Simulating impacts on UK air quality from net-zero forest planting scenarios

Gemma Purser[1,2*], Mathew R. Heal[2], Edward J. Carnell[1], Stephen Bathgate[3], Julia Drewer[1], James I.L Morison[4], Massimo Vieno[1]

1. UK Centre for Ecology & Hydrology, Bush Estate, Penicuik, EH26 0QB, UK
2. University of Edinburgh, School of Chemistry, David Brewster Rd, Edinburgh EH9 3FJ, UK
3. Forest Research, Northern Research Station, Bush Estate, Roslin EH25 9SY, UK
4. Forest Research, Alice Holt Lodge, Wrecclesham, Farnham GU10 4LH, UK

*corresponding author: gepurse25@ceh.ac.uk

Abstract

The UK proposes additional bioenergy plantations and afforestation as part of measures to meet net-zero greenhouse gas emissions, but species and locations are not yet decided. Different tree species emit varying amounts of isoprene and monoterpene volatile organic compounds that are precursors to ozone and secondary organic aerosol (SOA) formation, the latter of which is a component of $PM_{2.5}$. The forest canopy also acts as a depositional sink for air pollutants. All these processes are meteorologically influenced. We present here a first step at coupling information on tree species planting suitability and other planting constraints with data on UK-specific BVOC emission rates and tree canopy data to simulate via the WRF-EMEP4UK high spatial resolution atmospheric chemistry transport model the impact on UK air quality of four potential scenarios. Our 'maximum planting' scenarios are based on planting areas where yields are predicted to be ≥50% of the maximum from the Ecological Site Classification Decision Support System (ESC-DSS) for *Eucalyptus gunnii*, hybrid aspen (*Populus tremula*), Italian alder (*Alnus cordata*) and Sitka spruce (*Picea sitchensis*). The additional areas of forest in our scenarios are 2.0 to 2.7 times current suggestions for new bioenergy and afforestation landcover in the UK. Our planting scenarios increase UK annual mean surface ozone concentrations by 1.0 ppb or 3% relative to the baseline landcover for the highest BVOC emitting species (e.g., *E. gunni*). Increases in ozone reach 2 ppb in summer when BVOC emissions are greatest. In contrast, all the additional planting scenarios lead to reductions in UK annual mean $PM_{2.5}$ – ranging from -0.2 μg m^{-3} (-3%) for Sitka spruce to -0.5 μg m^{-3} (-7%) for aspen – revealing that $PM_{2.5}$ deposition to the additional forest canopy area more than offsets additional SOA formation. Relative decreases in annual mean $PM_{2.5}$ are greater than the relative increases in annual mean ozone. Reductions in $PM_{2.5}$ are least in summer, coinciding with the period of maximum monoterpene emissions. Although only a first step in evaluating the impact of increased forest plantation on UK air quality, our study demonstrates the need for locally relevant data on landcover suitability, emissions and meteorology in model simulations.

1. Introduction

Forest areas currently comprise around 3.21 Mha (13%) of UK landcover. Under suggested measures to meet UK net-zero greenhouse gas emissions by 2050, forested areas could increase by 1.2 Mha to 4.4 Mha (18%) (Climate Change Committee, 2020). An additional 0.7 Mha of land could also be used to grow bioenergy crops. These could be perennial energy crops (Miscanthus), short-rotation coppice (willow) or short-rotation forest. The latter would likely comprise single-species plantations of fast-growing broadleaf tree species such as aspen, alder and eucalyptus (McKay, 2011). This increased afforestation and bioenergy crop planting has the potential to sequester an additional 14 $MtCO_2$ every year from 2024 (based on planting 30,000 trees annually) (Climate Change Committee, 2020).

In addition to being a sink for CO_2, terrestrial vegetation has long been known to emit biogenic volatile organic compounds (BVOCs) (Went, 1960). Explanations for BVOC emissions include being by-products of metabolism, relief from heat stress, defence against herbivory and disease, and communication (Dudareva et al., 2006; Laothawornkitkul et al., 2009). A very important class of BVOCs comprises isoprene (2-methyl-1,3-butadiene) (a hemiterpene) and monoterpenes. These are secondary metabolic products of photosynthesis whose emissions vary predominately in response to changes in light and temperature (Sharkey et al., 1996). Reactions of VOCs in the atmosphere impact on air quality. In areas with high nitrogen oxide (NO_x) concentrations, usually as a result of anthropogenic sources, emissions of additional VOCs lead to increased concentrations of ozone (O_3). Ground-level ozone is detrimental to agriculture and natural ecosystems because its toxicity to foliage reduces plant growth and crop yields (Fares et al., 2013; Felzer et al., 2007; Emberson, 2020). It is also a human respiratory pollutant (COMEAP, 2015), and a greenhouse gas (UNEP/WMO, 2011). Other reactions of VOC in the atmosphere, and particularly those of isoprene and monoterpenes, lead to formation of secondary organic aerosols (SOA) (Wyche et al., 2014; Carlton et al., 2009). These particles contribute to the substantial negative impact of airborne particulate matter (PM) on human health (WHO, 2013).

Research in the UK on domestic tree planting for carbon sequestration and biomass has previously focused on carbon uptake capacity, land availability, land suitability and biomass yield (Aylott et al., 2008; Tallis et al., 2013; Hastings et al., 2014; Wang et al., 2014). More recent studies have also sought to align locations for bioenergy crops with end-use facilities such as electricity and heat generating stations, particularly those that could be linked with carbon capture and storage capabilities (Albanito et al., 2019; Donnison et al., 2020). However, exactly where in the UK trees will be planted to provide a domestic source of biomass, or as part of afforestation schemes, is still largely undefined. In addition, very few studies have focused on the impacts of forest planting on UK air quality using individual tree species data. Those that have divide into three categories. Firstly, those that use simple empirical calculations to estimate the increase in UK emissions of a particular atmospheric BVOC (Eller et al., 2012; Graus et al., 2013; Morrison et al., 2016; Purser et al., 2021a, b). Secondly, those that extract lower spatial resolution data on changes to UK air quality from European-scale atmospheric chemistry transport models (ACTMs) (Ashworth et al., 2015, 2012; Porter et al., 2015; Zenone et al., 2016).

95 Thirdly, those that use higher spatial resolution ACTM simulations but simulate
96 arbitrary or only local variations in tree cover (Nemitz et al., 2020; Donovan et al.,
97 2005). An important additional issue is that the magnitude of isoprene and
98 monoterpene emissions varies by orders of magnitude between different tree
99 species, and with geographical location due to meteorology, so it is imperative that
100 models use relevant emissions data (Bäck et al., 2012; Staudt et al., 2004; Purser et
101 al., 2021b).

103 Here we improve on what has been undertaken before for the UK by presenting high
104 spatial resolution (5 km) air quality simulations which use (a) UK-wide afforestation
105 planting scenarios that take account of tree species ecological suitability data and (b)
106 BVOC emissions variables measured in UK bioenergy plantations. The former uses
107 the Ecological Site Classification-Decision Support System (ESC-DSS) to define
108 locations where planting is potentially possible for a given tree species, and the latter
109 uses data for the four tree species of interest – *Eucalyptus gunnii*, hybrid aspen
110 (*Populus tremula L. × P. tremuloides* Michx.), Italian alder (*Alnus cordata*) and Sitka
111 spruce *(Picea sitchensis)* – from Purser et al. (2021b, a). We use the EMEP4UK
112 ACTM (Simpson et al., 1999a, 2012; Vieno et al., 2010, 2014, 2016). The advantage
113 of an ACTM is that it tracks the full process of emissions, reaction and deposition of
114 chemical components in space and in time, allowing the changes in atmospheric
115 composition to reflect how increases in afforestation change all relevant processes.
116 For example, not only do forests affect BVOC emissions, and hence ozone and SOA
117 formation chemistry, but trees also affect ozone and PM removal via deposition
118 (Nemitz et al., 2020). Trees also enhance removal of other gaseous components
119 such as NO_x and ammonia (NH_3) which reduces their contribution to formation of
120 secondary inorganic aerosol components of PM. Our study is a first step in
121 evaluating the potential impact on UK air quality of large-scale single-species tree
122 planting under potential maximum planting scenarios using relevant measured field
123 data.

2. Methods

2.1   Estimating suitable areas for planting

129 To determine locations in the UK suitable for afforestation for a given tree species
130 we used the Ecological Site Classification Decision Support System (ESC-DSS)
131 (Pyatt and Suarez, 1997; Pyatt et al., 2001). In its normal operational mode, ESC-
132 DSS outputs a suitability score as yield potential (%) or as a fraction of yield, for a
133 range of possible tree species at a given location using local variables based on
134 climate (wind, temperature, rainfall), soil moisture regime and soil nutrient regime
135 (Pyatt et al., 2001). However, in this work we used the four pre-selected species of
136 interest to generate planting suitability maps for the whole of the UK based on
137 present climate (Figure 1). The aspen (*Populus tremula L. × P. tremuloides* Michx.),
138 eucalyptus (*E. gunnii)* and alder (*Alnus cordata*) species used in the scenarios are
139 examples of the successful tree species in UK trials of monoculture forest plantations
140 for bioenergy (Purser et al., 2021b, a). A Sitka spruce (*Picea sitchensis*) scenario is
141 also included because this species is highly productive and already accounts for
142 25% of the forest areas in Great Britain (Forest Research, 2022). ESC-DSS does not
143 cover Northern Ireland, so the tree planting scenarios formulated here are strictly for

144 Great Britain only, but as Northern Ireland comprises <6% of the area of the UK use
145 of 'UK' is retained.
146
147 The suitability of each 250 m x 250 m grid in ESC-DSS is categorised according to
148 the fraction of the potential for growth or yield for each species into very suitable
149 (≥75%), suitable (50-74%), marginal (30-49%) or unsuitable (<30%). Since there was
150 not a complete dataset for Italian alder in ESC-DSS, common alder (*Alnus glutinosa*)
151 was used as a substitute to generate the alder planting scenario. This is anticipated
152 to have negligible impact on the planting map since Italian alder has no significant
153 climatic limitations in the UK and can tolerate as broad a range of soil types as
154 common alder (Wilson et al., 2018).
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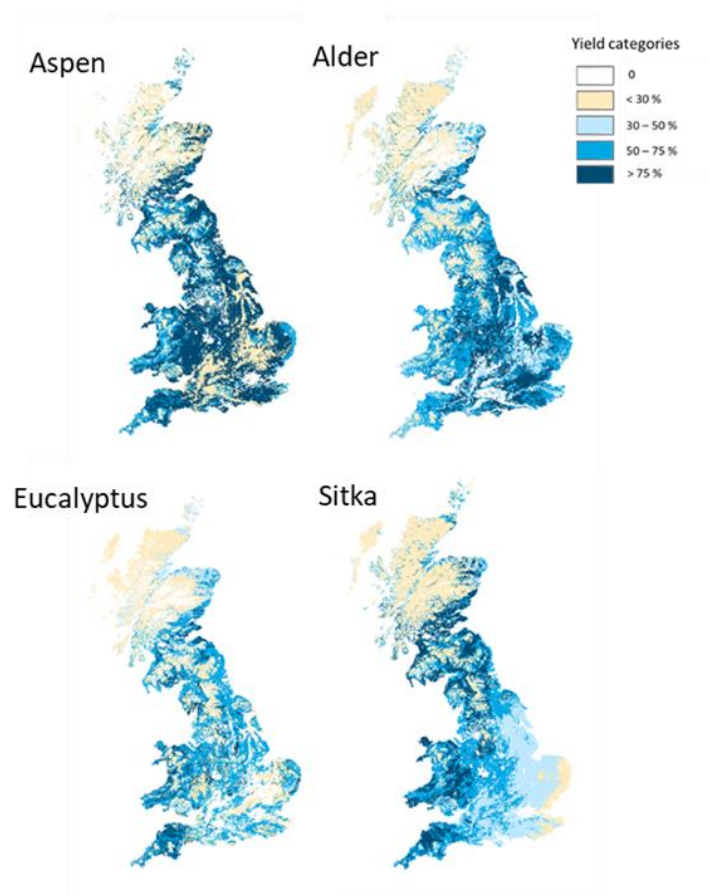
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159 *Figure 1: Yield maps for aspen, common alder, Eucalyptus gunnii and Sitka spruce, derived*
160 *from the Ecological Site Classification Decision Support System for UK meteorology and soils.*
161 *Locations where yields are ≥50% are shown in dark and medium blue colours. Based on data*
162 *from Forest Research.*
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2.2   Application of other planting constraints

Locations for the expansion of bioenergy crops or afforestation in the UK have been discussed but not yet formalised (House of Commons, 2021) although schemes that encourage tree planting exist (Woodland grants and incentives overview table - GOV.UK (www.gov.uk). The use of low grade and marginal agricultural land, in particular, has been suggested as most favourable for developing both bioenergy planting and afforestation (Lovett et al., 2014; Thomson et al., 2020). In addition, Lovett et al. (2014) listed the following nine constraints on where bioenergy crops (including short-rotation forests) should not be planted: slopes greater than 15%; high organic carbon soils; urban areas, roads, rivers, lakes; existing woodland; cultural heritage sites; designated areas (national parks, areas of outstanding natural beauty); natural and semi-natural habitats; and those areas which were given high value based on their habitat being similar to areas of outstanding natural beauty and national parks. We layered the constraint map by Lovett et al. (2014) over the species suitability maps (Section 2.1) to produce the landcover planting scenarios for each species shown in Figure 2. Only areas where ECS-DSS predicted tree yields ≥50% of potential for a given species were included in these new planting scenarios. The figure shows that suitability varies spatially, for example, with drier areas in the east being more suitable for aspen than for Sitka.

Data in Table 1 show that the increases in forest cover under these potential maximum planting scenarios range between 3.85 Mha for Sitka spruce to 5.35 Mha for *E. gunni*. These additional areas correspond to increases of 120% and 164%, respectively, on the 2018 baseline forest cover of 3.21 Mha (the latter being 13% of UK land area). Table 1 also illustrates how the additional forest covers distribute across the different categories of agricultural land that each scenario replaces. These distributions are very similar: ~20% of each scenario has replaced excellent quality agriculture land, ~60% has replaced good quality agriculture land and the remainder has replaced poor, unsuitable or unknown land. However, as noted above, the absolute amounts of each land category converted to forest differs; the distributions of the underlying agricultural land classes replaced in each additional SRF planting scenario are shown in Figure 3. Forest planting on the highest quality agriculture land is unlikely but is included here to simulate the impacts on air quality from the maximum possible forest cover for these four species in the UK.

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221 Table 1. Total additional land cover converted to forest in the four planting scenarios,
222 and the proportions of different categories of agricultural land that each scenario
223 replaces. Agricultural land classification systems differ between England and Wales,
224 and Scotland, so land quality was assigned to one of the three descriptors of
225 excellent, good and poor as specified in the table.

226

	England & Wales land class	Scotland land class	Land quality descriptor	Planting scenario			
				Sitka spruce	Eucalyptus gunnii	Italian alder	Hybrid aspen
% of additional land converted to forest by agricultural land class	Grade 1 & 2	1 to 3.1	Excellent	18.7	21.2	21.4	21.3
	Grade 3a & 3b	3.2 to 4.2	Good	62.3	60.5	60.6	61.4
	Grade 4 & 5	5.1 to 7	Poor	15.6	13.3	13.0	13.6
	Unsuitabe/ unknown			3.4	5.0	5.1	3.8
Total additional land converted to forest / km^2 (Mha)				38,472 (3.85)	52,501 (5.25)	47,657 (4.77)	52,218 (5.22)
% increase in forest relative to the baseline forest of 3.21 Mha				120	164	149	163
Additional forest as a multiple of the 1.9 Mha 2050 additional planting proposed				2.03	2.76	2.51	2.74

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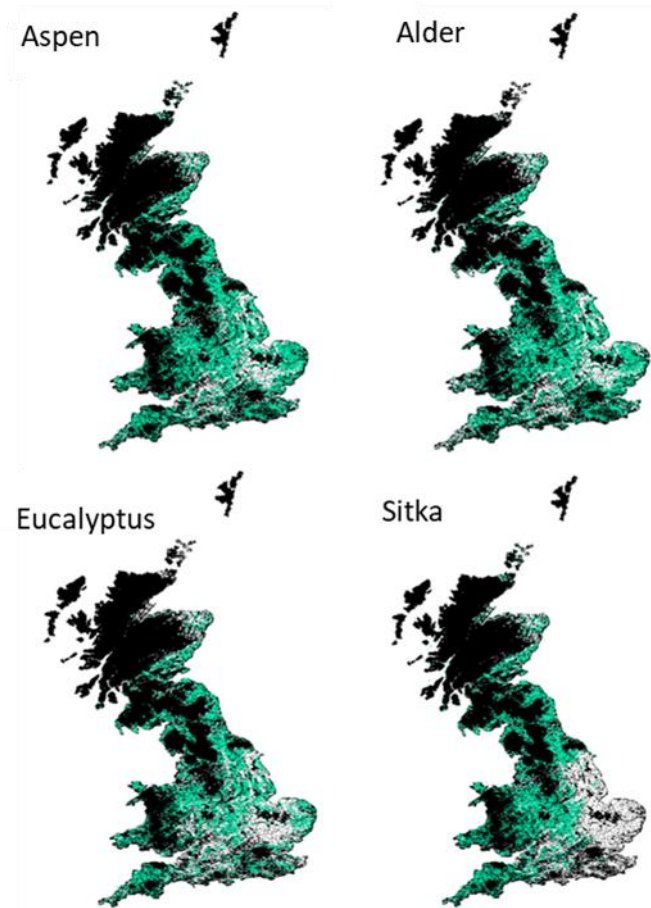
233
234 *Figure 2: Additional SRF planting scenarios developed in this study for aspen, common alder,*
235 *Eucalyptus gunnii and Sitka spruce, shown in green. These are areas classified as very*
236 *suitable or suitable (tree yields ≥50%) for that species, whilst also avoiding areas identified by*
237 *Lovett et al. (2014) where no bioenergy crops could or should be planted, shown in black.*
238 *White shows areas classified as unsuitable for planting the species (yield <50%).*
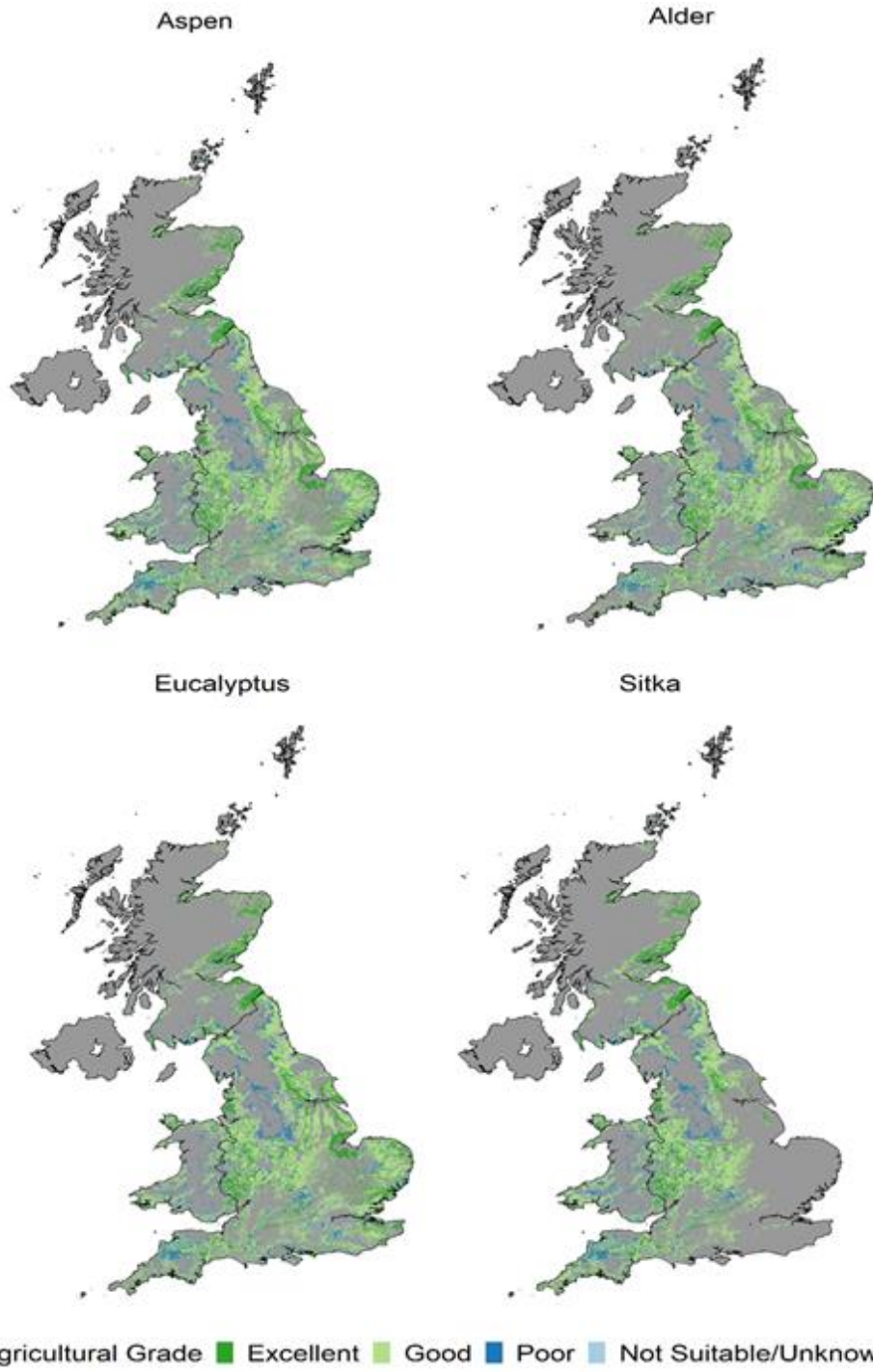
Figure 3: Underlying agricultural land class replaced in each additional SRF planting scenario for aspen, common alder, Eucalyptus gunnii and Sitka spruce. Grey areas show where there is no additional planting for that species.

2.3 EMEP4UK model simulations

2.3.1 Baseline model set-up

Simulations were undertaken at 5 km × 6 km horizontal resolution (and hourly temporal resolution) with EMEP4UK ACTM version rv4.34,(Vieno et al., 2014; Nemitz et al., 2020; Vieno et al., 2010, 2016). This is a nested version of the EMEP MSC-W model described in Simpson et al. (2012, 2020) in which the higher resolution British Isles domain is nested within an extended Europe domain that is simulated at ~50 km × 50 km horizontal resolution. The auxiliary files for this version can be downloaded from GitHub (https://github.com/metno/emep-ctm/releases/tag/rv4_34). The EMEP modelling suite is routinely validated against measurements and is widely used for air quality scenario simulations (see, for example, online tools and annual reports at www.emep.int/mscw/ and Vieno et al. (2014, 2010, 2016). The EMEP4UK model was driven by meteorology from WRF version 4.1.5 (Skamarock et al., 2008) which includes data assimilation (Newtonian nudging) of the numerical weather prediction model meteorological reanalysis from the US National Center for Environmental Prediction (NCEP)/National Center for Atmospheric Research (NCAR) Global Forecast System (GFS) at 1° resolution every 6 h (NCEP, 2000). The meteorology used in the baseline and planting scenarios is for 2018.

Anthropogenic emissions of NO_x, NH_3, SO_2, CO, NMVOC (non-methane VOC), $PM_{2.5}$ and PM_{CO} (coarse particulate matter) for the UK were taken from the 2018 National Atmospheric Emissions Inventory (NAEI, 2020). For the rest of the extended European domain in which the British Isles domain is nested the official EMEP emissions fields were applied (https://www.ceip.at). Emissions of dimethyl sulfide (DMS), lightning and soil NO_x, and wind-derived dust and sea salt were set as reported in Simpson et al. (2012, 2020). Vegetation fire emissions were also included (Wiedinmyer et al., 2011), although these very rarely impact atmospheric composition over the UK. Isoprene and other biogenic emissions for the baseline model runs were set as described in Simpson et al. (2012) Dry deposition of gas and aerosol species is simulated utilizing deposition velocity as described in Simpson et al. (2012). For wet deposition, all $PM_{2.5}$ particle components have the same in-cloud wet scavenging ratio and below-cloud size-dependent collection efficiency by raindrops, whilst coarse particles are divided into two groups (coarse sea salt and other coarse particles) with their own sets of parameters (Simpson et al., 2012). The baseline landcover for the UK was derived by remapping the UKCEH Landcover Map 2007 (LCM2007) (Morton et al., 2011) to the seven existing landcover classes of the EMEP model (deciduous forest, coniferous forest, crops, semi-natural land, water, desert and urban). Elsewhere, the EMEP landcover dataset was used.

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298 **2.3.2 Additional planting scenarios model set-up**
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300 Since the desert landcover type in the ACTM is redundant for the UK it was adopted
301 to create a new landcover class to represent the new forest planting areas shown in
302 Figure 3. The landcover data used by EMEP4UK is at a grid resolution of 0.01 x 0.01
303 degree (~1 km) resolution with values representing percent cover of each land cover
304 type. The ECS-DSS yield data was converted to the same spatial resolution (0.01
305 degree) and projection system as the land cover data (as %/grid cell). These
306 datasets were then combined to estimate a new land cover values. If the yield map
307 for a given model grid is favourable for a given tree species, then it replaced the
308 existing landcover. New forest created is additional forest. Minor variations in
309 percentage coverage of land covers exist between the planting scenarios and the
310 baseline due to projecting the land cover scenarios from British National Grid to
311 WGS84 coordinate reference system.
312
313 The tree variables used in the model for the new planting scenarios are summarised
314 in Table 2. The leaf area index (LAI) values are those measured in 9-year-old trial
315 SRF stands at East Grange, UK (Purser et al., 2021b) and 8-year-old stands of
316 regrown short-rotation coppice at Daneshill, UK (Purser et al., 2021a), the same
317 forests in which the BVOC emissions were measured. The biomass density (g m$^-$
318 $^2_{ground}$) data are derived from measurements of LAI and leaf mass area as discussed
319 in Purser et al. (2021b). BVOC emissions in the ACTM are driven by the algorithms
320 of Guenther et al. (1993) and Simpson et al. (2012). The standardised mean
321 emission rates for isoprene (E_{iso}) and total monoterpenes (E_{mtp}) (μg g$_{dw}^{-1}$ h^{-1}) given in
322 Table 2 for the four tree species investigated in this work derive from field
323 measurements of the emissions under 'real-world' UK conditions as reported in
324 Purser et al. (2021a, b). No appropriate above-canopy flux measurements were
325 available for the tree species in this study. The emissions were therefore based on
326 chamber studies conducted on single-species branches. Further information on the
327 methodology used to derive emission potentials, and a comprehensive comparison
328 against other literature values, is given in Purser et al. (2021). The values for the
329 same model variables and the standardised mean emission rates for different
330 woodland types, grassland and cropland used in the baseline scenario are also
331 given in Table 2 for comparison. In the monoterpene emission algorithm, a different
332 fraction of the emission of an individual monoterpene compound (e.g., α-pinene, d-
333 limonene) may be attributed to a de-novo source or a storage pool source. However,
334 in this study the monoterpene emissions from the four tree species investigated were
335 assigned to pool emissions (E_{mtp}) only as no separate light-driven fractions (E_{mtl})
336 were reported. (The latter are available for existing landcover vegetation.) The
337 EMEP4UK simulations of monoterpene chemistry utilise a 'lumped' reaction
338 mechanism in which 'total monoterpene' is represented by a single monoterpene
339 (Simpson et al., 2012).
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Table 2 Tree species model input parameters

Tree species or other land cover	No. days leaves present	LAI_{Min} / m^2 m^{-2}	LAI_{Max} / m^2 m^{-2}	Vegetation height (m)	Biomass density / g m$^{-2}_{ground}$	E_{iso}* / µg C g$_{dw}^{-1}$ h^{-1}	E_{mtp}* / µg C g$_{dw}^{-1}$ h^{-1}	E_{mtl}* / µg C g$_{dw}^{-1}$ h^{-1}
Aspen[†]	307	0	4.24	20	329	22.8	0.17	0
Alder[†]	307	0	3.25	20	315	0.03	0.86	0
Eucalyptus[†]	366	2.0	2.0	20	429	7.5	1.16	0
Sitka spruce[†]	366	3.14	3.14	20	619	10.9	3.4	0
Grassland	366	2	3.5	0.3	400	0.2	0.2	0.3
Cropland	213	0	3.5	1	700	0.2	0.2	0.3
Deciduous woodland	307	0	4	20	320	26	3.4	2
Conifer woodland	366	5	5	20	1000	1.7	0.85	2

[†] Based on measurements conducted by Purser et al.,(2021a, b)

*30 °C and 1000 µmol m^{-2} s^{-1}

3. Results

Table 3 presents, for each planting scenario, the changes relative to the baseline in UK total isoprene and monoterpene emissions, together with the simulated changes in UK annual mean surface concentrations of ozone, SOA and PM$_{2.5}$. (The SOA presented here is SOA produced from UK emissions of VOC and does not include SOA transported from outside the inner model domain.) Each of these changes are discussed in further detail in Sections 3.1-3.5. Population-weighted annual mean surface concentrations, and their changes, for each planting scenario are given in Table 4. The table shows that the relative changes in UK mean surface concentrations induced by each planting scenario differed little whether expressed as an area mean or as a population-weighted mean.

Table 3 Annual UK emissions of isoprene and total monoterpenes, and UK annual mean surface concentrations of O$_3$, SOA and PM$_{2.5}$ for the 2018 baseline and the four additional forest planting scenarios.

	UK annual emissions		UK annual mean concentration			Absolute (and % relative) change from baseline				
	Isoprene / kt y^{-1}	Monoterpene / kt y^{-1}	Ozone / ppb	SOA / µg m^{-3}	PM$_{2.5}$ / µg m^{-3}	Isoprene / kt y^{-1}	Monoterpene / kt y^{-1}	Ozone / ppb	SOA / µg m^{-3}	PM$_{2.5}$ / µg m^{-3}
Baseline	63.9	120.8	30.4	0.42	7.0	-	-	-	-	-
Eucalyptus	97.7	147.8	31.4	0.44	6.7	33.8 (53%)	27.0 (22%)	1.0 (3%)	0.02 (5%)	-0.3 (-4%)
Alder	54.9	127.2	30.8	0.41	6.6	-9.0 (-14%)	6.4 (5%)	0.4 (1%)	-0.01 (-2%)	-0.4 (-6%)
Sitka spruce	120.8	233.9	31.0	0.55	6.8	56.9 (89%)	113.1 (94%)	0.6 (2%)	0.13 (31%)	-0.2 (-3%)
Aspen	150.3	110.8	30.9	0.38	6.5	86.4 (135%)	-10.0 (-8%)	0.5 (2%)	-0.04 (-10%)	-0.5 (-7%)

Table 4 Population-weighted UK annual mean surface concentrations of O$_3$, SOA and PM$_{2.5}$ for the 2018 baseline and the four additional forest planting scenarios.

	UK population-weighted annual mean concentration			Absolute (and % relative) change from baseline		
	Ozone / ppb	SOA / $\mu g\ m^{-3}$	PM$_{2.5}$ / $\mu g\ m^{-3}$	Ozone / ppb	SOA / $\mu g\ m$	PM$_{2.5}$ / $\mu g\ m^{-3}$
Baseline	28.9	0.44	8.6	-	-	-
Eucalyptus	29.6	0.47	8.2	0.7 (2%)	0.03 (7%)	-0.4 (-5%)
Alder	29.1	0.44	8.1	0.2 (1%)	0.00 (0%)	-0.5 (-6%)
Sitka spruce	29.4	0.58	8.4	0.5 (2%)	0.14 (32%)	-0.2 (-3%)
Aspen	29.2	0.41	8.1	0.3 (1%)	-0.03 (-7%)	-0.5 (-7%)

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3.1   Changes in isoprene emissions

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373 The baseline (2018) annual UK emissions of isoprene are 63.9 kt y^{-1} (Table 3), of the
374 same order as the 44 kt y^{-1} reported from the JULES land surface model (Hayman et
375 al 2017). Figure 4 illustrates the magnitude and spatial distributions of UK isoprene
376 emissions for the baseline and the four planting scenarios and the differences
377 between the latter and the former. The baseline emissions are those from the current
378 UK landcover. The highest emissions (in red), which exceed 1800 mg m^{-2} y^{-1}, are in
379 the south where there are existing forests that are dominated by mixed broadleaf
380 species. The broadleaf forest landcover type that is used to represent these forests
381 in the model is assigned an emission potential of 26 µg C g$_{dw}$$^{-1}$ h^{-1} (Table 2). This
382 value is derived from a weighted sum of emission potentials of species that
383 contribute to this landcover type in the UK, such as oak (*Quercus* spp.), beech
384 (*Fagus* spp.), birch (*Betula* spp.) and ash (*Fraxinus* spp.), and from aggregated
385 landcover class maps (Köble and Seufert, 2001), because the EMEP landcover
386 scheme cannot currently handle large numbers of tree species (Simpson et al.,
387 1999b, 2012). These broadleaf species represent the range of broadleaf woodlands
388 that can be found in this region of England. In the rest of the UK, isoprene emissions
389 are in the range 800 to 1400 mg m^{-2} y^{-1} (green to orange colours in Figure 4). The
390 emissions of isoprene in northern England, north Wales and south and west
391 Scotland are predominately driven by the conifer forests in these parts of the UK.
392 The coniferous woodland landcover type used to represent these areas in the model
393 is assigned an emission potential of 1.7 µg C g$_{dw}$$^{-1}$ h^{-1}, which again represents a
394 weighted sum of individual species emission potentials.

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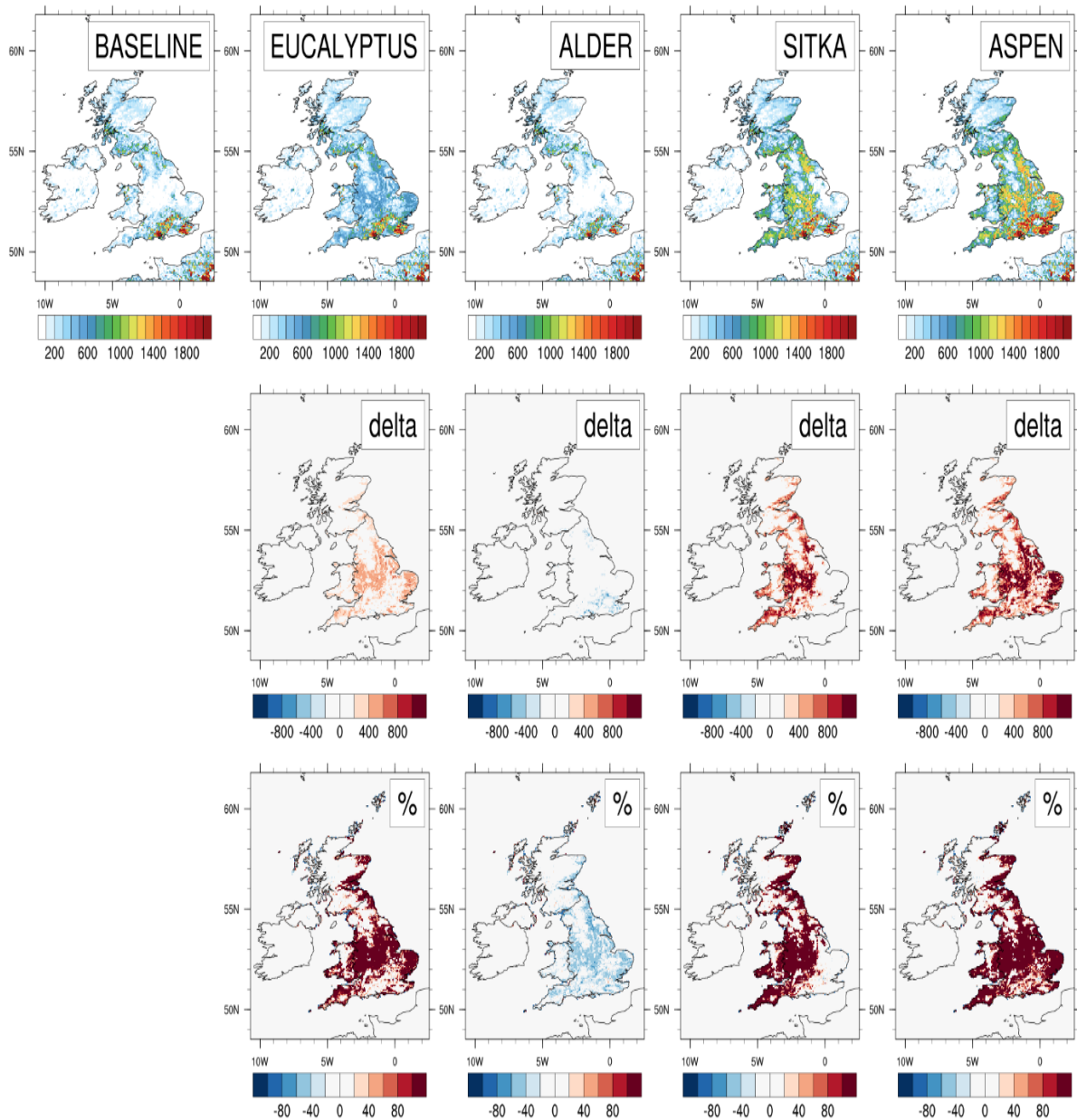
Figure 4: Modelled isoprene emissions for current UK landcover (baseline) and for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and hybrid aspen. Row 1 shows the annual isoprene emissions (mg m⁻²) for each scenario. Rows 2 and 3 respectively show the absolute and relative differences between each planting scenario and the baseline, with blue colours representing decreases and red colours representing increases.

414 Table 3 shows that annual UK isoprene emissions are simulated to increase by 86.4
415 kt (135%), 56.9 kt (89%) and 33.8 kt (53%) for the aspen, Sitka spruce and
416 eucalyptus planting scenarios, respectively, relative to the baseline isoprene
417 emissions of 63.9 kt y^{-1}. However, for the alder planting scenario, annual UK
418 isoprene emissions decrease by 9.0 kt to 56.9 kt y^{-1} because the isoprene emission
419 potential for alder (0.03 µg m^2 h^{-1}) is lower than that of the grassland and agricultural
420 land (both 0.2 µg m^2 h^{-1}) that the new planting replaces (Table 2).
421
422 For the aspen and Sitka spruce scenarios, isoprene emissions of up to 800-1000 mg
423 m^{-2} y^{-1} are evident in Figure 4 from the additional forests, particularly in the Midlands
424 and north of England where conditions to grow these moderately isoprene-emitting
425 species are favourable based on ESC-DSS information. The eucalyptus planting
426 scenario produces only about half the additional isoprene emissions annually as the
427 aspen and Sitka spruce scenarios, with emissions of around 400-600 mg m^{-2} y^{-1} in
428 areas where forests are added. There is a decrease in isoprene emissions of up to
429 200-400 mg m^{-2} y^{-1} relative to the baseline in the alder planting scenario (Figure 4).
430
431 For all tree species, the emissions of isoprene are predominately driven by solar
432 radiation and temperature and the presence of foliage (Monson and Fall, 1989).
433 Consequently, isoprene emissions were highest in July and lowest in December
434 (Figure 5). (By way of example data, sunshine hours in the UK for summer (June –
435 August) 2018 averaged 625 hours compared to 191 hours in winter (December-
436 February) (Met Office, 2018). Emissions of isoprene in summer account for the
437 majority, 63%, of the annual isoprene emissions in each tree planting scenario.
438 Spring (March – May), autumn (September-November) and winter isoprene
439 emissions account for 20%, 15% and 3% of the annual isoprene emissions
440 respectively. Maps showing the spatial emissions of isoprene each month and
441 monthly emission data tables are presented in Supplementary Material S1 and S2,
442 respectively.
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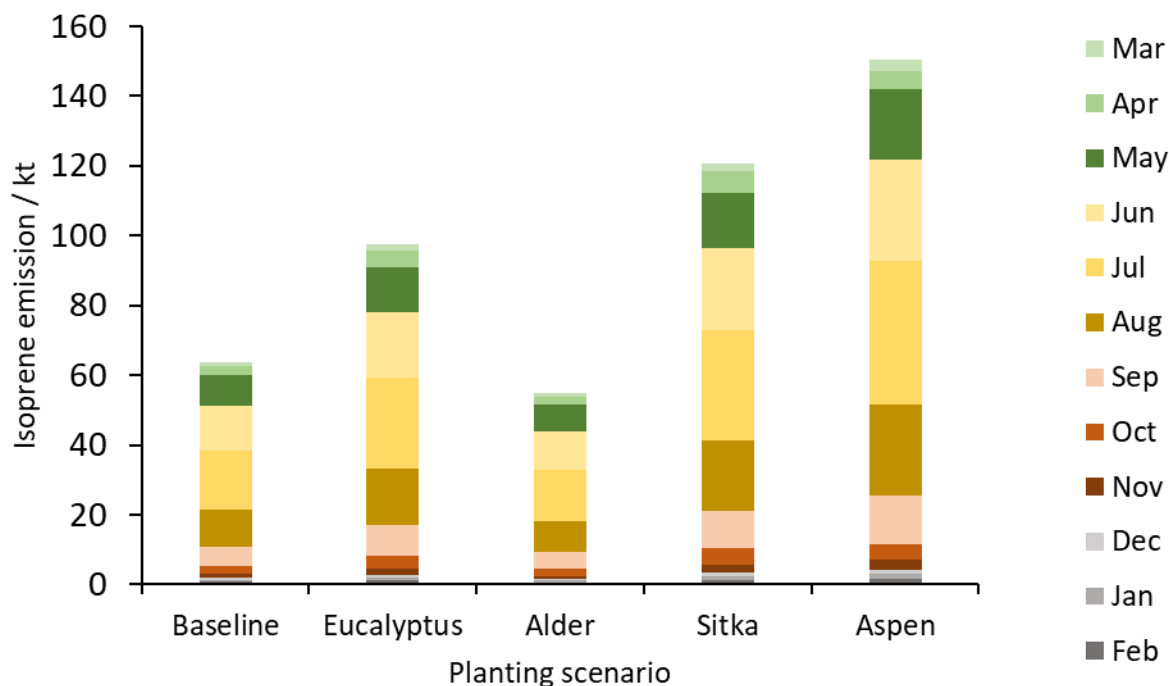
Figure 5: Total monthly isoprene emissions (kt) for current UK landcover (baseline) and for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and hybrid aspen.

3.2 Changes in total monoterpene emissions

The baseline annual UK total monoterpene emissions are 120.8 kt y^{-1} (Table 3), comparable with the 125 kt y^{-1} reported using the JULES land surface model (Hayman et al., 2017). Annual UK emissions of total monoterpenes are simulated to increase by 113.1 kt (94%), 27.0 kt (22%) and 6.4 kt (5%) relative to the baseline emissions of 120.8 kt y^{-1} for the Sitka spruce, eucalyptus and alder planting scenarios, respectively (Table 3). In contrast, total monoterpene emissions for the aspen scenario are simulated to decrease by 10.0 kt y^{-1} (8%) relative to the baseline. The highest monoterpene emissions for the baseline landcover are in Scotland, Wales and a small patch in eastern England. Emissions exceed 1800 mg m^{-2} in these areas and derive from the presence of conifer plantations.

Figure 6 shows the spatial heterogeneity of the monoterpene emissions across the UK associated with the four planting scenarios. Sitka spruce is a high monoterpene emitter, with monoterpene emissions increasing substantially, 1000-1200 mg m^{-2}, in those areas where this scenario replaces existing landcover. The increases in monoterpene emissions in the new planting areas in the eucalyptus scenario are much lower than for the Sitka spruce planting scenario, with increases in the new planting areas of 200-400 mg m^{-2} relative to the baseline. Changes in absolute monoterpene emissions for the alder scenario are negligible.
However, even though increases in monoterpene emissions nationally are relatively modest for the eucalyptus and alder planting scenarios (22% and 5%, respectively), even for the alder planting scenario local emissions of monoterpene could still

482 increase by more than 20% in many areas (Figure 6). For the eucalyptus scenario,
483 local monoterpene emissions would more than double in some areas.
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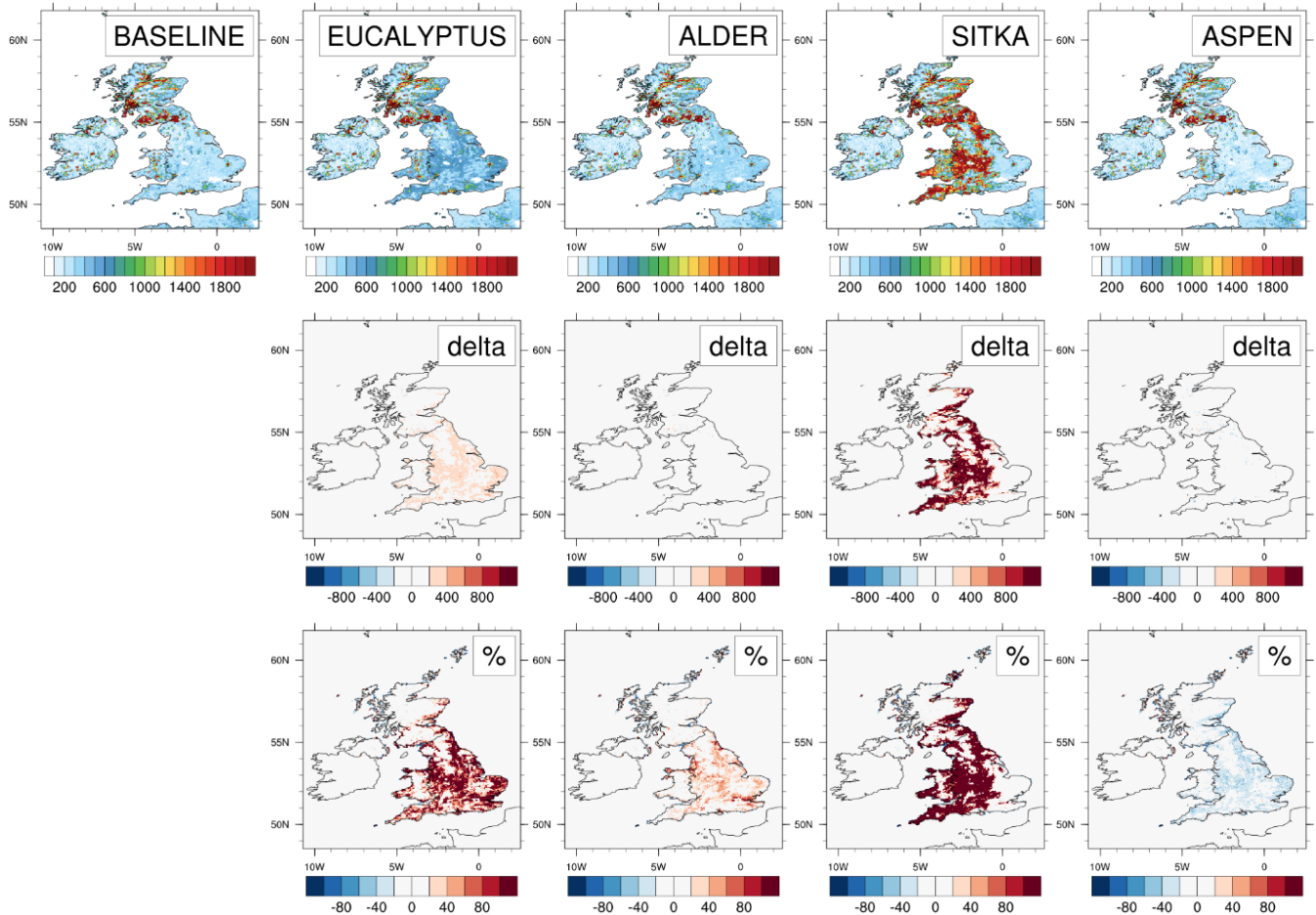
488
489
490 *Figure 6: Modelled total monoterpene emissions for current UK landcover (baseline) and for the*
491 *additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and hybrid aspen.*
492 *Row 1 shows the annual total monoterpene emissions (mg m^{-2}) for each scenario. Rows 2 and*
493 *3 respectively show the absolute and relative differences between each planting scenario and*
494 *the baseline, with blue colours representing decreases and red colours representing increases.*
495
496

497 The decrease in monoterpene emissions under the aspen planting scenario arises
498 because aspen has a monoterpene emission potential (0.17 μg m^2 h^{-1}) that is lower
499 than those from the grassland (0.2 μg m^2 h^{-1}) and agricultural land (0.2 μg m^2 h^{-1})
500 that the tree planting replaces (Table 2). Reductions in monoterpene emissions of up
501 to 40% occur in areas with new aspen planting (Figure 6). This is a similar effect to
502 that observed for changes in isoprene emissions in the alder scenario (Figure 4),
503 when a low BVOC emitting species replaces higher BVOC-emitting vegetation cover.
504
505 Total monoterpene emissions are highest in July and lowest in January for all
506 scenarios (Figure 7). There is relatively small difference in emissions between the
507 summer months (June – August) because total monoterpene emissions are driven

508 by temperature and average temperatures in the UK for these months are similar.
509 For example, the average UK temperatures in June, July and August 2018 were
510 14.8, 17.3 and 15.3 °C respectively (Met Office, 2018). Summer contributes most to
511 annual total monoterpene emissions (43%, seasonal mean temperature 15.8 °C),
512 followed by spring and autumn (22% each, mean temperatures of 8.1 °C and 9.8 °C,
513 respectively) and winter (13%, 3.6 °C). Maps showing the spatial emissions of total
514 monoterpenes each month and monthly emission data tables are presented in
515 Supplementary Material S3 and S4, respectively.
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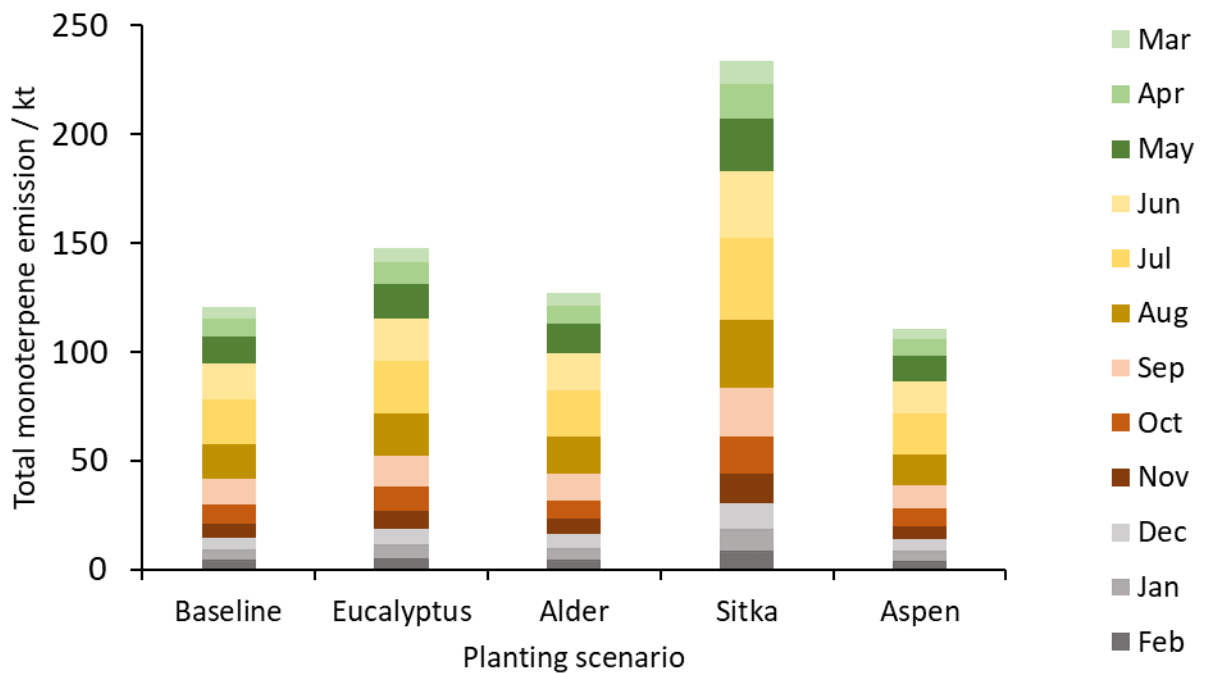
519
520
521 ***Figure 7: Total monthly total monoterpene emissions (kt) for current UK landcover (baseline)***
522 ***and for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and***
523 ***hybrid aspen.***
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3.3 Changes in surface ozone concentrations

Annual mean surface ozone concentrations are simulated to increase slightly in all scenarios of additional afforestation (Figure 8). The UK averaged annual mean ozone concentrations increase by 1.0 ppb (3%), 0.4 ppb (1%), 0.6 ppb (2%) and 0.5 ppb (2%) relative to the baseline UK averaged concentration of 30.4 ppb for the eucalyptus, alder, Sitka spruce and aspen planting scenarios, respectively (Table 3). Increases in annual mean surface ozone are much larger in some areas than the corresponding UK average (Figure 8). In the eucalyptus scenario, annual mean ozone is simulated to increase by more than 1 ppb (6%) over most of England (except in upland areas where eucalyptus cannot be planted) and in small areas in Wales and Scotland (again not in upland areas which are not suitable for eucalyptus) (Figure 2). The alder and aspen planting scenarios lead to smaller increases in local annual mean ozone, although still reaching 0.6 ppb or more across much of England. The increased ozone in these areas is driven not only by the enhanced BVOC emissions from the additional forest plantings, but by the greater anthropogenic NO_x emissions (required for ozone production) that are also associated with these higher population density areas of the UK.
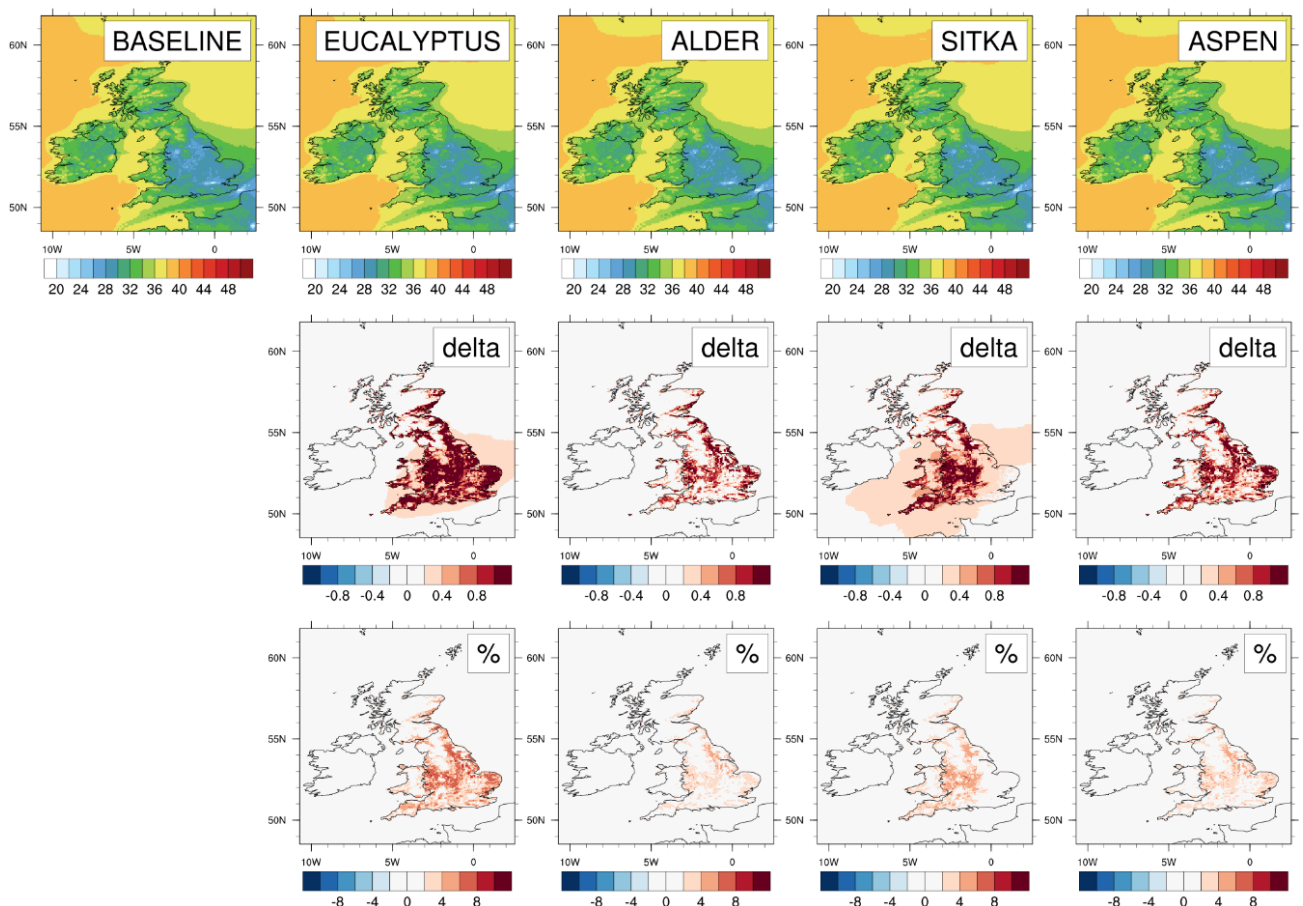
Figure 8: Modelled annual mean surface ozone concentrations for current UK landcover and for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and hybrid aspen. Row 1 shows the ozone concentrations (ppb) for each scenario. Rows 2 and 3 respectively show the absolute and relative differences between each planting scenario and the baseline, with blue colours representing decreases and red colours representing increases.

568 Monthly mean ozone concentrations peak in April and May in the UK and then
569 decrease during the summer months and into autumn and winter (Figure 9).
570 (Monthly versions of the ozone maps shown in Figure 8 are presented in
571 Supplementary Material S5.) This annual cycle is driven by many factors including
572 seasonal changes in vegetation (which affects both ozone formation via BVOC
573 emissions and ozone loss via deposition), hemispheric background ozone and ozone
574 transport (AQEG, 2021). The additional tree planting leads to greatest enhancement
575 of ozone during summer (June-August), reflecting the dominant contribution of
576 isoprene and monoterpene emissions in these months in the planting scenarios
577 (Figures 5 and 7). The simulations indicate that the impact of additional BVOC
578 emissions on ozone concentrations in summer are larger than the additional canopy
579 depositional sink for ozone. The eucalyptus planting scenario yields the largest
580 changes in ozone concentrations, peaking at 2 ppb in July), presumably a
581 consequence of eucalyptus being both a moderate isoprene and moderate
582 monoterpene emitter.
583
584 Interestingly, the aspen planting scenario has a lower impact on ozone concentration
585 changes in the summer, only 1 ppb, despite being a higher emitter of isoprene than
586 eucalyptus and Sitka spruce (Table 3 and Figure 4). Both isoprene and
587 monoterpenes are precursors for the formation of tropospheric ozone, and aspen
588 does not emit monoterpenes, whereas eucalyptus and Sitka spruce are significant
589 emitters of monoterpenes (Table 3 and Figure 6). Comparison of the aspen and
590 alder scenarios reveal an interesting phenomenon. Although the alder scenario leads
591 to a decrease in isoprene emissions compared with the baseline (Figure 4), the
592 increased monoterpene emissions from alder (Figure 6) offset the decreased
593 isoprene emissions to yield similar increases in ozone concentrations overall (Table
594 3). The reverse is true for the aspen scenario: the effect on ozone of a decrease in
595 monoterpene emissions is more than offset by the increase in isoprene emissions
596 from this species. The comparison of the effect on ozone across these three species
597 (Figures 8 and 9) therefore indicates the importance of monoterpene emissions as
598 well as isoprene emissions.
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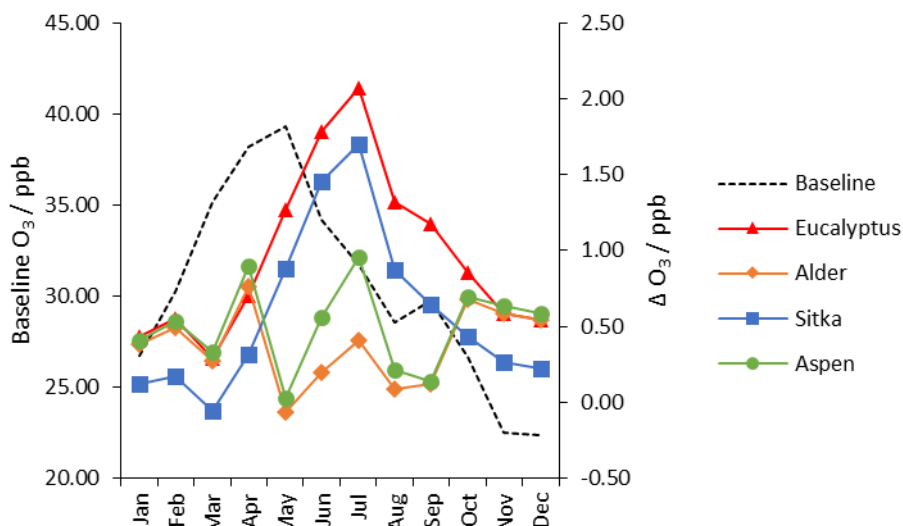
Figure 9: Monthly mean UK averaged concentrations of surface ozone (ppb) for baseline UK landcover (left-hand scale) and the monthly changes in ozone (right-hand scale) under the additional planting scenarios for Eucalyptus gunnii (red line), Italian alder (orange line), Sitka spruce (blue line) and hybrid aspen (green line).

These net impacts on ozone concentration are driven not only by the different ozone formation propensities of isoprene and monoterpenes (which in turn are influenced by local NO and NO_2 concentrations), but also by the different rates of ozone dry deposition across the different tree species. Our model simulations explicitly include these changes in ozone dry deposition. The relevant variables in the model are the biomass density, leaf area index and tree height. For all four planting scenarios the enhanced chemical production of ozone due to increased BVOC emissions is larger than the loss through increased in ozone dry deposition to the additional forest landcover (Table 3 and Figures 8 and 9). Aspen has the largest LAI of the four tree species, and a wider geographical range for planting; both these factors contribute to a greater depositional sink for ozone to aspen than for the other species and additionally explains why the aspen scenario yields smaller increases in ozone compared with the Sikta spruce and eucalyptus scenarios despite giving rise to large increases in BVOC emissions.

3.4 Changes in surface SOA concentrations

UK averaged annual mean surface SOA decreases by 0.04 µg m^{-3} (10%) and by 0.01 µg m^{-3} (2%) relative to the baseline SOA concentration of 0.42 µg m^{-3} for the planting scenarios involving the two broadleaf species, aspen and alder, respectively (Table 3). In contrast, UK averaged SOA increases by 0.13 µg m^{-3} (31%) and 0.02 µg m^{-3} (5%) for the Sitka spruce and eucalyptus scenarios, respectively. Note that the SOA data presented here is SOA derived from UK VOC emissions and do not include SOA derived from outside the UK. Most UK SOA derives from biogenic rather than anthropogenic VOC (Redington and Derwent, 2013) and the main biogenic precursors for SOA formation are monoterpenes. Aspen and alder are relatively low monoterpene emitters (Table 2), whilst eucalyptus and Sitka spruce are medium and high emitters of monoterpenes that contribute more substantially to the

642 formation of SOA. However, the exact impact of a particular species on SOA
643 concentration is the net effect of its roles in SOA formation and deposition.
644
645 The spatial distribution of these increases or decreases in SOA are heterogeneous
646 and therefore larger than the annual UK mean for SOA in some cases (Figure 10).
647 For the eucalyptus scenario there are up to 10% (0.08 µg m^{-3}) increases in SOA in
648 some locations, whilst for the aspen scenario there are reductions in SOA up to 10%
649 (0.08 µg m^{-3}), related to the distribution of new planting (Figure 3). The Sitka spruce
650 scenario yields the greatest increases in SOA, reaching up to 50% in central
651 England. As already noted, Sitka spruce is a high emitter of monoterpenes.
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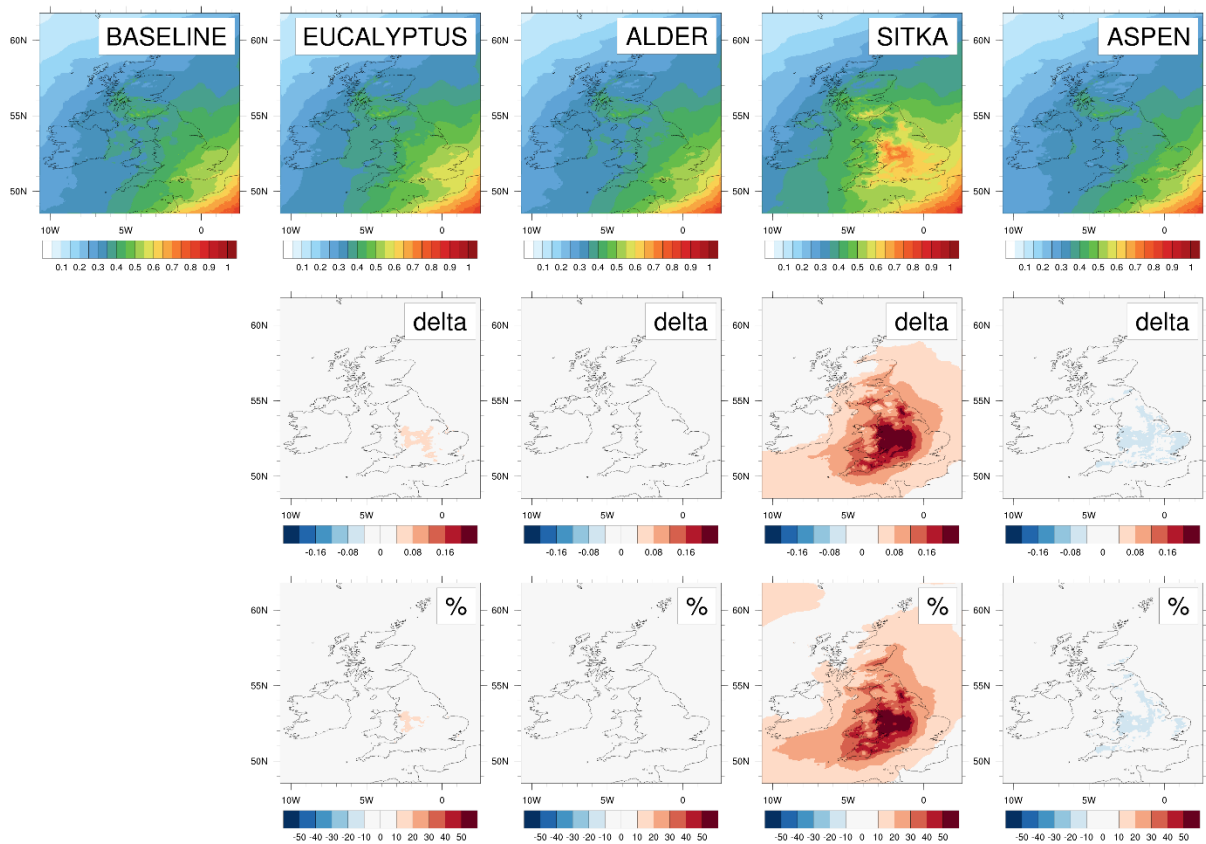
654
655 *Figure 10: Modelled annual mean surface SOA concentrations for current UK landcover*
656 *and for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce*
657 *and hybrid aspen. Row 1 shows the SOA concentrations (µg m^{-3}) for each scenario. Rows*
658 *2 and 3 respectively show the absolute and relative differences between each planting*
659 *scenario and the baseline, with blue colours representing decreases and red colours*
660 *representing increases.*
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665 Monthly mean concentrations of SOA for the baseline (Figure 11) confirm that, as
666 expected, SOA is greatest during spring and summer, peaking in May (0.32 µg m^{-3}),
667 and negligible in autumn and winter. (Monthly concentration data for the SOA shown

668 in Figure 11 are presented in Supplementary Material S7.) For the Sitka spruce
669 planting scenario, additional SOA concentrations relative to baseline peak in July
670 when the monoterpene emissions are greatest (Figure 7). This suggests that the
671 planting of high monoterpene emitters could extend the period over which SOA
672 concentrations are at their highest. The eucalyptus scenario follows a similar
673 seasonal trend to the Sitka spruce scenario but the contribution to additional SOA
674 concentration overall is lower. The most benefit in reduction in SOA concentration is
675 observed in the aspen and alder scenarios when foliage is present in May but when
676 temperatures and monoterpene emissions are relatively low.

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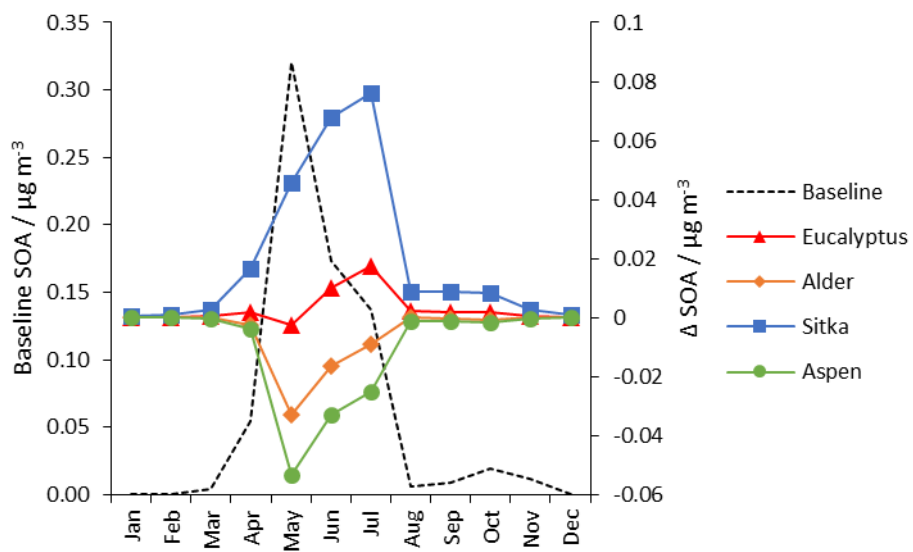
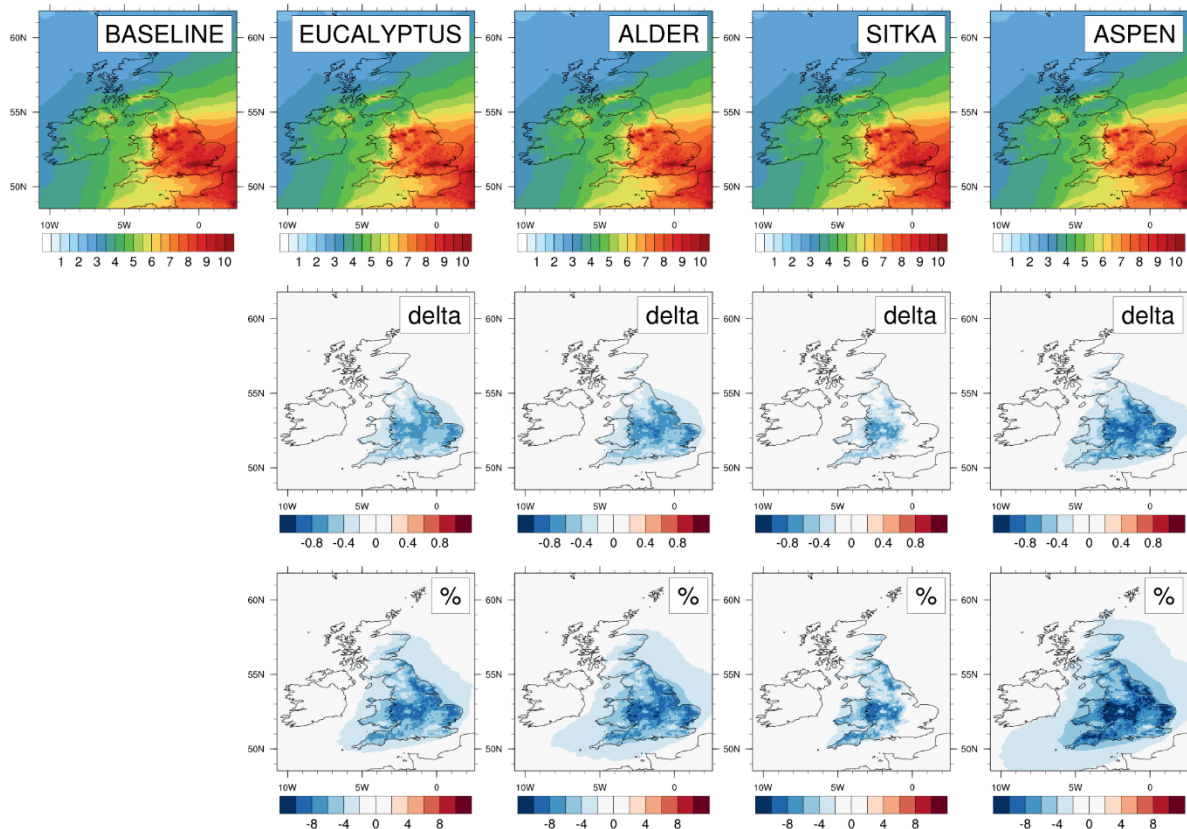

678
679
680 *Figure 11: Monthly mean UK averaged concentrations of surface SOA (µg m^{-3}) for baseline UK*
681 *landcover (left-hand scale) and the monthly changes in SOA (right-hand scale) under the*
682 *additional planting scenarios for Eucalyptus gunnii (red line), Italian alder (orange line), Sitka*
683 *spruce (blue line) and hybrid aspen (green line).*

684

3.5 Changes in surface PM$_{2.5}$ concentrations

686
687 In contrast to the situation for ozone, reductions in annual mean surface PM$_{2.5}$
688 concentrations relative to the baseline are simulated for all four additional
689 afforestation scenarios (Figure 12). The UK averaged annual mean PM$_{2.5}$
690 concentrations decrease by 0.3 µg m^{-3} (4%), 0.4 µg m^{-3} (6%), 0.2 µg m^{-3} (3%) and
691 0.5 µg m^{-3} (7%), relative to the baseline concentration of 7.0 µg m^{-3} for the
692 eucalyptus, alder, Sitka and aspen planting scenarios, respectively (Table 3).
693

694

695 *Figure 12: Modelled annual mean surface PM₂.₅ concentrations for current UK landcover and*
696 *for the additional planting scenarios for Eucalyptus gunnii, Italian alder, Sitka spruce and hybrid*
697 *aspen. Row 1 shows the PM₂.₅ concentrations (µg m⁻³) for each scenario. Rows 2 and 3*
698 *respectively show the absolute and relative differences between each planting scenario and the*
699 *baseline, with blue colours representing decreases and red colours representing increases.*

700

701

702 The decreases in annual mean PM₂.₅ under the planting scenarios are
703 geographically heterogeneous. Reductions exceeding 0.6 µg m⁻³ (6%) are simulated
704 across central and eastern England, particularly under the aspen planting scenario.
705 The spatial distribution of PM₂.₅ decreases corresponds to the locations of additional
706 afforestation shown in the planting maps (Figure 2) and is driven by the enhanced
707 dry deposition of particles to the trees relative to the baseline landcover type that the
708 trees have replaced (predominantly agricultural land, Figure 3). Although the new
709 planting areas for aspen and eucalyptus are of similar magnitude (approx. 52,000
710 km²) (Table 1) and distributed similarly over the UK (Figure 2), the differences in
711 PM₂.₅ deposition is larger for the aspen scenario (Figure 12) because the modelled
712 aspen area has a LAI double that of eucalyptus, even though the biomass density of
713 eucalyptus is higher than aspen (Table 2). The impact of additional tree cover on
714 PM₂.₅ via enhanced deposition outweighs new SOA formation from enhanced BVOC
715 emissions (Section 3.4).

716

717 Baseline monthly PM₂.₅ concentrations (Figure 13) display an increase in spring
718 (April-May) which is often observed in the UK, and which is related to ammonia
719 emissions from agricultural fertilisation enhancing secondary inorganic aerosol
720 formation and to meteorological conditions promoting long-range transport of PM₂.₅
721 from continental Europe (Vieno et al., 2014; Tang et al., 2018). (Monthly
722 concentration data for the PM₂.₅ map shown in Figure 12 are presented in

723 Supplementary Material S6.) In summer, PM$_{2.5}$ concentrations are lower because
724 combustion-related emissions are lower, higher temperatures promote ammonium
725 nitrate volatilisation, the boundary layer is on average deeper and there is greater
726 dry deposition to tree foliage (AQEG, 2012).
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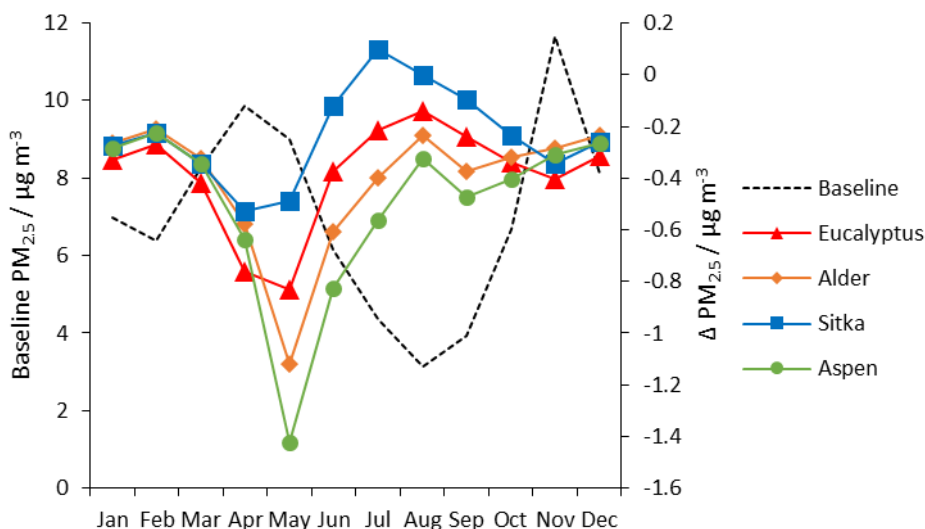
730
731
732 *Figure 13: Monthly mean UK averaged concentrations of surface PM$_{2.5}$ (µg m^{-3}) for baseline UK*
733 *landcover (left-hand scale) and the monthly changes in PM$_{2.5}$ (right-hand scale) under the*
734 *additional planting scenarios for Eucalyptus gunnii (red line), Italian alder (orange line), Sitka*
735 *spruce (blue line) and hybrid aspen (green line).*
736
737
738 The greatest reductions in surface PM$_{2.5}$ arising from the additional foliage due to
739 tree planting occurs in April and May in all four scenarios (Figure 13), suggesting
740 afforestation may help to reduce the burden of agricultural contributions to PM$_{2.5}$.
741 The aspen planting scenario showed the greatest reductions, which is likely due to
742 this tree species having the largest LAI in the model (Table 2). All planting scenarios
743 show reductions in monthly PM$_{2.5}$ in all months but reductions in PM$_{2.5}$ are smallest
744 in July and August. The Sitka spruce scenario shows a slight increase in PM$_{2.5}$ in
745 July. The trend arises because monoterpene emissions, the precursor to biogenic
746 SOA, are greatest in the summer and Sitka spruce is a particularly large emitter of
747 monoterpene; greatest monoterpene emissions from Sitka spruce occur in July
748 (Figure 7), in turn leading to greatest additional SOA concentrations in July (Figure
749 11).
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4. Discussion

The model scenarios presented suggest the scale of changes in atmospheric composition that may occur across the UK in response to planting substantial areas of land with different tree species as part of measures to meet net-zero greenhouse gas emissions. Proposals for possible pathways to achieve net-zero published to date have suggested additional planting of 1.9 Mha through both afforestation and bioenergy schemes (Climate Change Committee, 2020). For this study, however, we deliberately investigated the maximum planting scenarios possible for our four target tree species using only areas that had =>50% of potential yield, taking local climate and soil suitability and other land-use constraints into account using an ecological decision model. These scenarios result in additional areas of forest cover (Figure 2) that are 2 to 2.7 times greater than the 1.9 Mha currently being considered (Table 1). Less extensive planting schemes will lead to smaller changes in atmospheric composition than simulated here but, given that where the planting will occur in the UK is still undecided, our study highlights the spatial relationships between land suitable for new forest and the resultant impacts (via natural and anthropogenic emissions and deposition) on atmospheric composition.

Importantly, we also quantified the amounts of different categories of agricultural land that each planting scenario would replace. We show in order to provide good productivity that only 13-16%, or 0.6-0.8 Mha, of our maximum planting scenarios could take place on land classed as agriculturally 'poor' (Table 1). Although this area is comparable to that suggested so far for bioenergy crops, our analysis shows that any additional afforestation would have to displace agricultural land of higher quality. In our species suitability scenarios, the majority (~60%) of new planting would occur on 'good' quality agricultural land. Our dataset therefore provides important information for decision-making on the locations of land-use change resulting from different extents of new planting (Figure 3).

In all four of our individual tree species planting scenarios surface ozone concentrations were simulated to increase and surface $PM_{2.5}$ concentrations to decrease (Table 3). The changes in SOA concentration were dependent upon tree species, with those that were high monoterpene emitters, Sitka spruce in particular, yielding increased SOA (Figures 10 and 11).

The increases in UK averaged annual mean ozone were small, ranging between 0.4 and 1.0 ppb (1 and 3%), even under these maximum possible tree-planting scenarios which contribute large increases in emissions of isoprene and/or monoterpenes (Figures 4 and 6). In some localities, however, particularly in central and eastern England where large areas of land were assumed planted in these scenarios and where there are high emissions of anthropogenic NO_x, increases in annual mean ozone concentrations of 6% are simulated. For comparison, previous modelling work by Ashworth et al. (2015) investigating the impact on ozone levels in Europe of a range of poplar hybrids (*Populus* spp.), and focusing specifically on isoprene emissions, found similar increases of annual mean ozone concentration, although much higher increases in the Mediterranean (12-36%, up to 18 ppb) where higher temperatures drive much higher BVOC emissions. Our simulations also show strong seasonality in the increases of ozone under the planting scenarios (Figure 9).

808 Under the eucalyptus and Sitka spruce scenarios, UK averaged monthly mean
809 ozone increases exceed 1.5 ppb in summer (June-Aug) when BVOC emissions are
810 at their maximum (Figures 5 and 7). Ozone also dry deposits efficiently to vegetation,
811 but our simulations show that the chemical impact of the enhanced BVOC emissions
812 on ozone formation exceeds the enhanced ozone sink for each species investigated.
813
814 Our simulated reductions in UK averaged annual mean $PM_{2.5}$ concentrations ranged
815 between 0.2 and 0.5 µg m^{-3} (3 and 7%) (Table 3). However, reductions across much
816 of central and eastern England are larger and exceed 0.6 µg m^{-3} (6%). It is clear
817 from our simulations that the increase in $PM_{2.5}$ due to SOA formed from the
818 additional isoprene and monoterpenes is more than offset by the enhanced
819 deposition of $PM_{2.5}$ to the additional forest vegetation. Biogenic SOA formation as a
820 result of the simulated large expansion of high monoterpene emitting tree species
821 such as Sitka spruce could lead to an increase of 0.13 µg m^{-3} (31%) in annual mean
822 SOA relative to the baseline UK annual mean SOA concentration of 0.42 µg m^{-3}
823 (Table 3). However, SOA formation from BVOC sources within the UK remains a
824 relatively minor component of UK $PM_{2.5}$. For the two species investigated that
825 promote SOA formation, Sitka spruce and eucalyptus, the increase in SOA
826 concentration occurs solely in summer (Figure 11), coincident with the timing of the
827 monoterpene emissions. In other parts of the year, and for species that are low or
828 zero emitters of monoterpenes, the additional particle deposition sink provided by the
829 additional forest cover leads to net decreases in SOA and $PM_{2.5}$ overall compared to
830 the baseline landcover. Vegetation differences, such as those driven by biomass
831 density (by leaf area index in particular), are the important determinants in the
832 magnitudes of both isoprene and monoterpene emissions, and ozone and $PM_{2.5}$
833 depositions.
834
835 Localised environmental conditions may result in differences in specific leaf area for
836 a given tree species which then impacts on the leaf mass area that the model uses
837 to calculate the biomass density. In this study, UK-specific field data is used to derive
838 these terms (Purser et al. (2021b). The biomass density numbers we used are
839 comparable to other modelling studies (Keenan et al., 2009). As LAI is dependent on
840 forest structure (which is effected by plantation, density and management, for
841 example) and age we use values measured in UK bioenergy plantation trials (Purser
842 et al., 2021a, b). The EMEP4UK model does not yet incorporate the differences in
843 small-scale leaf deposition processes for individual tree species beyond
844 differentiating between different landcover types. This should be a consideration for
845 future model developments as different leaf surfaces have different particle capture
846 efficiencies, with coniferous species being the most efficient (Räsänen et al., 2013).
847
848 Although we apply a set of constraints on where each of our four species may be
849 planted, we recognise that our planting scenarios, although feasible, are large scale.
850 In reality, land assigned to new forest cover will be smaller and be a mixture of
851 monospecific plantations, as simulated here, and mixed species woodlands. Other
852 factors such as landowner preference, timber yields, biodiversity considerations,
853 aesthetics and tree species availability will all play a role in what tree species are
854 planted and where in the UK.
855
856 Our scenarios are based on UK field data for four tree species already performing
857 well in short-rotation bioenergy trials or, in the case of Sitka spruce, already widely

858 planted; but other species may be planted also. However, the species we use in our
859 simulations are representative of the range of possible impact that tree species have
860 on atmospheric composition. Thus, our four species span the forest functional types
861 of deciduous broadleaf (aspen and alder), evergreen broadleaf (eucalyptus) and
862 evergreen coniferous (Sitka spruce), which have different impacts on gas and
863 particle deposition. These species also include both low and high emitters of
864 isoprene and monoterpenes. In order to mitigate uncertainties in the emission
865 potentials of isoprene E_{iso} and monoterpenes E_{mtp}, as well as the temperature, light
866 and humidity dependence of the BVOC emissions, we use data from UK-specific
867 measurements to underpin the model simulations. The default emission potentials
868 for landcover types in the model are not assigned an uncertainty as they are derived
869 from a weighted sum of emission potentials of species based on literature values. All
870 measurements of emission potentials are subject to uncertainties, and potentially
871 more so when using plants grown and measured under field conditions. The
872 uncertainties of emission potentials used in this study are given in the
873 Supplementary Material S8. Detailed discussions of these individual uncertainties
874 are given in Purser et al. (2021a) and (2021b). Both monoterpene and isoprene
875 emission factors may also be impacted by a range of other variables in the field such
876 as biotic factors e.g. herbivory or plant disease (Rieksta et al., 2020; Blande et al.,
877 2007), effect of precipitation; genetic differences within each tree species (van
878 Meeningen et al., 2017; Duncan et al., 2001; Bäck et al., 2012); flooding, drought
879 and heat stress (Copolovici and Niinemets, 2010; Seco et al., 2015; Bonn et al.,
880 2019). The full range of variables found in the field currently cannot be replicated in
881 the necessarily simplified model environment. It is also possible that the collection of
882 such emission data using the enclosure technique could have an influence on the
883 measured emissions. The ranges in isoprene and monoterpene emissions from our
884 four species also indicate the sensitivity of surface atmospheric composition to
885 uncertainties in BVOC emissions.
886
887 A huge diversity of monoterpenes and other BVOCs are emitted from trees in nature,
888 the emissions and subsequent reactions of which can affect atmospheric
889 composition but are not included in atmospheric models (Faiola et al., 2018). Model
890 chemistry schemes are usually simplified to lump monoterpene emissions and
891 chemistry into a total monoterpene function with emissions representing the sum of
892 the most frequently measured monoterpenes in the field such as α-pinene, β-pinene,
893 limonene, myrcene and δ-3- carene. This is the approach used in the EMEP4UK
894 model we used in this study but is also the case in other widely used ACTMs (Monks
895 et al., 2017; Emmons et al., 2020; Arneth et al., 2008). Some chemistry schemes are
896 becoming more advanced (Schwantes et al., 2020) and may produce further
897 insights.
898
899 We are interested in the changes in atmospheric composition associated with new
900 forest planting, rather than the absolute atmospheric concentrations, so use the
901 same meteorological year (2018) in our simulations. Interannual differences in
902 temperature, cloudiness and weather patterns will influence the magnitude of BVOC
903 emissions and will also influence other variables affecting UK ozone and $PM_{2.5}$ each
904 year, such as photolysis rates, wet and dry deposition, boundary-layer height and
905 long-range transport. However, as an example, although changing, variances in UK
906 annual climate conditions assessed through changes in total rainfall, mean
907 temperature and total sunshine hours, over the past 11 years (2011-2021) have

908   been small (relative stand deviation of 9, 4 and 4% respectively). Therefore, given
909   that small changes to surface ozone occur in our simulations for 2018 based on
910   large additional forest planting it may suggest that relative changes to ozone under
911   other meteorological years may be similar (Met Office, 2022). The impact of the
912   planting scenarios on surface $PM_{2.5}$ has been shown to be dominated by the
913   enhanced deposition to the additional forest canopy which will be much less
914   influenced by interannual variations in meteorology than the BVOC emissions.
915   Perhaps more relevant to the impacts of forest planting on future atmospheric
916   composition in the UK is the trajectory of UK anthropogenic NO_x emissions, which
917   may reduce further under net-zero pathways that include widespread adoption of
918   green electricity. On the one hand, lower NO_x emissions can reduce photochemical
919   production of ozone, but on the other they will reduce the chemical loss of ozone.
920   Future climate change itself will also change air quality through many different
921   pathways (Doherty et al., 2017) including that increased surface temperature will
922   increase BVOC emissions and reduce stomatal deposition of ozone (Vieno et al,
923   2010). For example, Stewart et al. (2003) suggested a 1°C temperature rise would
924   increase summer isoprene emissions in the UK by 14%. Most of these effects are
925   difficult to quantify, and even where known are currently beyond incorporation at the
926   high spatial resolution required in regional ACTMs. Hence the simulations presented
927   here are based on current meteorology and emissions in order to concentrate
928   directly on the impact of the forest planting scenarios.
929
930   In addition, a substantial proportion of both ozone and $PM_{2.5}$ in the UK is
931   transboundary in origin (AQEG, 2021, 2013). If continental Europe and elsewhere
932   adopt similar large-scale afforestation, it might be anticipated that the perturbations
933   to UK ozone and $PM_{2.5}$ simulated here would be magnified.
934
935   Increases in ozone are detrimental to crops and vegetation (AQEG, 2021, 2013;
936   Emberson, 2020). Therefore, any increase in ozone, however small, leads to
937   increased adverse human health and ecosystem impacts. Conversely, any decrease
938   in $PM_{2.5}$ will lead to a decrease in health impact. Table 4 shows that the relative
939   decreases in UK population-weighted annual mean $PM_{2.5}$ concentrations are greater
940   than the relative increases in UK population-weighted annual mean ozone
941   concentrations across the four scenarios, and Figures 8 and 12 show that the
942   changes in both predominantly occur in the areas of the UK with greater population
943   density. Given the consensus that health burdens from $PM_{2.5}$ are greater than from
944   ozone (Cohen et al., 2017), our simulations suggest there could be a net decrease in
945   health burden overall in the UK from these scenarios. However, net health burden is
946   very sensitive to the details of the concentration changes in annual and daily means
947   in locations where people live and on assumed concentration response functions for
948   the full range of adverse health outcomes to both pollutants. Similarly, for
949   quantification of ecosystem impacts from air quality. This detail is well beyond the
950   purpose of this study, whose aim is to present a first simulation of the scale of
951   changes in UK air quality associated with potential planting scenarios of certain tree
952   species being considered for afforestation. Nevertheless, our study shows it is
953   essential that assessment of additional forest planting on air quality uses
954   atmospheric chemistry transport models that account for the multiple ways forests
955   can impact on atmospheric composition.
956

5. Conclusions

The extent, geographical distribution and species of bioenergy plantations and afforestation that the UK will implement as part of measures to achieve net-zero greenhouse emissions has yet to be resolved. Our study presents a step at coupling information on tree species planting suitability and other planting constraints with data on UK-specific BVOC emissions and tree canopy data to simulate via the WRF-EMEP4UK high spatial resolution atmospheric chemistry transport model the impact on UK air quality of four potential planting scenarios. We deliberately investigate maximum possible planting scenarios: the additional areas of forest in our scenarios exceed current suggestions for new bioenergy and afforestation land cover in the UK by a factor 2.0 to 2.7.

Our simulations show that the changes in isoprene and total monoterpene emissions from such widespread new planting of trees slightly increase UK averaged annual mean surface ozone concentrations by 1.0 ppb or 3% relative to baseline for the highest BVOC emitting tree species such as eucalyptus. Increases in ozone reach 2 ppb in summer when BVOC emissions are greatest. Even planting of minor BVOC emitting species such as alder result in small increases in ozone. In contrast, the additional planting scenarios lead to reductions in UK averaged annual mean $PM_{2.5}$ regardless of the tree species planted, ranging from -0.2 $\mu g\ m^{-3}$ (-3%) for Sitka spruce to -0.5 $\mu g\ m^{-3}$ (-7%) for aspen. The decreases in annual mean $PM_{2.5}$ are of greater relative magnitude than the relative increases in annual mean ozone. Reductions in $PM_{2.5}$ were greatest in late spring, coinciding with the seasonal maximum in UK $PM_{2.5}$ concentrations, and least in summer, coinciding with the period of maximum monoterpene emissions. The simulations show that the additional depositional sink for $PM_{2.5}$ from the additional forest canopy more than offsets additional secondary organic aerosol (SOA) formation. We show how locally-relevant tree species data, BVOC emissions potentials and meteorology should, in principle, improve the simulations by atmospheric chemistry transport models of the complex interactions between additional forest planting and impacts on surface atmospheric composition.

Access to code

This study used two open-source global models: the European Monitoring and Evaluation Programme Meteorological Synthesizing Centre – West atmospheric chemistry transport model (EMEP MSC-W, 2020, version 4.34, source code available at https://doi.org/10.5281/zenodo.3647990) and the Weather Research and Forecasting meteorological model (WRF, version 4, https://www.wrf-model.org doi:10.5065/D6MK6B4K, (Skamarock et al., 2021)). The ECS-DSS model is available at http://www.forestdss.org.uk/geoforestdss/.

Data availability

The annual and monthly emissions and concentration data are in the Supplementary Material.

Author contributions

GP designed the study, GP provided experimental VOC and LAI data. SB provided tree species suitability data from ECS-DSS. EC provided spatial data conversions for model runs and spatial data calculations. MV provided model data using EMEP4UK. GP, MRH, MV contributed to the data interpretation. GP prepared the initial manuscript with input from MRH. GP, MRH, MV, EC, JD, SB, JILM contributed to the discussion, writing and editing of the article.

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Competing interests

The contact author has declared that neither they nor their co-authors have any competing interests.

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