J. For. Res. (2019) 30(6):1981–1996
https://doi.org/10.1007/s11676-019-00916-x

REVIEW ARTICLE

Towards an integrative approach to evaluate the environmental ecosystem services provided by urban forest

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Received: 3 November 2018 / Accepted: 15 January 2019 / Published online: 22 March 2019

Abstract As a Nature-Based Solution, urban forests deliver a number of environmental ecosystem services (EESs). To quantify these EESs, well-defined, reliable, quantifiable and stable indicators are needed. With literature analysis and expert knowledge gathered within COST Action FP1204 GreenInUrbs, we proposed a classification of urban forest EESs into three categories: (A) regulation of air, water, soil and climate; (B) provisioning of habitat quality; and (C) provisioning of other goods and services. Each category is divided into EES types: (a) amelioration of air quality; restoration of soil and water; amelioration of the microclimate; removal of CO_2 from the air; (b) provision of habitat for biodiversity; support for resilient urban ecosystems; provision of genetic diversity; and (c) provision of energy and nutrients; provision of grey infrastructure resilience. Each EES type provides one or more benefits. For each of these 12 benefits, we propose a set of indicators to be used when analyzing the impacts on the identified EESs. Around half of the 36 indicators are relevant to more than one single benefit, which highlights complex interrelationships. The indicators of wider applicability are tree and stand characteristics, followed by leaf physical traits and tree species composition. This knowledge is needed for the optimization of the EESs delivered by urban forests, now and in the future.

Keywords Urban forests · Environmental ecosystem services · Indicators

Project funding: The work was financially supported by COST Action FP1204 GreenInUrbs, PRIN project EUFORICC, and Ministry of Education and Science of the Russian Federation (the Agreement No. 02.A03.21.0008).

The online version is available at http://www.springerlink.com

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Introduction

Urban areas are projected to accommodate 68% of the world's population by 2050 (United Nations 2018). Urban forests (including individual trees and shrubs, parks and forests) play a crucial role in improving the environmental quality of cities and urban dwellers (Roy et al. 2012; Shwartz et al. 2014). Conventional urban greening management primarily aims at enhancing amenity values (Pandit et al. 2013) and maintaining biodiversity (Llausàs and Roe 2012), but growing interest has been focusing on carbon (C) management perspectives (Grimm et al. 2008) and other environmental ecosystem services (EESs). Ecosystem services (ES) are defined as benefits that humans obtain from ecosystem functions (De Groot et al. 2002), or as direct and indirect contributions from ecosystems to human well-being (TEEB 2010). Many ES types have been identified and grouped into three (provisioning, regulating, and cultural services, Maes et al. 2016) or four categories (the former three, plus supporting services; TEEB 2010). A meta-analysis on urban ESs concluded that most studies were undertaken in Europe, North America and China, and that almost 50, 20, 11 and 15% of the studies were about regulating, supporting, provisioning and cultural ESs, respectively (Haase et al. 2014). As compared to other ecosystems like wetlands or natural forests, the attention given to urban ESs is still insufficient (Gómez-Baggethun and Barton 2013), especially when the focus is on the urban forest alone (24% of studies in the meta-analysis by Haase et al. 2014). In the analysis carried out in this paper we established that EESs include all ESs sensu stricto, except the cultural services (Fig. 1).

Because EESs are biodiversity-based, species composition and community structure occurring in urban forests are crucial for the delivery of any EES in cities (Cardinale et al. 2012). Air- and climate-related EESs comprise air purification, climate regulation and C sequestration (McDonald et al. 2007; Armson et al. 2012; Lafortezza and Chen 2016). Other goods and services comprise the delivery of energy, food, non-timber forest products (NTFPs), fresh and clean water, regulation of water runoff and erosion (Guo et al. 2000; Roy et al. 2012). These EESs act at the local level, i.e. at street or neighborhood scale, but may also exert an impact at the regional level, for example, in relation to climate- or water-regulating effects (Guo et al. 2000). Several urban forest EESs are coupled to each other. As an example, water availability can influence the cooling effect of urban forests, but can also affect the interaction of trees with air pollutants. Air pollution in urbanized areas can result in polluted soils (Davidson et al. 2006), surface waters (Le Pape et al. 2012) and groundwater (Gallo et al. 2012). Moreover, the different environmental conditions often observed within the urban environment as compared to its surroundings can affect the physiology of plants (Sicard et al. 2016) and consequently their capacity to provide EESs (Calfapietra et al. 2015). Science-based evidence on urban forest EESs is needed to identify and assess trade-offs, disservices (environmental, social and financial) and complex interactions among different EESs. For instance, certain tree species (e.g. poplars) might take up great amounts of ozone, which is beneficial in terms of air quality, but the same species might be a strong emitter of biogenic volatile organic compounds (BVOCs) and thus contribute to the formation of ozone itself (Calfapietra et al. 2013).

To quantify the urban forest EESs and untangle the complex interrelationships among them in a changing climate and under different socio-cultural conditions, EES

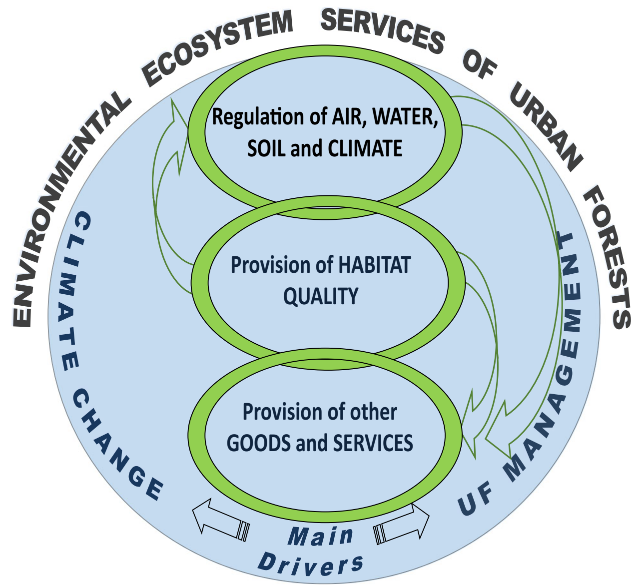

Fig. 1 Type, relationships and main drivers of the environmental ecosystem services (EESs) of urban forests (UF). We classify UF EESs into three categories: (1) regulation of air, water, soil and climate; (2) provisioning of habitat quality; and (3) provisioning of other goods and services, with habitat quality as central to the other two. Two main drivers (UF management and climate change) affect all the three EESs

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indicators and their benchmark definitions should be identified. The EES indicators should be well defined, measurable, reliable and stable parameters that are directly or indirectly related with one or more ecosystem processes and underlying services. The identification and monitoring of indicators help to link decision making in urban planning and management with relevant scientific knowledge (Maes et al. 2016). Such is the case for biodiversity, where accounting for the needs of multiple biological groups is a challenge that can be addressed with the proper indicators (Pinho et al. 2016).

There is an urgent need to understand the complex relationships highlighted above, in particular considering the ongoing climatic and socio-economical changes that affect worldwide urbanized areas in an unprecedented way. Therefore, the aim of this paper is to provide an improved classification of urban forest EESs, discuss their benefits, specificity and relevance in the context of other EESs, and define a set of suitable indicators. This work is an outcome of the activities of the COST Action FP1204 'Green Infrastructure approach: linking environmental with social aspects in studying and managing urban forests' (GreenInUrbs).

The environmental ecosystem services of urban forests

Classification of urban forest EESs

We classified urban forest EESs into three main categories: regulation of air, water, soil and climate; provisioning of habitat quality; and provisioning of other goods and services. As shown by the green arrows in Fig. 1, habitat quality, in the broader sense of its definition, as the quality of the above and belowground space where urban trees live, is central for the provisioning of all the other EESs, while air, water, soil and climate EESs also affect the delivery of other goods and services (Chen et al. 2016; Giannico et al. 2016; Mariani et al. 2016; Pesola et al. 2017). A high habitat quality of the living space of urban trees will positively affect their growth, survival and reproduction, enhancing their potential provision of multiple functions and services. The main benefits and interlinkages are summarized in Fig. 2 by different colours. Fifty-three percent of the 36 indicators suggested herein are relevant to more than one single benefit, which highlights the complex interrelationships among different urban forest EESs. The indicators of wider applicability are tree and stand characteristics (e.g., density and continuity of the plant cover, tree age, architecture, diameter at breast height (DBH), leaf area index (LAI), canopy height, tree height), followed by leaf physical traits (shape, persistence, orientation, wettability, hairiness, roughness, toughness, albedo) and tree species composition (species identity and relative abundance). The relative importance of EESs, i.e. either the impact on the environment and the general perception by the population, differs along a rural–urban transect (Fig. 3). Amelioration of air quality and microclimate, as well as building preservation, are of utmost importance in cities given the density of population and built infrastructures, while CO_2 sequestration, provision of energy, nutrients and resources, and restoration of soil and water are quantitatively more important for forests in rural areas. Habitat quality for biodiversity is extremely important in both urban and rural forests.

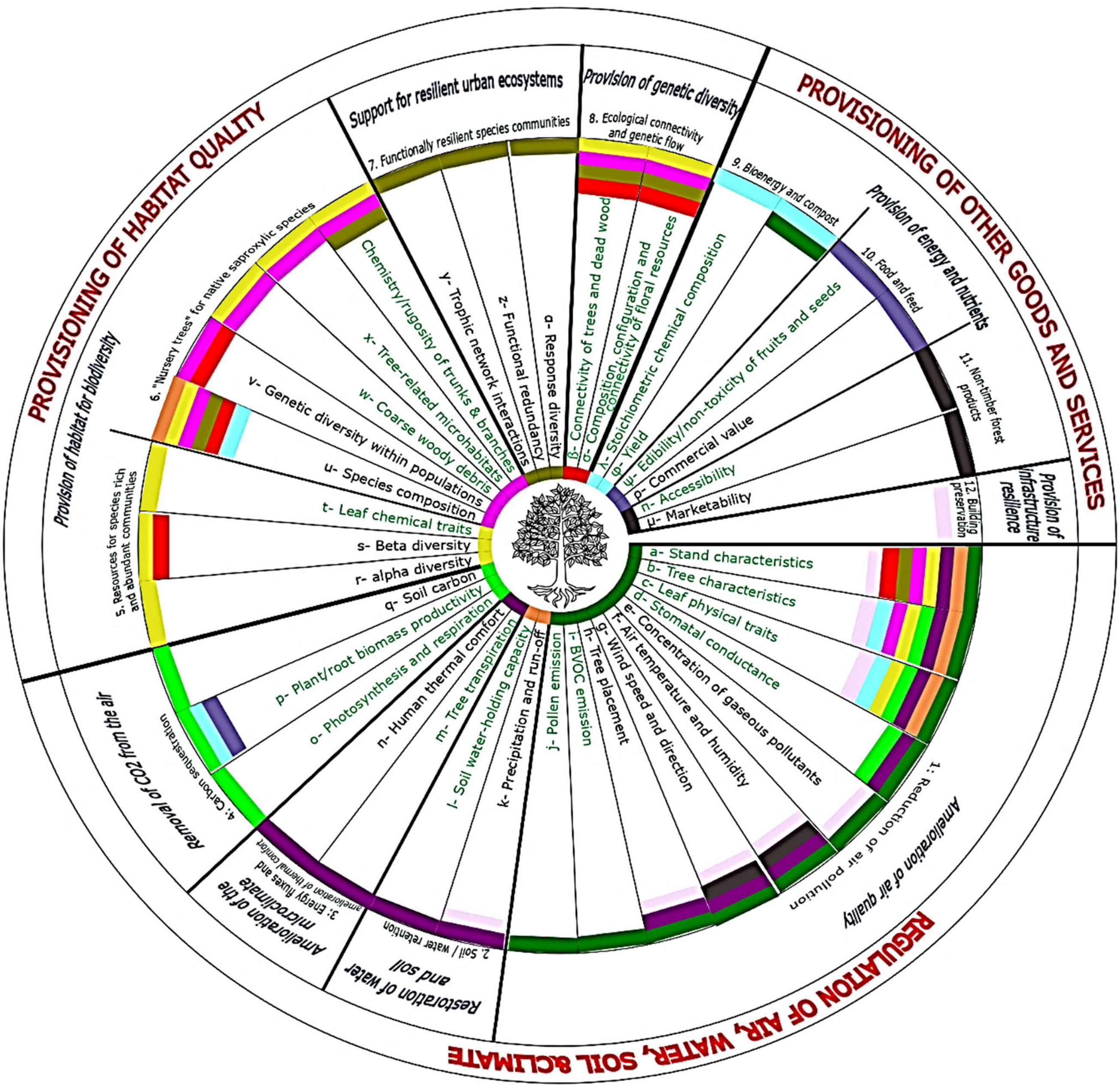
Regulation of air, water, soil and climate

By interacting with the atmosphere, urban forests provide valuable EESs to society. These services can be classified into three major types: (1) amelioration of air quality; (2) restoration of water and soil; (3) amelioration of the microclimate and removal of the greenhouse gas (GHG) CO_2 from the atmosphere (Fig. 2). These EESs provide a number of benefits that are summarized as: (1) reduction of air pollution; (2) soil/water retention; (3) amelioration of thermal comfort; and (4) carbon sequestration (the numbers refer to the indicators in Fig. 2). For each benefit, a variety of indicators are suggested and discussed below. The indicators can be either characteristics of individual trees and tree canopies (written in light green in Fig. 2) or characteristics of the environment where the trees are located (in black in Fig. 2).

Amelioration of air quality

Urban air pollution is a major threat to citizens' health (Pascal et al. 2013). A plethora of primary pollutants, e.g. nitrogen oxides (NO_x), sulphur dioxide (SO_2) and particulate matter (PM), are directly emitted by combustion in industrial processes and road and non-road transport (EEA 2015). Secondary pollutants, e.g. ozone (O_3) and secondary organic aerosols (SOA), are formed by reactions of precursors such as NO_x and VOCs (Lelieveld and Dentener 2000). Among the major air pollutants, PM, nitrogen dioxide (NO_2) and ground-level O_3 are still the most serious pollutants in terms of impacts on the health of European population (European Environment Agency 2018), and should be priorities in urban context.

Air pollution mitigation is a relevant EES provided by urban forests due to air pollution deposition and filtering (Nowak et al. 2006; Escobedo et al. 2011). Urban forests have been argued to phytoremediate the air by removing PM and gases (Nowak 2006; Grote et al. 2016; Sicard et al. 2018). Areas with high urban forest density, in fact, have



lower PM than other sites (Irga et al. 2015). Although most of the estimates suggest quite low mitigation potentials in terms of atmospheric concentrations, such small percentages translate into significant savings in terms of human health (Nowak et al. 2015, 2018).

Urban forests can reduce the concentration of pollutants in the atmosphere by direct deposition on plant surfaces and stomatal uptake of gases inside the leaves (Cieslik et al. 2009; Niinemets et al. 2014). Non-stomatal removal processes include physical deposition on any external surface, and apply to both gas and particle phases, although it is important to consider that gas exchange always occurs in a bidirectional manner (Niinemets et al. 2014). Larger tree crowns have a greater potential of ameliorating air quality by maximizing pollutant deposition (Paoletti et al. 2004), even though edges of shrubs and smaller trees can be planted extremely near the source of pollution (i.e. road traffic) and thus also maximize pollutant filtering (Popek et al. 2013; Mori et al. 2015). Therefore, the characteristics of the tree cover, in particular density and continuity of crowns, size and architecture of individual tree crowns, and leaf area per unit ground surface area (LAI), are recommended as indicators for this EES. Also, leaf surface characteristics, for example, cuticular morphology, leaf

◀**Fig. 2** Classification of the three main environmental ecosystem services (EESs) provided by urban forests (in red in the outer layer), their types and benefits (1–12) (next layer), and indicators (wedges). Interlinks of each indicator to a benefit are marked by the same colour. Indicators of environment or biodiversity are written in black. Indicators of trees or forest stands are written in light green. For a description of each indicator: (a) Main stand characteristics are density, continuity and age (Vilhar and Simončič 2013; Frehner et al. 2005); (b) Main tree characteristics are architecture, DBH, LAI, canopy height, tree height (Tiwary et al. 2016; Nowak et al. 2002; Colding and Barthel 2013); (c) main leaf physical traits are shape, persistence, orientation, wettability, hairiness, roughness, toughness, albedo (Llorens and Domingo 2007); (d) (Li et al. 2007); (e) (Kumar and Imam 2013); (f) Use of vapour pressure deficit recommended (Tiwary and Kumar 2014); (g) (Tiwary and Kumar 2014); (h) Main tree placement characteristics are distance to road, arrangement, orientation (Amorim et al. 2013; Salmond et al. 2013; Vos et al. 2013; Gromke and Blocken 2015); (i) (Calfapietra et al. 2013); (j) (Cariñanos et al. 2016; Ziello et al. 2012); (k) (Gallo et al. 2012; Le Pape et al. 2012; Tiwary and Kumar 2014); (l) (Klein et al. 2014); (m) (Pataki et al. 2011); (n) (Pearlmutter et al. 2007; Shashua-Bar et al. 2011); (o) (Fu et al. 2015); (p) or plant carbon storage (BISYPLAN 2012); (q) (Bae and Ryu 2015); (r) including number of species and alpha diversity (Handley et al. 2015; Ikin et al. 2015; Oishi and Tabata 2015); (s) (Knop 2016); (t) Main leaf chemical traits are chemistry and palatability (Backhaus et al. 2014); (u) Main species composition characteristics are species identity and relative abundance (Karp et al. 2011; Ikin et al. 2015; Oishi and Tabata 2015); (v) (Sadanandan and Rheindt 2015); (w) Main coarse woody debris characteristics are CWD, i.e., density of dead wood and decay stages (Lehvavirta and Rita 2002; Le Roux et al. 2014); (x) Main tree-related microhabitats characteristics are densities of cavities, cracks, bark loss, and dead branches (Lehvavirta and Rita 2002; Terho and Hallaksela 2008; Le Roux et al. 2014); (y) Main topologies of trophic network interactions are diversity, nestedness, and connectance (Baldock et al. 2015; Harrison and Winfree 2015); (z) Main functional redundancy characteristics are number of functionally redundant species within an effect group (Elmqvist et al. 2003; Laliberté et al. 2010); (α) (Elmqvist et al. 2003; Laliberté et al. 2010); (ß) (Turrini and Knop 2015); (σ) (Braaker et al. 2014a; Baldock et al. 2015; Harrison and Winfree 2015); (λ) (AIEL 2008); (φ) (Lambin and Meyfroidt 2011); (ψ) (Bhat et al. 2010); (ρ) (Lambin and Meyfroidt 2011); (π) (McLain et al. 2012); (μ) (Ticktin and Shackleton 2011)

wettability and hairiness, affect PM (Kardel et al. 2012) and O_3 deposition (Li et al. 2018) thus representing a suitable indicator.

Another important leaf trait affecting the EES air quality is the persistence of foliage throughout the year (evergreen species) or only during the growing season (deciduous species). Stagnant atmospheric conditions combined with higher primary PM emissions from residential combustion often lead to high PM events during winter (e.g., European Environment Agency 2018). Therefore, and without considering other eventual effects, evergreen species are recommended for maximizing the PM deposition in winter. In contrast, gaseous pollutants, and in particular O_3, increase during the growing season (Paoletti 2006, 2009); thus, deciduous species are better suited for filtering gaseous air

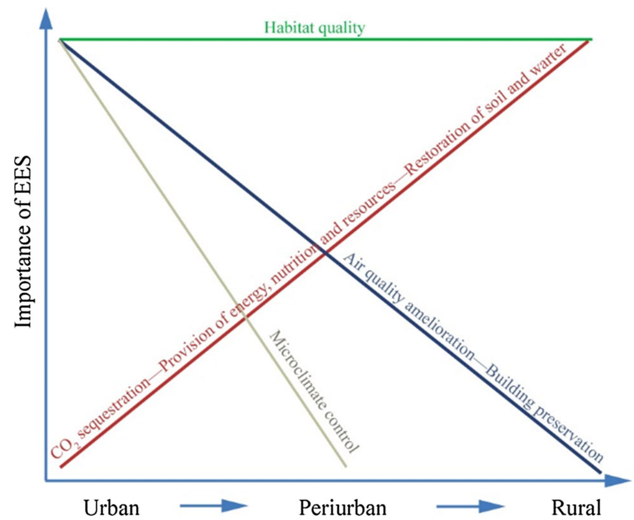


Fig. 3 Schematic representation of the main types of environmental ecosystem services (EESs) and how their relative importance changes in a rural to urban transect. The EESs provided by urban forests are similar to those provided by forests in rural areas, while the relative importance does differ except for habitat quality, which is equally important along the urban–rural gradient

pollutants provided they do not emit BVOCs or emit them at a low rate, as BVOCs favour the O_3 formation (Calfapietra et al. 2013). In addition, deciduous species may have higher stomatal conductance than evergreen species (Larcher 1995) and thus maximise the air pollutant uptake through the stomata. Direct deposition on leaf surfaces may be rather limited under dry conditions, while on wet canopies this process may represent a major sink. Leaf wetness, however, is a difficult parameter to measure and cannot be proposed as an indicator, whereas the vapor pressure deficit of the air (VPD) is a simple parameter combining air temperature and relative humidity (RH) and can be recommended as a proxy of leaf capacity for wet deposition (or release once the leaves dry out and the volatiles are gassed out).

The capacity of trees to remove pollutants may largely differ under stress conditions (Sicard et al. 2018). Species selection for air pollution mitigation should thus consider the ability of a tree species to adapt to local conditions; for example, sclerophyllous species adapt better to warm environments (Larcher 1995). A water-saving approach by actively controlling stomatal conductance is also typical in warm climates (most commonly in evergreen needle-leaved species), whereas a water-spending strategy is more typical in temperate/cold regions (in deciduous broad-leaved species) (Lo Gullo and Salleo 1988; Karabourniotis et al. 2014).

Inside leaves, gaseous pollutants can be scavenged by antioxidant enzymes (Niinemets et al. 2014). Although such scavenging ability is species-specific, extensive individual variability, broad seasonality and numerous compounds involved in scavenging make the assessment of this parameter unsuitable as an indicator of the EES air quality. Non-stomatal removal processes also include chemical deposition resulting from gas-phase reactions between pollutants (mostly O_3 and NO_x) and BVOCs emitted from the ecosystem (e.g., plants or soil) (Fares et al. 2010). The emission of BVOCs is species-specific and is light- and temperature-dependent (Grote et al. 2013; Niinemets et al. 2011). High BVOC emitters are typically broadleaf plants, such as the genus *Eucalyptus*, *Populus* and *Quercus*, with the highest emissions occurring during spring and summer (Benjamin and Winer 1998; Calfapietra et al., 2009). The downside is that BVOCs themselves can promote higher air pollution due to the production of secondary pollutants (Calfapietra et al. 2013). Adopting low BVOC-emitting species in urban forests is thus crucial for the EES air quality (Benjamin and Winer 1998; Ren et al. 2014). In addition to the species-specific BVOC emission factor, however, the amount of emitting leaves affects the total production of BVOCs by trees. Therefore, larger crowns constitute a negative indicator of air quality in BVOC-emitting species.

A trade-off (or environmental disservice) of urban forests is pollen emission. In Europe, 113 million individuals suffer from allergic rhinitis and 68 million from allergic asthma (EFA 2011); both numbers will likely increase due to climate change (Forsberg et al. 2012). Pollen affects human health by triggering such allergic reactions (Bartra et al. 2007; Cariñanos et al. 2016). Pollen deposition on leaf surfaces, however, helps to abate pollen concentrations in the air (Dzierzanowski and Gawronski 2011; Terzaghi et al. 2013) likely by mechanisms similar to those regulating PM deposition. The urban forest pollution-pollen nexus, however, is rather complex. For instance, recent evidence suggests that air pollution can change the protein profile of allergenic pollens and this may increase the symptomatic response in sensitive people (Hinge et al. 2017). The review by Grote et al. (2016), produced within COST Action FP1204, points out to crucial knowledge gaps associated with the emission of pollen and VOCs, stressing the need of holistic investigations of these inter-related processes.

Research has shown that the presence of roadside trees in street-canyons can disrupt the upwards transport of air pollutant emissions, increasing their storage within the canyon and reducing the penetration of clean air from aloft. As a result, higher pollutant concentrations may be observed at pedestrian level within vegetated canyons (Amorim et al. 2013; Salmond et al. 2013; Vos et al. 2013; Gromke and Blocken 2015), which can also constitute a disservice of the urban forests. A recent study ranked 95 urban plant species based on the ability to maximize air quality, by removing O_3, NO_x and PM, and minimize the associated disservices (Sicard et al. 2018). However, further investigation of the inter-relations between plant characteristics, microclimate, street configuration and pollutant emissions is still warranted.

Restoration of soil and water

Urbanization can negatively impact stream and drinking water quality by increasing the loads of nutrients, metals and organic pollutants to surface and ground water (Gallo et al. 2012; Le Pape et al. 2012). Fresh water provisioning and water purification are of topical interest in urbanized areas (Vilhar and Simončič 2013). Urbanization, causing soil sealing and a reduction of vegetated surfaces, increases flooding frequency and duration due to enhanced partitioning of rainfall into runoff (Gallo et al. 2012). Urban forests contribute to stormwater harvesting through the interception of precipitation (Llorens and Domingo 2007), thereby playing a major role in sustaining urban water balance and reducing stormwater runoff (Pataki et al. 2011). Owing to their levels of influence, canopy and leaf characteristics (including canopy structure and density; shape and orientation of the leaves) are suitable indicators for this benefit.

The influence of forest cover on soil water retention capacity depends on the medium- and long-term improvement of soil conditions (Frehner et al. 2005). Urban forests can increase this capacity especially in the case of slowly drained to very wet soils, whereas improvement in well drained soils is much less (Lee 1980). Infiltration due to urban forests depends mainly on the density and continuity of tree canopies and ground vegetation cover, and is of little importance for individual street-trees (Vilhar and Simončič 2013). Soil water retention capacity is closely linked to the root system performance of trees and includes the water-holding capacity of organic and mineral soil, as well as of the litter layer (Klein et al. 2014). In cities, however, litter is often removed, e.g., to reduce the spread of plant diseases. Therefore, the indicators for this benefit are: leaf traits (shape and orientation; evergreen vs. deciduous), species composition, tree and plant cover characteristics (root system, density, continuity, and placement), precipitation and runoff, and soil water-holding capacity.

Amelioration of the microclimate

A well-known effect of urban development is the warmer temperature in cities compared to the surrounding rural areas, known as the urban heat island (UHI) (Oke 1982). The magnitude of warming that characterizes a given city UHI, with impacts on e.g. human comfort and energy demand, is highly variable over time and space. On average, urban temperatures are about 1–3 °C warmer than the surrounding rural environments, but under warm summer conditions they can be > 10 °C warmer (Mills et al. 2010).

Vegetation contributes to the reduction of heat stress and wind speed even in the surrounding areas, with a cooling effect that can extend horizontally for up to 1000 m into the built-up area (Bowler et al. 2010; Klemm et al. 2015). The largest cooling impact of trees is observed on clear and hot summer days (Bowler et al. 2010). In winter, air temperature is slightly reduced by evergreen trees, with no clear impact on thermal comfort (Cohen et al. 2012). Cooling of air temperature due to urban forests is higher during the day than during the night in summer and the opposite is true in winter (at least in the dry-warm conditions of the study by Cohen et al. (2012)), whereas cooling of surface temperature is greater during the day in summer and nearly similar during the night throughout the year. Surface temperature within a greenspace may be 15–20 °C lower than that of the surrounding urban area, giving rise to urban air temperatures cooler by 0.3–9.5 °C (Saaroni et al. 2018). The review paper by Saaroni et al. (2018), also supported by COST Action FP1204, showed no apparent correlation of this park-induced cooling within the climatic region, while a tendency for larger green sites (> 2 ha) to induce a stronger cooling effect (> 4 °C) was observed. However, the extent and magnitude of the impacts on human comfort and building energy demand are far from being effectively quantified especially due to their site-specific nature.

Urban forests ameliorate the microclimate essentially via evapotranspiration and shading (Dimoudi and Nikolopoulou 2013). Shading by trees reduces both short-wave (direct, diffuse and reflected) and long-wave radiation emitted from surfaces (Saaroni et al. 2018). When leaves are heated by solar radiation, the cooling process takes place by evapotranspiration. Evaporation occurs from wet plant surfaces (e.g., after rainfall), while transpiration takes place through stomata. Also, shading by trees produces a net cooling effect in the below-canopy region (Akbari et al. 2001). Plant functional traits play a major role in this sense, since crown architecture and shape, leaf clumping and orientation, and the amount of foliage (often expressed as LAI) determine the interception of solar radiation (Cescatti and Niinemets 2004; Disney 2015). Evergreen trees impact the microclimate throughout the year, whereas the effect of deciduous broadleaves is mostly restricted to the growing season. This may represent a benefit in temperate regions where the penetration of sunrays through the canopy is beneficial during the winter. In fact, extensive areas of shadow are usually not desired in outdoor urban environments at high latitude cities, which creates an additional challenge for architects and urban planners (Lindberg et al. 2014).

The most relevant environmental indicators in the amelioration of the urban microclimate are air temperature and RH, as they contribute to the thermal comfort of people (Susca et al. 2011), and wind speed and direction. The Universal Thermal Climate Index (UTCI) and the Physiologically Equivalent Temperature (PET) are useful biometeorological indices (Matzarakis et al. 1999; Bröde et al. 2012), but were considered too complex for being included in Fig. 2 as indicators.

Sequestration of carbon dioxide (CO_2)

Another important benefit of urban forests is their capacity to remove CO_2 from the atmosphere. This might occur directly through photosynthesis and indirectly through energy savings triggered by microclimatic amelioration (Rosenfeld et al. 1998). In cities, the anthropogenic emission of CO_2 is far above the capacity of trees to sequester and store it (Briber et al. 2013). For instance, urban forests in the USA were estimated to store 712 Mt C, corresponding to a gross sequestration of 23 Mt C annually (Nowak and Crane 2002), i.e., 16% of the C stored in natural forests (Woodbury et al. 2007). Conversely, urban forests emit CO_2 to the atmosphere via respiratory processes. However, as the removal usually exceeds the emission, forests act as a net sink of atmospheric C (Thornton et al. 2002). Photosynthesis takes place under favourable conditions for plants, while stress caused by pruning or severe drought may promote respiratory processes, which in some cases induce trees to act as a net C source. In general, CO_2 uptake capacity per leaf area unit is species-specific, being higher in deciduous broadleaves and lower in evergreen species (Wright et al. 2004). Total uptake depends on both the photosynthetic capacity and total amount of foliage, as well as on the radiation available within tree canopies (Niinemets 2015). All these traits represent suitable indicators for this EES. Sequestration capacity varies also with the type of green space, being higher in roadside vegetation that is closer to the traffic source (Wu et al. 2010). The C removed from the atmosphere is then stored in above- and below-ground biomass and in soil organic matter, which are other relevant indicators of the long-term capacity of urban forests to store atmospheric C. Increasing soil organic matter, however, is negligible for individual street-trees in the short term (Vidal-Beaudet et al. 2015).

Provisioning of habitat quality

Provision of habitat for biodiversity

Forests are among the most species-rich terrestrial ecosystems (Croci et al. 2008; Jim and Liu 2001; Kühn et al. 2004). This extended multi-dimensional green space represents a resource for urban biodiversity, including wood-dwelling organisms (Grimm et al. 2008). If well designed and managed, urban forests are expected to favour the overall green connectivity across fragmented and densely built areas enhancing the permeability of the urban matrix for forest- and wood-dwelling species and for biodiversity in general (Vergnes et al. 2012; LaPoint et al. 2015).

In the last decades, biodiversity has emerged as one of the most important environmental concerns (Hooper et al. 2005). There is a growing consensus that biodiversity determines ecosystem functions and underlying services (Isbell et al. 2011; Cardinale et al. 2012) while contributing to the overall resilience of ecosystems. Thus, urban biodiversity constitutes not only a matter of scientific interest but also an object of increasing public concern, and could be particularly sensible to climate change (Puppim de Oliveira et al. 2014). An approach based on functional diversity can provide the link between biodiversity and EESs by taking into account the common functions that species perform in ecosystems or the way species respond to an environmental constraint (Pinho et al. 2016).

The EESs that urban forests provide to habitat quality can be classified into three major types: (i) provision of habitat for biodiversity; (ii) support for resilient urban ecosystems, and (iii) provision of genetic diversity (Fig. 2). These EESs provide a number of benefits that are summarized as: (5) provision of resources for species-rich and abundant communities; (6) provision of "nursery" trees for native saproxylic species; (7) support of functionally resilient species communities, and 8) provision of ecological connectivity and genetic flow (the numbers refer to Fig. 2). The benefit indicators can be either features of individual trees and forest patches or characteristics of the environment where trees and forests are located, or taxonomic and functional metrics reflecting different components of biodiversity (Fig. 2).

Species richness and population size are important components of biodiversity, which reflect the amount and complexity of available niches (e.g., type of leaves and branch architecture) and ecosystem productivity. Complex vegetation structures are likely to increase the availability of foraging and breeding resources for a multitude of organisms, enhancing the number and diversity of species within (alpha diversity) and across (beta diversity) forest stands (McElhinny et al. 2006; Tassicker et al. 2006). Spatial and temporal heterogeneity of urban forests can be obtained by maximizing species composition and demographic structures, thus affecting the type of leaves, branch architecture, tree-related microhabitats and coarse woody debris.

Urban forests can maintain and even increase overall richness thanks to the presence of exotic plant/animal species (Pyšek 1998; Celesti-Grapow et al. 2006; Nobis et al. 2009), whose number is higher in urban environments compared to rural areas (Czech et al. 2000; McKinney 2002, 2006; Sattler et al. 2011, but see Cadotte et al. 2017). The presence of exotic animal species, especially arthropods, in the urban environment might be underestimated due to the lack of information and reference species lists. However, it is well documented that the introduction of alien species can cause negative effects on forest ecosystems, by modifying habitats and potentially disrupt some natural interactions among species, which can alter key functions in such ecosystems (McAfee et al. 2006). On the other hand it can also provide a resource for organisms from different trophic levels, thus taking over the functional role of endemic species that got locally extinct (Finerty et al. 2016; Gray and van Heezik 2016;).

Besides favouring species that use urban forests as a source of food and shelter, urban forests provide habitat for obligate canopy- and wood-dwelling organisms, also defined as saproxylic species. These species, such as fungi, mosses, lichens (Speight 1989), are dependent upon decayed wood during part of their life cycle. "Nursery" trees for saproxylic species are related to the concept of "habitat trees" defined as dead or living, very large and old microhabitat-bearing trees (Bütler et al. 2013). The presence of woody debris and the number of tree-related habitats influence the species composition and genetic diversity of these organisms, and improve the functional connectivity between forest patches and old trees (Vandekerkhove et al. 2013). Single habitat trees or parts of urban forest stands need to be well connected with floral feeding resources in sunny open green areas distributed within a few hundred meters.

Support for resilient urban ecosystems Urban forests are usually dominated by open environments and are often designed to accomplish recreational, aesthetic and regulation functions (Nowak 2006). The characteristics of urban habitats and the similarity in urban greens suggest that the species living in cities converge toward a subset of rather homogenous species sharing a common suite of traits (Haase et al. 2014; Knapp et al. 2008). This effect is recognized as 'biotic homogenization' (McKinney 2006), and have deep implications for conservation (but see Colding 2007; Lepczyk et al. 2017 when urban green space heterogeneity is included in the models). Species living in urban areas tend to be more generalist than species living in rural areas or into the wild (Clavel et al. 2011). The increase of ecologically similar species can lead an increase of redundancy of the assemblages (different species with similar traits, ecological roles and functions) (Alberti and Marzluff 2004). Since many ecosystem functions ultimately rely on interactions between primary producers (plants) and other trophic levels (e.g. pollinators, soil decomposers, and herbivores), the redundancy and resilience framework (e.g., Elmqvist et al. 2003) should be extended to multi-trophic systems (Lavorel et al. 2013) and include metrics of biotic interactions (for measuring functional redundancy and response diversity) as indicators of functionally resilient urban ecosystems (e.g., Frey et al. 2018; Tresch et al. 2019).

Provision of genetic diversity

Heterogeneous and structurally complex urban forests might not be enough to maintain viable populations and promote functionally resilient species communities if trees and forest patches remain isolated or too small (Turrini and Knop 2015). Communities in small and isolated patches tend to contain fewer species and to become increasingly homogenous in taxonomic and functional composition (McKinney 2006; Knapp et al. 2008; Turrini and Knop 2015), therefore reducing the response diversity of species communities (Elmqvist et al. 2003). The spatial configuration of urban forests and, in particular, habitat connectivity, defined as the connectedness of habitat patches for a given species (Fischer and Lindenmayer 2007), play important roles in enhancing taxonomic and functional stability and resilience. They favour trophic interactions, successful reproduction, dispersal and genetic exchange, and provide refuge from predators (Taylor et al. 2006), while enhancing meta-population and meta-community dynamics (Leibold et al. 2004; Vergnes et al. 2012; LaPoint et al. 2015). This is particularly important in view of new climate change stressors. The spatial scale at which ecological connectivity varies depends on species composition (LaPoint et al. 2015). For sessile and low dispersal organisms, such as plants, soil fauna and ground dwellers, the spatial configuration of urban forests acts within tens and hundreds of meters, while it expands up to a few kilometers for flying organisms such as bees, birds and bats (Sattler et al. 2010; Braaker et al. 2014a, b). Indicators of well-connected populations of forest-dwelling species and communities need to be further investigated. Movement patterns, the genetic diversity of model species and functional resilience of communities, including topologies of multi-trophic network interactions, are promising tools (La Point et al. 2015).

Provisioning of other goods and services

Provision of energy and nutrients

Another major category of EESs provided by urban forests is the provisioning of other goods and services (Gómez-Baggethun and Barton 2013; Hansen and Pauleit 2014). These EESs can be classified into the following two types: (i) provision of energy and nutrients; and (ii) provision of grey infrastructure resilience (Fig. 2). These EES types provide a number of benefits: (9) bioenergy and compost; (10) food and feed; (11) non-timber forest products (NTFPs); and (12) building preservation (the numbers refer to Fig. 2). Individual indicators characterizing the provisioning of different goods and services from trees or the environment are suggested and discussed below.

Two possibilities for resource recovery are usually considered: nutrient recovery via composting the foliage and/or energy recovery via biomass productivity (in terms of calorific value). Woody vegetation is an important renewable resource for bioenergy, alleviating the growing demand for cropped biofuels (de Richter et al. 2009). Large-scale commercial plantations of trees such as poplar and willow, mainly in urban parks and peri-urban woodlands, may fulfil the current drive for energy sustainability from renewable biomass (Djomo et al. 2015; Seidel et al. 2015). For example, sustainably grown tree biomass is projected to provide up to 10% of the UK energy needs by 2050 and to significantly contribute to the reduction of GHG emissions (DECC 2012). The biomass potential is evaluated by multiplying the primary production by the residue-to-product ratio, which is a tree-specific indicator (BISYPLAN 2012). The recovery of bioenergy, mainly as heat from the combustion of managed urban forests, is obtained from its heating value on a dry basis. The heating value (expressed in wet tons) is related to its typical stoichiometric chemical composition (AIEL 2008). Therefore, suitable indicators of bioenergy provision potential include tree-specific structural characteristics, biomass yield and stoichiometric chemical composition. Important limitations are the many energy crops, such as willow and poplar, which are strong emitters of allergens and BVOCs (Olofsson et al. 2005; Owen et al. 2013), and biomass combustion that emits considerable amounts of PM (Lim et al. 2015), both of which could adversely affect air quality.

Urban forests have the potential to offer a source of nutrition to urban populations (e.g., food and feed). Community scale and individual initiatives are gaining popularity for securing sustainable food production while ensuring minimal environmental footprints (Lambin and Meyfroidt 2011). Although there is a huge potential for urban forest EESs to provide a sustainable supply of nutrition from trees (e.g. fruits, seeds, roots), there are limitations posed by their toxicity and edibility for human consumption driven by pollution of the urban habitats (Bhat et al. 2010). Therefore, toxicity and the commercial value of products were identified as the key indicators for this benefit. In addition, productivity is considered as a measure of the volume of supply that can be acquired to meet the demand. These indicators are applicable towards ensuring cost-competitiveness and widespread sustainability incentives in both the developed and developing worlds.

Most resource management decisions are strongly influenced by urban forest EESs entering markets. Values associated with NTFPs (e.g. litter, flowers, leaves, bark, cones, galls, resins, spring buds, fungi, honey) account for approximately 25–96% of the total economic value of forests (Palahi et al. 2008). The cost-competitiveness of these products, however, largely depends on their accessibility (McLain et al. 2012) and marketability (Ticktin and Shackleton 2011), which have been identified as key indicators for assessing this benefit, together with the climatic conditions that affect NTFP growth.

Provision of grey infrastructure resilience

Incorporating urban forests into the urban built space is gaining popularity as a cost-effective and long-term measure for mitigating climate change impacts associated with proliferating grey infrastructure (CABE 2010; Hamdouch and Depret 2010; Wang et al. 2018). The role of urban forests in developing resilience and environmental stewardship in cities (Colding and Barthel 2013) and in the preservation of buildings (including bridges, car parks and historical monuments) is an emerging EES. This is mainly attributed to the altered micro-meteorological profile and chemical withering of building materials caused by air pollution and the changing climate (Kumar and Imam 2013; Tiwary and Kumar 2014). As this is influenced by an interplay between plant morphological, biophysical and chemical traits, the suitable indicators of this EES benefit include tree canopy architecture and position relative to building infrastructures, leaf physical traits, micro-meteorology and the concentration of pollutants.

Conclusions

Ecosystem services play a crucial role in the optimization of life quality in cities. Urban forests can reduce air pollution and greenhouse gas emissions, sequester carbon, regulate air temperature, mitigate stormwater runoff, reduce noise, as well as provide recreational, social, psychological and aesthetic benefits. In this study, we provide 36 indicators that can be used for quantifying urban forest EESs, predicting climate change effects on urban forests, and developing scientifically sound strategies for optimizing the management of urban forests and maximizing their EESs. These indicators may also be combined to develop summary indices, such as the "pollution flux potential" index (Tiwary et al. 2016). Around two-thirds of them are indicators of trees and forest stands and can be obtained by measurements or literature data, while one-third are indicators of environmental or biodiversity characteristics and can only be measured in situ. All these indicators allow efficient quantitative assessments of urban forest EESs on large areas and across cities, even though they still need to be adequately and rigorously tested, especially when applied across taxa, processes and services. Such an improved understanding is needed to increase the willingness of public entities (Vuletić et al. 2010) and private companies (Glück et al. 2010) to acknowledge urban forest EESs.

Cities are constantly evolving (Stott et al. 2015). Differences in the urbanization process depend on the historical background, urban design and available green spaces in and around cities. Therefore, guidelines on the optimization of EESs and the related choice of species and planting architecture are case-specific and must rely on EES indicators. Because different EESs are strongly interconnected and sometimes show complex and contrasting interactions, the choice and design of urban forests require a local but science-based approach. A "species selector" should be developed in each continent as a tool to help policy-makers to define suitable urban forest management, including proper tree species selection, by simultaneously optimizing different EESs. An emerging approach in this regard is to consider urban forests as Nature-Based Solutions in the urban environment and include them in the city management and planning. The integration of Nature-Based Solutions is recognized as a way to achieve several environmental, social, cultural, economic, policy and planning aims (Hansen and Pauleit 2014; Madureira and Andresen 2014), and as a tool to maximize city resilience to climate change while minimizing the associated disservices (e.g., maintenance costs, infrastructure damage/degradation, allergic reactions). There is still little information available on the EESs provided by urban forests and to what extent urban forests play a key role towards the optimization of EESs. It is important to recognize that urban forest EESs confer substantial economic value to human societies and activities in urban areas (Haase et al. 2014). Accurate calculations of urban forest EES capacity, based on the recommended indicators, will provide a basis for the economic evaluations of changes and enable stakeholders to estimate the trade-offs between EESs and other services, such as agricultural food production (MEA 2005) or urbanization (Dobbs et al. 2011).

Acknowledgements This work was carried out within COST Action FP1204 GreenInUrbs. We thank the participants of our conferences for sharing their expert knowledge in defining the urban forest EES indicators and the effects of climate change and management. The publication was financially supported by the Ministry of Education and Science of the Russian Federation (the Agreement No. 02.A03.21.0008). We acknowledge ARCHES Conseils for drafting the Fig. 2. Publication was carried out within the PRIN project EUFORICC.

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