Natural capital accounting of the air quality regulating service by vegetation in the UK

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Abstract

Urban areas are particular hotspots receiving primary and secondary pollutants from rural areas, as well as locally originated pollutants from industry and traffic. Furthermore, high population densities lead to the greatest exposure of the population to air pollution in urban areas. Vegetation, and trees in particular, provide an ecosystem service by removing some of this pollution from the air, thereby reducing exposure of the population and leading to health benefits in terms of avoided mortality and morbidity. However, the magnitude of these benefits is contested. While a number of studies have quantified the amount of pollution removal by trees most use a static approach, assessing pollutant removal and the resulting benefit in situ and for pollutants in isolation. These approaches usually do not take account of the dynamic interactions among pollutants and meteorology, which govern spatial patterns of pollutant removal, or pollutant transport, which determines where the benefits are received. In this study, we apply an atmospheric transport model under two scenarios to calculate the pollution removed by vegetation at a UK scale. We present and discuss initial results on quantities of pollution removed and the resulting change in pollutant concentrations (i.e. pollutant exposure) in the context of other studies.

Keywords

Air pollution, Dispersion modelling, Urban trees

Introduction

Air pollution is increasingly recognised as a global hazard with severe impacts on human health and the environment. In the UK, poor air quality is estimated to result in 40,000 equivalent attributable deaths every year and is a major cause of morbidity (RCP, 2016).

In Western Europe, some atmospheric pollutants have reduced in concentration over the last few decades (such as sulphur dioxide and nitrogen dioxide) due to emission controls on point sources (e.g. large power plants, industry), and vehicle exhausts via catalytic converters. However, the concentrations of other pollutants, especially particulate matter (PM_{10} and $PM_{2.5}$) remain high. In rural areas, ammonia emissions primarily from agriculture, which are a substantial precursor to aerosol formation contributing to $PM_{2.5,}$ remain relatively unchanged. Ammonia reacts in the atmosphere with other gases like NO_x and SO_2 to form ammonium sulphates and nitrates (Bauer et al. 2016). These secondary inorganic aerosols often represent 10–20% of fine particle mass in densely populated areas in Europe (Brunekreef et al, 2015). Other secondary pollutants such as ozone are also a concern due to long-range hemispheric transport of precursor chemicals. They are therefore harder to control by local measures (Maas and Grennfelt, 2016; Maione et al., 2016).

Urban areas form a hot-spot of pollution exposure: i) they receive secondary pollutants blown in from rural areas; ii) the high traffic volumes and stop-start nature of urban driving mean that there is also substantial local generation of air pollution; iii) pollutant concentrations in the air at ground-level are often poorly dispersed due to restricted air movement in many urban locations; perhaps most importantly, iv) the high population densities in urban areas result in the greatest exposure of the general public to elevated levels of air pollution.

Vegetation can play a role in reducing air pollutant concentrations (UKNEA, 2011) and there is considerable interest in the role of urban greenspace in reducing exposure in urban areas. Trees along with other vegetation can provide this air quality regulating service by capturing airborne pollutants and removing them from the atmosphere through two main mechanisms: stomatal uptake and dry gaseous deposition to leaf surfaces (Bignal et al., 2004).

Trees are particularly effective scavengers of air pollutants due to their large surface area (Beckett et al 2000, Nowak 2000) and high roughness length (and lower aerodynamic resistance R_a), which aids mechanical turbulence and promotes dry deposition to the surface. The deposition velocity is the rate at which a compound deposits to a leaf surface and incorporates uptake of pollutants into the plant leaves via stomatal openings, as well as direct deposition to the leaf surface. It is dependent on plant characteristics such as the number of stomata and the area and chemistry of leaf surfaces. Dry deposition rates to trees exceed those to grassland by typically a factor of 3–20 (Gallagher et al., 2002, Fowler et al., 2004).

Several previous studies have shown the effectiveness of trees in capturing pollutants (e.g. $PM_{10,\ 2.5}$) in relation to improving urban air quality. For example Nowak et al. (2013) modelled $PM_{2.5}$ removal by trees and associated health effects in ten US cities. McDonald et al., (2007) modelled the potential of urban tree planting to mitigate PM_{10} across two UK conurbations. Nowak et al. (2006), used meteorological and air pollution data to calculate the removal of O_3, PM_{10}, NO_2, SO_2, CO by urban trees and shrubs across the United States. Some studies have looked at the suitability and pollutant capture efficiency of particular trees. For example, Becket et al., 2001 showed in wind tunnel experiments that coniferous species, and broadleaf trees with hairy leaves, had a greater effectiveness at capturing particles than other broadleaf trees.

The simplest approaches to calculate this ecosystem service tend to use an annual average concentration field of a pollutant overlaid on a land-cover grid. The pollutant removed within each grid cell is calculated as a function of the deposition velocity appropriate to the vegetation type, which may be adjusted for seasonality effects or regional climatology (e.g. Powe & Willis 2004).However, these simpler approaches fail to capture a number of key processes and interactions. Firstly, this approach considers pollutants in isolation and does not consider interactions between pollutants and their chemical transformations. These interactions can substantially alter deposition velocities of other pollutants and, therefore, their concentrations. For example, chemical interactions between ammonia and sulphur aerosols mean that changes in sulphur concentrations affect the temporal and spatial deposition of ammonia. There are also complex photochemical reactions between ozone, NO and NO_2.

In addition, some plant species can provide a dis-service by releasing Non-Methane Volatile Organic Compounds (NMVOCs) which are a precursor for ozone pollution and for formation of secondary organic aerosols (Maas and Grennfelt, 2016). Together with secondary inorganic aerosols, these organic aerosols contribute roughly half of roadside $PM_{2.5}$, and up to 90% of rural $PM_{2.5}$. Thus, the emissions of organic compounds by vegetation can also influence PM concentrations. A second issue is the dynamic interaction between air pollution and meteorology. Deposition velocities can be highly dependent on climate factors, but this differs by pollutant. Much of the literature on PM focuses on the number of dry days per year as the sole meteorological controlling factor, since pollution removal is not mediated by vegetation on wet days. However, wind-speed plays a far greater role in dry deposition of some pollutants such as ammonia (Asman, 1998). Lastly, there is often a spatial disconnect between where air pollution is generated, where the ecosystem service of pollution removal by vegetation occurs and where the benefit in terms of reduced exposure of people to pollution is realised (Figure 1). A model that does not consider atmospheric transport would estimate the air pollutant absorption (tonnes) of vegetation in Area A (Figure 1) to be minimal despite its high levels of air pollution, because there is limited vegetation there. Additionally, it would show Area B to have high levels of air pollution absorption due to substantial vegetation, but low economic benefit because the local population is small. Finally, it would show Area C to have low pollution removal but higher benefit despite the low quantity of vegetation within the area, because the benefitting population is large. In reality, all of these locations are connected through atmospheric transport of pollutants. As a consequence, failure to account for atmospheric transport and for the chemical interactions among pollutants and meteorology can lead to under- or over-estimates of the service provided by vegetation, depending on the local context.

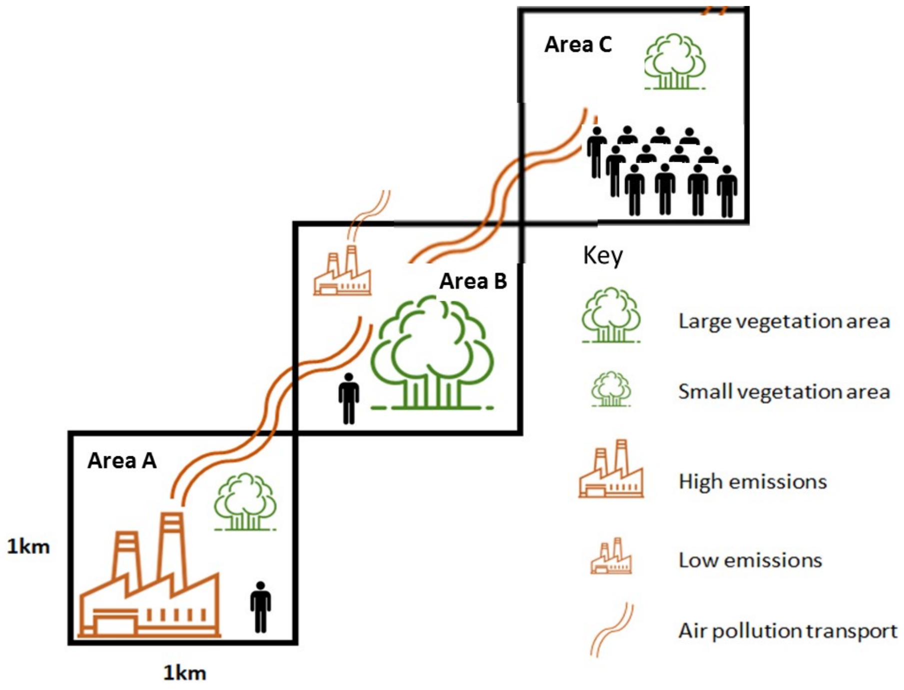

Figure 1. Spatial disconnects between locations of pollution emission (Area A), pollution removal (Area B) and receiving the benefit (Area C).

This study demonstrates the use of an atmospheric transport model to calculate the quantity of air pollution removed by vegetation, at a UK scale. It also calculates the resulting change in concentrations, which will lead to health benefits through reduced exposure to poor quality air.

Methods

The EMEP4UK atmospheric chemistry transport model

EMEP4UK incorporates aspects of chemical transport and transformation, and dynamic interactions with meteorology and land cover at an hourly time step. It is capable of representing UK atmospheric composition in greater detail than larger i.e. European-scale models, with the ability to simulate hourly air pollution interactions over decadal time scales using a 5km grid or finer, down to 2km. The Weather Research Forecast (WRF) model is used as the main meteorological driver.

The current operational version of EMEP4UK is rv4.4 (Vieno et al. 2016), based on the EMEP MSC-W rv4.4 (Simpson et al. 2011). This model is used to support European policy development by the UNECE Convention on Long-range Transboundary Air Pollution (CRLTAP) and the European Commission. The model core code is open source and available for download from the EMEP website. The EMEP4UK is thus an ideal tool to analyse the impact of policies in the UK, with the benefit of higher resolution which is critical to account for the spatial allocation of wet deposition.

EMEP4UK simulates hourly to annual average atmospheric composition and deposition of various pollutants; including PM_{10}, PM_{fine} (broadly equivalent to $PM_{2.5}$), secondary organic aerosols (SOA), elemental carbon (EC), secondary inorganic aerosols (SIA), SO_2, NH_3, NO_x, and O_3. Both dry and wet deposition of pollutants are calculated. In the model, PM_{fine} concentrations from both primary and secondary sources are calculated based on primary industrial and agricultural emissions of precursor compounds within the UK, import of precursors from abroad via hemispheric transport as well as VOC emissions from vegetation and other sources. More information, including extensive information on validation and model performance, can be found on the EMEP4UK website.

Scenarios

To calculate quantities of pollutant removed by vegetation in a complex transport model, the effect of vegetation must be isolated through scenario analysis. This creates alternative scenarios with different vegetation cover, while keeping pollutant emissions, meteorology and all other factors constant. To assess the amount of pollution removed by vegetation at a UK scale, two scenarios were created. The reference scenario included current UK land cover and the alternative scenario replaced all vegetated land classes with a neutral land surface equivalent to bare soil. The beneficial effects in terms of pollution removal by vegetation were then calculated from the difference between the two model runs.

Data inputs

The model was run for 2015. Land cover was based on the Corine land cover 2012, at 0.25 km resolution. Land cover was aggregated to seven broad land cover types for which deposition velocities for a suite of pollutants are available. The land cover classes were:

- coniferous woodland
- deciduous woodland
- heather & grass
- crops
- bare soil
- water
- urban

Results

Deposition velocities

Pollutant deposition depends on the deposition velocity, which is dynamic according to a range of factors. Figure 2 illustrates spatial variability in the annual average deposition velocity (Vd) for two of the outputs from the model, for (a) ammonia and (b) sulphur dioxide using deposition to deciduous forest. The ammonia Vd ranged from 6 to > 20 mm s^{-1} and was largely governed by the wind speed pattern due to orography. Sulphur dioxide Vd ranged from <6 to > 30 mm s^{-1} and was strongly increased in areas of high NH_3 emission, where the neutralising effect of NH_3 on leaf surfaces is most effective. The annual average deposition velocity for a number of pollutants is shown in Table 1. This comprises a spatial average across the UK and temporal average across the year, effectively accounting for all the covarying factors such as meteorology and other pollutants which contribute to the variability in space and time.

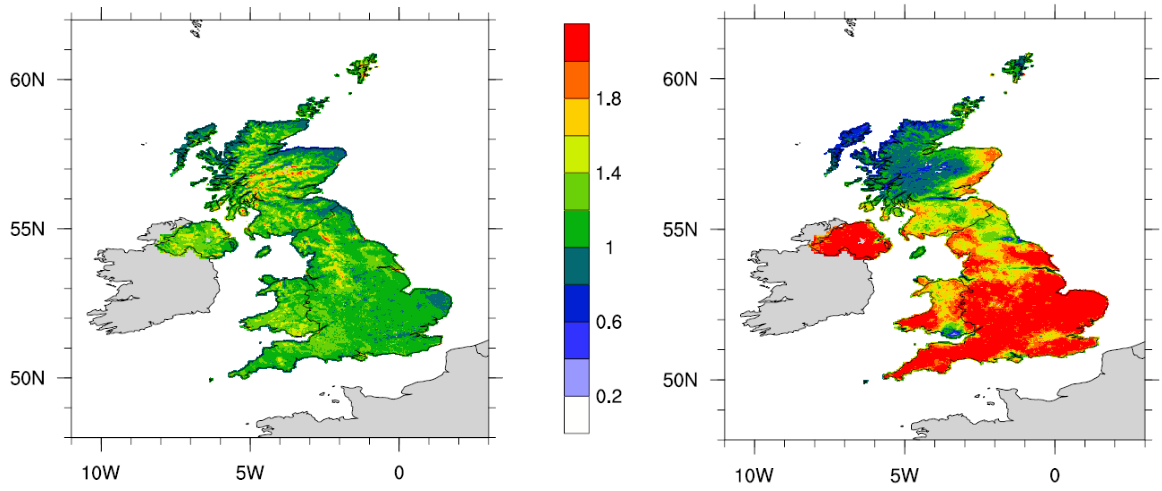

Figure 2 . Annual average dry deposition [cm s^{-1}] to deciduous forest of (a) ammonia and (b) sulphur dioxide.

Table 1. Annual average deposition velocities extracted from EMEP4UK (mm/s)

	Coniferous	Deciduous	Heather & grass	Crops	Water
PM_{10}	6.85	4.68	2.07	1.92	1.99
PM_{fine}	5.40	3.58	0.89	0.66	0.71
SO_2	14.37	17.11	7.13	5.40	7.44
NH_3	11.75	11.81	6.15	3.05	6.35

Removal of PM_{fine} by vegetation

Dry deposition of PM_{fine} under the 'vegetation' and 'no vegetation' scenarios is shown in Figure 3. The difference map (Figure 3c) therefore shows the quantities of pollutants removed by vegetation, with positive values showing pollution removed and negative values showing additional pollutant deposition. Areas showing high pollutant removal are primarily where deciduous and conifer woodland dominate the land cover. Major UK urban areas show relatively little pollutant removal compared with the surrounding countryside. Table 2 shows the total quantities of pollutants removed by vegetation in Great Britain, with available values from Powe and Willis (2004) for comparison. The resultant change in average concentration across Great

Britain is provided in Table 3 and reveals a 0.77 decrease in PM_{fine} concentrations when vegetation is considered.

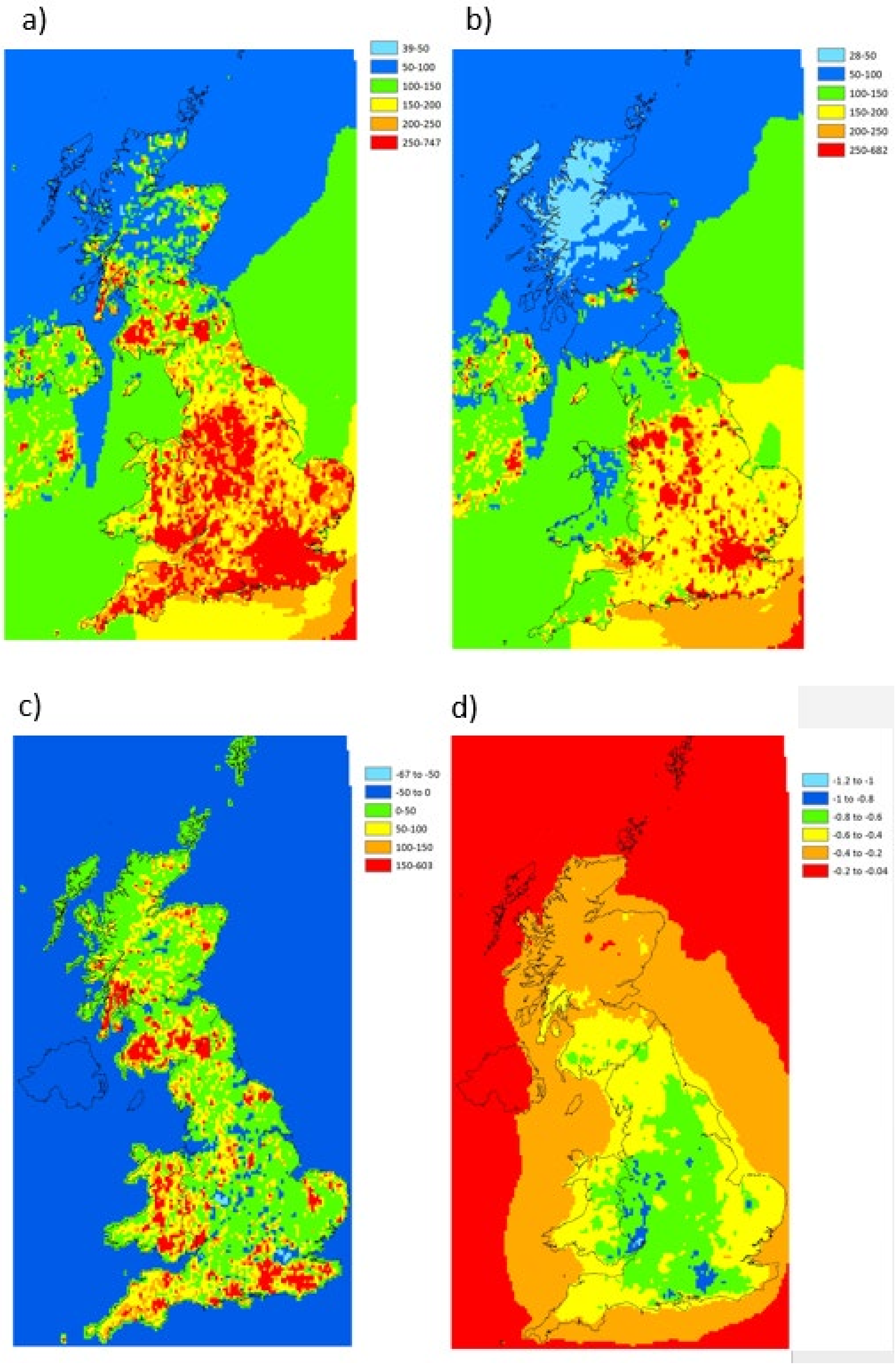

Figure 3. Maps of pollutant PM_{fine} removal by vegetation under two model scenarios using EMEP4UK, showing a) Pollutant removal with current UK vegetation, b) Pollutant removal assuming no UK vegetation, c) Difference map showing amount of

pollutant removed by vegetation (red values show areas of greatest removal of PM_{fine} by vegetation), d) Resulting change in PM_{fine} concentrations (blue values show greatest reduction).

Table 2. Estimated pollutant uptake by GB vegetation (kt pollutant yr^{-1})

	Powe & Willis (2004)	**EMEP4UK**
PM_{10}	385.7	
PM_{fine}		9.97
SO_2	7.7	29.62
NH_3		31.28
NH_4^+		2.33
NO_3^-		4.07

Table 3. Average concentrations simulated with the EMEP4UK model (ug m^{-3})

Pollutant	**Current landcover**	**No vegetation**	**Absolute difference**	**Rel. difference**
PM_{fine}	4.70	5.47	-0.77	-14%
SO_2	0.81	1.10	-0.29	-26%
NH_3	1.28	1.58	-0.30	-19%
NO_2	5.35	5.36	-0.005	-0.1%
O_3	68.2	77.7	-9.5	-12%

Discussion

This study demonstrates the use of an atmospheric transport model and land cover scenarios to calculate pollutant removal by vegetation at a UK scale. Although there are few previous studies against which the results can be directly compared, the quantities of pollutant removal are within similar orders of magnitude.

The annual average deposition velocities for trees revealed in the EMEP4UK model outputs are approximately five times higher than for other vegetation types. This reflects their greater surface area and increased canopy roughness. Thus, in urban areas, trees will remove more pollutants than other vegetation types. However, the total amount of pollutant removal is scaled by the area of each vegetation type, so in the wider countryside, a greater amount of pollution is removed by other vegetation types since they occupy a greater area.

The only comparable study for the whole UK, Powe & Willis (2004) estimated pollutant removal for PM_{10}, not PM_{fine}, and pollutant removal of SO_2. However, their calculations included a double-counting error which involved scaling deposition velocity by surface area index of the vegetation. This leads to double counting because the deposition velocity for a given vegetation type already incorporates a measure of leaf surface area (Simpson et al. 2011). Nonetheless, for the SO_2 removal, our approach estimates a greater quantity of pollutant removal, probably because the deposition velocity for sulphur is dependent on the relative balance of sulphur and ammonia concentrations, which has changed over time as sulphur emissions have declined (RoTAP 2011).

Our initial estimates suggest that vegetation across the UK leads to a reduction of PM_{fine} concentrations of ~14%. This difference is somewhat greater than estimates in the USA which suggest air pollution improvements due to trees across rural and urban areas in the conterminous USA are below 1% (Nowak et al. 2014).

When applied to trees within the urban extent, the total amount of pollution removed will be smaller because urban areas comprise < 7% of the UK land area (UKNEA 2011) and there are generally fewer trees in urban areas than in the countryside. Nowak et al. (2006) found that trees within cities in the USA reduced air pollution concentrations by 0.2 – 1%, this value was mainly dependent on the proportional area of trees within each city. In the UK, Tiwary et al. (2009) calculated removal of PM_{10} under different urban greenspace scenarios within a 10x10 km area of London, showing a removal of 90.4 t yr^{-1}. The next steps for this research are therefore to develop and apply urban-only scenarios, to estimate the benefits derived solely from all aspects of urban natural capital, including trees and grassland areas (green infrastructure) and the blue infrastructure such as lakes, rivers and canals.

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