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Functional traits of urban trees: air pollution mitigation potential

Rüdiger Grote^{1*}, Roeland Samson², Rocío Alonso³, Jorge Humberto Amorim⁴, Paloma Cariñanos⁵, Galina Churkina⁶, Silvano Fares⁷, Didier Le Thiec⁸, Ülo Niinemets⁹, Teis Norgaard Mikkelsen¹⁰, Elena Paoletti¹¹, Abhishek Tiwary¹², and Carlo Calfapietra^{13,14}

In an increasingly urbanized world, air pollution mitigation is considered one of most important issues in city planning. Urban trees help to improve air quality by facilitating widespread deposition of various gases and particles through the provision of large surface areas as well as through their influence on microclimate and air turbulence. However, many of these trees produce wind-dispersed pollen (a known allergen) and emit a range of gaseous substances that take part in photochemical reactions – all of which can negatively affect air quality. The degree to which these air-quality impacts are manifested depends on species-specific tree properties: that is, their “traits”. We summarize and discuss the current knowledge on how such traits affect urban air pollution. We also present aggregated traits of some of the most common tree species in Europe, which can be used as a decision-support tool for city planning and for improving urban air-quality models.

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Urban forests and trees contribute to human health and well-being through a multitude of services, primarily by removing pollutants and particulate matter (PM) from the air (Weber 2013). This benefit has encouraged urban planners to increase the areal extent of

green space in cities (Figure 1), including an expansion of street tree populations (Llausàs and Roe 2012). However, at present, several key knowledge gaps – including whether these air-quality benefits are species-specific, whether the improvements in air quality come at the expense of other ecosystem services, and how to select the most suitable tree species for particular settings – complicate the realization of this initiative.

Pollution removal by plants occurs through a combination of two pathways: deposition to plant surfaces and/or stomatal uptake. Deposition includes filtering of air by foliage or bark and, in cases of reactive air pollutants such as ozone (O₃), also the decomposition in the gas-phase that is triggered by plant-emitted substances (Kurpius and Goldstein 2003; Janhäll 2015). Apart from air pollution concentrations and meteorological conditions, deposition rates depend on three types of resistance: (1) air movement in the crown space, (2) transfer through the boundary layer adjacent to surfaces, and (3) absorption capacity of surfaces themselves, which also depend on stomatal conductance (Wesely and Hicks 2000). These resistances are controlled at different scales by the following vegetation properties: community organization (eg single trees, green corridors, parks, and forests), canopy structure (crown size, shape, and density), and foliage characteristics (eg leaf shape, surface properties, and physiology). Here, we focus on the species-specific tree properties (ie traits) that determine canopy and foliage interaction with major air pollutants, mainly in the context of European cities. In addition, we compare the traits related to air pollution mitigation with other traits that provide important ecosystem services (such as cooling by shading or by evapotranspiration), or that are vital for planting success (stress tolerance). However,

In a nutshell:

- Typical groups of urban tree species are identified for northern, central-eastern, and southern Europe; some species are ubiquitous whereas others occur only in specific regions
- The tree traits considered most relevant for air pollution mitigation are canopy density, foliage longevity, water-use strategy, and emission of reactive compounds
- A holistic evaluation of tree traits associated with air pollution mitigation is needed; for example, canopy density can be potentially beneficial for specific ecosystem services (eg shading, pollution removal) but detrimental for others (eg water use)
- There are crucial knowledge gaps associated with exacerbated emissions of pollen and volatile organic compounds, which may increasingly contribute to tropospheric ozone and particle formation under future climatic conditions

¹Institute of Meteorology and Climate Research (IMK-IFU), Karlsruhe Institute of Technology, Garmisch-Partenkirchen, Germany * (Ruediger.Grote@kit.edu); ²Research Group Environmental Ecology and Applied Microbiology (ENdEMIC), University of Antwerp, Antwerp, Belgium; ³Ecotoxicology of Air Pollution (CIEMAT), Research Center for Energy, Environment and Technology, Madrid, Spain; ⁴Air Quality Research Unit, Swedish Meteorological and Hydrological Institute, Norrköping, Sweden; ⁵Department of Botany, University of Granada, Granada, Spain; continued on last page



Figure 1. High BVOC-emitting *Platanus* trees planted for shading and cooling the boulevard La Rambla, in Barcelona, Spain.

consideration of feedbacks from plants to local air pollution concentrations – attributed primarily to the spatial locations of trees (both individuals and groups) or the complementary effects of different tree sizes and species composition – are beyond the scope of this article.

It is increasingly recognized that trees may also negatively affect air quality by emission of primary organic particles and biogenic volatile organic compounds (BVOCs) (Churkina *et al.* 2015). All parts of the tree produce such BVOCs, which vary in chemical properties and may have different impacts; BVOCs are already known to play a role in the formation of O₃, secondary organic aerosols, and PM in urban environments (Calfapietra *et al.* 2013a). As trees respond to elevated urban temperatures, pollutant levels, and atmospheric carbon dioxide (CO₂) concentrations projected under future climate-change scenarios, their associated BVOC emissions may intensify substantially (Calfapietra *et al.* 2013b). In addition, primary organic particles such as pollen may act as allergens and are possibly more potent in combination with other urban pollutants (Beck *et al.* 2013). Because the importance of BVOCs and pollen to air quality, especially in urban settings, has only recently been acknowledged, urban planners have yet to fully incorporate these features into tree selection criteria.

■ Abundance of street tree species

Before discussing tree and leaf traits, we provide a brief overview of species abundance of trees in streets, parks, and gardens in European cities. By joining inventories from southern (Chaparro and Terradas 2009; Soares *et al.* 2011), northern (Sæbø *et al.* 2003; Sjöman

et al. 2012), and central (Halajova and Halaj 2014) Europe, we find that some tree species emerge as highly abundant in most areas (eg *Tilia* spp) or at least frequently present (*Acer* and *Platanus*). Others differ in their regional importance; for instance, *Pinus* species decrease in abundance from south to north whereas *Prunus* species are distributed the other way round (Table 1). Our list includes only those species that contribute to at least ~1% of the total tree numbers within cities of any of the three aforementioned regions. Although similar to Yang *et al.*'s (2015) compilation, taken from globally distributed inventories, our version shows a greater abundance of *Tilia* species and *Aesculus hippocastanum*. Compared to the global inventories, *Ulmus* is underrepresented because its populations were reduced by Dutch elm disease in the 1970s and 1980s. Across Europe, broadleaved trees are more common as street trees than conifers, which usually do not exceed 1% of the total tree population. The most abundant evergreen trees are *Pinus* spp; while *Picea abies* and *Pseudotsuga menziesii* occur occasionally in central and northern regions, *Quercus ilex* as well as some varieties of *Brachychiton* often appear in the south.

■ Trees as air pollution sinks – the influence of species-specific traits

Air-flow impact by tree crowns

Most studies on the air-flow impacts of urban tree crowns have been conducted for street environments (eg Amorim *et al.* 2013). Constituent vegetation traits (eg crown geometry, foliage distribution) determine turbulence

Table 1. Abundance of urban tree species in European cities, classified by region

Latin name	Common name	Northern	Central-East	Southern
<i>Tilia</i> spp	Linden/ Lime	+++	+++	+++
<i>Acer</i> spp	Maple	+++	+++	+++
<i>Platanus</i> spp	Plane	++	++	+++
<i>Quercus</i> spp*	Oak	++	+++	+
<i>Aesculus hippocastanum</i>	Horse chestnut	+	+	+
<i>Fraxinus</i> spp	Ash	+	+	+
<i>Pinus</i> spp	Pine	+	0	+++
<i>Prunus</i> spp	Cherry	++	+	0
<i>Populus</i> spp	Poplar	+	0	+
<i>Ulmus</i> spp	Elm	+	0	+

Notes: *only deciduous oaks considered. High = red, medium = green, unevenly distributed = white; +++, ++, + = among the top 3, 7, 11 species in the region; 0 = more than 1% of tree number. Data are from Chaparro and Terradas (2009), Halajova and Halaj (2014), Sæbø *et al.* (2003), Sjöman *et al.* (2012), and Soares *et al.* (2011).

properties such as deceleration or acceleration of wind, as well as qualitative changes in air flow. However, it is difficult to reach universal conclusions on the local air-quality response to modified ventilation patterns because of the complex feedbacks between tree traits, “urban street canyon” configurations, and local meteorological conditions (Amorim *et al.* 2013). For example, low turbulence within a dense canopy, which increases the residence time of the air, favors chemical reactions between reactive gaseous pollutants and emitted BVOCs simply because there is more time to react. These conditions may lead not only to enhanced deposition of O₃ but also to increased concentrations of nitrous oxide (N₂O) and carbon monoxide (CO) (Harris and Manning 2010). Furthermore, deposition and emission properties depend on multiple plant-specific traits that vary during the season as well as during plant development.

Capturing and holding air pollutants

The majority of gaseous and particle deposition happens at the leaf surface, particularly under conditions where the stomata are closed (see below). Particle deposition in urban woodlands has been extensively studied. For example Brantley *et al.* (2014) estimated a reduction of total black carbon (fraction of PM ≤ 2.5 μm consisting of pure carbon particles) by *Acer* and *Quercus* of about 12%. However, there was a 10–20 fold difference between tree species (reviewed by Janhäll 2015), and many reasons have been suggested for this. For instance, the complexity of leaf structure is positively correlated with potential deposition (Beckett *et al.* 2000), and the presence of hairs or waxes on the leaf surface (Figure 2), which differs considerably between species, will also have a major influence (Kardel *et al.* 2012). Waxes have been found to almost double PM deposition in *Tilia* compared to *Platanus* (Dzierzanowski *et al.* 2011). This is partly attributed to the influence of leaf wettability, since considerable amounts of PM can be removed by reaction with wet surfaces, and the rate of deposition increases with the occurrence of waxes, salts, and ions (Altimir *et al.* 2006). If the pollutant is water soluble, as in the case of nitrogen dioxide (NO₂) or sulfur dioxide (SO₂), direct dissolution in a water film on the plant surface is also possible. Measurements suggest that particulates may be encapsulated – and thereby immobilized – within the wax layer during the growing season (Hofman *et al.* 2014).

Uptake of pollutants – other than those that merely bind to or are destroyed at the outer surface – into leaves occurs through the stomata; these pores can open and close, thereby regulating the intercellular concentration of CO₂ and thus controlling photosynthesis while preventing excessive water loss. In turn, stomatal uptake depends on photosynthetic activity and turgor pressure, which are determined by environmental variables. For instance, stomatal uptake in a Mediterranean evergreen

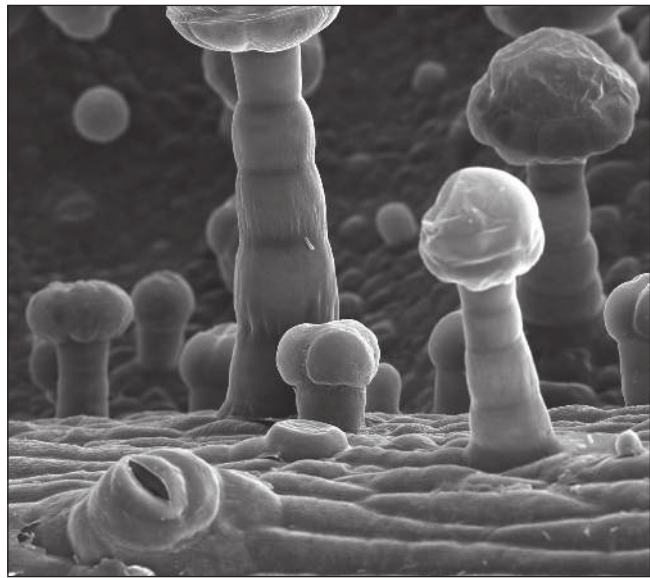


Figure 2. Lower leaf surface of black walnut (*Juglans nigra*), with trichomes (hairs that enhance particle capture) and stomata (electron microscope: ZEISS962 SEM).

forest was considerably higher in spring, when water supply and vapor pressure were greater than during summer (Fares *et al.* 2014). Differences between tree species can be attributed to BVOC emissions that influence non-stomatal deposition processes (Fares *et al.* 2008), as well as to different water-use strategies: anisohydric tree species, which keep their stomata open over extended periods, are more efficient at pollution uptake than isohydric species, which tend to close their stomata early in response to decreasing water availability. Thus, it is not only the short-term response to drought that affects O₃ uptake but also the selection of anisohydric species such as *Populus* or deciduous oaks – in contrast to isohydric species such as *Pinus* or *Platanus* trees – that increases stomatal uptake.

Pollutant uptake through stomata is high as long as the respective compounds are quickly removed from the intercellular spaces. For example, O₃ and NO₂ are almost immediately metabolized, which means that the uptake increases with increasing outside concentrations as long as photosynthesis and membrane permeability are not seriously damaged by the inflow of pollutants. The effectiveness of leaf defense mechanisms can therefore also be considered as a species-specific trait affecting deposition. In the case of O₃ and nitrogen oxides (NO_x), the primary mechanism is the detoxification potential in the apoplast (extracellular space that surrounds the cell contents); as for SO₂, the transport resistances within cells and the ability to counteract changes in pH are crucial (Dizengremel *et al.* 2009).

Using physiological as well as anatomical species-specific traits, Nowak and Crane (2000) developed a model to calculate deposition rates of SO₂, NO_x, CO, O₃, and PM per leaf area and per tree from climatic and air pollution boundary conditions. This model has been

Table 2. Estimated removal of gaseous pollutants and particles as well as ozone-forming potential for several urban trees in Newcastle-upon-Tyne, UK

Latin name	LAI	PM10	GP	OFP
<i>Betula populifolia</i>	2.0	88	388	23
<i>Quercus phellos</i>	2.3	200	1392	12277
<i>Platanus x acerifolia</i>	2.4	181	619	5490
<i>Prunus serotina</i>	2.4	100	574	20
<i>Acer pseudoplatanus</i>	2.8	170	997	379
<i>Liquidambar styraciflua</i>	3.6	70	299	2827
<i>Tilia cordata</i>	3.9	76	520	0
<i>Fraxinus pennsylvanica</i>	4.1	97	593	21
<i>Aesculus hippocastanum</i>	5.6	320	2268	861
<i>Fagus sylvatica</i>	6.1	549	2934	441
<i>Picea abies</i>	9.8	224	4335	11648

Notes: Gaseous pollutants (GP) = SO₂, NO_x, CO, and O₃. Particles = PM10; ozone-forming potential = OFP. Data from Tiwary et al. (2016). The values were calculated with the UFORE model (Nowak and Crane 2000) based on local weather conditions and air-quality data and on a tree inventory of the city center. The leaf area index (LAI, expressed as one-sided leaf area [square meters] per unit ground area [square meters]) is the average for a tree of the respective species within this sample. For the OFP, we weighted the calculated biogenic emissions according to Benjamin and Winer (1998) as described in the text (deciduous in blue, coniferous in green; PM, GP, and OFP values all in grams per tree per year).

applied in several European case studies to evaluate the impact of different tree species on air pollution removal (and other ecosystem services) under the respective environmental conditions. Results from such a simulation for Newcastle-upon-Tyne, UK (Table 2; Tiwary et al. 2016), demonstrate that deposition is not necessarily correlated with leaf area index and highlight the importance of differentiated analyses. Yang et al. (2015) followed a simpler approach for an integrated analysis, by ranking seven species-specific properties relevant to deposition (foliage longevity, growth, height, crown density, leaf complexity, leaf length, and other leaf characteristics) from 1 to 3 and calculating the sum of all values. These sums are shown for the most relevant tree species in our study (Figure 3).

■ The other side of the bi-directional exchange

Pollen and other biological particles

Biogenic particulate matter (BPM) is emitted mainly by fungi (spores) and flowering plants (pollen). BPM size ranges from 0.1–5 μm for small fungal spores to about 90 μm for large pollen grains. Despite its size, most of this material can be deposited far from the emission source. Some particles (ruptured pollen grains) exist as lower-micron-sized fragments but their allergenic

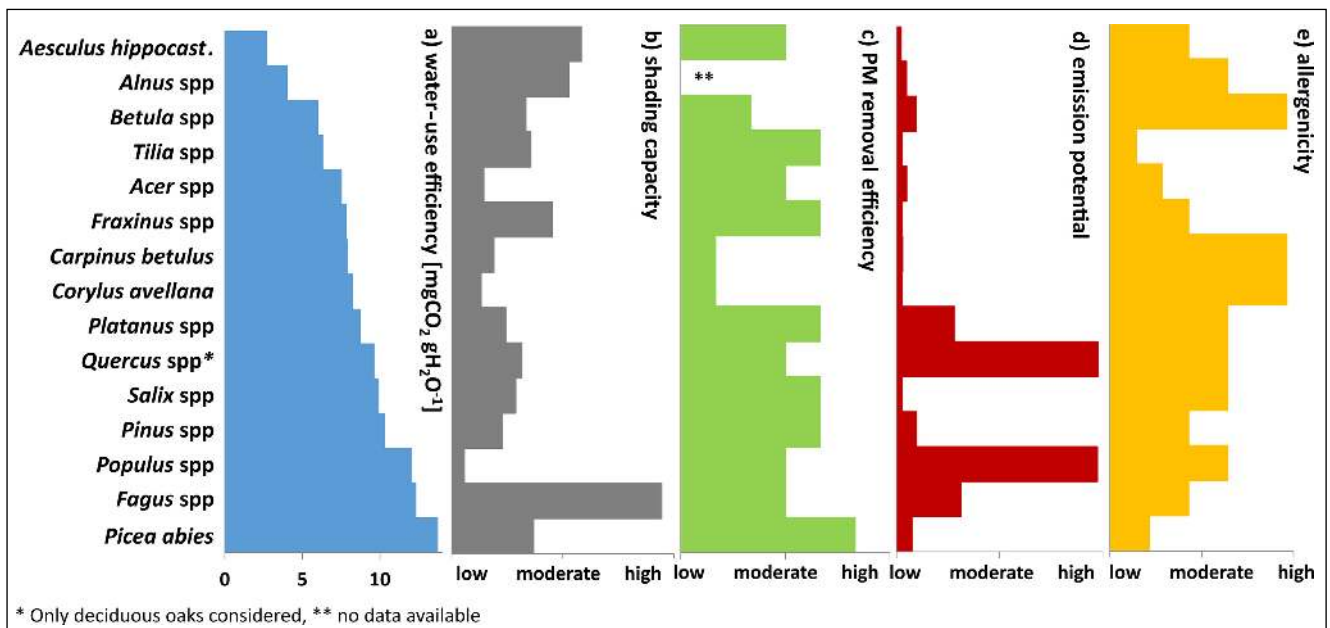


Figure 3. Selection criteria listed for the most common urban tree species. (a) Water-use efficiency (WUE) taken from Wang et al. (2013) and other sources; (b) shading capacity calculated as leaf area index × relative leaf abundance throughout the year (based on Tiwary et al. 2016) × crown width/tree height (based on <https://www.horticopia.com/hortpip/index.shtml>); (c) PM removal efficiency based on relative numbers given by Yang et al. (2015) as described in the text; (d) lumped isoprene and monoterpene emission potentials under standard conditions (based on units of micrograms per grams dry weight per hour; Karl et al. 2009); and (e) allergenicity calculated as pollination duration × intensity × toxicity (Cariñanos et al. 2016). All values except WUE are scaled between lowest and highest values obtained in the dataset.

activities remain intact (Cariñanos *et al.* 2001). Although pollen emissions are considered as one of the key ecosystem disservices of urban vegetation, the specific allergenicity of pollen grains is seldom investigated and is therefore not yet considered as a selection criterion in urban planning (Cariñanos *et al.* 2016). Emission intensity is related to temperature and wind, and allergenicity of pollen – despite being species-specific – is modified by atmospheric pollutants. The latter is triggered by larger amounts of allergenic proteins or by changes in lipid composition under polluted conditions (Beck *et al.* 2013). For city planners, this might represent a dilemma since the general practice of placing trees as closely as possible to the pollutant source in order to improve pollutant removal efficiently may at the same time increase the allergenicity of pollen grains. When selecting trees, it is therefore important to consider and assess these potential allergenic impacts (Figure 4).

Gaseous emissions of volatile organic compounds

The BVOC release from vegetation is governed by environmental conditions (eg sunlight, temperature, and water availability) and is highly species-specific; for example, many urban trees such as *Populus* and *Quercus* emit large amounts of isoprene (Karl *et al.* 2009; Churkina *et al.* 2015). Although airborne BVOCs can also be deposited through stomatal uptake and surface degradation (Nguyen *et al.* 2015), this process is considered to be negligible. Given a sufficiently high level of NO_x , isoprene can contribute substantially to O_3 formation in the atmosphere, to such an extent that it surpasses the O_3 deposition capacity of the trees (Calfapietra *et al.* 2013a). We illustrate this concept by calculating the “ozone-forming potential” (OFP), introduced by Benjamin and Winer (1998), for selected urban tree species in Newcastle-upon-Tyne, as well as by estimating the capacity of those trees to remove gaseous pollutants and particles from the air (Table 2). These emissions (not shown) are multiplied with a “maximum incremental reactivity” that is compound-specific (9.1 g of O_3 per gram of isoprene, 3.8 g of O_3 per gram of monoterpene) and that indicates the amount of O_3 produced under optimum climatic and air chemistry conditions. Considering that about one-half of the indicated gaseous deposition is attributed to O_3 , it requires the realization of only about 5% of the OFP of *Quercus* or *Platanus* to compensate for the O_3 uptake of these tree species. Furthermore, the formation of secondary organic aerosols and thus PM is related to the presence of monoterpenes and sesquiterpenes, which are emitted in particular by genera such as *Pinus*, *Betula*, and *Aesculus* (Derwent *et al.* 1996). Regional modeling has shown that switching between an isoprene- and a monoterpene-emitting scenario may lead to major changes in tropospheric O_3 concentration (Fares *et al.* 2013).



P. Cariñanos

Figure 4. Cypress (*Cupressus*) produces high amounts of pollen, which is one of the leading causes of respiratory allergy in southern Europe.

Flowering and plant stress induce the emission of various oxygenated compounds, as well as some benzenoids and terpenoids (Misztal *et al.* 2015). Such emissions play important roles in mitigating plant oxidative stress, establishing communication networks with insects, and mediating photochemical reactions. Consequently, heavily flowering plants may not always be the preferable choice for parks and gardens (Niinemets and Peñuelas 2008). Other gaseous emissions from trees are relatively small and so contribute negligibly to air pollution. For example, several reactive gases such as nitric oxide (NO), CO, N_2O , and methane (CH_4) can be synthesized in an ultraviolet-induced photochemical reaction at the leaf surface or may be emitted from woody tissue (Fraser *et al.* 2015). CO production might also be stimulated by abiotic stresses, given that CO can alleviate oxidative damage by up-regulating antioxidant defenses (He and He 2014).

■ Tree traits – a moving target

The spatial and temporal plasticity of traits

In deciduous species, time-dependent variations in traits occur during the growing season as new leaves develop, mature, and age. In particular, specific leaf area, leaf

nitrogen content, photosynthetic activity, and BVOC emission capacity increase in developing foliage; remain relatively stable in mature, non-senescent tissue; and rapidly decline in senescing leaves (Wilson *et al.* 2000). During these stages, the composition of emission compounds is also changing, which might be related to specific requirements regarding communication or stress mitigation (eg to attract pollinators or predators of parasites) (Niinemets *et al.* 2013). In the case of evergreens, analogous changes occur during leaf development, and the maximum foliage physiological capacities are typically observed in spring and summer, depending on water availability (Gratani and Bombelli 2000). In addition to physiological changes, wettability of young leaves is higher than that of mature leaves but also increases in older leaves, reflecting the time-dependent accumulation of cuticular lesions (Wang *et al.* 2013).

In general, leaf structural and physiological traits also vary within plant canopies, reflecting the acclimation of foliage properties to light gradients; however, the degree of variation primarily depends on the species and on plant functional types (Niinemets 2015). Moreover, in highly polluted areas, excessive deposition of particles onto leaf surfaces could effectively block light that would otherwise be available for photosynthesis (Delegido *et al.* 2014). Since concentrations of urban air pollutants, especially vehicular-traffic-derived PM, decrease with increasing height (Hofman *et al.* 2013), vertical gradients of traits in urban trees are likely to be different from those in trees within less disturbed environments.

Impacts of environmental changes

Climate scenarios project increasing atmospheric CO₂ concentrations and air temperatures, which will compound already elevated concentrations and temperatures experienced by trees growing in urbanized environments. By the end of this century, global atmospheric CO₂ concentrations could approach 800 parts per million and temperatures will likely rise by 1.7° to 4°C. The frequency and severity of heat waves are also expected to increase, which in turn amplifies drought stress but favors O₃ formation in cities (Sicard *et al.* 2013).

Higher temperatures are known to extend the length of the growing season. In spring, elevated air temperatures accelerate bud burst, flowering, and stem elongation; in autumn, they may postpone litter fall, unless adverse effects cause premature senescence (Cleland *et al.* 2007). The urban heat island effect is therefore altering the length of the growing season within cities as compared with rural areas (Jochner and Menzel 2015). In addition, other factors, such as atmospheric CO₂, air pollutant concentrations, and nitrogen deposition, change along the urban–rural gradient (Calfapietra *et al.* 2015). BVOC emissions are expected to rise with higher temperatures and decrease with elevated CO₂ (Lahr *et al.* 2015). The latter effect depends on nutrient availability. Well-fertilized plants – which are common

in gardens and urban green spaces – are expected to enhance emissions in response to elevated CO₂ (Sun *et al.* 2013), which also increases leaf dry mass, leaf area, and water-use efficiency (WUE) but decreases stomatal conductance and stomatal density. These changes vary considerably between species (Woodward *et al.* 2002). Plants respond to drought by adjusting stomatal conductance, thereby improving WUE in a way that resembles the response to higher CO₂ concentrations. Furthermore, intensive drought is inducing leaf shedding; decreasing leaf growth, size, and branching; and increasing cuticle thickness and wax abundance. Interestingly, such changes in leaf properties might also impose a feedback on reflectance, thereby indirectly affecting leaf temperatures and, in turn, WUE. In the same way, elevated temperatures and CO₂ levels stimulate the production and growth of pollen (Hamaoui-Laguel *et al.* 2015) and – more importantly – their allergenicity (Ahlholm *et al.* 1998).

Although pollen viability decreases with increasing air pollution, O₃ is another agent that has been found positively correlated to the allergenicity of pollen, supporting the notion that an unhealthy link exists between air pollution and allergen toxicity (Beck *et al.* 2013). High O₃ concentrations also decrease photosynthesis and thus stomatal conductance, and impair plants' ability to withstand drought stress (Alonso *et al.* 2014; Hoshika *et al.* 2014). In contrast, BVOC emissions are initially enhanced under O₃ exposure, but chronic exposure leads to decreased emissions (Calfapietra *et al.* 2013b). Other air pollution impacts are similarly complex albeit generally less intense. For example, NO_x has been recognized as either detrimental (due to its oxidative impact) or beneficial (as a potential source of nutrients) for plant development. Moreover, CO affects diverse physiological processes in plants, from seed germination and dormancy to stomatal closure and regulation of multiple environmental stresses (He and He 2014).

Climate change will shift species' geographic ranges, leading to increased abundance of “new” tree species or varieties, mostly those that have adapted to warmer conditions (Holmes *et al.* 2013). This has important implications for estimating the impacts of urban trees, because these species – as compared with indigenous plants – may exhibit different characteristics (including growth patterns and leaf longevity), emit unfamiliar allergens, and release more, or more reactive, BVOCs.

■ The way forward

In this review, we have concentrated on tree traits that directly influence air pollution; in doing so, we neglected the numerous other ecosystem services and disservices provided by urban forests (eg Escobedo *et al.* 2011). Since improving air quality is not the only objective of city managers, other tree properties such as heat mitigation potential, stress tolerance, water consumption, and aesthetic beauty deserve consideration

(Salmond *et al.* 2016). Some traits inhibit services (eg a high deposition capacity improves air quality but impairs plant health) while other traits reinforce services (eg increased leaf area is positively correlated with cooling the environment and reducing air pollutants). However, since ecosystem services are functions of many traits, the suitability of a tree species for a particular combination of demands is highly case-specific. The relative trade-off or synergistic benefit of different traits also depends on the trees in the immediate vicinity and the importance of the respective ecosystem service. For instance, *Aesculus* might be favored for its shading ability but its abundance is restricted to sites with ample water supply because its WUE is low. *Pinus* species tend to be favored in southern Europe, because they are efficient at removing pollution and are relatively stress tolerant; in particular, they are drought resistant, a trait that is comparatively less relevant in northern regions. Stress tolerance may be the first selection criterion in polluted areas, even if the associated gain in ecosystem services is small. Ecosystem services are sometimes indirectly related to each other; for instance, services that effectively cool the microclimate help to curb commercial and residential energy consumption, leading to reduced anthropogenic emissions of greenhouse gases. The complexity of ecosystem service valuation has prevented holistic investigations for specific cities or regions, although model approaches that combine at least some aspects into an integrated analysis are available. We encourage the further development of such models and the collection of relevant data in urban areas, with a particular focus on tree physiological responses to changing environmental conditions.

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⁶Institute of Advanced Sustainability Studies, Potsdam, Germany;

⁷Council for Agricultural Research and Economics, Research Centre for the Soil–Plant System, Rome, Italy; ⁸Ecologie et Ecophysiologie Forestières, UMR INRA, Champenoux, France;

⁹Department of Plant Physiology, Estonian University of Life Sciences, Tartu, Estonia; ¹⁰Department of Environmental Engineering, Technical University of Denmark, Kongens Lyngby, Denmark;

¹¹Sustainable Plant Protection Institute, National Research Council, Florence, Italy; ¹²Faculty of Engineering and the Environment, University of Southampton, Southampton, UK;

¹³Institute of Agro-Environmental & Forest Biology, National Research Council, Rome, Italy; ¹⁴Global Change Research Centre, Brno, Czech Republic