# Functional traits of urban trees in relation to their air pollution mitigation potential: A holistic discussion

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## Abstract

In a world of increasing urbanization, air pollution mitigation is one of the most important issues of city planning. Urban trees are of central importance for this issue because they facilitate the deposition of various gases and particles and affect microclimate and air turbulence. In addition, many trees emit allergens as well as a range of gaseous substances which take part in photochemical reactions. The degree to which direct and indirect effects are manifested depends on species-specific tree properties - or traits. Here we summarize and discuss the current knowledge on how such traits impact air pollution. We also present aggregated traits of the most common tree species in Europe that can be used as a decision support tool for city planning and for improving the parameterization of urban air quality models.

## "In a nutshell"

- Typical groups of urban tree species are designated for northern, central-east, and southern Europe. Some species are ubiquitous while others occur only in specific regions.
- The dominating tree traits regarding air pollution are canopy density, foliage longevity, water use strategy, and emission potential. Particularly the emission of compounds which contribute to ozone and particle formation might get more important under future climatic conditions.
- Some trees traits such as foliage density are positively affecting specific ecosystem services (e.g. shading, pollution removal), but are detrimental for others (e.g. water use). Traits need thus to be combined to groups that are indicative for services.

## Introduction

Urban forests and trees contribute to human wellbeing due to a multitude of services, of which the most studied is the positive effect on air quality that is expected to improve human health by removing gaseous air pollutants and particulate matter (PM) from the air (Weber 2013). Therefore, a prominent measure in urban development plans is to increase the number of street trees. But how does the positive impact of these trees on local air quality depend on species specific traits? Are potential tradeoffs connected to these traits that might decrease other environmental services? How can the best suitable tree species be selected?

Pollution removal by plants follows two pathways: deposition at the foliage surface or stomatal uptake. Dry and wet deposition includes scavenging of pollutants by the leaf or bark and - in the cases of reactive air pollutants such as ozone – also in the gas-phase due to emitted reactive substances (Wesely and Hicks 2000; Janhäll 2015). Apart from air pollution concentrations and

meteorological conditions, stomatal and non-stomatal deposition rates depend on three bulk resistances: air movement in the crown space, transfer through the boundary layer adjacent to surfaces, and absorption capacity of surfaces themselves which includes stomatal conductance (Wesely and Hicks 2000). These resistances are controlled by vegetation properties on different scales: community (e.g. single trees, green corridors, parks, and forests), canopy (i.e. crown size, shape and density) and foliage structure (including leaf shape, surface properties and physiology). This review focuses on the species-specific properties or traits that determine canopy and foliage interaction with major air pollutants in European cities. It neglects the feedback from the community scale.

Our work was stimulated by an increasing awareness that trees do also affect air quality by emission of primary particles and BVOCs (biogenic volatile organic compounds) (Churkina *et al.* 2015). BVOCs are already known to take part in the formation of ozone, secondary organic aerosol and PM (Fuentes *et al.* 2000). As BVOC emissions are likely to respond to future higher temperatures, pollutant and $CO_2$ concentrations, this trait has been suggested to play an increasingly important role for air quality in the future (Calfapietra *et al.* 2013). 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). These are relatively new findings that are not yet established as selection criteria by urban planners.

## ■ Street tree species abundance

Before discussing tree and leaf traits, a brief overview is given about what species are currently most abundant 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 Europe (Halajova and Halaj 2014), some tree species emerge as ubiquitously highly abundant (e.g. Linden) or at least frequently present (Maples and Plane trees). Others differ in their regional importance as for example pines decrease in abundance from the South to the North and Cherries are distributed the other way round **(Table 1)**. Our list considers only species that contribute at least about 1 % of the total tree number at any of the regions. It is similar to the one compiled from globally distributed inventories by Yang et al. (2015) but shows a higher importance of Linden (*Tilia* sp.) and Horse chestnut (*Aesculus hippocastanum*), the latter being a native European species despite it is now also planted in North America. Compared to the global inventories, Elms are underrepresented because their abundance has been largely decreased by the Dutch Elm disease in the 70's and 80's. All over Europe, broadleaved trees are more common as street trees than conifers. The most abundant evergreen trees are Pines, while Norway spruce (*Picea abies*) and Douglas fir (*Pseudotsuga menziesii*)

occur frequently in central and northern regions, and Holm oak (*Quercus ilex*) as well as some varieties of Kurrajong (*Brachychiton populneus*) and Privet (*Ligustrum lucidum*) often appears in the South, but do not exceed 1% of the total tree number.

## ■ How tree traits affect deposition of pollutants

### *Air flow impact by tree crowns*

The majority of studies on air flow impacts of urban tree crowns have been conducted for street environments (Freer-Smith *et al.* 2005; Pugh *et al.* 2012; Gromke and Blocken 2015). Constituent vegetation traits (i.e. crown geometry, foliage distribution, etc.) are determining turbulence properties such as deceleration or acceleration of wind, as well as qualitative changes in the flow (Gromke and Blocken 2015). In particular, trees with dense crowns are prominent obstacles to airflow in poorly ventilated streets and also reduce vertical air exchange (Pugh *et al.* 2012). Only very recently, however, an index termed 'pollution flux potential' (PFP) has been suggested that combines canopy density (expressed as leaf area per ground surface LAI and inter-annual leaf cover IAL) with specific net exchange properties trying to merge different traits into one number (Tiwary *et al.* 2016). The calculation is based on literature derived annual BVOC emission ($P_{emit}$) vs. pollutant deposition ($P_{dep}$ ) estimates, both expressed as kg yr-1, with $P_{dep}$ including ozone, $SO_2$ $NO_2$, CO and PM less than 10 um and $P_{emit}$ isoprene, monoterpenes and other BVOCs.

We calculated this index for some of the most common street trees in Europe for which we were able to gather the necessary data (**Table 2**). From this, it can immediately be derived that coniferous trees tend to remove more pollution from the air because they feature a continuous canopy cover, a high LAI, and relatively large deposition velocities. However, the variability between species groups is large as are the uncertainties in the different traits. For example low turbulence within a dense canopy favors contactless deposition of reactive gaseous pollutants in the presence of reactive compounds (Kurpius and Goldstein 2003). In contrast, CO deposition decreases under such conditions because a low reactive air pollutant can only be oxidized by radicals that are scavenged by BVOCs (Baumgardner *et al.* 2002). Furthermore, as will be discussed below, deposition and emission properties depend on multiple plant-specific traits that are variable during the season and plant development.

*Capturing and holding air pollutants with leaves and needles*

The majority of gaseous and particle deposition happens at the leaf surface **(Figure 1)**, particularly under conditions where stomata are closed. Particle deposition on urban woodlands has been extensively studied (see e.g. reviews by Janhäll 2015). Brantley et al. (2014) estimated the reduction of PM by *Acer* and *Quercus* at about 12 %. Assuming that PM reduction ability and PFP are equivalent, this value might be at the lower edge of deposition capacities **(Table 2)**. Particularly the genera with more complex leaf structures such as *Pseudotsuga* for evergreen or *Fraxinus* for deciduous trees, are able to take up significantly higher quantities of PM than other genera with similar or higher leaf area (Beckett *et al.* 2000; Freer-Smith *et al.* 2005). In addition, particle deposition depends on the occurrence of hairs or the availability of waxes at the leaf surface which are different between species and in case of waxes have been found to almost double PM deposition in *Tilia* and *Acer* compared to *Platanus* (Dzierzanowski *et al.* 2011). This is partly attributed to the influence on leaf wettability: considerable amounts can be removed by reaction at wet surfaces with the rate of deposition increasing with the occurrence of waxes, salts, and ions (Altimir *et al.* 2006). If the pollutant is water soluble as for example $NO_2$ or $SO_2$, also direct dissolution in a plant surface water film is possible. Measurements also suggest encapsulation of particulates during the growing season in the wax layer, thereby immobilizing particles (Hofman *et al.* 2014).

*Stomatal uptake is driven by physiological properties*

Stomata regulate the intercellular concentration of $CO_2$ and thus control photosynthesis. In turn, stomatal uptake depends on photosynthesis activity and turgor pressure, which are determined by environmental variables. For example, stomatal uptake accounts for 20 - 90 % of total ozone deposition in a Mediterranean evergreen forest (Fares *et al.* 2014) while only for 20 – 50 % in a Danish conifer forest (Mikkelsen *et al.* 2004). The differences can be attributed to the dryer conditions in the Mediterranean climate but also to two different water use strategies: anisohydric tree species which maintain high conductance as long as possible are more efficient in pollution uptake than “water saving” 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 affect pollution uptake but also the selection of isohydric species such as *Pinus* or *Platanus* over those with anisohydric regulation (e.g. poplars and deciduous oaks) that defines water consumption. In addition, stomatal density is positively correlated with the capacity to sequestrate ozone from the atmosphere (Ollinger *et al.* 1997).

Pollutant uptake through stomata is high as long as the respective compounds are quickly removed from the intercellular spaces. For example, ozone and $NO_2$ are almost immediately metabolized,

which means that the uptake increases with outside concentrations in accordance with the Fick's law as long as photosynthesis and membrane permeability are not seriously damaged. Thus, also the effectiveness of defense mechanisms can be considered as a leaf trait modifying deposition. In case of ozone and NOx, this is particularly the scavenging potential in the apoplast, while for $SO_2$ mesophyll resistance and the capability to counteract pH changes are of primary importance (Dizengremel *et al.* 2009).

## ■ The other side of the bi-directional exchange

### *Pollen and other biological particles*

BPM are PM of biological origin emitted mainly by fungi (spores) and plants during flowering (pollen). Their size ranges from 0.1 - 5 µm of some fungal spores up to about 90 µm for large pollen. Despite its relatively large size, however, most of this material can be deposited far from the emission source. Part of the BPM appear in lower micron sized fragments due to the rupture of pollen grains while their allergenic 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 thus not yet considered as a selection criteria for urban planning (Cariñanos *et al.* 2015). However we know that 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 to place trees as closely as possible to the pollutant source in order to increase pollutant removal efficiently may at the same time increase the allergenicity of pollen grains. When selecting trees it is, therefore, important to assess and consider these potential allergenic impacts in order to avoid unsuitable combinations.

### *Gaseous emissions - the dawn of volatile organic compounds*

BVOC emissions are generally expressed as a function of environmental conditions that modify compound- and species-specific emission potentials. Although BVOCs can also be deposited through stomatal uptake and surface degradation (Nguyen *et al.* 2015), this process is considered to be small and thus generally neglected. Plant species strongly differ in their capacity to release BVOCs, for example, many urban trees such as *Populus* and *Quercus* are intensely emitting isoprene (Churkina *et al.* 2015). Given a sufficiently high NOx level, isoprene can significantly contribute to the formation of ozone in the atmosphere and this effect may be predominant over the capacity to sequester ozone.

Secondary organic aerosols and thus PM formation is more closely related to the presence of monoterpenes and sesquiterpenes, which are emitted by species such as *Pinus, Betula* and *Aesculus* (Derwent *et al.* 1996).

Flowering and plant stress induces the emission of various oxygenated compounds as well as some terpenoids (Xu *et al.* 2012). Such emissions are important to mitigate plant oxidative stress or to establish communication networks with insects but also take part in photochemical reactions. Having this in mind, heavily flowering plants may not always be the preferable choice for parks and gardens (Niinemets and Peñuelas 2008). Other gaseous emissions are small and thus play only a minor role in air pollution. For example NO is formed in a UV-induced photochemical reaction with nitric acid or nitrate at the leaf surface or can be emitted from woody tissue (Raivonen *et al.* 2009). Also CO production might be stimulated by abiotic stresses as CO can alleviate oxidative damage by up-regulating antioxidant defense (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 season as new leaves develop, mature and age. In particular specific leaf area, leaf nitrogen content, photosynthesis activity and BVOC emission capacity increase in developing foliage, after which they are 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 the specific requirements regarding communication or stress mitigation (e.g. to attract pollinators or parasite predators) (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 summer (Gratani and Bombelli 2000). In addition to physiological changes, wettability of young leaves is higher than that of mature leaves but it also increases in older leaves, reflecting time-dependent accumulation of cuticular lesions (Wang *et al.* 2013).

Leaf structural and physiological traits also vary within plant canopies reflecting the acclimation of foliage properties to light gradients (Van Wittenberghe *et al.* 2014; Niinemets 2015). Despite the generality of this feature, the degree of variation varies significantly among species and plant functional types (Niinemets 2015). Moreover, particle deposition at the leaf surface might reduce the light available to photosynthesis due to the creation of a shadow layer on the leaf surface in highly polluted environments (Delegido *et al.* 2014). It should be noted that concentrations of urban air

pollutants, especially traffic derived PM, decreases with height (Hofman *et al.* 2013). Thus, in urban trees, vertical gradients of traits are likely to be different than in natural environments.

*Impacts of environmental changes*

Climate scenarios suggest increasing atmospheric $CO_2$ concentrations and air temperatures, which will come on top of the already elevated concentrations and temperatures experienced by urban trees compared to those growing in more rural environments. Temperature will rise by 1.7 to 4 °C and global atmospheric $CO_2$ concentrations will approximately double from the currently 400 ppm by the end of this century. Additionally, frequency and severity of drought stress and heat waves are expected to increase, which in turn favors ozone formation in cities (Sicard *et al.* 2013).

A known effect of higher temperatures is an extension of growing season length. 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). Seasonality is thus already different inside than outside urban areas (Jochner and Menzel 2015). BVOC emissions are expected to rise with higher temperatures, and decrease with elevated [$CO_2$] (Possell and Hewitt 2011). However, the latter effect depends on nutrient availability, and well-fertilized plants - as common in gardens and urban green spaces - are expected to rather enhance emissions in response to elevated [$CO_2$] (Sun *et al.* 2013). Increasing [$CO_2$] also increases leaf dry mass and leaf area while decreasing stomatal conductance, stomatal density, and water use efficiency (WUE) - although these changes vary considerably between species (Woodward *et al.* 2002). Plants respond to drought by adjusting stomatal conductance, thereby increasing WUE in a similar way as in response to higher [$CO_2$]. Intensive drought is furthermore inducing leaf shedding, decreasing leaf growth, size and branching, and increasing cuticle thickness and wax abundance. Interestingly, such a change in leaf properties might also impose a feedback on reflectance thus indirectly affecting leaf temperatures and WUE (Monneveux and Belhassen 1996). Temperature and $CO_2$ also increases the amount and size of produced pollen (Bartra *et al.* 2007; Hamaoui-Laguel *et al.* 2015) as well as - more importantly - their allergenicity (Ahlholm *et al.* 1998).

Although the viability of pollen decreases with increasing air pollution, ozone is another agent that has been found to increase the allergenicity of pollen, giving rise to the assumption that an unhealthy link exists between air pollution and allergen toxicity (Bartra *et al.* 2007; Beck *et al.* 2013). High [$O_3$] also decreases photosynthesis and thus stomatal conductance, but chronic $O_3$ exposure impairs stomata and restricts their ability to close rapidly in response to drought (Hoshika *et al.* 2014). In contrast, BVOC emissions are initially enhanced under ozone exposure but chronic exposure leads to

decreased emissions (Calfapietra *et al.* 2013). Other air pollution impacts are similarly complex although generally less intense. For example NOx has been recognized as either detrimental (due to their oxidative impact) or beneficial (as a potential source of nutrients) for plant development (Takahashi *et al.* 2011). Also 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).

An indirect impact of imposed environmental changes is the increasing abundance of new tree species or varieties, mostly coming from warmer climates (Holmes *et al.* 2013). This has important implications for estimates about urban tree impacts because these species may have other growth patterns, leaf longevity, emit new allergens and – in particular – more reactive BVOC emissions than endogenous plants.

## ■ How to get on?

In this review we concentrated on tree traits that directly influence air pollution and neglected the numerous other ecosystem services and disservices (see e.g. 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, and beauty amongst others have also to be considered (Tiwary *et al.* 2016). Some of the underlying traits are acting in opposite directions for specific services as for example uptake capacity increases air quality but decreases plant health while others such as a leaf area are cooling the environment and at the same time reduce air pollutants **(Figure 2)**. It should also be mentioned that ecosystem services are sometimes indirectly related for example by modifying the microclimate and thus energy consumption which then reduces anthropogenic emissions. The complexity of the matter has prevented holistic investigations for specific cities or regions, although model approaches that combine at least some aspects into an integrated analysis are already available (Nowak *et al.* 2008). We encourage the further development of such tools, particularly the consideration of physiological responses to changing environmental conditions, including air pollution.

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Tables:

Table 1: Abundance of urban tree species in European cities classified by region (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)

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

* only deciduous oaks considered

Table 2: Potential for air pollution (PFP) removal calculated for selected deciduous (blue) and evergreen (green) urban tree species. *Picea abies* and *Pseudotsuga menziesii* are not abundant in southern cities but are added to illustrate the properties of conifers. (LAI = leaf area index, IAL = intra annual leaf cover, Pemit = emission influence calculated from emission factors for isoprene and monoterpenes, Pdep = deposition factor calculated from deposition velocity, PFP = plant flux potential (the higher the value, the better the impact on air quality; for calculation of Pemit and Pdep see Tiwary et al. (2016)). The higher the PFP, the higher is the removal potential.

| Latin name | LAI | IAL | Pemit | Pdep | PFP |
|---|---|---|---|---|---|
| *Acer sp.* | 2.8 | 0.8 | 0.2 | 1.2 | 1.7 |
| *Fraxinus sp.* | 4.1 | 0.6 | 0.1 | 0.7 | 2.1 |
| *Populus sp.* | 3.5 | 0.6 | 1.8 | 0.9 | -2.0 |
| *Quercus sp.* * | 2.3 | 0.6 | 1.5 | 1.6 | 0.1 |
| *Tilia sp.* | 3.9 | 0.6 | 0.1 | 0.6 | 1.9 |
| *Picea abies* | 9.8 | 1.0 | 2.4 | 4.6 | 4.7 |
| *Pinus sp.* | 5.0 | 1.0 | 0.3 | 1.1 | 3.6 |
| *Pseudotsuga menziesii* | 8.1 | 1.0 | 0.0 | 1.7 | 8.1 |

* only deciduous oaks considered

Figures:

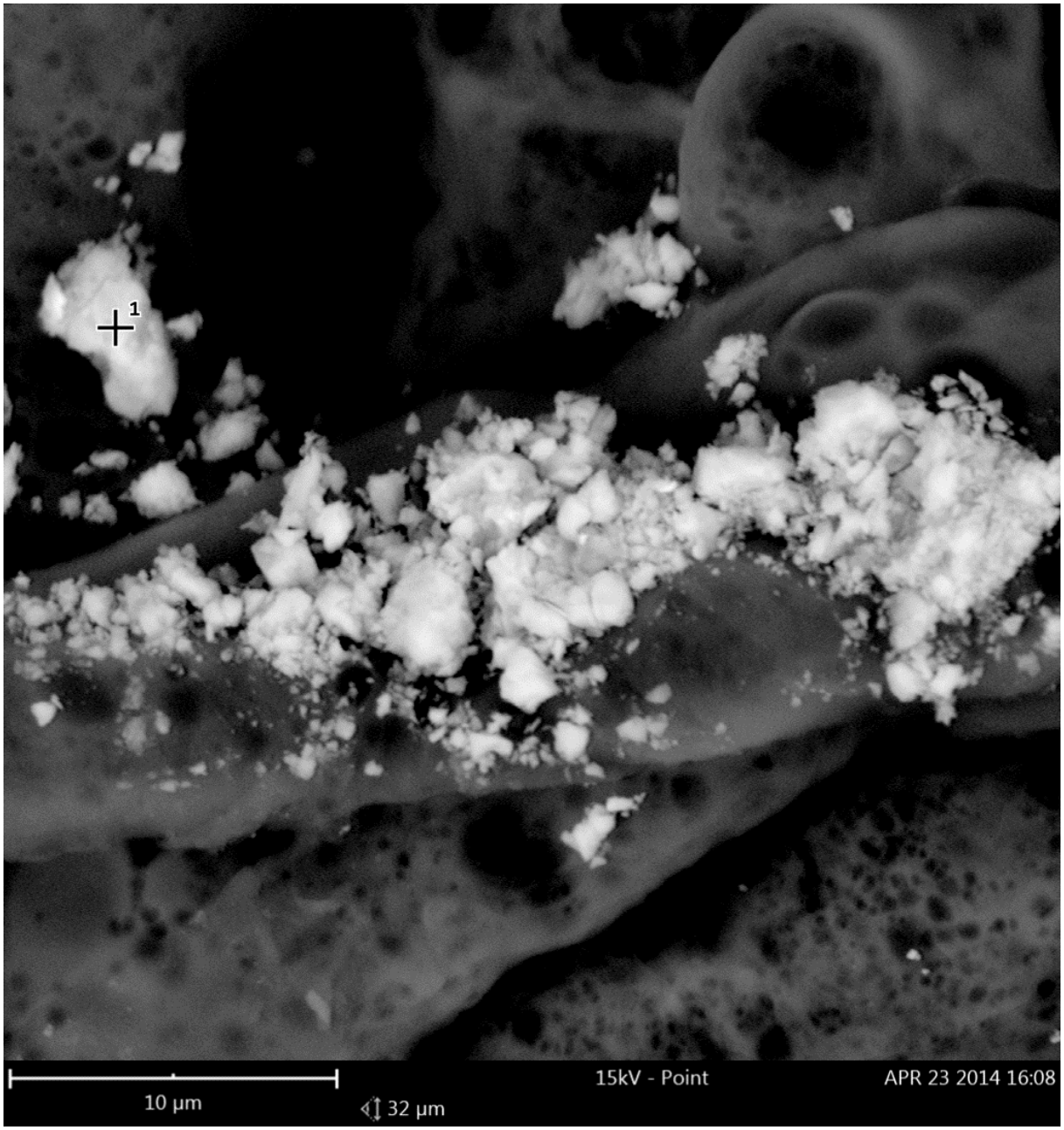


Figure 1. Iron particles detected with SEM-EDX analysis from filters obtained from *Quercus ilex* leaves in the industrial city of Terni, central Italy. Details of the case study are available in Sgrigna et al. (2015). Photo Credit: C. Baldacchini and G. Sgrigna.

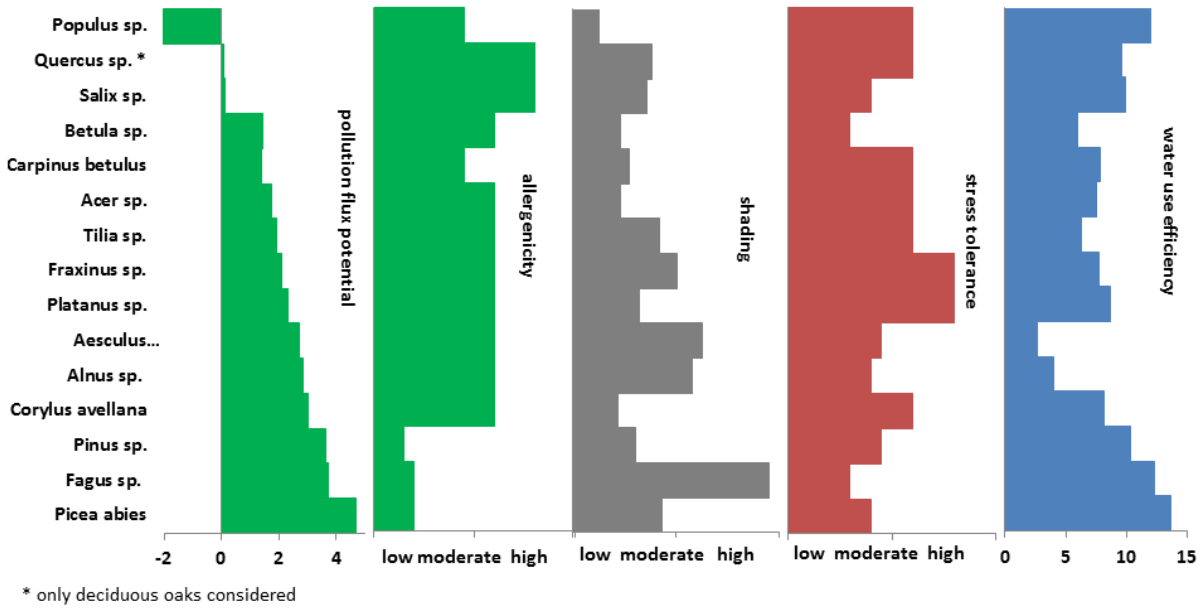


Figure 2. Ecosystem services of the most important urban tree species sorted by benefit for air quality. The indices are derived from morphological and physiological properties (allergenicity = pollination intensity (1-4) * toxicity (1-4), shading = LAI/IAL * width/height) and general information from literature (stress tolerance = drought tolerance (1-4) * air pollution tolerance (1-4)).