cm	0.11	38	8.7	99	24	37
Edgbaston SSSI, 0–5 cm	2.6	160	31	210	34	480
Edgbaston SSSI, 5–10 cm	2.5	200	35	240	43	480
Edgbaston SSSI, 10–20 cm	1.4	120	23	160	36	270

Table 5: Heavy metal concentrations in soils of the West Midlands (From Fowler et al 2004)

5 Biogenic volatile organic compound emission from urban tree planting

5.1 Introduction

Natural or biogenic emissions of volatile organic compounds (BVOCs) from vegetation are the dominant global source of reactive carbon in the atmosphere (Goldstein and Galbally, 2007). The most abundant BVOC is isoprene, but many hundreds of other VOCs are also emitted by vegetation (Guenther *et al*. 1995). Other classes of BVOCs include monoterpenes, (C_{10} hydrocarbons, those with 10 carbon atoms in a molecule), sesquiterpenes (C_{15}) and di-terpenes (C_{20}), green leaf volatiles such as hexanol and hexenel, and small oxygenated compounds including acetone and methanol. In general terms the atmospheric reactivity of biogenic hydrocarbons is higher than hydrocarbons emitted from fossil fuel combustion processes (Carter, 1994).

The atmospheric oxidation of BVOCs, when in the presence of nitrogen oxides, leads to the formation of ozone. The secondary organic oxidation products formed on oxidation can also form new particles or condense to existing PM. The potential urban air quality impacts of BVOCs have been known for many years, see for example estimates of the BVOC contribution to ozone in US cities by Rasmussen (1972), Chameides *et al.* (1988) and Geron *et al.* (1995).

5.2 Considerations for urban tree planting

Any practical policy interventions that substantially increase the vegetative cover in an urban environment should pay due attention to the potential effects that may arise from changes in emission of BVOCs and the consequential impacts on ozone and PM arising. BVOCs participate in the same atmospheric chemical degradation processes as anthropogenic VOCs, reacting primarily with OH but also O_3, and with NO_3 during the night. Their rates of reaction with these oxidants are often faster than comparable carbon number fossil fuel aliphatic hydrocarbons, a consequence of weak and accessible hydrogen double bonds in BVOCs. In turn their atmosphere lifetimes are often therefore shorter than anthropogenic hydrocarbons. In the case of some sesiquiterpenes and diterpenes (C_{15} and C_{20} structures) reaction with O_3 is so fast that their lifetimes in the atmosphere can be as short as a few minutes or less. The most abundance biogenic VOC isoprene has a daylight lifetime in summer of around 20-30 minutes. When considering impacts of tree planting and changes to VOCs it is important therefore to consider not only any changes to the total *mass* of VOC that the additional trees may release but the atmospheric *reactivity* of those VOCs. In general terms higher reactivity VOCs have the potential to generate higher concentrations of secondary pollutants closer to the point of emission.

Even given the reactivity of many BVOCs the formation of ozone or generation of secondary aerosols is not instantaneous, and the majority of effects from additional BVOC emissions would occur some distance downwind, with regional increases in air pollution rather than substantial BVOC-induced changes in urban centres themselves (see for example Mackenzie *et al.* 1991). These spatial effects of urban emissions of BVOCs are illustrated by Nowak *et al*. (2000), who modelled that changing tree coverage in cities (from 20 to 40%) led to modest decreases in urban ozone (~1 ppb) and increases (0.26 ppb) in the wider regional domain.

The emission rate of BVOCs from trees and plants increases with ambient temperature, or both temperature and photosynthetically active radiation (PAR) (Geron *et al.* 1994), with highest BVOC emission rates generally associated with trees in warm climates. Emission rates, and the response to temperature and PAR vary greatly, even between trees of very similar species. “Wounding events”, such as grass cutting and crop harvesting, also give rise to BVOC emissions (including aldehydes, ketones and alcohols). Such sources can be relevant for rural ozone formation, but are unlikely to be large enough to make a significant contribution to a city centre environment. In general terms BVOC emissions from most plant species are low below 20 °C, with peak emission rates at around 35 °C before plateauing and then declining above 40 °C. Previous literature has focused in particular on BVOC - air quality interactions in US cities such as Houston, Los Angeles and Atlanta and also in the Mediterranean, locations with high mean summertime air temperatures and substantial regional vegetation.

The temperate nature of the UK climate has meant that historically BVOCs have not been considered a substantial contributor to overall VOC emissions or consequential ozone production. Owen *et al.* (2003), made an estimate of BVOC emissions for various landscape types in the West Midlands, apportioning land-cover to trees and other vegetation and then assigning emissions based on known species type. The conclusion was that under conditions of the study period and region, BVOCs were a very minor contributor (< 1%) to overall UK VOC emissions, when considered on an annual basis. It is worth noting that since that study anthropogenic VOCs have declined further and the fractional contribution from BVOC is likely to have increased.

Stewart *et al*. (2003) generated the first UK BVOC emissions inventory. This required the assignment of BVOC emission rates and response curves to more than 1,000 plant species, combined with highly detailed and spatially resolved vegetation cover data for the UK. Results from this study concluded that Sitka spruce species were the dominant source of BVOCs in the UK as a whole, with emissions arising predominately from coniferous areas in Northern England and Scotland. Stewart *et al.* also identified poplar (*Populus* spp.) as a notable isoprene source in eastern England. The peak in both isoprene and monoterpene emissions was estimated to be in the summer months in this inventory based on established temperature and PAR relationships.

The current influence of BVOCs on the UK atmosphere is not straightforward to discern directly since there are very few observations that allow for a quantification of BVOC abundances or emissions. The Defra Automatic non-methane hydrocarbon network has collected a long time-series of VOCs that includes isoprene, but the more extensive diffusion tube network does not include any BVOC measurements. In many urban locations the majority of urban isoprene can be attributed to fuel and combustion sources, evident through the close correlations with other combustion tracers such as 1,3-butadiene. However in the summer months a second uncorrelated temperature dependent source of isoprene has been detectable in the UK, becoming most obvious on the warmest days. This has been strongly indicative of an existing vegetative isoprene source in most UK cities (von Schneidemesser *et al.* 2011). Fluxes, as well as concentrations, of isoprene were measured in central London by Valach *et al.* (2015) between August and December 2012. In August and September, measured isoprene fluxes correlated strongly with temperature and PAR, and the magnitude of the fluxes agreed well with model predictions of the urban biogenic emissions.

The scale of possible impacts of BVOCs on UK ozone were studied in some detail following the warm summer of 2003, a year which is often taken as a proxy for potential summertime conditions in future climate scenarios with increased anticyclonic blocking. 2003 had extended multi-day periods where the UK urban environment was more similar in climate to BVOC-air quality conditions from North America, with daytime ozone reaching mixing ratio of 100 ppb and air temperature >30 °C. Lee *et al.* (2006) reported isoprene concentrations of > 2 µg m^{-3} in South East England during the hottest days of August 2003, more than 10 times the typical mean UK summertime values. Vieno *et al*. (2010) reported a modeling study of the same event which showed that model performance simulating UK BVOCs was in general good, but with increasing underestimates of BVOC emissions at higher temperatures, and up to a factor of 5 too low on the hottest day studied.

There are very few atmospheric observations of monoterpenes in a UK urban context. Dunmore *et al.* (2015) observed a constant background of around 20 ppt α-pinene in central London during UK winter, but a temperature-dependent diurnal profile during summertime, with peak daytime values of around 150 ppt. During the wintertime BVOCs contributed around 1% of the total primary OH reactivity to organic compounds (around 0.025 s^{-1}); this value increased to around 10% of organic-OH reactivity during the summertime (~ 0.5 s^{-1}, out of a total ~5 s^{-1}) in central London at the North Kensington AURN site.

There is evidence therefore to show that BVOCs from trees are already present in the UK urban environment, and most abundant on the warmest days in summer. Their contribution to the urban OH reactivity is however currently very small compared to the effects from anthropogenic VOCs. Vieno *et al.* (2010) varied ambient temperature by up to 5°C as a means to drive changes in BVOC across the UK in the EMEP air quality model. This found incremental regional ozone changes no greater than 10 ppb, and with the majority of ozone deriving from trans-boundary sources (although that trans-boundary ozone had a continental BVOC contribution). Donovan *et al.* (2005) reported regional-scale simulations of air quality changes under a range of different vegetation and climatic scenarios in the UK. Under the most extreme scenarios where there were substantial urban and regional increases in the number of high BVOC emitting trees in the UK, coupled with an average 2°C temperature increase, incremental increases in regional ozone of 6% were simulated. Where low BVOC-emitting trees were planted instead, the simulation showed reductions of the order 1-2% in ozone, even under +2 °C temperature scenarios.

Whilst the potential impacts on regional ozone from an increase in the density and number of trees in cities appears in principle small, the size and sign of the effect is clearly dependent on the type of vegetation planted. The emissions of BVOCs from different species vary over several orders of magnitude, and indeed many plant types have no measureable BVOC emissions at all. For example, perennial rye grass is the most abundant vegetation type by peak biomass in the UK and it has no measureable BVOC emissions (Hewitt and Street 1992). There is huge reported variation in emissions from different tree types. For example oak is estimated to have approximately ten times the isoprene emission potential of Sitka spruce, when expressed as kg BVOC emissions per kg dry weight of biomass per unit time. If all other environmental factors are unchanged (e.g. temperature, rainfall, CO_2 etc.) then

BVOC emissions can to some degree be controlled by manipulation of land-cover and vegetation type through the selection of low emitting species.

Benjamin and Winer (1998) made a comprehensive assessment of the ozone impacts arising from different tree species in California, covering more than 300 different varieties. Whilst many of these species are unlikely to be relevant for tree planting in a UK context, it is illustrative of the potential range of impacts that different tree selections might have. Using local atmospheric conditions and literature BVOC emission values Benjamin and Winer estimated a mass formation of ozone per tree per day. The highest potential was from oil palm (*Elaeis guineesis*) at more than 400 g_{ozone} $tree^{-1}$ day^{-1}, followed by weeping willow (134 g $tree^{-1}$ day^{-1} and coast live oak (114 g $tree^{-1}$ day^{-1}). In broad terms palms, willows, eucalypts, gums and oaks all had high ozone forming potential from their BVOC emissions. In contrast almost 100 tree species were estimated to have no ozone forming potential since they had no measurable BVOC emissions (for example species such as juniper, myrtle, hickory and walnut). The potential ozone changes induced by these low-emitting species would likely then be net negative through ozone deposition and removal, although this wasn't quantified in the study. The study does highlight that there are far more variety of tree that are low BVOC-emitting than high.

Donovan *et al.* (2005) made an assessment of the impacts on air quality arising from changing the tree coverage in the West Midlands and in different temperature scenarios. Although not explicitly quantifying the relative BVOC emissions potential from each species, it can be inferred from the resultant ozone changes calculated for a particular tree type scenario. From a BVOC emissions perspective English oak, white willow, aspen, sessile oak, red oak and goat willow were modelled as having the highest emissions, with an increase in ozone detectable in the regional domain (range +0.8 to + 2.9%), whilst planting species such as Austrian pine, larch, silver birch and maple led to small (0.3 – 0.8%) reductions in regional ozone, since BVOC emissions were insignificant and the trees act as a surface for enhanced deposition.

6 Summary

In summarising the effects of urban vegetation on ambient concentrations of particulate matter and gaseous pollutants, there are potential benefits of vegetation in changing dispersion and deposition processes and also potential problems. For dispersion, locally (tens to hundreds of square metres) the planting of trees may enhance or reduce dispersion; this redistributes pollution but does not remove it. Where vegetation acts as a barrier close to a source, concentrations immediately behind the barrier owing to that source are reduced typically by a factor of about 2 relative to those which would occur without the barrier, whereas on the source side of the barrier concentrations are increased.

Effects of vegetation removing pollutants from urban air by deposition, and thereby reducing concentrations and population exposure to particulate matter have been demonstrated in field measurements and using models. However, the magnitude of the reduction in concentration by realistic planting schemes, using trees, is small and in the range 2% to 10% for primary PM_{10} and ambitious plantings. For practical planting schemes and PM from all sources, the scale of reductions is expected to be no more than a few percent. For NO_2, vegetation is not a very efficient sink, and as the deposition occurs in daytime, and primarily in the warmer months, there is little benefit for air quality for most of the time that NO_2 is a problem.

BVOC are already present in small amounts in UK cities from existing vegetation emissions, and are highest during warm summer weather. At present regional BVOC emissions from the UK make only very minor contributions to ambient ozone, and the specific contribution from city centre vegetation is too small to be isolated in modelling studies. Increasing tree cover in cities has the potential to increase BVOC emissions, with impacts felt through small increases in ozone and possibly aerosols downwind. The reactivity of BVOC emissions can be higher than similar carbon number fossil fuel derived VOCs and this reactivity should be considered (and minimised) along with any potential change in total mass of VOC emissions. Of potential relevance to UK planting, oak, aspen and willow species should be avoided since these are estimated to being highest BVOC emitting species. The potential ozone increase from additional urban tree planting appears entirely avoidable however through selection of low BVOC emitting species, of which many varieties are reported in literature.

7 Valuing the benefits of vegetation as a sink for air pollutants

The UK office of National Statistics has in recent years estimated the value of natural capital as a way of measuring economic progress. Within this exercise the asset value of UK woodlands for the filtration of atmospheric particulate matter (PM_{10}) and SO_2 was for the first time included in ONS-Defra statistics and valued at £4.5 billion in 2012, creating a total asset value of £114 billion (AECOM, 2015). The exercise was useful in recognising an important property of vegetation. However, the very simplistic approach used to value this ecosystem service was subject to considerable uncertainty.

A follow-up study (Jones et al., 2017) developed a more sophisticated approach for valuing the service, based on a Chemistry and Transport Model (CTM), now comparing the effect of deposition to vegetation to that of bare soil. This approach takes account of chemical interactions, the interaction with wet deposition, and enables a more direct impact valuation based on human exposure rather than on the amount removed. It also extended the approach to additional pollutants ($PM_{2.5}$, NH_3, NO_2, O_3, $PM_{2.5}$). Assessments were carried out for 2007, 2011, 2015 and, based on emission projections, with 2015 meteorology, for 2030.

It should be noted that overall, vegetation was estimated to have a detrimental net effect on NO_2 concentrations. The prime reason is likely that vegetated soils present a larger source of soil nitrogen oxides (as NO) than bare (desert) soil. There might have been other effects which were not quantified in isolation, such as interactions with changes in biogenic volatile organic compounds and O_3.

Overall, UK vegetation is estimated to remove 1,354 ktonnes of $PM_{2.5}$, SO_2, NO_2 and O_3, with an annual value of £1.00 billion (2015, at 2012 prices), presenting an asset value of £34.5 billion (2015, incl. income uplift). The study also developed initial, more detailed accounts, for urban areas.

The study suggested that for 2015, the total existing UK vegetation reduces the average annual surface concentration by about 10% for $PM_{2.5}$, 6% for PM_{10}, 13% for O_3, 24% for NH_3 and 30% for SO_2, but did not markedly change NO_2 concentrations. Woodland dominated the removal of PM, whilst agricultural land (accounting for 4.3 times as much land area), dominated the removal of gaseous pollutants.

Whilst the overall amount of pollutant removed was broadly consistent with an alternative approach, i-tree Eco London study (Rogers et al., 2015) that assessed the value of trees in London only, but the split across pollutants differed significantly. This is partly due to difficulties in comparing a national with a much more regional estimate, which is dominated by a different land cover. It also reflects differences in how deposition is calculated in the two studies, each approach with some advantages and disadvantages. The i-tree study used a more detailed tree cover database which is not available for the whole of the UK and includes parameterisations to deal with smaller woodland features and single trees. By contrast, it fails to reproduce the feedback of pollutant removals on concentrations downwind in a way a CTM can. There are also important uncertainties in the parameterisations of deposition between CTMs. For example, in a European study Flechard et al. (2011)

demonstrated that deposition estimates of nitrogen compounds can vary by a factor of 3 for NH_3, 7 for NO_2 and 10 for aerosol, depending on the approach used.

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