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Should we see urban trees as effective solutions to reduce increasing ozone levels in cities?☆

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ABSTRACT

Outdoor air pollution is considered as the most serious environmental problem for human health, associated with some million deaths worldwide per year. Cities have to cope with the challenges due to poor air quality impacting human health and citizen well-being. According to an analysis in the framework of this study, the annual mean concentrations of tropospheric ozone (O_3) have been increasing by on average 0.16 ppb year^{-1} in cities across the globe over the time period 1995–2014. *Green urban infrastructure* can improve air quality by removing O_3. To efficiently reduce O_3 in cities, it is important to define suitable urban forest management, including proper species selection, with focus on the removal ability of O_3 and other air pollutants, biogenic emission rates, allergenic effects and maintenance requirements. This study reanalyzes the literature to i) quantify O_3 removal by urban vegetation categorized into trees/shrubs and green roofs; ii) rank 95 urban plant species based on the ability to maximize air quality and minimize disservices, and iii) provide novel insights on the management of urban green spaces to maximize urban air quality. Trees showed higher O_3 removal capacity (3.4 g m^{-2} year^{-1} on average) than green roofs (2.9 g m^{-2} year^{-1} as average removal rate), with lower installation and maintenance costs (around 10 times). To overcome present gaps and uncertainties, a novel Species-specific Air Quality Index (S-AQI) of suitability to air quality improvement is proposed for tree/shrub species. We recommend city planners to select species with an S-AQI>8, i.e. with high O_3 removal capacity, O_3-tolerant, resistant to pests and diseases, tolerant to drought and non-allergenic (e.g. *Acer* sp., *Carpinus* sp., *Larix decidua*, *Prunus* sp.). Green roofs can be used to supplement urban trees in improving air quality in cities. Urban vegetation, as a cost-effective and nature-based approach, aids in meeting clean air standards and should be taken into account by policy-makers.

1. Introduction

Nowadays, 54% of the world population lives in cities, a number which is expected to rise up to 70% by 2050 (United Nations, 2014). In a typical urban environment, citizens are exposed to about 200 air pollutants or classes of air pollutants (Sicard et al., 2011; Goudarzi et al., 2017; Khaniabadi et al., 2017). Air pollution is a major environmental issue in urban areas, which can affect the life quality and well-being of citizens (Manning, 2013). Exposure to air pollutants such as particulate matter (PM), nitrogen oxides (NOx), sulfur dioxide (SO_2), and surface ozone (O_3) associates with respiratory and cardiovascular diseases and mortality (Sicard et al., 2011; Lee et al., 2014; Khaniabadi et al., 2016). Despite the reductions in the emissions of common air pollutants since the early 1990s, millions of people are still exposed to concentrations above the critical levels associated with increased risks for cardiovascular and respiratory diseases (Yang et al., 2005; Foley et al., 2014; World Health Organization, 2016a). Outdoor air pollution associates

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with about 8 million deaths worldwide per year (World Health Organization, 2016a).

Ozone and PM are the most threatening air pollutants in cities in terms of harmful effects on human health (Pascal et al., 2013; Guerreiro et al., 2014; Miranda et al., 2016; European Environment Agency, 2017) as well as NO_2 (Weinmayr et al., 2010), with concentrations above the World Health Organization (WHO) air quality guidelines in 98% of cities in low- and middle-income countries (> 100,000 inhabitants) and in 56% of cities in high-income countries (World Health Organization, 2016a). Ozone is considered as the most damaging air pollutant in terms of adverse effects on natural vegetation and cultivated crops in Europe (Sicard et al., 2016c; Ochoa-Hueso et al., 2017) and short- and long-term exposures to O_3 have been associated with increased numbers of hospital admissions and premature mortality for respiratory and cardiovascular diseases, reduced worker productivity and increased medication use (Jerrett et al., 2009; Anderson et al., 2012; Orru et al., 2013; Krmpotic et al., 2015; Nuvolone et al., 2017). The global impact of long-term outdoor O_3 exposure contributed to 1.2 million premature respiratory deaths in 2010, i.e. one in five of all respiratory deaths (Malley et al., 2017).

The mean global background O_3 concentration is expected to increase from the current level of 35–50 ppb (Lefohn et al., 2014) to 85 ppb by 2100 at mid-latitudes of the Northern Hemisphere (IPCC, 2014). The current O_3 levels in cities in many regions of the world regularly exceed the target value set by WHO for the protection of human health, i.e. 50 ppb as daily 8-h average concentration (World Health Organisation, 2008), and the critical level for the protection of vegetation in Europe is 9000 ppb h for the exposure to O_3 hourly concentrations exceeding 40 ppb over the daylight hours of the growing season in Europe (Kroeger et al., 2014; Saitanis et al., 2015; Anav et al., 2016; García-Gómez et al., 2016; Churkina et al., 2017), while different critical levels are suggested for different vegetation types by UNECE (2017). In cities, the rising background O_3 levels appear to be a major air quality issue and become a public health concern despite control efforts (Bogaert et al., 2009; Sicard et al., 2013; World Health Organisation, 2013; Lefohn et al., 2018).

Urban green spaces can reduce air pollution and greenhouse gas emissions (Escobedo and Nowak, 2009; Pataki et al., 2011; Baró et al., 2014; Tong et al., 2016), sequester carbon (Tsay et al., 2015; Anav et al., 2016; Proietti et al., 2016), regulate air temperature (Bowler et al., 2010; Manes et al., 2012), mitigate storm-water runoff (Pataki et al., 2011), reduce noise (Cohen et al., 2014; Klingberg et al., 2017a), as well as provide recreational, social, psychological, and aesthetic benefits (Nowak et al., 2006; Dadvand et al., 2015; World Health Organization, 2016b). The *green urban infrastructure*, defined as a network of natural and semi-natural green spaces such as forests, parks, green roofs and walls, can provide nature-based and cost-effective solutions (Connop et al., 2016) and contribute to ecosystem resilience and human well-being through ecosystem services from urban centers to peri-urban areas (Lafortezza et al., 2013; Fusaro et al., 2015; Davies et al., 2017; Jayasooriya et al., 2017). Previous studies demonstrated the ability of *green urban infrastructure* to ameliorate urban air quality (including PM, NOx, SO_2) across conurbations worldwide (Vos et al., 2013; Morani et al., 2014; Baró et al., 2015; Silli et al., 2015; Nowak et al., 2018) and thereby enhance citizen well-being (Tzoulas et al., 2007). In urban densely populated areas, where it is not always easy to plant trees, green walls and green roofs can be used to supplement the use of urban trees in air pollution control (Yang et al., 2008; Rowe, 2011; Gregory et al., 2016). The vegetation facilitates deposition of PM and gases on plant surfaces, absorbs gaseous pollutants through leaf stomata (Yang et al., 2015; Jayasooriya et al., 2017; Nowak et al., 2018) and regulates the transport of pollutants, for instance by reducing ventilation of street canyons (Salmond et al., 2013).

Among the most cost-effective approaches, the direct removal of O_3 and one of its precursors (NOx) by urban green spaces is of particular interest. The potential of *green urban infrastructure* to mitigate O_3 pollution is well documented by field measurements and model estimates (Nowak et al., 2000; Harris and Manning, 2010; Manes et al., 2012; Hu et al., 2016; Klingberg et al., 2017a; Yli-Pelkonen et al., 2017a, b).

To reduce O_3 levels in cities, it is important to define efficient city planning. For that, municipalities need to clearly understand the magnitude of *green urban infrastructure* effects on air pollution as well as its disservices (environmental, social and financial) (Escobedo et al., 2011; Fantozzi et al., 2015; Selmi et al., 2016; Bottalico et al., 2017). Disservices of urban forests can be i) environmental, such as the introduction of invasive species, an increase in water use in semi-arid areas and the release of volatile organic compounds (VOC) and secondary aerosol emissions leading to ground-level O_3 formation (Nowak et al., 2002; Paoletti, 2009); ii) social (e.g. allergenic pollen, fear of crime) and iii) financial, as additional costs incurred by urban forest management such as pruning, replacement, green waste, control of pests and diseases, irrigation and damage to urban infrastructure caused by vegetation (Nowak and Dwyer, 2007; Escobedo et al., 2011). Trees can also be net CO_2 emitters when accounting for maintenance activities, pruning, watering, etc. (Nowak et al., 2002). However, the benefits provided by urban forests usually overcome their disservices (Escobedo et al., 2011).

Through an extensive literature review and the evaluation of the O_3 risk, for human and vegetation health, at urban stations across the world, the main goals of this paper are to: i) quantify O_3 removal capacity by urban vegetation categorized into trees/shrubs and green roofs, ii) rank common urban plant species based on the ability to improve air quality and minimize disservices, and iii) provide novel insights on the management of urban green spaces to maximize urban air quality. The effect of *green urban infrastructure* on energy conservation, storm-water management, water quality, noise pollution, carbon sequestration, habitat for wildlife and the urban heat island is beyond the scope of this review paper.

2. Materials and methods

2.1. Determining global trends in ground-level ozone

The international Tropospheric Ozone Assessment Report (TOAR), initiated by the International Global Atmospheric Chemistry Project, established a database of global *in-situ* hourly O_3 observations over the time period 1970–2015 from almost 10,000 sites around the world (Schultz et al., 2017; Lefohn et al., 2018). From this database, the stations with more than 75% of validated hourly data per year were selected over the time period 1995–2014. Finally, 319 urban and 306 rural stations, distributed across 35 countries and 6 continents, were selected for the calculation of annual O_3 averages.

For each country, a national-averaged trend magnitude (in ppb per year) was calculated. The non-parametric Mann-Kendall test was applied to detect a monotonic trend, increasing or decreasing, within the time series. To estimate the trend magnitude, the Sen (1968) non-parametric estimator was used. Results were considered significant at $p < 0.05$.

2.2. Literature review

To quantify the effects of green infrastructure on urban O_3 concentrations, approximately 150 peer-reviewed articles and technical reports spanning over the period 1990–2017 were retrieved from literature databases (Science Direct, Web of Science, Silver Platter, and Google scholar). Details on the used keywords are provided in the supplementary material. From each peer-reviewed article, the following parameters were extracted: country; location; climate; time period; time of the year; duration of study; method (passive or active O_3 monitors or UFORE/i-Tree model); mean O_3 concentration over the study period (ppb); plant species; age of the plants; urban forests and green roofs area; average annual O_3 removal rate per square meter of tree canopy cover (g m^{-2} of tree cover $year^{-1}$ and kg ha^{-1} $year^{-1}$) and annual O_3 removal (tons). Details on the calculation of the last two parameters are given in section 2.2.1. When O_3 concentrations over the study period were not available in the article, the TOAR database was used. For the plant species, a search was done in the literature for details on: BVOC emission potential; ozone forming potential (OFP); and O_3, drought, pest and disease tolerance (section 2.2.2).

2.2.1. Quantification of ozone removal capacity

To assess the O_3 removal capacity of green infrastructure, both direct measurements and modeling techniques are available in the literature. Widely applied techniques utilize duplicate passive diffusion samplers inside and outside urban forest canopy at local scale (Harris and Manning, 2010; Grundström and Pleijel, 2014; Fantozzi et al., 2015; García-Gomez et al., 2016; Yli-Pelkonen et al., 2017b) and dry deposition models at larger scale (Yang et al., 2008).

The mostly used dry deposition model for studying the effects of urban vegetation on air pollution is the Urban FORest Effects (UFORE) model and following developments (i-Tree). Another model used is the Fine Resolution Atmospheric Multi-species Exchange, FRAME model (McDonald et al., 2007). UFORE/i-Tree estimates the deposition rates (both stomatal and non-stomatal deposition) for all pollutants (O_3, SO_2, NO_2, CO and PM_{10}), while FRAME estimates only PM_{10} removal.

The UFORE model was developed by the U.S Forest Service to quantify multiple urban forest ecosystem services (such as deposition of air pollutants, emission of BVOC, annual carbon sequestration) by using the concepts of dry deposition modeling (i.e. pollution removal during non-precipitation periods), and identify tree species which are most effective in improving local air quality (Nowak and Crane, 2000; Nowak et al., 2006; Saunders et al., 2011). To assist urban forest planners and managers, an open-source model combining the UFORE model and a Geographical Information System, namely i-Tree, was introduced. This model allows using GIS-based systems to find areas where reforestation would be beneficial for air quality (Hirabayashi et al., 2011, 2012; Janhäll, 2015). Such models have been widely applied across the world (Nowak et al., 2006; Yang et al., 2008; Baró et al., 2014; Selmi et al., 2016; Bottalico et al., 2017) and are parameterized to model stomatal conductance based on the eco-physiological responses of plants to environmental conditions (Nowak et al., 2014). A good agreement (least sum square $= 5.6 \times 10^{-4}$) was observed between i-Tree modelled and measured (Eddy Covariance technique) O_3 fluxes in a peri-urban Mediterranean forest near Rome (Morani et al., 2014).

For each city, the article provided the downward O_3 flux per square meter of tree canopy cover (F; g m^{-2} s^{-1}) calculated using the UFORE/i-Tree methods detailed in the supplementary material. The downward O_3 flux (F) was then accumulated over the accumulation period (e.g. over a year, leaf-on and leaf-off) and multiplied by the total tree cover (m^2) to estimate the total O_3 removal in a city over a year, expressed in metric tons (Nowak et al., 2006). The hourly percent improvement was calculated as: quantity of O_3 removed (in grams) divided by "quantity removed + quantity in atmosphere" where quantity in atmosphere = measured concentrations (g m^{-3}) x boundary layer height (m) x city area (m^2). To estimate the boundary-layer height in the study area, mixing-height measurements from the nearest location were used (Nowak et al., 2006).

For urban areas where mean O_3 concentrations were not available in the article, local pollution monitoring data were retrieved from the TOAR database (Schultz et al., 2017) and used to calculate the average O_3 concentration in the urban area over the study period. As proposed by Nowak et al. (2006), the O_3 removal rates (g m^{-2} of tree cover) were standardized to the average O_3 concentration in the city (g m^{-2} per ppb) to account for concentration differences among urban areas (Tables1S–2S). In addition, we reported O_3 removal capacity in "g m^{-2} of tree cover day^{-1}" for a better comparison between tree species (Table 3S).

2.2.2. Ranking plant species

We ranked the plant species in the articles according to their effectiveness in removing O_3, NO_2 (O_3 precursor) and another major urban air pollutant i.e. PM_{10}, and according to their capacity of emitting biogenic VOCs (BVOCs). BVOCs are the most significant contributor to O_3 formation over land areas in the Northern Hemisphere (Zare et al., 2014; Monks et al., 2015). Standardized species-specific ozone forming potentials were proposed by Benjamin and Winer (1998) and calculated for many species (Simpson and McPherson, 2011; Baraldi et al., 2018) on the basis of the species-specific emission factors of isoprene/monoterpenes and the plant biomass at maturity. The total BVOC emission rates were standardized under specific conditions, at 30 °C and with a Photosynthetic Photon Flux Density of 1000 mmol m^{-2} s^{-1} (Guenther et al., 1995; Chaparro and Terradas, 2009).

For each plant species, a value from the range 1–3 (1 = low, 2 = medium, 3 = high) was attributed to each removal capacity for O_3, NO_2 and PM_{10} and for OFP (Table 4S). Tolerance to O_3, pest, disease and drought could damage vegetation or lead to stomatal closure, thus reducing O_3 removal ability (Escobedo and Nowak, 2009; Hoshika et al., 2017). Also pollen release is a serious air quality issue in the cities (Carinanos et al., 2014). As air pollutant removal capacity and OFP are the most important factors to be considered, we attributed a single score (1–3) to the combination of all four secondary criteria "pollen allergenicity, O_3 sensitivity and tolerance to drought, and pest and diseases", i.e. we attributed a value of 1–3 (low, medium, high) to each criterion and, then we calculated an average score. Finally, we summed up the five scores and the total ranged between 5 (minimum) and 15 (maximum). For each plant species, we adjusted by cross-multiplication the numerical scale from the range 5–15 to 1–10 to facilitate interpretation, and called this summary index as "Species-specific Air Quality Index" (S-AQI), for categorizing urban tree/shrub species for air quality planning from "recommended" (>8) to "not recommended" (<4).

3. Results and discussion

3.1. Urban ozone pollution is still increasing

The background O_3 concentrations increased by, on average, + 0.16 ± 0.22 ppb $year^{-1}$ at 89% of 319 urban stations and decreased by - 0.05 ± 0.21 ppb $year^{-1}$ at 65% of 306 rural stations globally over the time period 1995–2014 (Fig. 1). The background O_3 increase in cities is confirmed by other studies covering global trends from 2000 onwards in Europe and the United States (Saavedra et al., 2012; Wilson et al., 2012; Sicard et al., 2013, 2016b; Paoletti et al., 2014; Simon et al., 2015; Lefohn et al., 2018). These increasing O_3 background levels in the cities can be mainly attributed to a lower O_3 degradation by NO due to the reduction in local NOx emissions (Sicard et al., 2016a; Lefohn et al., 2018). The current background O_3

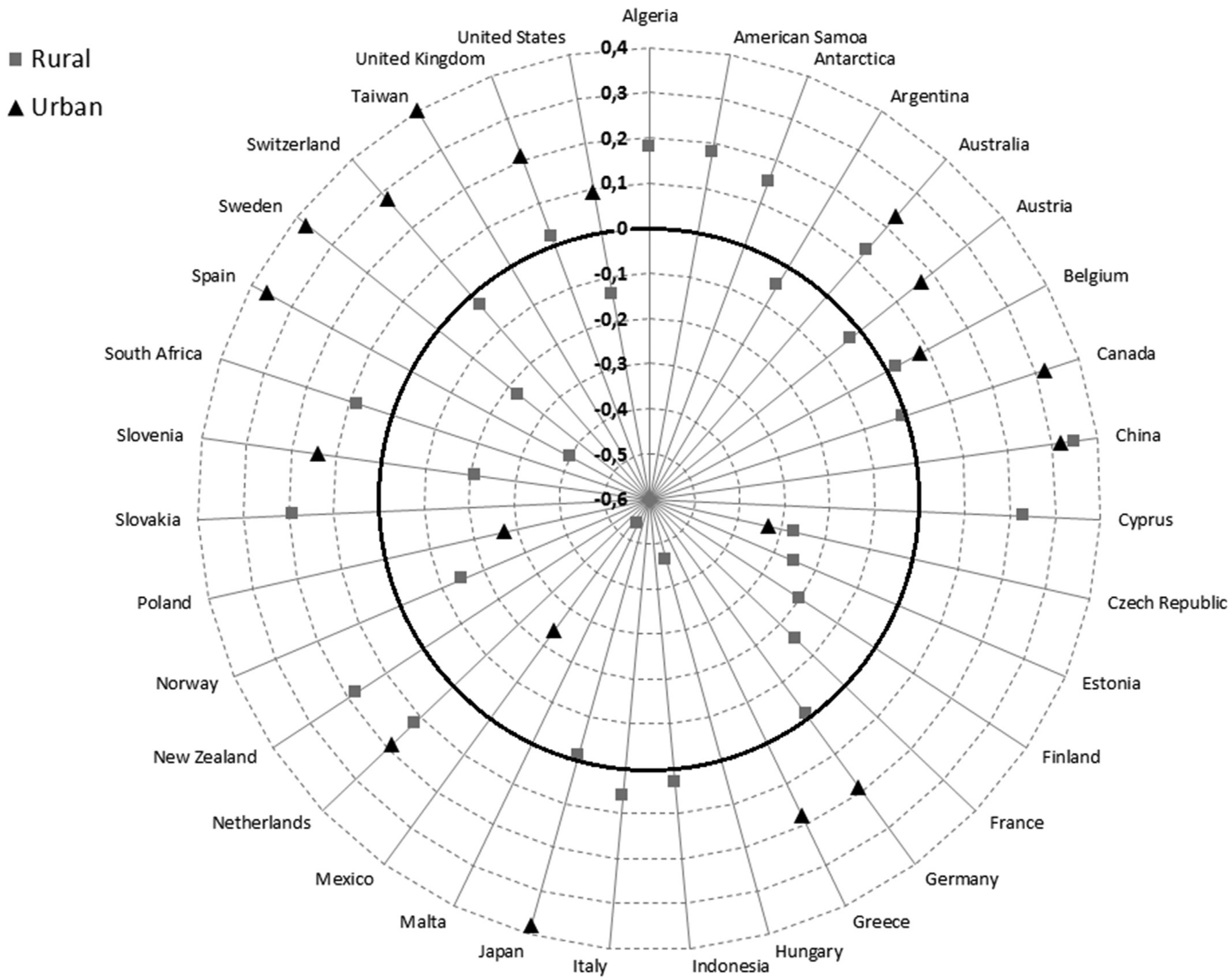

Fig. 1. Country-averaged annual trend magnitude (ppb per year) over the time period 1995–2014, by applying the Mann-Kendall test and the Sen's method to annual ozone averages from rural and urban background monitoring stations included in the Tropospheric Ozone Assessment Report database. Points below the thick line show a decrease in ozone annual averages, while points above the thick line show an increase.

levels may thus still exceed target values for the protection of human health and annual and perennial vegetation (Ashmore, 2005; Paoletti, 2006; Emberson et al., 2013; Sicard et al., 2017).

3.2. Ozone removal by green roofs

In the last decade, in response to rising air pollutants, many cities increased the urban green spaces to mitigate air pollution; for instance by transforming urban roofs into green roofs (Andersson-Skold et al., 2015). Roofs can represent up to 20–30% of the horizontal surface of built-up areas, giving an opportunity for green roofs to be implemented on a large scale (Oberndorfer et al., 2007; Rowe, 2011; Jacobson and Ten Hoeve, 2012). Vertical greening can also be a barrier between a high source of air pollutants, such as roads, and a building (Gregory et al., 2016). Green roofs are defined as continuous surfaces of short grass, tall perennial herbaceous plants, and occasionally, shrubs and small trees (Yang et al., 2008).

By applying dry deposition models (UFORE/i-Tree), a number of studies showed that the annual O_3 removal by green roofs ranged from 1.2 to 4.4 g m^{-2} year^{-1} (Deutsch et al., 2005; Currie and Bass, 2008; Yang et al., 2008; Jayasooriya et al., 2017) with standardized O_3 removal rates ranging from 0.07 to 0.23 g m^{-2} per ppb of O_3 (Table 1S). For example, 109 ha of green roofs removed 3.1 tons of O_3 in Toronto (Currie and Bass, 2008) and about 202 ha of green roofs removed 6.0 tons of O_3 from the air in Washington, equivalent to approximately 25,000 street trees (Deutsch et al., 2005). In Putrajaya (Malaysia), an experimental comparison showed that diurnal O_3 levels above an open roof (without plant) can be up to 50% higher than those above a green roof (Sam and Sohaili, 2016). In parallel, green roofs can conserve energy, mitigate the urban heat island, reduce noise pollution, provide a more aesthetically pleasant environment, and improve dwelling investment (Oberndorfer et al., 2007; Rowe, 2011).

3.3. Ozone removal by urban forests

There are a variety of different scales to consider when studying the effect of urban forests on the urban air quality. Passive samplers are used at street level, as point measurements, while dry deposition models are used at larger scale (i.e. city averaged). Urban forests are defined as the sum of all urban trees and shrubs (Escobedo et al., 2011).

By applying the UFORE/i-Tree dry deposition model to estimate the total dry deposition (stomatal and non-stomatal O_3 deposition), a number of studies showed that public trees, i.e. trees managed by the municipal authorities, removed 305,100t and 523,000t of O_3 in 55 cities in the U.S. in 1994 and 2010, respectively (Nowak et al., 2006, 2014), 12,370t in 86 cities in Canada in 2010 (Nowak et al., 2018), and 30,014t in 10 cities in Italy in 2003 (Manes et al., 2016) (Table 2S). At city scale, the annual O_3 removal depends on local conditions (e.g. O_3 concentrations, meteorology, phenology, type of forest cover); from 0.7t in Florence (Italy) by a single park in 2004 to 6137t over the whole city of Turin (Italy) in 2003. In Strasbourg, the urban forests (57%) and parks (28%) removed more O_3 pollution than residential, industrial and agricultural areas (Selmi et al., 2016).

The average annual O_3 removal in 2010 was 5.4 g m^{-2} of tree cover year^{-1} in 55 U.S cities (2.1–7.6 g m^{-2} year^{-1}), 3.7 g m^{-2} year^{-1} in 86 Canadian cities (2.1–11.1 g m^{-2} year^{-1}) and 3.3 g m^{-2} year^{-1} in 10 Italian cities in 2003 (Table 2S). The O_3 removal rate widely vary worldwide: Melbourne (0.9 g m^{-2} year^{-1}), Stockholm (1.3 g m^{-2} year^{-1}), Montréal (3.9 g m^{-2} year^{-1}), Beijing (5.5 g m^{-2} year^{-1}), Florence (7.3 g m^{-2} year^{-1}) and San Diego (7.6 g m^{-2} year^{-1}). The highest standardized O_3 removal rates (>0.30 g m^{-2} per ppb of O_3) are observed in Los Angeles, San Diego, Chicago, Toronto and Perth while the lowest removal rates (<0.10 g m^{-2} per ppb of O_3) are found in Santiago, Mexico, Québec, Melbourne and Northern Europe (Table 2S). The standardized removal rates differ among cities according to the amount of air pollution, length of in-leaf season, precipitation and other meteorological variables (Nowak et al., 2006).

The mean annual reduction in hourly concentrations due to urban trees and shrubs varied from 0.1% in Tucson (U.S) in 1994 to 1.5% in Florence (IT) in 2013 (Table 2S). Other examples include: Rotterdam 0.1%, Barcelona, Berlin, Chicago, Denver and San Diego ≈0.3%, Santiago 0.2–0.5%, Boston, Dallas, Strasbourg and Washington 0.6%, and Stockholm 1.0% (Nowak et al., 2006; Yang et al., 2008; Chaparro and Terradas, 2009; Baró et al., 2015; Selmi et al., 2016; Bottalico et al., 2017). In 2010, the annual O_3 reduction ranged from 0.1 to 0.8% with an average of 0.36% over 55 U.S cities (Nowak et al., 2014) and 0.25% over 86 Canadian cities (Nowak et al., 2018). In 2000, a reduction of 0.61% was observed for O_3 during the daytime of the in-leaf season across 14 U.S cities (Nowak and Dwyer, 2007). The urban forest in Florence (Mediterranean climate) removed monthly up to 5% of O_3 in 2013 (Bottalico et al., 2017). The maximum short-term O_3 reduction (1-h) from trees is reached during daytime of the in-leaf season, e.g. 14.8% on average in 2000 across 14 U.S cities and 16.4% reduction of O_3 peaks over heavily forested areas in Toronto in 2007 (Nowak and Dwyer, 2007; Nowak et al., 2013). An increase of tree cover by 10% (about 200 km^2) in New York could reduce 1-h O_3 peaks by 4 ppb in 1995 (Luley and Bond, 2002).

To quantify the impact of urban and peri-urban forests on surface O_3 levels, few studies with O_3 passive samplers, above- and below-forest canopies, are available (Bytnerowicz et al., 1999; Harris and Manning, 2010; Fantozzi et al., 2015; García-Gomez et al., 2016; Yli-Pelkonen et al., 2017b). The differences in O_3 concentration between above- and below-forest canopies, in urban parks or near traffic routes, ranged from 2% to 40% improvement depending on season, location, and timescale. For example, a 40% improvement (hourly mean) by broad-leaf deciduous species was observed in San Bernardino Mountains in 1996 (Bytnerowicz et al., 1999), a 20% (daily mean) improvement was observed over *Acer rubrum* stands in Springfield, USA, in summer 2009 (Harris and Manning, 2010), and a 5–7% (annual mean) improvement was observed over *Quercus ilex* stands over the period 2011–2013 in Barcelona and Madrid (García-Gomez et al., 2016). Likewise, a 18% (monthly mean) improvement was observed over *Q. ilex* stand at a 10 m distance from road in Siena (Italy) between June 2011 and October 2013 (Fantozzi et al., 2015) and a 10% (daily mean) improvement by mixed broadleaved deciduous trees was observed in Baltimore (USA) in May 2016 (Yli-Pelkonen et al., 2017b). Overall, the mean annual percent improvement by the UFORE/i-Tree models is usually less than 2% of O_3 levels, while studies based on passive samplers suggest that local reductions up to 40% in urban parks. Such discrepancies between model estimates and direct measurements may suggest that the dry deposition models are not yet able to incorporate the actual effects of vegetation on O_3 reductions.

The temporal variation of urban trees effects on O_3 is strictly linked to meteorological-dependent plant features such as stomatal conductance, LAI, tree transpiration, dry deposition velocities, and length of growing season (Yang et al., 2005; Nowak and Dwyer, 2007; Wang et al., 2012; Baró et al., 2014; Nowak et al., 2018). The optimal effect is observed during the daytime of the in-leaf season (Selmi et al., 2016; Nowak et al., 2018) while the O_3 removal by trees at night is limited due to stomatal closure (Nowak et al., 2002). At mid-latitude in the Northern Hemisphere, monthly O_3 removal is lower in November–January (2–3% of annual O_3 removal) and maximal in April–September (78–81% of annual O_3 removal) during the in-leaf season (spring-summer) due to greater leaf area and often higher O_3 concentrations (Yang et al., 2005; Baró et al., 2014; Selmi et al., 2016).

3.4. Urban trees are a more efficient and cost-effective nature-based solution than green roofs

By comparing the results in the previous two sections, green roofs are less effective than urban trees to remove O_3, in particular due to lower species-specific stomatal conductance, surface roughness and LAI (Speak et al., 2012). On average, the standardized removal rates by green roofs (0.13 g m^{-2} per ppb of O_3 on average) are lower than the removal rates for tree species (0.19 g m^{-2} per ppb of O_3 on average). In the same place (Chicago) and time period while the rate for green roofs is 1.2 g m^{-2} year^{-1} (Table 1Svs Table 2S).

Two comparative studies, one in Toronto (Currie and Bass, 2008) and one in Melbourne (Jayasooriya et al., 2017), analyzed the air quality improvement through three *green urban infrastructure* scenarios (urban trees, green roofs, and green walls) by using the UFORE/i-Tree model to define a sustainable city planning for air pollution and climate change mitigation. In Melbourne, the existing urban forests (10 trees per ha) remove 246 kg O_3 per year. By adding new trees (up to 80 trees per ha), 1885 kg O_3 will be removed annually. In addition to existing trees, replacing the roof areas of industrial buildings (28.9 ha) through green roofs will remove 357 kg O_3, and replacing the building walls by green walls will take up 298 kg O_3 (Jayasooriya et al., 2017). In Toronto, the existing trees and shrubs remove 10.7t O_3 per year (7.4t by trees, 3.3t by shrubs). The green walls will remove 1.1t O_3. In addition to existing trees and shrubs, adding green roofs on commercial, residential, and institutional buildings will remove 1.3t O_3. If trees and shrubs were augmented with grass on all available roof surface areas across Midtown, the gain was 3.1t O_3 (Currie and Bass, 2008). By combining green roofs or green walls with trees and shrubs, this slightly reduces O_3 levels (Currie and Bass, 2008; Jayasooriya et al., 2017).

Interestingly, the installation costs of green roofs are more expensive than planting trees (Yang et al., 2008; Speak et al., 2012; Jayasooriya et al., 2017). The annual life cycle cost is considered as a good indicator to assess the annual recurrent costs of different scenarios, i.e. green roofs, green walls or urban trees installation

(Jayasooriya et al., 2017). The cycle costs (including planting and maintenance) are \$300 to \$500 per tree depending on the species and height (Akbari, 2002). A medium size tree (planting costs: \$400, on average) can remove the same amount of air pollutants as 19 m^2 of green roof in one year (installation around \$3000) (Deutsch et al., 2005; Yang et al., 2008). The annual costs (operation, maintenance) is around \$300 per kg O_3 removed (urban trees), \$600 per kg O_3 (green roofs) and \$1300 per kg O_3 (green walls). In summary, urban trees provide the highest potential of O_3 removal (and other gaseous pollutants) and installation and annual costs are lower for urban trees compared to both green roofs and green walls (Currie and Bass, 2008; Yang et al., 2008; Li and Yeung, 2014; Jayasooriya et al., 2017). For every \$1 invested in tree management, citizens receive \$1.4-\$4.5 in benefits (property value, carbon dioxide, energy saving, storm-water, air quality) in E.U and U.S. cities (Soares et al., 2011).

3.5. Species-specific ozone removal capacity

By analysing the standardized O_3 removal rates (Table 3S), we can observe that Mediterranean evergreen broadleaf (0.088×10^{-2} g m^{-2} per ppb of O_3) and deciduous broadleaf (0.093×10^{-2} g m^{-2} per ppb of O_3) tree species removed more O_3 than conifers (0.054×10^{-2} g m^{-2} per ppb of O_3) in Madrid over 2003. In Florence, over 2013, Mediterranean evergreen broadleaves (0.147×10^{-2} g m^{-2} per ppb of O_3) removed more O_3 than deciduous broadleaves (0.037×10^{-2} g m^{-2} per ppb of O_3). Based on standardized O_3 removal rates, estimated by dry deposition models and expressed per leaf area and day of growing season, the tree species considered as top rated species for reducing O_3 pollution are: *Magnolia liliflora* (0.218×10^{-2} g m^{-2} per ppb of O_3), *Aesculus chinensis* (0.182×10^{-2} g m^{-2} per ppb of O_3), *Ginkgo biloba* (0.141×10^{-2} g m^{-2} per ppb of O_3), *Liquidambar* sp. (0.138×10^{-2} g m^{-2} per ppb of O_3) and *Fraxinus excelsior* (0.114×10^{-2} g m^{-2} per ppb of O_3). On the other hand, *Picea abies, Quercus ilex* and *Robinia pseudocacia* are considered as low rated species (Table 3S). Overall, broadleaf tree species remove more O_3 than conifers because of higher stomatal uptake (Bottalico et al., 2017; Hoshika et al., 2018); Mediterranean evergreen tree species remove more O_3 than deciduous tree species because of a longer growing season (Hoshika et al., 2018).

Compared to stomatal O_3 flux into the leaves measured by sap flow measurements (Wang et al., 2012; Hu et al., 2016), by direct measurements of stomatal conductance (Hoshika et al., 2018) or by stomatal flux models such as Deposition of Ozone and Stomatal Exchange (DO3SE) model (Manes et al., 2016; Bottalico et al., 2017), the UFORE/i-Tree modeling approach estimates the total dry deposition, including both stomatal and non-stomatal O_3 uptake, leading to higher standardized O_3 removal rates e.g. *Fagus sylvatica* (0.048×10^{-2} g m^{-2} per ppb of O_3 as stomatal deposition and 0.097×10^{-2} g m^{-2} per ppb of O_3 as total deposition) or *Pinus* sp. (0.015×10^{-2} g m^{-2} per ppb of O_3 as stomatal deposition and 0.066×10^{-2} g m^{-2} per ppb of O_3 as total deposition). Dry deposition processes account for about 25% of the total O_3 removed from the troposphere (Hardacre et al., 2015). At vegetated surfaces, 30–90% of O_3 dry deposition occurs though stomata, i.e. O_3 stomatal uptake (Cieslik, 2004; Fares et al., 2008; Hoshika et al., 2018).

4. Research directions and perspectives

4.1. Model improvements

The annual O_3 removal rates vary across the world according to the amount of tree cover, tree features (such as foliage, LAI, height, species, stomatal conductance), surface O_3 concentrations, duration of in-leaf season, and meteorological parameters affecting tree transpiration and deposition velocities (Paoletti, 2009; Baró et al., 2014; Hardacre et al., 2015; Nowak et al., 2018). The large discrepancy among cities in standardized O_3 removal rates (e.g. 0.11 and 0.24 g m^{-2} per ppb of O_3 in Florence or 0.42 g m^{-2} per ppb of O_3 in Perth vs. 0.06 g m^{-2} per ppb of O_3 in Melbourne) is mainly due to the difference between the actual ground measurements of forest related traits versus assumed parameters in UFORE/i-Tree model, such as LAI and canopy resistance and local environmental conditions such as solar radiation and precipitation (Escobedo and Nowak, 2009). In Manes et al. (2016), the total potential O_3 removal was estimated from the stomatal O_3 fluxes, considered as 30% of total O_3 removal, this may explain the difference observed in Florence. The UFORE/i-Tree results are based on well-watered plants, therefore, the lower rate in Melbourne than in San Diego is not upon soil water availability but mainly upon weather conditions (such as temperature, humidity, solar radiation, precipitation) and tree features.

The UFORE/i-Tree model has a number of assumptions, some of which could be addressed by the use of a site-specific deposition model (Tiwary et al., 2009). The UFORE/i-Tree model calculates the dry deposition to tree canopies and does not take into account wet deposition; therefore the total deposition is underestimated (McDonald et al., 2007; Tiwary et al., 2009; Kerckhoffs, 2014). In UFORE/i-Tree model, generic dry deposition values are assigned to trees due to a lack of empirical deposition data for specific tree species accounting for large variations in deposition velocity (Beckett et al., 2000; Tiwary et al., 2009).

The UFORE/i-Tree model is based on tree cover data instead of land cover data, and then the variation in large-scale O_3 dry deposition might be due to very limited land cover scheme (Hardacre et al., 2015; Anav et al., 2018). Urban vegetation could indirectly affect photochemical O_3 formation through reductions in ambient air temperature by changing the albedos of urban surfaces and by evapotranspiration cooling, although these effects are also largely unquantified (Pataki et al., 2011).

The discrepancy between modelled and observed O_3 dry deposition fluxes is generally driven by the modelled O_3 dry deposition velocity rather than by surface O_3 concentrations (Hardacre et al., 2015). In UFORE/i-Tree model, the canopy resistance is calculated either based on multi-layer canopy deposition model or derived from literature (Nowak and Crane, 2000; Yang et al., 2005) and the assumed canopy resistance (1.74 cm s^{-1} used in UFORE/i-Tree) can be significantly smaller than field measurements (2.81 cm s^{-1} in Beijing, Wang et al., 2012). The canopy surface resistance differs considerably between global scale chemistry climate models (Anav et al., 2018). The canopy surface resistance is regulated by the stomatal uptake, which relies on stomatal conductance, as well as external plant surfaces (Hoshika et al., 2017). In both mostly used models, biases in the diurnal cycle of deposition O_3 fluxes and partitioning between stomatal and non-stomatal fluxes have a significant effect on annual O_3 dry deposition (Hardacre et al., 2015).

Small differences in O_3 dry deposition fluxes can lead to large differences in tropospheric O_3 concentrations (Hardacre et al., 2015). The standard version of the UFORE/i-Tree model does not consider the drought-induced stomata closure that is limiting the O_3 uptake (Saunders et al., 2011; Morani et al., 2014). Hence, the estimated O_3 removal can be overestimated in cities with no irrigation system or in Mediterranean area (De Marco et al., 2016). A better quantification of O_3 removal by urban trees could be obtained by improving constraints on deposition velocities, a better quantification of the stomatal O_3 uptake by the inclusion of drought effect on leaf-level stomatal conductance and by taking into account wet deposition.

The leaf-level stomatal conductance can be well estimated by using the DO3SE model (Emberson et al., 2000). The stomatal flux-based approach estimates the amount of O_3 that is absorbed into the leaves through stomata and integrates the effects of multiple climatic factors, vegetation characteristics and local and phenological inputs on O_3 uptake through some limiting functions over the growing season (Paoletti and Manning, 2007; Anav et al., 2018). The limiting functions f_{phen}, f_{light}, f_{temp}, f_{VPD} and f_{SWC} consider the variation in the maximum stomatal conductance with phenology, photosynthetic photon flux density, air temperature, vapor pressure deficit and volumetric soil water content, respectively (Anav et al., 2018). The soil-water limitation function (f_{SWC}) should be included in the model because it is critical for environments characterized by summer water stress, e.g. Mediterranean region (De Marco et al., 2015, 2016).

Plant phenology regulates the gas exchange between the vegetation and the atmosphere (Anav et al., 2017). However, model-based phenology fails to capture local-to regional-scale variability specific to plant species (Melaas et al., 2016). Reliable phenological models are needed to model the phenological response to climate forcing and to define the proper growing season for the O_3 accumulation into the leaves, for instance in the DO3SE model. In this regard, remote sensing is a valuable tool for plant phenology estimation (Anav et al., 2017) though the temporal variation in the normalized difference vegetation index (NDVI) and LAI.

Patterns in dry O_3 deposition are correlated with the seasonal biologic cycle of trees, in particular the LAI (Nowak et al., 2006). The UFORE/i-Tree model estimates the leaf area using regression equations (Nowak et al., 2006) with field measurements as input data. If the species details are not available, the model uses values that are defined by genus or family, and an average LAI of 6 is applied. This is problematic if the model is applied to regions outside U.S., for which the parameters were designed (Pace et al., 2018), or in cities where the LAI measured is lower, e.g. in Santiago (LAI = 3; Escobedo and Nowak, 2009). In Nowak et al. (2014), Manes et al. (2016) and Nowak et al. (2018), maximum (mid-summer) and average LAI values were derived from satellite data for the growing season. In areas where LAI values were missing, a mid-summer LAI value of 4.9 was used based on the average LAI in urban areas. The CHIMERE dry deposition model estimates the seasonal LAI variation from a phenology function: minimum LAI till start date of the growing season and maximum LAI during the growing season (Alonso et al., 2011). In Bottalico et al. (2017), seasonal LAI was spatially estimated using a regression model between in-field LAI survey and Airborne Laser Scanning data, and the LAI values were kept constant during the in-leaf season for deciduous species and throughout the year for evergreen species. The refined LAI and the seasonal variation also explain the higher O_3 removal rates reported in Florence, compared to other cities (Table 2S).

In each city, application of a generic deciduous forest (Nowak et al., 2006; Grote et al., 2016) instead of a refined land cover scheme results in a systematic bias of O_3 deposition fluxes. However, initializations using remote sensing data have an inevitable degree of uncertainty concerning species differentiation (Parmehr et al., 2016). Further studies are needed to derive a transfer function between reflectance values of the acquired satellite images and biophysical variable measurements in cities (such as LAI, tree height) as model inputs.

The tree species categorization, as well as an accurate estimation of LAI, under both leaf-on and leaf-off conditions are challenging (high-density stands, logistical challenges, cost issues) to refine air pollution removal estimation by urban trees. To quantify ecosystem services such as air quality regulation, the LAI is an important index (Klingberg et al., 2017b). The urban areas are heterogeneous with a wide range of plant species and LAI values, emphasizing the importance of accurate estimates of LAI variation in cities in different climates for assessments of ecosystem services of urban forests (Alonzo et al., 2015; Klingberg et al., 2017b; Ren et al., 2018). The UFORE/i-Tree models estimate urban LAI from allometric relationships (Nowak et al., 2018), however, adjustments of LAI logarithms may be performed for crowns with dimensions beyond the limits for which the equations were developed, for instance for crowns with leaf loss, pest-diseases and/or after pruning (Alonzo et al., 2015). It is challenging to measure leaf area in cities, but different methods can be used to characterize the LAI heterogeneity in urban areas and map LAI at high spatial resolution: ground measurements (e.g. LAI-2200 LI-COR), hemispherical photography and/or remote sensing. Over larger areas, ground-based measurements are time consuming and costly, making remote sensing techniques more attractive (Alonzo et al., 2015; Klingberg et al., 2017b). LAI can be estimated from LiDAR data, using a Beer-Lambert law based approach, or by remote sensing to provide near-continuous observations at a high temporal and spatial resolution (Pleiades).

Although the UFORE/i-Tree models are practical, the properties of the atmosphere and surface are oversimplified (Hardacre et al., 2015; Anav et al., 2018). The ground measurements of forest related traits, such as phenology, tree distribution, surface area, tree height, LAI, canopy surface resistance, species-specific stomatal conductance, BVOC inventory, are important to run UFORE/i-Tree because these input data define the parameters used for calculating O_3 removal rates and standardized BVOC emission rates.

4.2. Planning the green urban infrastructure

Two local-scale studies in hemi-Boreal region, by using duplicate passive diffusion samplers, reported a limited effect of urban trees on O_3 between tree-covered and open near-road areas (<1 ppb lower in tree-covered areas) in Gothenburg (Sweden) and Helsinki (Finland) during September–October 2010 and June–July 2014, respectively (Grundström and Pleijel, 2014; Yli-Pelkonen et al., 2017a). For O_3, no substantial effect was detected due to the short study period (35 days on average), low background O_3 levels (12 and 18 ppb, respectively) and counteracting processes (down-mixing from air layers above the canopy). Cohen et al. (2014) found even 6% higher daily O_3 mean concentrations in an urban park with 85% tree coverage (dominant tree species, *Ficus microcarpa*) than in an urban square in Tel-Aviv (Israel) on 22 and 23 June 2006. *Ficus microcarpa* is classified as “high” OFP (Benjamin and Winer, 1998; Chaparro and Terradas, 2009). Higher O_3 levels in the tree-covered area is likely due to photochemical reaction of BVOCs (mainly isoprene) emitted by *Ficus* sp. This phenomenon is more pronounced during the summer due to high solar radiation (Cohen et al., 2014). To maximize the effect of urban trees on O_3, proper species selection should be carefully carried out by local authorities.

For an optimal city-averaged air quality improvement, the choice between tall or short and dense or sparse vegetation determines the effect on air pollution (Tallis et al., 2011; Janhäll, 2015). At the street level, roadside urban vegetation may induce a local increase in pollutant concentrations, depending on vegetation structure (e.g. tree, hedge) and design (e.g. density, planting distance). The reduced air flow within tree-covered areas can increase air pollutant levels, in particular fine particles but also O_3, while in open areas the air mass is mixed and diluted by wind (Vardoulakis et al., 2003; Gromke and Ruck, 2009; Salmond et al., 2013; Vranckx et al., 2015). In street canyon, larger tree spacing ensures better air ventilation and reduced air pollution (Gromke and Ruck, 2009; Vos et al., 2013; Wania et al., 2012). Vegetation has to be dense and porous enough to offer large deposition surface area and allow penetration of the air stream (Wania et al., 2012; Salmond et al., 2013; Janhäll, 2015; Vranckx et al., 2015).

In urban areas, where it is often difficult to plant trees in a densely populated city, peri-urban reforestation can be used to supplement the use of urban trees in air pollution control, such as in Shanghai (Zhao et al., 2013) and Mexico City (Baumgardner et al., 2012), and help meet clean air standards. Further studies are needed to evaluate the "plantable area" and to evaluate the effects of a peri-urban reforestation on air quality in the city. To be effective, nature-based strategies to abate O_3 pollution in cities should be implemented at broader spatial scales (i.e. metropolitan area).

For improving urban air quality, by maximizing O_3 removal throughout the year, tree planting programs and management strategies in urban areas need to: a) plant and sustain healthy trees in polluted and populated areas as a priority, thus selecting a diversity species that are well adapted to the local conditions and protecting them from stressors; b) avoid O_3 sensitive species; c) use low VOC and pollen emitting trees; d) supply ample water to vegetation; e) select species adapted to local climate conditions and to urban environment; f) use long-lived and low maintenance species (Beckett et al., 2000; Nowak, 2000; Yang et al., 2005; Manes et al., 2012; Speak et al., 2012; Janhäll, 2015; Nowak et al., 2018). The basic premise of a green city is "*right tree - right place*", i.e. associate tree planting programs to other strategies taking into account the urban environment characteristics (such as architecture, street design).

4.3. Ranking plant species for air quality improvement

A major requirement for comprehensive guidelines to improved O_3 air quality via green infrastructure is an enhanced database of plant species at global scale. To date, studies performed in 196 cities were mainly carried out in North America (44% in Canada, 39% in U.S) and Europe (13%) and only 1% in Asia.

Here we propose the S-AQI index as a tool for ranking common urban plant species worldwide on the basis of their ability to improve air quality. We consider major disservices (pollen and BVOC emission) and benefits (capacity of filtering air pollutants, and tolerance to pollution, pests, pathogen and drought).

Trees release pollen at the time of blooming (Garcia-Mozo et al., 2006). Environmental factors, such as air pollution, may increase the development of allergenic pollen through a direct influence on the pollen grains (shape, size, porosity, physiologic features, proteins, enzymes), especially in cities (Obtułowicz, 1993; Hinge et al., 2017). The allergen pollen numbers can increase by O_3, despite the allergen protein can decrease, thus, increasing the allergen exposure; especially elevated CO_2 can increase pollen independently from O_3 (Albertine et al., 2014; Schiavoni et al., 2017). Climate changes have an effect on pollen allergenicity, e.g. air temperature, O_3 (under elevated CO_2) and CO_2 increase determines higher allergen contents in pollen (Albertine et al., 2014; Schiavoni et al., 2017). Estimating the allergenic potential of urban green spaces is of great interest, because 30–40% of world's population is affected by some form of allergy such as asthma exacerbation, sneezing, nasal congestion (Cariñanos et al., 2014). Sensitive groups of citizens suffering from asthma (children, elders, and vulnerable adults) can be particularly sensitive to allergenic pollen (Thibaudon, 2017). A prolonged pollen season with higher amount of pollen released in the air is occurring in particular in Mediterranean countries (Sicard et al., 2012). To overcome this issue, city planners may preferentially favor female plants over male plants in the case of dioecious species (Cariñanos et al., 2014). Breeding of low-pollen production clones, limiting the introduction of exotic species, preferring species associated with low-to-moderate pollen production and increase plant diversity are other useful management opportunities (Cariñanos and Casares-Porcel, 2011).

The local O_3 production depends on the ratio between VOCs and NOx (Markakis et al., 2014): VOC-limited (ratio < 4); optimum (15 > ratio > 4); and NOx-limited conditions (ratio > 15). Different simulations showed that BVOC contribute substantially to O_3 formation (Monks et al., 2015), as 12% on average of the daily maximum 1-h O_3 concentration in Berlin in summer (Churkina et al., 2017), 16% in Osaka Prefecture in 2015 (Ren et al., 2017), up to 30% in Beijing in 2006 (Pang et al., 2009), and 50–75% in summer in Italy (Duane et al., 2002). Due to higher NOx and OH concentrations in urban areas, the BVOC contribution to O_3 formation is higher than in rural NOx-poor areas (Khedive et al., 2017). In non-attainment O_3 areas and urban areas, characterized by "VOC-limited" conditions, O_3 production is not reduced by the reduction in NOx, but rather increased (Kang et al., 2004; Calfapietra et al., 2009; Duncan et al., 2010; Ren et al., 2017). In cities, an effective strategy to mitigate O_3 levels would be to combine significant reductions in VOC emissions with slight changes in NOx emissions (Nowak et al., 2000; Calfapietra et al., 2009; Im et al., 2011; Huszar et al., 2015). Therefore, the use of low BVOC-emitting species may help to keep this ratio low and thus maintain lower O_3 levels (Calfapietra et al., 2013).

The amount of BVOC emissions depends on tree species, LAI, air temperature and other environmental factors (Nowak et al., 2002). The environmental effects were removed, through standardization based on light and temperature conditions, for VOC emission rates and OFP. The algorithm proposed by Guenther et al. (1995) exploits the empirical dependencies from environmental factors and corrects a standardized emission rate measured in standard conditions by light- and temperature-dependent correction factors.

Broad-leaved species emit predominantly isoprene whereas conifers (e.g. *Pinus* sp., *Cedrus* sp.) emit predominantly monoterpenes (Calfapietra et al., 2009). *Quercus* sp. are among the strongest BVOC emitters (Loreto et al., 2009). Isoprene and α-pinene are among the BVOC which are mostly emitted by vegetation (Guenther et al., 2006), with reactivity factors of 9.1 $gO_3(gVOC)$ and 3.3 $gO_3(gVOC)$, respectively (Carter, 1994). The strongest isoprenoids emitters throughout the season, with a high OFP, are species from the genus *Casuarina*, *Eucalyptus*, *Nyssa*, *Populus*, *Quercus*, *Robinia*, and *Salix* (Guenther et al., 1994; Calfapietra et al., 2009; Keenan et al., 2009; Alonso et al., 2011; Khedive et al., 2017; Ren et al., 2017). Benjamin and Winer (1998) estimated the actual OFP of urban trees and shrubs (Table 4S). Plant species producing less than 1 g O_3 $tree^{-1}$ day^{-1} are considered as "low" OFP (e.g. *Acer* sp., *Cupressus* sp., *Fraxinus* sp., *Platanus* sp., *Prunus* sp.), species producing 1–10 g O_3 $tree^{-1}$ day^{-1} are "medium" OFP (e.g. *Betula* sp., *Malus* sp., *Pinus halepensis, Pinus pinea, Pinus ponderosa, Pinus sylvestris*) and those producing greater than 10 g O_3 $tree^{-1}$ day^{-1} are "high" OFP (e.g. *Ficus elastica, Picea abies, Populus* sp., *Quercus* sp.). Tree species with OFP <10 g O_3 $tree^{-1}$ day^{-1} are highly recommended for tree planting program aiming at improving urban air quality (Benjamin and Winer, 1998).

As the production of BVOCs depends on the temperature, and the isoprene-emitter plant species are less adapted than monoterpene-emitter species to aridity (Loreto et al., 2014), isoprenoid-emitting taxa will become more abundant with an increase of O_3 levels in a climate change context (Nowak et al., 2000; Paoletti, 2009; Escobedo et al., 2011).

The most effective species for O_3 removal, i.e. a net O_3 rate lower than 0 g O_3 per g of leaf biomass, are: *Acer* sp., *Ailanthus* sp., *Carpinus* sp., *Crataegus* sp., *Euonymus* sp., *Fagus* sp., *Fraxinus* sp., *Gleditsia triacanthos, Jacaranda* sp., *Juglans* sp., *Hibiscus* sp., *Larix decidua, Liriodendron* sp., *Metasequoia glyp.*, *Morus* sp., *Prunus* sp., *Pyrus* sp., *Sequoia* sp., *Sophora japonica* and *Tilia* sp. (Table 4S). Although *Fagus* sp. and *Sophora japonica* are effective in removing O_3, they have high OFP. During urban land use planning, selection of genera like *Casuarina, Eucalyptus, Ficus, Liquidambar, Myrtus, Mahonia, Platanus*, *Populus* and *Quercus* should be avoided (Table 4S).

More studies are needed to evaluate whether green-roof plant species are well suited to ameliorate air quality. Baraldi et al. (2018) evaluated the species-specific BVOC emission capacity and OFP of green roof species and highlighted that almost all the investigated species were low BVOC emitters (<1 μg g leaf dry weight^{-1} h^{-1} of BVOC) with the exception of *Satureja repandens, Origanum vulgare* and *Campanula persicifolia* (>2 μg g leaf dry weight^{-1} h^{-1} of monoterpenes) considered as moderate emitters and *Hypericum moserianum* (>20 μg g leaf dry weight^{-1} h^{-1} of isoprene) considered as high BVOC emitter.

Elevated O_3 can be harmful to plants by affecting growth, crown defoliation, photosynthesis, and leaf senescence (Karnosky et al., 2007; Paoletti et al., 2009; Mills et al., 2011; Büker et al., 2012; Fares et al., 2013; Sicard and Dalstein-Richier, 2015; Sicard et al., 2016a, c). As O_3 is increasing in cities, the benefits of healthy urban trees for decreasing O_3 levels can be affected by O_3 pollution itself (Paoletti et al., 2014; Sicard et al., 2016b). For areas with high O_3 levels, such as Southern Europe (Sicard et al., 2013), tree species should be selected as non-sensitive to acute and chronic exposure to O_3 (Paoletti, 2006), for instance Mediterranean evergreen broadleaves due to their sclerophyllous leaves and low gas exchange rates (Grulke and Paoletti, 2005).

Moricca et al. (2018) presented an up-to-date list of pathogens and insect pests affecting urban greening as well as a description of symptoms and key diagnostic elements for proper and timely phytosanitary management. In cities, tree species should be selected as tolerant to pests and diseases due to the limited use of phytosanitary products. Since water stress impairs transpiration

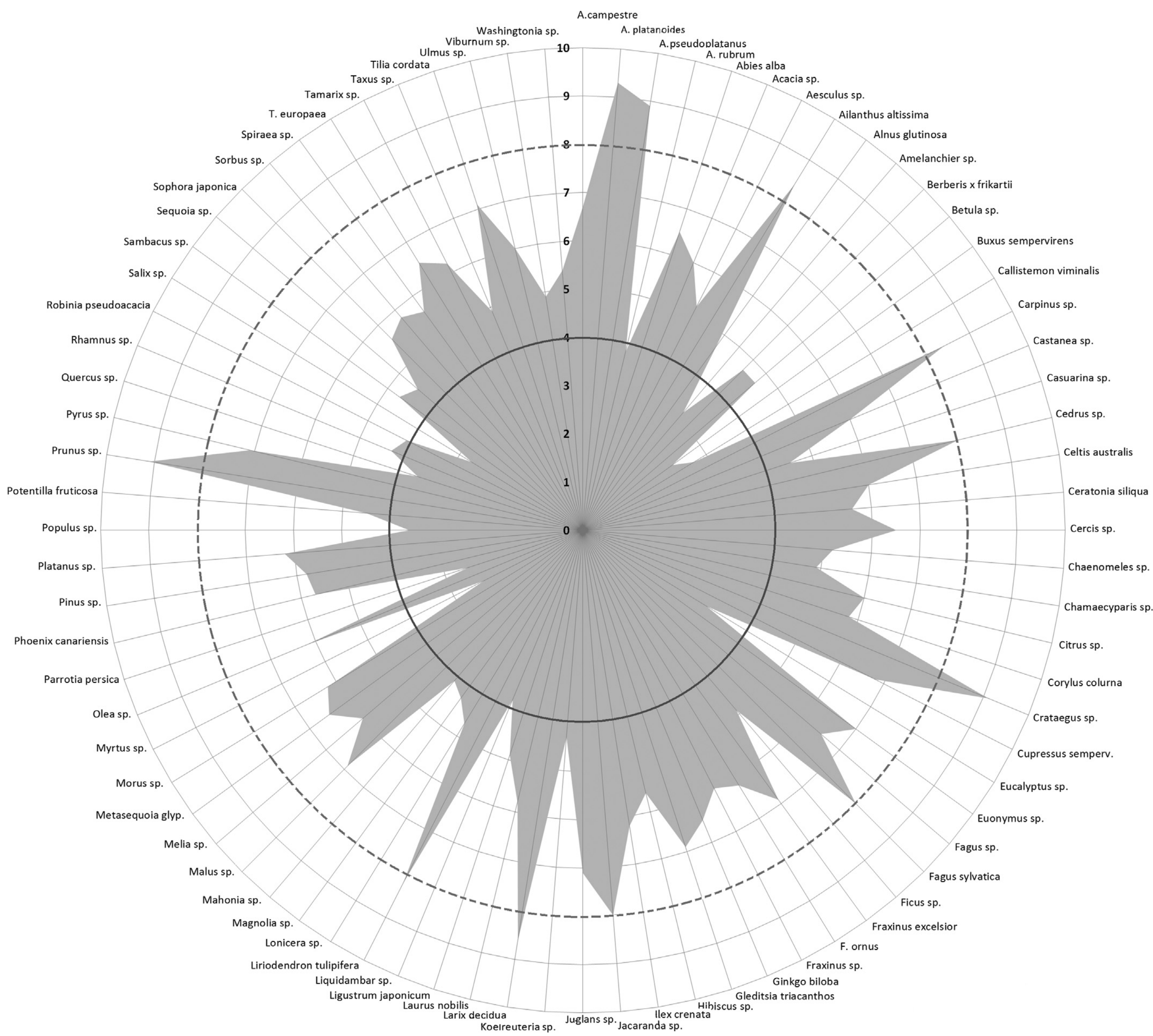

Fig. 2. Species-specific Air Quality Index (S-AQI). S-AQI: 1-4: not recommended (below the tick line); 8-10: recommended plant species for city planting program (over the dotted line).

and stomatal conductance to O_3 (Hoshika et al., 2017), tree species also should be drought tolerant (e.g. *Quercus ilex*) to keep high capacity in removing O_3 through stomatal flux (Escobedo and Nowak, 2009; Fares et al., 2014; Fusaro et al., 2015).

In cities, the combined effects of threatening air pollutant deposition (PM, NO_2 and O_3), disservices (pollen and BVOC emission) and resistance (diseases, pests, drought, O_3) from the urban forest must be considered (Donovan et al., 2005) to rank tree species according to their effectiveness of improving urban air quality. By combining all the above recommendations, the shrubs and tree species with S-AQI exceeding 8 (i.e. top 10) were arbitrarily considered as the most effective to improve air quality in cities: *Acer* sp., *Ailanthus altissima, Carpinus* sp., *Cedrus* sp., *Crataegus* sp., *Fagus sylvatica, Larix decidua, Liriodendron tulipifera* and *Prunus* sp, i.e. 8 broadleaf tree species and 2 conifers (Fig. 2). Even if evergreen broadleaf tree species remove more O_3 than deciduous broadleaf tree species, top 3 of recommended species are deciduous broadleaf tree species which have high O_3, PM_{10} and NO_2 removal efficiency, low OFP and good suitability for urban environments, i.e. robust, low allergenicity of pollen, tolerant to drought, pest and diseases, medium-sized tree and narrow growth habit: *Acer* sp., *Crataegus* sp. and *Prunus* sp. Both *Ace*r sp. and *Prunus* sp. are among the 10 most frequently occurring tree species in cities worldwide (Yang et al., 2015). The less effective plant species to improve air quality in cities are mainly shrubs (such as *Myrtus* sp., *Buxus sempervirens* and *Callistemon viminalis*) due to lower roughness and LAI (Speak et al., 2012) and *Quercus* sp., *Populus* sp. and *Eucalyptus* sp. for tree species, mainly due lower O_3 removal efficiency and higher BVOC emission capacity. The most widely occurring species in cities, *Robinia pseudoacacia* (Yang et al., 2015), have a low efficiency in air pollution removal (S-AQI = 4.1) because in the past, urban tree species were mainly selected for their aesthetic values (Yang et al., 2015).

We believe that the proposed S-AQI index warrants further evaluation, improvement and validation. Expanding the selection criteria to assessments of ecosystem services (e.g. air pollution reduction) versus disservices allows city planners to make better decisions. In order to refine the S-AQI, we recommend including additional selection criteria, for example tree traits considered most relevant for air pollution mitigation such as ultimate size, canopy density, foliage longevity and water-use strategy (Grote et al., 2016), maintenance cost, future VOC and pollen emission rates in a climate change context, while limiting detrimental effects for other ecosystem services (e.g. water use, storm-water runoff).

5. Conclusions

More than 80% of people living in cities are exposed to levels exceeding WHO guidelines for $PM_{2.5}$, PM_{10}, and O_3 (World Health Organization, 2016a). Urban areas represent 0.13% of the global surface area and 52.2% of the urban area is vegetated soil (Jacobson and Ten Hoeve, 2012). For strengthening a sustainable city planning, nature-based solutions where cities are “living labs” and actors of innovation are a major topic on the EU policy agenda (Davies et al., 2017). Until the last decade, the potential of urban green spaces in helping meet clean air standards has been neglected by policy-makers (Nowak et al., 2006; Escobedo et al., 2011; Baró et al., 2014). Mounting research highlights that tree planting could be a viable strategy to improve air quality and is beneficial for citizens' well-being; therefore the tree benefits have been included as key strategies for reducing climate change impacts and improving air quality in urban areas by the U.S. Environmental Protection Agency in 2004 (Yang et al., 2008). In addition to air quality improvement, the advantage of tree planting in cities is also to answer to social needs, e.g. recreation, cultural, aesthetic (Selmi et al., 2016). In addition to traditional air pollution abatement strategies, urban vegetation can be considered into policy options as a cost-effective and nature-based approach (Saunders et al., 2011), especially in the regions where climate change is expected to be more pronounced (Sicard et al., 2013).

Based on ground measurements and modeling studies, integrating deposition and emission effects of trees, this review stresses the key role of urban forest in regulating O_3 air quality and highlights the need for proper species selection for urban greening. Key results are:

- The amount of O_3 removal depends on the type and structure of tree cover, study area, climatic and environmental conditions, and local-to-regional O_3 concentrations.
- Broadleaf tree species remove more O_3 than conifers. Evergreen broadleaf tree species remove more O_3 than deciduous broadleaf tree species.
- The average annual percent air quality improvement due to urban trees and shrubs is less than 2%.
- The ability of urban vegetation to remove O_3 is critical in climates with a long growing season and high O_3 levels.
- Urban forests hold a key role in O_3 removal and provide a perspective for achieving healthier cities.
- Green roofs can be used to supplement the use of urban trees to improve air quality in a densely populated city.
- To maximize the beneficial environmental effects of urban forests, careful design, planning and cost–benefit analysis is required.
- Considering the specificities of each tree species and climate zone, a tool for a species-specific quantification of air quality amelioration (S-AQI) is provided herein.
- There is a need to incorporate local-scale urban forest structure and LAI seasonal variation in the modeling of urban forest impacts on air quality.
- Urban vegetation can be considered as a cost-effective and nature-based approach to improve citizens' well-being.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2018.08.049.

References

Akbari, H., 2002. Shade trees reduce building energy use and CO_2 emissions from power plants. Environ. Pollut. 116, 119–126.
Albertine, J.M., Manning, W.J., DaCosta, M., Stinson, K.A., Muilenberg, M.L., Rogers, C.A., 2014. Projected carbon dioxide to increase grass pollen and allergen exposure despite higher ozone levels. PLoS One 9, e111712.
Alonso, R., Vivanco, M.G., González-Fernández, I., Bermejo, V., Palomino, I., Garrido, J.L., Elvira, S., Salvador, P., Artíñano, B., 2011. Modelling the influence of peri-urban trees in the air quality of Madrid region (Spain). Environ. Pollut. 159, 2138–2147.
Alonzo, M., Bookhagen, B., McFadden, J.P., Sun, A., Roberts, D.A., 2015. Mapping urban forest leaf area index with airborne lidar using penetration metrics and

allometry. Rem. Sens. Environ. 162, 141–153.
Anav, A., Proietti, C., Menut, L., Carnicelli, S., De Marco, A., Paoletti, E., 2018. Sensitivity of stomatal conductance to soil moisture: implications for tropospheric ozone. Atmos. Chem. Phys. 18, 5747–5763.
Anav, A., Liu, Q., De Marco, A., Proietti, C., Savi, F., Paoletti, E., Piao, S., 2017. The role of plant phenology in stomatal ozone flux modeling. Global Change Biol. 24, 235–248.
Anav, A., De Marco, A., Proietti, C., Alessandri, A., Dell'Aquila, A., Cionni, I., Friedlingstein, P., Khvorostyanov, D., Menut, L., Paoletti, E., Sicard, P., Sitch, S., Vitale, M., 2016. Comparing concentration-based (AOT40) and stomatal uptake (PODY) metrics for ozone risk assessment to European forests. Global Change Biol. 22, 1608–1627.
Anderson, G.B., Krall, J.R., Peng, R.D., Bell, M.L., 2012. Is the relation between ozone and mortality confounded by chemical components of particulate matter? Am. J. Epidemiol. 176, 726–732.
Andersson-Sköld, Y., Thorsson, S., Rayner, D., Lindberg, F., Janhäll, S., Jonsson, A., Moback, U., Bergman, R., Granberg, M., 2015. An integrated method for assessing climate-related risks and adaptation alternatives in urban areas. Climate Risk Management 7, 31–50.
Ashmore, M.R., 2005. Assessing the future global impacts of ozone on vegetation. Plant Cell Environ. 28, 949–964.
Baraldi, R., Neri, L., Costa, F., Facini, O., Rapparini, F., Carriero, G., 2018. Ecophysiological and micromorphological characterization of green roof vegetation for urban mitigation. Urban For. Urban Green. https://doi.org/10.1016/j.ufug.2018.03.002 (in press).
Baró, F., Chaparro, L., Gómez-Baggethun, E., Langemeyer, J., Nowak, D.J., Terradas, J., 2014. Contribution of ecosystem services to air quality and climate change mitigation policies: the case of urban forests in Barcelona, Spain. Ambio 43, 466–479.
Baró, F., Haase, D., Gómez-Baggethun, E., Frantzeskaki, N., 2015. Mismatches between ecosystem services supply and demand in urban areas: a quantitative assessment in five European cities. Ecol. Indicat. 55, 148–156.
Baumgardner, D., Varela, S., Escobedo, F.J., Chacalo, A., Ochoa, C., 2012. The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. Environ. Pollut. 163, 174–183.
Beckett, K.P., Freer-Smith, P., Taylor, G., 2000. Particulate pollution capture by urban trees: effect of species and wind speed. Global Change Biol. 6, 995–1003.
Benjamin, M.T., Winer, A.M., 1998. Estimating the ozone-forming potential of urban trees and shrubs. Atmos. Environ. 32, 53–68.
Bogaert, P., Christakos, G., Jerrett, M., Yu, H.L., 2009. Spatio-temporal modelling of ozone distribution in the State of California. Atmos. Environ. 43, 2471–2480.
Bottalico, F., Travaglini, D., Chirici, G., Garfì, V., Giannetti, F., De Marco, A., Fares, S., Marchetti, M., Nocentini, S., Paoletti, E., Salbitano, F., Sanesi, G., 2017. A spatially-explicit method to assess the dry deposition of air pollution by urban forests in the city of Florence, Italy. Urban For. Urban Green. 27, 221–234.
Bowler, D.E., Buyung, A.L., Knight, T.M., Pullin, A.S., 2010. Urban greening to cool towns and cities: a systematic review of the empirical evidence. Landsc. Urban Plann. 97, 147–155.
Büker, P., Emberson, L., Falk, R., Briolat, A., Cinderby, S., Cambridge, H., Harmens, H., Mills, G., et al., 2012. Ozone pollution: impacts on carbon sequestration in Europe. In: Harmens, H., Mills, G. (Eds.), Report Prepared by ICP Vegetation. ICP Vegetation, CEH Bangor, UK, 90pp.
Bytnerowicz, A., Fenn, M.E., Miller, P.R., Arbaugh, M.J., 1999. Wet and dry pollutant deposition to the mixed conifer forest. In: Miller, P.R., McBride, J.R. (Eds.), Oxidant Air Pollution Impacts in the Montane Forests of Southern California: a Case Study of the San Bernardino Mountains, vol. 134, pp. 235–269. Ecol. Stud.
Calfapietra, C., Fares, S., Manes, F., Morani, A., Sgrigna, G., Loreto, F., 2013. Role of Biogenic Volatile Organic Compounds (BVOC) emitted by urban trees on ozone concentration in cities: a review. Environ. Pollut. 183, 71–80.
Calfapietra, C., Fares, S., Loreto, F., 2009. Volatile organic compounds from Italian vegetation and their interaction with ozone. Environ. Pollut. 157, 1478–1486.
Carinanos, P., Casares-Porcel, M., Quesada-Rubio, J.-M., 2014. Estimating the allergenic potential of urban green spaces: a case study in Granada, Spain. Landsc. Urban Plann. 123, 134–144.
Cariñanos, P., Casares-Porcel, M., 2011. Urban green zones and related pollen allergy: a review. Some guidelines for designing spaces with low allergy impact. Landsc. Urban Plann. 101, 205–214.
Carter, W.P.L., 1994. Development of ozone reactivity scales for volatile organic compounds. J. Air Waste Manag. Assoc. 44, 881–899.
Chaparro, L., Terradas, J., 2009. Ecological Services of Urban Forest in Barcelona. Centre de Recerca Ecològica i Aplicacions Forestals, Universitat Autònoma de Barcelona, Bellaterra, 103pp.
Churkina, G., Kuik, F., Bonn, B., Lauer, A., Grote, R., Tomiak, K., Butler, T.M., 2017. Effect of VOC emissions from vegetation on air quality in Berlin during a heatwave. Environ. Sci. Technol. 51, 6120–6130.
Cieslik, S.A., 2004. Ozone uptake by various surface types: a comparison between dose and exposure. Atmos. Environ. 38, 2409–2420.
Cohen, P., Potchter, O., Schnell, I., 2014. The impact of an urban park on air pollution and noise levels in the Mediterranean city of Tel-Aviv, Israel. Environ. Pollut. 195, 73–83.
Connop, S., Vandergert, P., Eisenberg, B., Collier, M.J., Nash, C., Clough, J., Newport, D., 2016. Renaturing cities using a regionally-focused biodiversity-led multifunctional benefits approach to urban green infrastructure. Environ. Sci. Pol. 62, 99–111.
Currie, B.A., Bass, B., 2008. Estimates of air pollution mitigation with green plants and green roofs using the UFORE model. Urban Ecosyst. 11, 409–422.
Dadvand, P., Nieuwenhuijsen, M.J., Esnaola, M., Forns, J., Basagana, X., Alvarez-Pedrerol, M., Rivas, I., Lopez-Vicente, M., De Castro Pascual, M., Su, J., Jerrett, M., Querol, X., Sunyer, J., 2015. Green spaces and cognitive development in primary schoolchildren. Proc. Natl. Acad. Sci. Unit. States Am. 112, 7937–7942.
Davies, H.J., Doick, K.J., Hudson, M.D., Schreckenberg, K., 2017. Challenges for tree officers to enhance the provision of regulating ecosystem services from urban forests. Environ. Res. 156, 97–107.
De Marco, A., Sicard, P., Fares, S., Tuovinen, J.-P., Anav, A., Paoletti, E., 2016. Assessing the role of soil water limitation in determining the Phytotoxic Ozone Dose (PODY) thresholds. Atmos. Environ. 147, 88–97.
De Marco, A., Sicard, P., Vitale, M., Carriero, G., Renou, C., Paoletti, E., 2015. Metrics of ozone risk assessment for Southern European forests: canopy moisture content as a potential plant response indicator. Atmos. Environ. 120, 182–190.
Deutsch, B., Whitlow, H., Sullivan, M., Savineau, A., 2005. Re-greening Washington, DC: a green roof vision based on environmental benefits for air quality and storm water management. In: Proc. 3rd North American Green Roof Conference: Greening Rooftops for Sustainable Communities, Washington, DC. 4-6 May 2005. The Cardinal Group, Toronto, Canada.
Donovan, R.G., Stewart, H.E., Owen, S.M., Mac Kenzie, A.R., Hewitt, C.N., 2005. Development and application of an urban tree air quality score for photochemical pollution episodes using the Birmingham, United Kingdom, area as a case study. Environ. Sci. Technol. 39, 6730–6738.
Duane, M., Poma, B., Rembges, D., Astorga, C., Larsen, B.R., 2002. Isoprene and its degradation products as strong ozone precursors in Insubria, Northern Italy. Atmos. Environ. 36, 3867–3879.
Duncan, B.N., Yoshida, Y., Olson, J.R., Sillman, S., Martin, R.V., Lamsal, L., Hu, Y., Pickering, K.E., Retscher, C., Allen, D.J., Crawford, J.H., 2010. Application of OMI observations to a space-based indicator of NOx and VOC controls on surface ozone formation. Atmos. Environ. 44, 2213–2223.
European Environment Agency, 2017. Air Quality in Europe - 2017 Report". EEA Report No 13/2017.
Emberson, L.D., Kitwiroon, N., Beevers, S., Büker, P., Cinderby, S., 2013. Scorched Earth: how will changes in the strength of the vegetation sink to ozone deposition affect human health and ecosystems? Atmos. Chem. Phys. 13, 6741–6755.
Emberson, L., Ashmore, M.R., Cambridge, H.M., Simpson, D., Tuovinen, J.P., 2000. Modelling stomatal ozone flux across Europe. Environ. Pollut. 109, 403–413.
Escobedo, F.J., Nowak, D.J., 2009. Spatial heterogeneity and air pollution removal by an urban forest. Landsc. Urban Plann. 90, 102–110.
Escobedo, F.J., Kroeger, T., Wagner, J.E., 2011. Urban forests and pollution mitigation: analyzing ecosystem services and disservices. Environ. Pollut. 159, 2078–2087.
Fantozzi, F., Monaci, F., Blanusa, T., Bargagli, R., 2015. Spatio-temporal variations of ozone and nitrogen dioxide concentrations under urban trees and in a nearby open area. Urban Climate 12, 119–122.
Fares, S., Savi, F., Muller, J., Matteucci, G., Paoletti, E., 2014. Simultaneous measurements of above and below canopy ozone fluxes help partitioning ozone deposition between its various sinks in a Mediterranean Oak forest. Agric. For. Meteorol. 198, 181–191.
Fares, S., Vargas, R., Detto, M., Goldstein, A.H., Karlik, J., Paoletti, E., Vitale, M., 2013. Tropospheric ozone reduces carbon assimilation in trees: estimates from analysis of continuous flux measurements. Global Change Biol. 19, 2427–2443.
Fares, S., Loreto, F., Kleist, E., Wildt, J., 2008. Stomatal uptake and stomatal deposition of ozone in isoprene and monoterpene emitting plants. Plant Biol. 10, 44–54.
Foley, K.M., Napelenok, S.L., Jang, C., Phillips, S., Hubbell, B.J., Fulcher, C.M., 2014. Two reduced form air quality modeling techniques for rapidly calculating pollutant mitigation potential across many sources, locations and precursor emission types. Atmos. Environ. 98, 283–289.
Fusaro, L., Salvatori, E., Mereu, S., Marando, F., Scassellati, E., Abbate, G., Manes, F., 2015. Urban and peri-urban forests in the metropolitan area of Rome: ecophysiological response of Quercus ilex L. in two green infrastructures in an ecosystem services perspective. Urban For. Urban Green. 14, 1147–1156.
García-Gómez, H., Aguillaume, L., Izquieta-Rojano, S., Valiño, F., Àvila, A., Elustondo, D., Santamaría, J.M., Alastuey, A., Calvete-Sogo, H., González-Fernández, I., Alonso, R., 2016. Atmospheric pollutants in peri-urban forests of Quercus ilex: evidence of pollution abatement and threats for vegetation. Environ. Sci. Pollut. Control Ser. 23, 6400–6413.
Garcia-Mozo, H., Galán, C., Jato, V., Belmonte, J., Diaz de la Guardia, C., Fernandez, D., Gutierrez, M., Aira, M.J., Roure, J.M., Ruiz, L., Trigo, M.M., Dominguez-Vilches, E., 2006. Quercus pollen season dynamics in the Iberian Peninsula: response to meteorological parameters and possible consequences of climate change. Ann. Agric. Environ. Med. 13, 209–224.
Goudarzi, G., Daryanoosh, S.M., Godini, H., Hopke, P.K., Sicard, P., De Marco, A., Dehdari Rad, H., Harbizadeh, A., Jahedi, F., Mohammadi, M.J., Savari, J., Sadeghi, S., Kaabi, Z., Khaniabadi, Y.O., 2017. Health risk assessment of exposure to the Middle-Eastern Dust storms in Iranian megacity of Kermanshah. Publ. Health 148, 109–116.
Gregory, D., McLaughlin, O., Mullender, S., Sundararajah, N., 2016. New Solutions to Air Pollution Challenges in the UK". Briefing Paper Produced by the London Forum for Science and Policy. published by the Grantham Institute. April 2016.
Gromke, C., Ruck, B., 2009. On the impact of trees on dispersion processes of traffic emissions in street canyons. Boundary-Layer Meteorol. 131, 19–34.
Grote, R., Samson, R., Alonso, R., Amorim, J.H., Cariñanos, P., Churkina, G., Fares, S., Thiec, D.L., Niinemets, Ü., Mikkelsen, T.N., Paoletti, E., Tiwary, A., Calfapietra, C.,

2016. Functional traits of urban trees: air pollution mitigation potential. Front. Ecol. Environ. 14, 543–550.
Grulke, N.E., Paoletti, E., 2005. A field system to deliver desired O_3 concentrations in leaf-level gas exchange measurements: results for Holm Oak near a CO_2 spring. Phyton 45, 21–31.
Grundström, M., Pleijel, H., 2014. Limited effect of urban tree vegetation on NO_2 and O_3 concentrations near a traffic route. Environ. Pollut. 189, 73–77.
Guenther, A., Zimmerman, P.R., Wildermuth, M., 1994. Natural volatile organic compound emission rates for U.S. woodland landscapes. Atmos. Environ. 28, 1197–1210.
Guenther, A., Hewitt, C.N., Erickson, D., Fall, R., Geron, C., Graedel, T., Harley, P., Klinger, L., Lerdau, M., McKay, W.A., Pierce, T., Scholes, B., Steinbrecher, R., Tallamraju, R., Taylor, J., Zimmerman, P., 1995. A global model of natural volatile organic compound emissions. J. Geophys. Res. 100, 8873–8892.
Guenther, A., Karl, T., Harley, P., Wiedinmyer, C., Palmer, P.I., Geron, C., 2006. Estimates of global terrestrial isoprene emissions using MEGAN (model of emissions of gases and aerosols from nature). Atmos. Chem. Phys. 6, 3181–3210.
Guerreiro, C.B.B., Foltescu, V., de Leeuw, F., 2014. Air quality status and trends in Europe. Atmos. Environ. 98, 376–384.
Hardacre, C., Wild, O., Emberson, L., 2015. An evaluation of ozone dry deposition in global scale chemistry climate models. Atmos. Chem. Phys. 15, 6419–6436.
Harris, T.B., Manning, W.J., 2010. Nitrogen dioxide and ozone levels in urban tree canopies. Environ. Pollut. 158, 2384–2386.
Hinge, V., Tidke, J., Das, B., Bhute, S., Parab, P., Apte, K., 2017. Air pollution exacerbates effect of allergenic pollen proteins of Cassia siamea: a preliminary report. Grana 56, 147–154.
Hirabayashi, S., Kroll, C.N., Nowak, D.J., 2012. I-tree Eco Dry Deposition Model Descriptions, p. 23.
Hirabayashi, S., Kroll, C., Nowak, D., 2011. Component-based development and sensitivity analyses of an air pollutant dry deposition model. Environ. Model. Software 26, 804–816.
Hoshika, Y., Carrari, E., Noguchi, I., Magrini, J., Forni, F., Paoletti, E., 2018. Estimating Pollution Removal by Individual Trees: A Species-Specific Case Study in Italy. Article in preparation.
Hoshika, Y., Fares, S., Savi, F., Gruening, C., Goded, I., De Marco, A., Sicard, P., Paoletti, E., 2017. Stomatal conductance models for ozone risk assessment at canopy level in two Mediterranean evergreen forests. Agric. For. Meteorol. 234, 212–221.
Hu, Y., Zhao, P., Niu, J., Sun, Z., Zhu, L., Ni, G., 2016. Canopy stomatal uptake of NOx, SO_2 and O_3 by mature urban plantations based on sapflow measurement. Atmos. Environ. 125, 165–177.
Huszar, P., Belda, M., Halenka, T., 2015. On the long term impact of emissions from central European cities on regional air-quality. Atmos. Chem. Phys. Discuss. 15, 32101–32155.
Im, U., Poupkou, A., Incecik, S., Markakis, K., Kindap, T., Unal, A., Melas, D., Yenigun, O., Topcu, S., Odman, T.M., Tayanc, M., Guler, M., 2011. The impact of anthropogenic and biogenic emissions on surface ozone concentrations in Istanbul. Sci. Total Environ. 409, 1255–1265.
IPCC, 2014. Intergovernmental panel on climate change: summary for policymakers. "Climate Change 2014: Impacts, Adaptation and Vulnerability". Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.
Jacobson, M.Z., Ten Hoeve, J.E., 2012. Effects of urban surfaces and white roofs on global and regional climate. J. Clim. 25, 1028–1044.
Janhäll, S., 2015. Review on urban vegetation and particle air pollution - deposition and dispersion. Atmos. Environ. 105, 130–137.
Jayasooriya, V.M., Ng, A.W.M., Muthukumaran, S., Perera, B.J.C., 2017. Green infrastructure practices for improvement of urban air quality. Urban For. Urban Green. 21, 34–47.
Jerrett, M., Burnett, R.T., Pope, C.A., Ito, K., Thurston, G., Krewski, D., Shi, Y., Calle, E., Thun, M., 2009. Long-term ozone exposure and mortality. N. Engl. J. Med. 360, 1085–1095.
Kang, D., Aneja, V.P., Mathur, R., Ray, J.D., 2004. Observed and modeled VOC chemistry under high VOC/NOx conditions in the Southeast United States national parks. Atmos. Environ. 38, 4969–4974.
Karnosky, D.F., Skelly, J.M., Percy, K.E., Chappelka, A.H., 2007. Perspectives regarding 50 years of research on effects of tropospheric ozone air pollution on U.S. Forests. Environ. Pollut. 147, 489–506.
Keenan, T., Niinemets, U., Sabate, S., Gracia, C., Peñuelas, J., 2009. Process based inventory of isoprenoid emissions from European forests: model comparisons, current knowledge and uncertainties. Atmos. Chem. Phys. 9, 4053–4076.
Kerckhoffs, J., 2014. "A Review of the Effects of Urban Vegetation on Air Quality". Master Thesis, Faculty of Medicine, Toxicology and Environmental Health. Institute for Risk Assessment Sciences, p. 14.
Khaniabadi, Y.O., Hopke, P.K., Daryanoosh, S.M., Goudarzi, G., Fanelli, R., Itai Kloog, I., Sicard, P., Godini, H., Babaei, A.A., Basiri, H., Mohammadi, M.J., Shikhrobat, M.V., De Marco, A., 2017. Hospital admissions in Iran for cardiovascular and respiratory diseases attributed to the Middle-Eastern Dust Storms. Environ. Sci. Pollut. Control Ser. 24, 16860–16868.
Khaniabadi, Y.O., Daryanoosh, S.M., Hopke, P.K., Ferrante, M., De Marco, A., Sicard, P., Conti, G.O., Goudarzi, G., Basiri, H., Mohammadi, M.J., Keishams, F., 2016. Acute myocardial infarction and chronic obstructive pulmonary disease attributed to ambient SO_2 in Iran. Environ. Res. 156, 683–687.
Khedive, E., Shirvany, A., Assareh, M.H., Sharkey, T.D., 2017. *In-situ* emission of BVOCs by three urban woody species. Urban For. Urban Green. 21, 153–157.
Klingberg, J., Broberg, M., Strandberg, B., Thorsson, P., Pleijel, H., 2017a. Influence of urban vegetation on air pollution and noise exposure - a case study in Gothenburg, Sweden. Sci. Total Environ. 599–600, 1728–1739.
Klingberg, J., Konarska, J., Lindberg, F., Johansson, L., Thorsson, S., 2017b. Mapping leaf area of urban greenery using aerial LiDAR and ground-based measurements in Gothenburg, Sweden. Urban For. Urban Green. 26, 31–40.
Krmpotic, D., Luzar-Stiffler, V., Herzog, P., 2015. Effects of urban ozone pollution on hospitalizations for exacerbation of chronic obstructive pulmonary disease. Eur. Respir. J. 46, PA3411.
Kroeger, T., Escobedo, F.J., Goodman, J.L., Varela, S., Delphin, S., Fisher, J.R.B., Waldron, J., 2014. Reforestation as a novel abatement and compliance measure for ground-level ozone. In: Proceedings of the National Academy of Sciences of the United States of America, Published Online September 8, 2014.
Lafortezza, R., Davies, C., Sanesi, G., Konijnendijk, C.C., 2013. Green Infrastructure as a tool to support spatial planning in European urban regions. iFor. Biogeosci. For. 6, 102–108.
Lee, B.J., Kim, B., Lee, K., 2014. Air pollution exposure and cardiovascular disease. Toxicological Research 30, 71–75.
Lefohn, A.S., Malley, C.S., Smith, L., Wells, B., Hazucha, M., Simon, H., et al., 2018. Tropospheric Ozone Assessment Report: global ozone metrics for climate change, human health, and crop/ecosystem research. Elementa: Science of the Anthropocene 6, 28. https://doi.org/10.1525/elementa.279.
Lefohn, A.S., Emery, C., Shadwick, D., Wernli, H., Jung, J., Oltmans, S.J., 2014. Estimates of background surface ozone concentrations in the United States based on model-derived source apportionment. Atmos. Environ. 84, 275–288.
Li, W.C., Yeung, K.K.A., 2014. A comprehensive study of green roof performance from environmental perspective. International Journal of Sustainable Built Environment 3, 127–134.
Loreto, F., Bagnoli, F., Calfapietra, C., Cafasso, D., De Lillis, M., Filibeck, G., Fineschi, S., Guidolotti, G., et al., 2014. Isoprenoid emission in hygrophyte and xerophyte European woody flora: ecological and evolutionary implications. Global Ecol. Biogeogr. 23, 334–345.
Loreto, F., Bagnoli, F., Fineschi, S., 2009. One species, many terpenes: matching chemical and biological diversity. Trends Plant Sci. 14, 416–420.
Luley, C.J., Bond, J., 2002. "A Plan to Integrate Management of Urban Trees into Air Quality Planning". Report to Northeast State Foresters Association. Davey Resource Group, Kent, OH, p. 73pp.
Malley, C.S., Henze, D.K., Kuylenstierna, J.C.I., Vallack, H.W., Davila, Y., Anenberg, S.C., Turner, M.C., Ashmore, M.R., 2017. "Updated global estimates of respiratory mortality in adults $\geq$30 Years of age attributable to long-term ozone exposure. Environ. Health Perspect. https://doi.org/10.1289/EHP1390.
Manes, F., Marando, F., Capotorti, G., Blasi, C., Salvatori, E., Fusaro, L., Ciancarella, L., Mircea, M., Marchetti, M., Chirici, G., Munafò, M., 2016. Regulating ecosystem services of forests in ten Italian metropolitan cities: air quality improvement by PM_{10} and O_3 removal. Ecol. Indicat. 67, 425–440.
Manes, F., Incerti, G., Salvatori, E., Vitale, M., Ricotta, C., Costanza, R., 2012. Urban ecosystem services: tree diversity and stability of tropospheric ozone removal. Ecol. Appl. 22, 349–360.
Manning, W.J., 2013. Urban environment: recognizing that pollution and social factors can create barriers to development of more healthy and liveable cities. Environ. Pollut. 183, 1.
Markakis, K., Valari, M., Colette, A., Sanchez, O., Perrussel, O., Honore, C., Vautard, R., Klimont, Z., Rao, S., 2014. Air-quality in the mid-21st century for the city of Paris under two climate scenarios; from regional to local scale. Atmos. Chem. Phys. 14, 95–136.
Mc Donald, A.G., Bealey, W.J., Fowler, D., Dragosits, U., Skiba, U., Smith, R.I., Donovan, R.G., Brett, H.E., Hewitt, C.N., Nemitz, E., 2007. Quantifying the effect of urban tree planting on concentrations and depositions of PM_{10} in two UK conurbations. Atmos. Environ. 41, 8455–8467.
Melaas, E.K., Friedl, M.A., Richardson, A.D., 2016. Multiscale modeling of spring phenology across deciduous forests in the eastern United States. Global Change Biol. 22, 792–805.
Mills, G., Hayes, F., Simpson, D., Emberson, L., Norris, D., Harmens, H., Büker, P., 2011. Evidence of widespread effects of ozone on crops and (semi-)natural vegetation in Europe (1990-2006) in relation to AOT40 and flux-based risk maps. Global Change Biol. 17, 592–613.
Miranda, A.I., Ferreira, J., Silveira, C., Relvas, H., et al., 2016. A cost-efficiency and health benefit approach to improve urban air quality. Sci. Total Environ. 569–570, 342–351.
Monks, P.S., Archibald, A.T., Colette, A., Cooper, O., Coyle, M., Derwent, R., Fowler, D., Granier, C., et al., 2015. Tropospheric ozone and its precursors from the urban to the global scale from air quality to short-lived climate forcer. Atmos. Chem. Phys. 15, 8889–8973.
Morani, A., Nowak, D., Hirabayashi, S., Guidolotti, G., Medori, M., Muzzini, V., Fares, S., Scarascia Mugnozza, G., Calfapietra, C., 2014. Comparing i-Tree modeled ozone deposition with field measurements in a periurban Mediterranean forest. Environ. Pollut. 195, 202–209.
Moricca, S., Bracalini, M., Croci, F., Corsinovi, S., Tiberi, R., Ragazzi, A., Panzavolta, T., 2018. Biotic factors affecting ecosystem services in urban and peri-urban forests in Italy: the role of introduced and impending pathogens and pests. Forests 9, 65.
Nowak, D.J., Hirabayashi, S., Doyle, M., McGovern, M., Pasher, J., 2018. Air pollution removal by urban forests in Canada and its effect on air quality and human health. Urban For. Urban Green. 29, 40–48.

Nowak, D.J., Hirabayashi, S., Bodine, A., Greenfield, E., 2014. Tree and forest effects on air quality and human health in the United States. Environ. Pollut. 193, 119–139.
Nowak, D.J., Hoehn III, R.E., Bodine, Allison R., Greenfield, Eric J., Ellis, Alexis, Endreny, Theodore A., Yang, Yang, Zhou, Tian, Henry, Ruthanne, 2013. "Assessing urban forest effects and values: Toronto's urban forest". U.S. Department of agriculture, forest service, Northern research station. Resource Bulletin NNRS- 79, 59pp.
Nowak, D.J., Dwyer, J.F., 2007. "Understanding the Benefits and Costs of Urban Forest Ecosystems". Urban and Community Forestry in the Northeast, pp. 25–46.
Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban For. Urban Green. 4, 115–123.
Nowak, D.J., Crane, D.E., Stevens, J.C., Ibarra, M., 2002. Brooklyn's urban forest, USDA forest service, Pennsylvania. Environ. Pollut. 116, 381–389.
Nowak, D.J., Crane, D.E., 2000. The Urban Forest Effects (UFORE) Model: quantifying urban forest structure and functions. In: Hansen, M., Burk, T. (Eds.), Integrated Tools for Natural Resources Inventories in the 21st Century, vol. 212. USDA Forest Service General Technical Report NC-, pp. 714–720.
Nowak, D.J., Civerolo, K.L., Rao, S.T., Sistla, G., Luley, C.J., Crane, D.E., 2000. A modeling study of the impact of urban trees on ozone. Atmos. Environ. 34, 1601–1613.
Nowak, D.J., 2000. The interactions between urban forests and global climate change. In: Abdollahi, K., Ning, Z.H., Appeaning, A. (Eds.), Global Climate Change and the Urban Forest. GCRCC and Franklin Press, Baton Rouge, LA, pp. 31–44.
Nuvolone, D., Petri, D., Voller, F., 2017. The effects of ozone on human health. Environ. Sci. Pollut. Control Ser. 25, 8074–8088.
Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R.R., Doshi, H., Dunnett, N., Gaffin, S., Köhler, M., Liu, K.K.Y., Rowe, B., 2007. Green roofs as urban ecosystems: ecological structures, functions, and services. Bioscience 57, 823–833.
Obtułowicz, K., 1993. Air pollution and pollen allergy. Folia Med. Cracov. 34, 121–128.
Ochoa-Hueso, R., Munzi, S., Alonso, R., Arróniz-Crespo, M., Avila, A., et al., 2017. Ecological impacts of atmospheric pollution and interactions with climate change in terrestrial ecosystems of the Mediterranean basin: current research and future directions. Environ. Pollut. 227, 194–206.
Orru, H., Andersson, C., Ebi, K.L., Langner, J., Aström, C., Forsberg, B., 2013. Impact of climate change on ozone-related mortality and morbidity in Europe. Eur. Respir. J. 41, 285–294.
Pace, R., Biber, P., Pretzsch, H., Grote, R., 2018. Modeling ecosystem services for park trees: sensitivity of i-tree eco simulations to light exposure and tree species classification. Forests 9, 89.
Pang, X., Mu, Y., Zhang, Y., Lee, X., Yuan, J., 2009. Contribution of isoprene to formaldehyde and ozone formation based on its oxidation products measurement in Beijing, China. Atmos. Environ. 43, 2142–2147.
Paoletti, E., De Marco, A., Beddows, D.C.S., Harrison, R.M., Manning, W.J., 2014. Ozone levels in European and USA cities are increasing more than at rural sites, while peak values are decreasing. Environ. Pollut. 192, 295–299.
Paoletti, E., 2009. Ozone and urban forests in Italy. Environ. Pollut. 157, 1506–1512.
Paoletti, E., Contran, N., Bernasconi, P., Günthardt-Goerg, M.S., Vollenweider, P., 2009. Structural and physiological responses to ozone in Manna ash (*Fraxinus ornus L.*) leaves in seedlings and mature trees under controlled and ambient conditions. Sci. Total Environ. 407, 1631–1643.
Paoletti, E., Manning, W.J., 2007. Toward a biologically significant and usable standard for ozone that will also protect plants. Environ. Pollut. 150, 85–95.
Paoletti, E., 2006. Impact of ozone on Mediterranean forest: a review. Environ. Pollut. 144, 463–474.
Parmehr, E.G., Amati, M., Taylor, E.J., Livesley, S.J., 2016. Estimation of urban tree canopy cover using random point sampling and remote sensing methods. Urban For. Urban Green. 20, 160–171.
Pascal, M., Corso, M., Chanel, O., Declercq, C., Badaloni, C., et al., 2013. Assessing the public health impacts of urban air pollution in 25 European cities: Results of the Aphekom project. Sci. Total Environ 449, 390–400.
Pataki, D.E., Carreiro, M.M., Cherrier, J., Grulke, N.E., Jennings, V., Pincetl, S., Pouyat, R.V., Whitlow, T.H., Zipperer, W.C., 2011. Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. Front. Ecol. Environ. 9, 27–36.
Proietti, C., Anav, A., De Marco, A., Sicard, P., Vitale, M., 2016. A multi-sites analysis on the ozone effects on Gross Primary Production of European forests. Sci. Total Environ. 556, 1–11.
Ren, Y., Qu, Z., Du, Y., Xu, R., et al., 2017. Air quality and health effects of biogenic volatile organic compounds emissions from urban green spaces and the mitigation strategies. Environ. Pollut. 230, 849–861.
Ren, Z., Du, Y., He, X., Pu, R., Zheng, H., Hu, H., 2018. Spatiotemporal pattern of urban forest leaf area index in response to rapid urbanization and urban greening. J. For. Res. 29, 785–796.
Rowe, D.B., 2011. Green roofs as a means of pollution abatement. Environ. Pollut. 159, 2100–2110.
Saavedra, S., Rodríguez, A., Souto, J.A., Casares, J.J., Bermúdez, J.L., Soto, B., 2012. Trends of rural tropospheric ozone at the Northwest of the iberian peninsula. The Scientific World Journal ID 603034 15pp.
Saitanis, C.J., Panagopoulos, G., Dasopoulou, V., Agathokleous, E., Papatheohari, Y., 2015. "Integrated assessment of ambient ozone phytotoxicity in Greece's Tripolis Plateau". J. Agric. Meteorol. 71, 55–64.
Salmond, J.A., Williams, D.E., Laing, G., Kingham, S., Dirks, K., Longley, I., Henshaw, G.S., 2013. The influence of vegetation on the horizontal and vertical distribution of pollutants in a street canyon. Sci. Total Environ. 443, 287–298.
Sam, L.C., Sohaili, J., 2016. Urban heat island mitigation by introducing green roof system. Environmental Engineering, Hydraulics and Hydrology 3, 13–24.
Saunders, S.M., Dade, E., Van Niel, K., 2011. An Urban Forest Effects (UFORE) model study of the integrated effects of vegetation on local air pollution in the Western Suburbs of Perth, WA. In: 19th International Congress on Modelling and Simulation, Perth, Australia, 12–16 December 2011.
Schiavoni, G., D'Amato, G., Afferni, C., 2017. The dangerous liaison between pollens and pollution in respiratory allergy. Ann. Allergy Asthma Immunol. 118, 269–275.
Schultz, M.G., Schröder, S., Lyapina, O., Cooper, O., Galbally, I., Petropavlovskikh, I., et al., 2017. Tropospheric ozone assessment report: database and metrics data of global surface ozone observations. Elementa: Science of the Anthropocene 5, 58.
Sen, P.K., 1968. Estimates of the regression coefficient based on Kendall's tau. J. of The Am. Statis. Ass. 6, 1379–1389.
Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L., Nowak, D.J., 2016. Air pollution removal by trees in public green spaces in Strasbourg city, France. Urban For. Urban Green. 17, 192–201.
Sicard, P., Anav, A., De Marco, A., Paoletti, E., 2017. Projected global tropospheric ozone impacts on vegetation under different emission and climate scenarios. Atmos. Chem. Phys. 17, 12177–12196.
Sicard, P., De Marco, A., Dalstein-Richier, L., Tagliaferro, F., Paoletti, E., 2016a. An epidemiological assessment of stomatal ozone flux-based critical levels for visible ozone injury in Southern European forests. Sci. Total Environ. 541, 729–741.
Sicard, P., Serra, R., Rossello, P., 2016b. Spatio-temporal trends of surface ozone concentrations and metrics in France. Environ. Res. 149, 122–144.
Sicard, P., Augustaitis, A., Belyazid, S., Calfapietra, C., De Marco, A., Fenn, M., Grulke, N., He, S., Matyssek, R., Serengil, Y., Wieser, G., Paoletti, E., 2016c. Global topics and novel approaches in the study of air pollution, climate change and forest ecosystems. Environ. Pollut. 213, 977–987.
Sicard, P., Dalstein-Richier, L., 2015. Health and vitality assessment of two common pine species in the context of climate change in Southern Europe. Environ. Res. 137, 235–245.
Sicard, P., De Marco, A., Troussier, F., Renou, C., Vas, N., Paoletti, E., 2013. Decrease in surface ozone concentrations at Mediterranean remote sites and increase in the cities. Atmos. Environ. 79, 705–715.
Sicard, P., Lesne, O., Alexandre, N., Mangin, A., Collomp, R., 2011. Air quality trends and potential health effects - development of an aggregate risk index. Atmos. Environ. 45, 1145–1153.
Sicard, P., Thibaudon, M., Besancenot, J.P., Mangin, A., 2012. Forecast models and trends for the main features of the Olea pollen season in Nice (South-eastern France) over the 1990-2009 period. Grana 51, 52–62.
Silli, V., Salvatori, E., Manes, F., 2015. Removal of airborne particulate matter by vegetation in an urban park in the city of Rome (Italy): an ecosystem services perspective. Ann. Bot. (Rome) 5, 53–62.
Simon, H., Reff, A., Wells, B., Xing, J., Frank, N., 2015. Ozone trends across the United States over a period of decreasing NOx and VOC emissions. Environ. Sci. Technol. 49, 186–195.
Simpson, J.R., McPherson, E.G., 2011. The tree BVOC index. Environ. Pollut. 159, 2088–2093.
Soares, A.L., Rego, F.C., McPherson, E.G., Simpson, J.R., Peper, P.J., Xiao, Q., 2011. Benefits and costs of street trees in Lisbon, Portugal. Urban For. Urban Green. 10, 69–78.
Speak, A.F., Rothwell, J.J., Lindley, S.J., Smith, C.L., 2012. Urban particulate pollution reduction by four species of green roof vegetation in a UK city. Atmos. Environ. 61, 283–293.
Tallis, M., Taylor, G., Sinnett, D., Freer-Smith, P.H., 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. Landsc. Urban Plann. 103, 129–138.
Thibaudon, M., 2017. "Pollen Allergy Potency for the Main Urban Plants". Scientific Advisor, Le Réseau National de Surveillance Aérobiologique. European Medical Journal Allergy Immunology 2, 59–60.
Tiwary, A., Sinnett, D., Peachy, C., Chalabi, Z., Vurdoulakis, S., Fletcher, T., et al., 2009. An integrated tool to assess the role of new planting in PM_{10} capture and the human health. Environ. Pollut. 157, 2645–2653.
Tong, Z., Baldauf, R.W., Isakov, V., Deshmukh, P., Zhang, K.M., 2016. Roadside vegetation barrier designs to mitigate near-road air pollution impacts. Sci. Total Environ. 541, 920–927.
Tsay, J.S., Ko, P.H., Chang, P.T., 2015. Carbon storage potential of avenue trees: a comparison of *Barringtonia racemosa, Cyclobalanopsis glauca*, and *Alnus formosana*. J. For. Res. 26, 307–314.
Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kazmierczak, A., Niemela, J., James, P., 2007. Promoting ecosystem and human health in urban areas using green infrastructure: a literature review. Landsc. Urban Plann. 81, 167–178.
United Nations, 2014. The 2014 Revision of the World Urbanization Prospects. Department of Economic and Social Affairs, Population Division. ST/ESA/SER.A/352.
U.S. Environmental Protection Agency, 2015. What Are the Six Common Air Pollutants? Available at: http://www.epa.gov/air/urbanair.
Vardoulakis, S., Fisher, B.E.A., Pericleous, K., Gonzalez-Flesca, N., 2003. Modelling air quality in street canyons: a review. Atmos. Environ. 37, 155–182.
Vos, P.E.J., Maiheu, B., Vankerkom, J., Janssen, S., 2013. Improving local air quality in cities: to tree or not to tree? Environ. Pollut. 183, 113–122.

Vranckx, S., Vos, P., Maiheu, B., Janssen, S., 2015. Impact of trees on pollutant dispersion in street canyons: a numerical study of the annual average effects in Antwerp, Belgium". Sci. Total Environ. 532, 474–483.

Wang, H., Zhou, W., Wang, X., Gao, F., Zheng, H., Tong, L., Ouyang, Z., 2012. Ozone uptake by adult urban trees based on sap flow measurement. Environ. Pollut. 162, 275–286.

Wania, A., Bruse, M., Blond, N., Weber, C., 2012. Analysing the influence of different street vegetation on traffic-induced particle dispersion using microscale simulations. J. Environ. Manag. 94, 91–101.

Weinmayr, G., Romeo, E., De Sario, M., Weiland, S.K., Forastiere, F., 2010. Short-term effects of PM_{10} and NO_2 on respiratory health among children with asthma or asthma-like symptoms: a systematic review and meta-analysis. Environ. Health Perspect. 118, 449–457.

Wilson, R.C., Fleming, Z.L., Monks, P.S., Clain, G., Henne, S., Konovalov, I.B., Szopa, S., Menut, L., 2012. Have primary emission reduction measures reduced ozone across Europe? An analysis of European rural background ozone trends 1996-2005. Atmos. Chem. Phys. 12, 437–454.

World Health Organization, 2016a. WHO's Urban Ambient Air Pollution Database - Update 2016. Public Health, Social and Environmental Determinants of Health Department, Geneva.

World Health Organization, 2016b. Urban green spaces and health. In: Egorov, A.I., Mudu, P. (Eds.), Copenhagen: WHO Regional Office for Europe. Braubach M. and Martuzzi M.

World Health Organisation, 2013. "Review of Evidence on Health Aspects of Air Pollution - REVIHAAP Project". Technical Report. World Health Organization, Regional Office for Europe, Copenhagen, Denmark.

World Health Organisation, 2008. "Health Risks of Ozone from Long-range Transboundary Air Pollution" WHO/Euro Product, ISBN 978 92 890 42895.

Yang, J., Chang, Y.M., Yan, P.B., 2015. Ranking the suitability of common urban tree species for controlling $PM_{2.5}$ pollution. Atmospheric Pollution Research 6, 267–277.

Yang, J., Yu, Q., Gong, P., 2008. Quantifying air pollution removal by green roofs in Chicago. Atmos. Environ. 42, 7266–7273.

Yang, J., Mc Bride, J., Zhou, J., Sun, Z., 2005. The urban forest in Beijing and its role in air pollution reduction. Urban For. Urban Green. 3, 65–78.

Yli-Pelkonen, V., Setälä, H., Viippola, V., 2017a. Urban forests near roads do not reduce gaseous air pollutant concentrations but have an impact on particles levels. Landsc. Urban Plann. 158, 39–47.

Yli-Pelkonen, V., Scott, A.A., Viippola, V., Setälä, H., 2017b. Trees in urban parks and forests reduce O_3, but not NO_2 concentrations in Baltimore, MD, USA. Atmos. Environ. 167, 73–80.

Zare, A., Christensen, J.H., Gross, A., Irannejad, P., Glasius, M., Brandt, J., 2014. Quantifying the contributions of natural emissions to ozone and total fine PM concentrations in the Northern Hemisphere. Atmos. Chem. Phys. 14, 2735–2756.

Zhao, M., Escobedo, F.J., Wang, R., Zhou, Q., Lin, W., Gao, J., 2013. Woody vegetation composition and structure in peri-urban Chongming Island, China. Environ. Manag. 51, 999–1011.