URBAN TREES, AIR QUALITY, AND ASTHMA: AN INTERDISCIPLINARY REVIEW

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Abstract

A "call to action" has been issued for scholars in landscape and urban planning, natural science, and public health to conduct interdisciplinary research on the human health effects of spending time in or near greenspaces. This is timely in light of contemporary interest in municipal tree planting and urban greening, defined as organized or semi-organized efforts to introduce, conserve, or maintain outdoor vegetation in urban areas. In response to injunctions from scholars and urban greening trends, this article provides an interdisciplinary review on urban trees, air quality, and asthma. We assess the scientific literature by reviewing refereed review papers and empirical studies on the biophysical processes through which urban trees affect air quality, as well as associated models that extend estimates to asthma outcomes. We then review empirical evidence of observed links between urban trees and asthma, followed by a discussion on implications for urban landscape planning and design. This review finds no scientific consensus that urban trees reduce asthma by improving air quality. In some circumstances, urban trees can degrade air quality and increase asthma. Causal pathways between urban trees, air quality, and asthma are very complex, and there are substantial differences in how natural science and epidemiology approach this issue. This may lead to ambiguity in scholarship, municipal decision-making, and landscape planning. Future research on this topic, as well as on urban ecosystem services and urban greening, should embrace epistemological and etiological pluralism and be conducted through interdisciplinary teamwork.

1.0 Introduction

In a 2014 volume of this journal, Sullivan et al. issued a "call to action" for scholars in landscape and urban planning, natural science, and public health to conduct interdisciplinary research on the human health effects of spending time in or near greenspaces. This was inspired by growing recognition of the health benefits of contact with nature (Frumkin et al., 2017; Hartig, Mitchell, de Vries, & Frumkin, 2014), as well as a policy issued by the American Public Health Association (APHA) entitled, "Improving Health and Wellness through Access to Nature" (Chawla & Litt, 2013). Our goal is to address this call to action by providing an interdisciplinary assessment of scientific literature regarding links between urban trees, air quality, and asthma; and to offer associated recommendations for future research and landscape planning practice.

The call for interdisciplinary research on this topic is also timely in light of contemporary interest in urban greening, defined as organized or semi-organized efforts to introduce, conserve, or maintain outdoor vegetation in urban areas (Eisenman, 2016; Feng & Tan, 2017). This includes a range of policies, incentives, and initiatives to vegetate the urban landscape (Beatley, 2016; Tan & Jim, 2017). In many cases, greening involves substantial tree planting. The number of street trees in Paris has increased since the late 1990s by more than 12% to over 100,000 today (Laurian, 2012). London has set a target to increase tree cover from 20% to 25% by 2025, equal to roughly 2 million additional trees (Ween, 2012). And in the United States, cities have established ambitious canopy cover goals and major tree planting programs, including municipal initiatives to plant a million trees (Locke, Romolini, Galvin, O'Neil-Dunne, & Strauss, 2017; Young, 2011).

Historically, trees were planted in the public realm of Western cities for aesthetics, civic improvement, expressions of power, and national identity (Campanella, 2003; Lawrence, 2006). But the primary rationale for urban trees has now shifted to provision of ecosystem services (Silvera Seamans, 2013; Young, 2013). Air pollution abatement is routinely portrayed as a key ecosystem service provided by urban vegetation (e.g., Chen, 2017; Grant, 2012; McDonald, 2015), and it is the most cited economic benefit of urban trees (Song, Tan, Edwards, & Richards, 2018).

Human health has been described as the “central aspect” of ecosystems services (see Figure 1 from the Millennium Ecosystem Assessment, 2005, p. 14); and the biennial conference A Community on Ecosystem Services (ACES, 2014) pledged to “explicitly and formally link ecosystem services with human health and well-being.” Yet, public health scholarship is still not integrated in ecosystem services literature, and this is particularly noteworthy in *urban* ecosystem services (UES) discourse (Eisenman, 2014, 2016). This disciplinary gap is also reflected in urban forestry science (e.g., Krajter Ostoić & Konijnendijk van den Bosch, 2015). So, while ecosystem services has become a prominent construct in certain scholarly domains, it is still an open frontier of research (Gómez-Baggethun & Barton, 2013); and its theorization and application to urban greening practice is at a very early stage (Chen, 2017).

Figure 1: Relationship between human health and ecosystem services (MEA 2005)

In response to these gaps, urban greening trends, and injunctions from scholars, this paper provides an interdisciplinary perspective on a commonly referenced benefit of urban trees: namely, improved air quality with a particular focus on asthma – a chronic respiratory disease linked to poor air quality that disproportionately affects urban populations (American Lung Association, 2013). Approximately 300 million people suffer from asthma worldwide, and this condition contributes to roughly 1 in 250 deaths (Masoli, Fabian, Holt, Beasley, & Global Initiative for Asthma, 2004). Asthma prevalence has been steadily increasing over the past decades in children and adults; and disease morbidity imposes a large economic burden (Asher et al., 2006; de Nijs, Venekamp, & Bel, 2013), especially among low-income, urban minorities. In the U.S., roughly 8% of adults and 10% of children are diagnosed with asthma (Akinbami et al., 2012); and in 2013, the economic burden of asthma including costs incurred by absenteeism and mortality was estimated at $81.9 billion (Nurmagambetov, Kuwahara, & Garbe, 2018).

Asthma incidence and prognosis is affected by a complex interplay of environmental (von Mutius, 2009), genetic, and psychosocial factors including stress (Lietzén et al., 2011). Air pollutants such as particulate matter (PM), ozone (O_3), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), and polycyclic aromatic hydrocarbons (PAHs) have been associated with asthma incidence, exacerbations, or decreased lung function (Delfino et al., 2014; Karimi, Peters, Bidad, & Strickland, 2015; Mar & Koenig, 2009; Patel et al., 2010; Qian et al., 2007; Sarnat et al., 2012). But air pollutant levels have not been uniformly linked to asthma burden, which can increase risk through synergistic interactions with allergenic pollen (Toh, Jariwala, Rosenstreich, & Zou, 2012; Toh et al., 2014). Allergic sensitization (i.e., documented reactivity to a given allergen based on laboratory and/or allergy skin test results) is also a risk factor for the subsequent development of symptomatic allergic disease and asthma (Porsbjerg, von Linstow, Ulrik, Nepper-Christensen, & Backer, 2006). Among allergic individuals, increased asthma incidence or severity have been associated with outdoor exposures such as pollen (Lovasi et al., 2013), and indoor exposures such as cockroaches, mice, pet dander, and dust mites, as well as mold spores, which can be indoor or outdoor exposures (Gaffin, Kanchongkittiphon, & Phipatanakul, 2014; Gent et al., 2012). These exposures trigger allergies and asthma symptoms due to the body's production of histamine. Plant-derived pollen allergy is, in turn, an important contributor to allergy and asthma symptoms (DellaValle, Triche, Leaderer, & Bell, 2012), particularly in urban populations where air pollutant exposure may amplify the health effects of pollen (Löhmus & Balbus, 2015).

Today, air quality improvement by urban trees is routinely cited in popular media (e.g., BBC, 2017; Hamblin, 2014; Wong, 2017), and promoted as a strategy to reduce asthma (City of New York, 2014; Nowak, Hirabayashi, Bodine, & Greenfield, 2014; STF, 2015; TNC, 2016; Nowak, Hirabayashi, Doyle, McGovern, & Pasher, 2018). We assess the scientific literature underlying these claims by first reviewing the biophysical interactions between urban trees and air quality, and associated studies extending findings to asthma reduction. This is followed by a review of public health scholarship addressing links between urban trees and asthma. When available we drew upon refereed literature reviews identified through Web of Science and PubMed databases, and the expertise of our team. The closing discussion addresses the implications of our review, focusing on interdisciplinary assessment, and recommendations for future research and urban landscape planning practice.

2.0 Biophysical Processes Linking Urban Trees, Air Quality & Asthma

There are four biophysical processes whereby urban trees may directly affect human health via outdoor air quality: 1) deposition of gaseous and PM pollution onto tree surfaces and uptake via leaf stomata (microscopic pores); 2) modification of air circulation – also known as dispersion – by tree surfaces; 3) formation of O_3 through emission of biogenic volatile organic compounds (BVOCs); 4) pollen production including synergistic interactions with anthropogenic air pollution. These processes are depicted in Figure 2. Other indirect processes can include temperature reduction, and resultant air quality improvements such as reduced O_3 production and reduced pollutant emissions from power generation for building temperature control systems. In the ensuing discussion, we focus on *direct* interactions between trees and air quality.

Figure 2: Links between urban trees and air quality.

Of these direct interactions, deposition and dispersion are the processes whereby urban trees may reduce anthropogenic air pollution. This can be assessed through modeling studies; or by empirical studies using field measurements in areas of varying tree canopy density and distance from pollution source, or measurements under laboratory conditions (e.g., wind tunnels). But unlike laboratory and field research, some model-based studies extend pollution reduction estimates to projections of human health effects, including reductions in asthma and respiratory disease (e.g., Nowak et al. 2013, 2018; Rao et al. 2014; TNC 2016).

From a mechanistic perspective, deposition and dispersion are discrete phenomena: the former is governed by gravity and the latter by air circulation. However, these processes are deeply interconnected, and it is a rare situation in the real world where one functions independent of the other. A more practical distinction – especially for municipal decision makers and landscape planners – may be the effect of urban trees upon air quality at city versus site scales. By extension, most studies address one or the other of these scales, and we strive to illuminate these differences. As pollen and BVOCs/O_3 can reduce air quality through emissions from urban trees, they are handled in separate subsections.

We ground our assessment by briefly summarizing refereed reviews, identifying the magnitude of reported pollution levels attributable to urban trees, and highlighting the research gaps and implications for urban planning identified in these reviews. We then review modeling that extends air pollution reduction by urban trees to actual asthma outcomes, and we include a subsection on empirical studies assessing links between urban trees and air pollution levels, as model verification is essential before an air quality model is applied in public policy (Gurjar, Molina, & Ojha, 2010).

2.1 Pollution Removal by Urban Trees: Reviews

Review papers addressing the pollution filtration capacity of urban trees focus on deposition and dispersion processes. Both PM and gaseous pollutants deposit on all environmental surfaces including buildings, vehicles, pavement, the ground, leaves, and other plant surfaces. If surfaces are wet or when humidity is high, soluble PM may dissolve in water and enter leaves through the cuticle or continuous water films that extend from leaf surfaces onto cells lining the sub-stomatal cavity (Burkhardt, 2010). In addition, gas molecules may enter plants by passive diffusion through stomata (Seinfeld & Pandis, 2016). Urban tree canopies can also concentrate or dilute air pollution by modifying air flow (Abhijith et al., 2017; Baldauf, 2017). In scholarly literature, dispersion refers to the movement of pollution away from a source, which under unobstructed conditions results in dilution. In urban conditions, trees and other objects can also concentrate pollution by reducing air circulation. While the term dispersion can thus be confusing, we use this term in our discussion.

Focusing on PM, an early review summarizing the potential magnitude of air pollution reduction via urban vegetation canopies found that average published deposition values (v_d in units of cm s^{-1}) corresponded to an estimated 1% reduction of PM_{10} across urban areas (Litschke & Kuttler, 2008). At the scale of a busy arterial road, this review noted plants can modify air flow (i.e., dispersion), thus increasing PM concentrations near emission sources such as roads. The authors suggested that large vegetation areas exceeding 10,000 m^2 would be needed to compensate for local emissions of PM_{10} based on the average v_d. This review also noted that contemporary interest in the pollution mitigation potential of urban trees reflects a discursive shift: through much of the 20th century, scientists addressed the damage caused to plants by polluted air, not the filtration effect of trees on pollution.

Assessing scientific literature on dry deposition of PM to vegetation, Petroff et al. (2008) found that differences among model predictions, especially for fine particles, differed by more than an order of magnitude. This review found large discrepancies between modeling results and empirical measurements. The authors emphasize that regardless of the model configuration, validation is crucial for identifying problematic assumptions and opportunities to improve accuracy. To meet this goal, leaf area index, canopy aerodynamics (e.g., roughness length, displacement height), leaf and canopy geometry, and leaf morphology must be quantified and explicitly considered.

Leung et al. (2011) reviewed literature on the effects of urban vegetation on urban air quality, and offered a descriptive summary of several numerical models. This review concluded that because some plant species produce pollen grains and fungal spores that are health hazards for allergic individuals, and some plants can increase O_3 by emitting BVOCs, careful planning and cost–benefit analysis should precede large-scale planting of urban trees. Janhäll (2015) reviewed literature on the effect of urban vegetation upon PM via deposition and dispersion, and offered additional caveats: research addressing deposition and dispersion must incorporate findings from one another before action is taken in urban planning; great caution should be exercised when transferring deposition models between different applications; and there is a need for site-specific measurements including detailed descriptions of measured parameters.

The site-specific air quality effect of vegetation along open roads and street canyons flanked by buildings has emerged as a topic of special interest (Baldauf, 2017; Shaneyfelt, Anderson, Kumar, & Hunt, 2017; Tong, Whitlow, MacRae, Landers, & Harada, 2015). A comprehensive review by Abhijith et al. (2017) concluded that in street canyons with an aspect ratio ≥ 0.5 – in other words, where the height of the solid or nearly solid façade of adjacent buildings is equal to or greater than half the street corridor width – tall growing vegetation such as trees generally reduces air quality by decreasing air pollution dispersion in the vertical axis. But improvement or deterioration in street canyon air quality via trees also depends on wind direction and speed, and the size, species, and placement of trees.

In open road conditions, naturally occurring or planted greenbelts have the potential to filter automobile emissions between highways and adjacent areas, but findings are mixed and depend on several factors. In these settings, wide, tall, and dense (e.g., conifers) vegetative buffers may lead to downwind pollutant reductions between 15%- 60% (ibid). But if density is too high, it can act like a solid barrier, forcing air up, over, and behind the buffer (Baldauf 2017). Moreover, gaps and sparse vegetation (e.g., deciduous trees during leaf-off season) can lead to no improvement or even reducing air quality by up to 15% depending on pollutant (Ghasemian, Amini, & Princevac, 2017; Hagler et al., 2012). This is because sparse foliage may allow pollutants to pass through or around the vegetation barrier, resulting in reduced air flow and stagnant conditions within and behind the barrier (Hagler et al., 2012). Indeed, many empirical studies in open road conditions show variable results, with PM concentrations lower, higher or essentially the same downwind as upwind of vegetation (Abhijith et al., 2017). Importantly, wind speed and direction are rarely measured even though these variables may have overriding effects (Viippola et al., 2018; Yli-Pelkonen, Setälä, & Viippola, 2017).

2.1.2 Asthma Reduction via Air Pollution Removal by Urban Trees: Deposition Modeling

Various models have assessed the capacity of urban trees to remove air pollution via deposition, and recent models assess combined effects with dispersion and temperature (e.g., Buccolieri et al. 2018; Santiago, Martilli, and Martin 2017). A review of such models is, however, beyond the scope of this paper. Importantly, most models do not extend findings to human health outcomes. Here, we address modeling that estimates asthma outcomes based on air pollution reduction via deposition onto urban trees.

One of the most widely used deposition models in research and practice is i-Tree Eco, formerly known as U-FORE (Driscoll et al., 2012; Timilsina, Beck, Eames, Hauer, & Werner, 2017). As of 2015, this urban forest ecosystem service model had over 36,000 registered users in some 120 countries (Nowak, 2015). In a systematic review addressing the economic benefits and costs of urban trees, 19 out of 21 studies applying valuation methods such as air pollution reduction were based on i-Tree software or its predecessor algorithms (Song et al., 2018). Air pollution reduction estimates from this deposition model are also sometimes layered with the EPA (2018) BenMAP model to generate estimates of asthma reduction (e.g., Nowak et al. 2013, 2014, 2018). For these reasons, i-Tree/BenMAP modeling deserves special attention in the discussion at hand.

Echoing Janhäll's (2015) caution about applying models to different applications, Timilsina et al. (2017) noted that many urban ecosystem service modeling studies are based on an assumption that relationships developed elsewhere are applicable to sites that vary in species, site, climate, and environmental conditions. To test this hypothesis, they compared the predictive accuracy of i-Tree with a local model using data from 74 trees in Stevens Point, Wisconsin. The sampling design was based on Nowak (1996) for Chicago, Illinois. Timilsina et al. found that predictions from a locally developed model predicting leaf area from DBH were much closer to observed values than Nowak (1996). For example, the Stevens Point diameter at breast height (DBH) model over-estimated leaf area by 6% of the mean whereas the Nowak (1996) DBH model over-estimated leaf area by 106% of the mean.

These findings raise questions about the generalizability of deposition-based modeling such as the commonly used i-Tree. For example, air pollution removal by urban trees and shrubs may, in some places, be less than the ~1% estimated by i-Tree in 55 U.S. cities (Nowak, Crane, & Stevens, 2006). Moreover, the model does not account for spatial variation or differences in tree species composition within and among cities (Saebo, Janhäll, Gawronski, & Hanslin, 2017). Potential errors associated with i-Tree may be further propagated when air pollution reduction estimates are layered with another model – the Environmental Benefits Mapping and Analysis Program (BenMAP) – to extend projections to human health outcomes including asthma. BenMAP is an open source computer program created by the U.S. Environmental Protection Agency (2018) that estimates the health benefits from improvements in air quality. Yet, this model appears to be based on a single study by Ostro et al. (2001), who reason that the strength of their results may have been over or underestimated due to characteristics of the included population and measurement methods. Moreover, BenMAP does not account for the mediating influence of urban trees upon air quality; and urban forest modeling that draws upon BenMAP does not include the complex etiology of a given disease such as asthma. Additional concerns with layered i-Tree/BenMAP modeling have been raised by others (Pataki et al., 2011; Whitlow et al., 2014). We concur that modeling research should provide easy-to-navigate documentation to underlying studies and clearly state caveats and concerns with modeling assumptions, especially when extending findings to actual human health outcomes including but not limited to asthma.

2.1.3 Asthma Reduction via Air Pollution Removal by Urban Trees: LUR Modeling

Studies using land use regression (LUR) models to account for landscape heterogeneity reach inconsistent conclusions. King et al. (2014) built a LUR model in New York City and found no difference in neighborhood levels of NO_2 and PM_{10} between leaf-on and leaf-off seasons despite the expectation that leaves would enhance pollutant removal. The researchers found that estimated total emissions of both pollutants are spatially disconnected from deposition, and they speculate that in this case, tree cover is a surrogate for the absence of pollution sources rather than pollution removal by trees. Indeed, air pollutant concentrations tend to decline rapidly with distance away from automobile tailpipe emissions (Carpentieri & Kumar, 2011; Zhu, Hinds, Kim, & Sioutas, 2002).

Rao et al. (2014) also built an LUR model based on measurements from 144 passive NO_2 samplers, and came to a different conclusion. In Portland, Oregon, their model found that trees accounted for a 10% reduction in NO_2, or roughly 10 times the pollution reduction predicted by i-Tree (Marritz, 2014). The researchers then used BenMAP to predict the effect of the LUR estimates on asthma-related endpoints, estimating approximately 21,000 fewer incidences and 7,000 fewer days of missed school due to asthma exacerbation for 4 to 12-year-olds; 54 fewer ER visits across people of all ages; and 46 fewer cases of hospitalization due to respiratory problems triggered by NO_2 in the elderly. Based on this combination of field measurements and layered models, this study estimated that NO_2 reduction by trees in Portland could provide a $7 million USD annual benefit. This study is sensitive to spatial scale resolution, but similar to studies that layer i-Tree and BenMAP models to project human health and asthma outcomes, the limitations of this approach are not discussed.

2.1.4 Pollution Removal by Urban Trees: Empirical Research

A limited but growing body of in-situ observational studies on links between urban trees and air pollution levels shows mixed findings. Fantozzi et al. (2015) and García-Gómez et al. (2016) found lower NO_2 levels under *Quercus ilex* canopy than open areas in Siena, Italy and three Spanish sites, respectively. In Shanghai, China, Yin et al. (2011) also observed lower NO_2 concentrations in parks with tree cover compared to a single reference site without tree cover. But in Sydney, Australia, Irga et al. (2015) found no observable trends in NO_2 concentrations between sites with different traffic and greenspace densities; and Yli-Pelkonen, Scott et al. (2017) found that NO_2 levels did not differ significantly between tree-covered and open habitats in Baltimore, Maryland. Likewise, two studies found no differences in gaseous pollutant concentrations between tree-covered and open near-road areas in hemiboreal zones (Setälä, Viippola, Rantalainen, & Pennanen, 2013; Yli-Pelkonen, Setälä, et al., 2017). Another study observed elevated gaseous PAH concentrations in road-side forests and parks compared to adjacent treeless areas during summer in Finland (Viippola, Rantalainen, Yli-Pelkonen, Tervo, & Setälä, 2016). Moreover, roadside greenbelts of mostly broadleaf trees did not reduce NO_2 levels in near-road environments, but yielded higher NO_2 levels in front of and inside greenbelts, regardless of season (Yli-Pelkonen, Viippola, Kotze, & Setälä, 2017).

Some field based studies do find PM reductions in areas with more canopy (Irga et al., 2015; Yli-Pelkonen, Setälä, et al., 2017). However, Viippola et al. (2018) found that this only applied to PM_{10}, whereas mean $PM_{2.5}$ levels were unaffected by trees near roads. It is noteworthy that deposition velocity follows a U-shaped (parabolic) curve that reaches a minimum between 0.1 and 1.0 µm (micrometers), meaning that small particles are more prone to stay airborne than the coarse 2.5-10 µm particles that dominate the PM10 class. The physics of this phenomenon are beyond the scope of this paper, but interested readers are encouraged to consult Seinfeld and Pandis (2016). Importantly, small particles less than 10 µm in diameter – and especially those under 2.5 µm – pose the greatest health problems including aggravated asthma, decreased lung function, and increased respiratory symptoms such as irritation of the airways, coughing or difficulty breathing (EPA, 2018).

Viippola et al. (2018) also note that local-scale effects of vegetation on air quality, as exemplified in their study, are seldom measured. Their findings support another study which concluded that reduction in particle levels at tree-covered sites did not relate to vegetation properties such as canopy closure, tree number and size, and ground vegetation (Yli-Pelkonen, Setälä, et al., 2017). Both studies suggest that reduced particle concentration probably results from lower wind speed and reduced air penetration under the canopy rather than deposition.

2.2 BVOC Emissions, O_3 Production, and O_3 Uptake

Ground-level ozone (O_3) has been linked to increased hospital admissions for respiratory problems such as asthma, even at levels below U.S. federal standards for O_3 (UCS, 2017). The relationship between urban trees and O_3 is complex, interacting in a feedback loop that can generate opposing air quality effects (see Figure 3). Urban trees can remove O_3 through gaseous uptake via leaf stomata and direct deposition onto plant surfaces (Calfapietra et al., 2013). But trees also release BVOCs and increase these emissions in response to stressors such as heat, drought, air pollution, or when plant tissue is damaged, e.g., after pruning or from herbivory by insects (Holopainen & Gershenzon, 2010). These BVOCs function as communication media within plant communities, between plants, and between plants and insects (Laothawornkitkul, Taylor, Paul, & Hewitt, 2009). BVOCs are a large class of compounds including but not limited to isoprene, monoterpenes, and sesquiterpenes. In the presence of sunlight, BVOCs interact with oxides of nitrogen to produce O_3. High O_3 levels, in turn, inhibit tree growth and survival (Gao, Calatayud, Garcia-Breijo, Reig-Arminana, & Feng, 2016; Guidi et al., 2017), which prompts the release of BVOCs (Llusià, Peñuelas, & Gimeno, 2002), and enables more O_3 production. All of this raises the pressing question of whether direct O_3 uptake by urban trees outweighs indirect O_3 production by urban trees through emission of BVOCs.

Figure 3: Interactions between urban trees and ground-level ozone (O_3)

2.2.1 BVOC Emissions, O_3 Production, and O_3 Uptake: Review

Calfapietra et al. (2013) reviewed some 35 papers using experimental or modeling methods to assess the role of BVOCs emitted by urban trees on O_3 concentration in cities. They found that realistic estimations of losses and gains of O_3 due to urban vegetation are challenging and highly dependent on local climate. They concluded that under low stress conditions, O_3 uptake is likely to dominate over O_3 formation via BVOC emissions; but in dry conditions, stomatal closure can make uptake of O_3 negligible. Importantly, the review did not report the magnitude of potential effects and it emphasized that field measurements are required to improve mechanistic understanding and to validate and improve models.

Studying the complex interactions between urban trees, BVOCs, NO_x, and O_3 in urban areas with measurements only is extremely difficult. Bonn et al. (2016) attempted to do so by investigating the effects of urban land cover types on levels of different pollutants. They demonstrated that O_3 levels were lowest near coniferous forests followed by deciduous and mixed forests. But they caution that this does not imply that increasing tree cover would reduce citywide O_3 levels. While O_3 levels near large stands of urban trees may be reduced through stomatal uptake of O_3, BVOC emissions from these stands can be transported elsewhere in the city and produce O_3 through reaction with NO_x. Additionally, Calfapietra et al. (2013) caution that BVOC emissions are stimulated by the simultaneous occurrence of high temperatures and drought – conditions that trees are often exposed to in urban settings and conditions that are likely to be more common in the future due to climate change.

2.2.2 BVOC Emissions, O_3 Production, and O_3 Uptake: Modeling

There are two principal types of models investigating links between urban trees and O_3: those that calculate O_3 levels using explicit simulation of meteorological conditions and air chemistry, and those that do not. Reflecting the latter, some studies derive O_3 formation potential by multiplying BVOCs with reactivity coefficients. For instance, Ren et al. (2017) used coefficients of incremental reactivity for different BVOCs, AVOCs, carbon monoxide, and methane to estimate their relative roles in O_3 formation potential in Beijing, China. But this study did not explicitly account for O_3 uptake. By contrast, the widely used i-Tree/UFORE model only estimates O_3 deposition and does not account for O_3 production (Nowak et al., 2006); although it has also been used in conjunction with an air chemistry model to estimate the net effect of urban trees on O_3 levels (Nowak et al., 2000).

The WRF-CHEM model, on the other hand, calculates O_3 levels by combining meteorological conditions and detailed air chemistry (Churkina et al., 2017; Grell et al., 2005). This approach is advisable because net O_3 levels depend on reactions between BVOCs and NO_x as well as on the O_3–NO_x conversion cycle (Bonn et al., 2016). Others reviewed additional models and cautioned that advanced modeling requires computational platforms and skills that may not be available in many urban forestry and planning offices (Leung et al., 2011).

2.2.3 BVOC Emissions, O_3 Production, and O_3 Uptake: Urban-Region Links

A significant challenge in assessing links between urban trees and O_3 is that this relationship depends on background levels and transport of VOCs and NO_x. While O_3 has historically been lower in urban centers than in rural areas downwind of cities (Gregg, Jones, & Dawson, 2003; Sartelet, Couvidat, Seigneur, & Roustan, 2012), time series of observations between 1990-2010 from Europe and the USA (Paoletti, De Marco, Beddows, Harrison, & Manning, 2014) and between 2000-2010 from the Mediterranean region in Europe (Sicard et al., 2013) show convergence of O_3 levels in urban and rural settings. While neither study could attribute a causal mechanism for these observations, the convergence may be partially explained by increasing emissions of BVOCs in cities in response to rising urban air temperatures, air pollutants, and tree planting.

Numerous studies show a strong influence of BVOC emissions upon diminished air quality in Asian, European, and North American urban and periurban areas with high NO_x concentrations during warm seasons (Churkina, Grote, Butler, & Lawrence, 2015). In Berlin, Churkina et al. (2017) found that BVOC emissions from urban vegetation accounted for O_3 increases up to 60% on particular days within an analyzed heat wave period, and between 6% and 20% due to seasonal increases in temperature from June to August. Likewise, Ren et al. (2017) found that despite a significant decline in air pollution emissions, the urban core of Beijing was the greatest source of regional BVOC emissions; and that said emissions from urban greenspaces played a more important role in O_3 and PM formation, and associated health impacts, than rural forests. They attribute this counter-intuitive finding to the introduction of high BVOC-emitting tree species plus high temperatures and low tree density (which allows more light penetration) in urban green spaces, which can augment BVOC emissions. These studies illustrate that our understanding of O_3 formation in urban and nonurban settings is undergoing reevaluation.

2.2.4 BVOC Emissions, O_3 Production, and O_3 Uptake: Summary

The relationship between urban trees and O_3 is very complex and considerable uncertainty remains in determining the net impact of urban vegetation on urban O_3 concentrations. Whether O_3 uptake by urban trees outweighs the O_3 produced by the reaction between BVOCs and NO_x depends on numerous factors including tree species types, regional land cover, season, climate, and air pollution levels. Our review adopts a cautionary stance and suggests that BVOC emissions of urban trees can play a potentially significant role in diminishing air quality – especially in a warming world (Hansen et al., 2016) where some three-quarters of people will be living in urban areas by the end of this century (Angel, 2012). BVOCs should be considered a potential risk factor in any discussion of links between trees, air pollution, and asthma; as well as other air quality related health problems (EPA, 2017). This is all the more relevant as some three-quarters of common or potential urban tree species in Europe have been identified to have moderate or high BVOC emissions (Samson, Ningal, et al., 2017); the same may be true elsewhere. Landscape planning and design implications are discussed in Section 4.2.

2.3 Pollen Production & Synergistic Links with Air Pollution

Pollen production is essential to the reproductive cycle of trees and vegetation. However, the emission of allergenic pollen particles into the atmosphere can add to the burden of inhaled irritants and compromise human health. In addition to being a major risk factor for asthma (as detailed in Section 3), pollen concentrations have been associated with deaths due to cardiovascular disease, chronic obstructive pulmonary disease, pneumonia (Brunekreef, Hoek, Fischer, & Spieksma, 2000; Weichenthal, Lavigne, Villeneuve, & Reeves, 2016), and even suicide (Qin, Waltoft, Mortensen, & Postolache, 2013). Some studies suggest that people living in urban areas are 20% more likely to suffer airborne pollen allergies (pollinosis) than people living in nonurban areas (D'Amato et al., 2007; P. Ogren, 2002); and lack of attention to plant species selection has been described as a factor contributing to "one of the most widespread diseases in urban populations: pollen allergy" (Cariñanos & Casares-Porcel, 2011, p. 205).

In Denmark, for example, urban areas are a significant source of birch pollen (Skjøth et al., 2008); and in the country's second largest city, the municipal council of Aarhus has halted the planting of birch trees in public places, as the species is a major culprit in provoking allergic reactions (BBC, 2015). According to one study, roughly 50% of common or potential urban tree species in Europe have moderate or high pollen allergenicity (Samson, Ningal, et al., 2017). A substantial review has, in turn, identified eight prominent causes for the increased pollen allergen load by urban plants, especially trees. This includes low species biodiversity at planting; overabundance of species that act as key pollen sources; planting of exotic species that are prompting new allergies in local human populations; choice of male, pollen-producing individuals in dioecious species (i.e., trees that are wholly male or wholly female); presence of invasive species; inappropriate garden management and maintenance activities; appearance of cross-reactivity between phylogenetically related species; and the interaction between pollen and air pollutants (Cariñanos & Casares-Porcel, 2011).

Another review found that pollen levels vary substantially within a given city (Weinberger, Kinney, & Lovasi, 2015), suggesting that both spatial and temporal variation in exposure may be needed to more precisely estimate pollen levels that individuals encounter. Most of the studies contributing to this review relied on data from only a few monitors; yet other studies with much denser monitoring networks have come to the same conclusion regarding the spatial variability of pollen levels (Hjort et al., 2016; Weinberger et al., 2018).

Moreover, airborne pollen grains mechanically interact with air pollutants, triggering release of allergen-containing granules (Löhmus & Balbus, 2015), which can increase the risk for allergic and asthmatic reactions. For example, gaseous pollutants (e.g. NO_2, SO_2) have been shown to damage pollen grains, thereby generating microscopic allergenic particles that can penetrate the lower respiratory tract (Ouyang, Xu, Fan, Li, & Zhang, 2016). Allergic reactions to tree pollen are also exacerbated by O_3 (Samson, Grote, et al., 2017), and studies consistently find that rodents exposed to pollen and PM are more likely to become allergic to pollen than exposure to pollen alone (Fernvik, Peltre, Sénéchal, & Vargaftig, 2002; Steerenberg et al., 1999).

Increased exposure to pollutants in urban regions may explain why residents in cities can be more likely to suffer from pollen allergies compared to their nonurban counterparts (Löhmus & Balbus, 2015). Pollutants can increase the allergenic properties of pollen, and along these lines, pollen allergenicity (i.e. the potential of pollen to cause allergic reactions) may be greater in urban than in nonurban areas (Armentia et al., 2002). There are several possible explanations for this: 1) pollutants such as NO_2 trigger chemical reactions (i.e. nitration) involving pollen and thereby stimulate immune mechanisms that contribute to allergies; 2) pollutants may induce the expression of allergenic proteins in pollen; and 3) pollutants (e.g. diesel exhaust particles) directly bind to pollen and trigger allergic reactions (Sedghy, Varasteh, Sankian, & Moghadam, 2018). These synergistic interactions between primarily urban pollutants and pollen may contribute to the rising prevalence of asthma among inner-city populations. Indeed, there is evidence of pollen and other aeroallergens interacting with air pollutants to increase asthma hospitalization (Cakmak, Dales, & Coates, 2012; Hebbern & Cakmak, 2015).

2.4 Biophysical Processes Linking Urban Trees, Air Quality & Asthma: Summary

Research assessing asthma reduction via air quality improvement by urban trees must account for a complex suite of interconnected biophysical processes. Deposition and dispersion are the mechanisms whereby urban trees may reduce anthropogenic air pollution levels. But in the real world, these mechanisms rarely function independent of one another, and deposition alone seems to yield a mere 1% reduction in air pollution, while trees in urban canyons with a building height to street width ratio ≥ 0.5 generally increase air pollutant concentrations by limiting air dispersion. Along open roads, literature is mixed: depending on a range of conditions, vegetation buffers may reduce downwind air pollution, but they may also lead to no improvement or even locally increased pollutant concentrations.

Moreover, the pollen production of urban trees, synergistic interactions of pollen with air pollution, and the contribution to O_3 pollution from BVOCs emitted by urban vegetation suggests that in some places, urban trees may have a net negative effect on air quality and associated links to asthma. Land use regression models arrive at contradictory findings; and deposition modeling is not consistently supported by empirical measurements, which also reach inconsistent conclusions. The commonly used i-Tree model, in turn, does not account for dispersion, pollen production, or O_3 production, which combined with potential errors in underlying assumptions raises questions about asthma projections via layered modeling with BenMAP. All of this reinforces the need to consult scholarship on observable, empirical links between asthma and urban trees, as discussed in the ensuing section.

3. Empirical Links Between Asthma & Urban Trees

Research that focuses on observable human health outcomes is generally conducted by epidemiologists and public health scholars. One of the first empirical studies to investigate the relationship between urban trees and asthma found a link between street trees and lower prevalence of early childhood asthma when comparing large neighborhood areas across New York City (Lovasi, Quinn, Neckerman, Perzanowski, & Rundle, 2008). However, this study had methodological gaps (Zandbergen, 2009) and was unable to account for all potentially important neighborhood differences in the characteristics of children and households. Pilat et al. (2012) subsequently found no link between childhood asthma and tree cover in Texas metropolitan statistical areas.

Other studies have reported links between more greenery and decreased prevalence of asthma in children and adults in urban populations (Donovan, Gatziolis, Longley, & Douwes, 2018; Maas et al., 2009; Sbihi, Tamburic, Koehoorn, & Brauer, 2015; Ulmer et al., 2016). Yet, these studies do not to the best of our knowledge identify air quality improvement as the mediating mechanism. For example, Donovan et al. (2018) hypothesize that observed benefits may be explained by greater and more diverse microbial exposure in vegetated spaces. Additionally, these studies did not consider if the study populations were allergic to tree pollen, or the allergenicity of local trees. These are important considerations as populations can have different sensitivity to pollen (Baxi & Phipatanakul, 2010), and the allergenicity of local vegetation may vary depending on a range of factors (Cariñanos & Casares-Porcel, 2011; Kuchcik, Dudek, z łażejczyk, Milewski, & z łażejczyk, 2016).

In contrast to these studies, Lovasi et al. (2013) conducted more detailed follow-up measurements than the area-level analysis of Lovasi et al. (2008), and arrived at a different conclusion. Enrolling individual children in disadvantaged areas of New York City (Northern Manhattan and the Bronx) from birth to 7 years of age, the authors found that local canopy cover offered no evidence of a protective association with an asthma diagnosis or symptoms, even among the subset *without* allergic sensitization to the assessed tree pollen mix (including common street tree species but not all allergenic species that are present). Moreover, they found that local tree cover was associated with a possible increase in asthma prevalence and a clear increase in allergic reaction to tree pollen.

Increased tree pollen has been consistently linked to seasonal peaks in adult and pediatric emergency department (ED) visits for asthma (Jariwala et al., 2011, 2014; Weinberger, Robinson, & Kinney, 2015), asthma hospitalizations (Dales, Cakmak, Judek, & Coates, 2008; Dales et al., 2004), allergic sensitization among children; and both over-the-counter purchases (Sheffield et al., 2011) and prescriptions (Ito et al., 2015) of allergy medications. For example, high concentrations of oak pollen measured at a site in Atlanta were associated with increased citywide ED visits for asthma and wheeze (Darrow et al., 2012). In the spring, the days with highest pollen counts in the Bronx were significantly associated with increased asthma ED visits, and the days with lower pollen counts were linked to relatively decreased asthma ED visits (Jariwala et al., 2014). While such studies consistently find a link between increased tree pollen and spring asthma exacerbations, the magnitude of this observed relationship likely varies based on differences in canopy coverage and the distribution of specific taxa. In New York City, tree pollen-related spikes in asthma ED visits were elevated in zip codes with a higher degree of tree canopy cover (Weinberger, Robinson, & Kinney 2015).

Moreover, a Bronx-based study assessing how air pollutants (NO_2, fine PM), humidity, and tree pollen together influence asthma ED visits yielded a noteworthy result. The highest quartile of daily tree pollen counts in the spring season resulted in consistently elevated asthma ED visits across pollutant or humidity levels. By contrast, on days when tree pollen counts were low, higher humidity and air pollution measurements were not significantly associated with increased asthma ED visits (Toh et al., 2014). Along these lines, Dadvand et al. (2014) did not observe any association between ambient levels of air pollutants (NO_2 and PM_{10}) at the home address of study participants and current asthma and allergic rhinoconjunctivitis. Yet, living close to parks in urban areas was associated with a 60% higher relative prevalence of current asthma. These results suggest that in some locales, and per seasonal effects, tree and plant pollen are major contributors towards increased asthma morbidity, especially among individuals that have allergic sensitization to corresponding types of pollen (DellaValle et al., 2012).

In contrast to these findings, Alcock et al. (2017) found that increased tree density was linked to reduced asthma hospitalizations amidst high air pollutant levels. These seemingly conflicting results may be explained in a couple of ways: 1) the aforementioned studies evaluate population-level datasets, and do not account for individual characteristics; and 2) the relative impacts of pollen and pollutants on asthma morbidity may differ by geographic region and are likely not generalizable. Other limitations of investigations linking urban trees to asthma, as well as place and health studies more broadly, is the reliance on residential location rather than alternatives such as activity space that may capture a more complete picture of an individual's environment. Most studies also fail to consider patient-level information such as an individual's predisposition to allergies and/or allergic sensitization status.

4. Discussion

4.1 Interdisciplinary Assessment

This interdisciplinary review finds that links between urban trees, air quality, and asthma are very complex and include numerous interacting processes. While anthropogenic air pollution deposits by gravity on all surfaces including trees, there is no scientific consensus that urban trees reduce citywide air pollution levels – there are inconsistencies within models, between models, among empirical studies, and between model and empirical studies. Both species composition and spatial arrangement have been noted as critical to pollution mitigation potential by urban trees (Saebo et al., 2017), and users of modeling tools are encouraged to account for this variability by collecting parameters for smaller spatial units rather than assuming homogenous conditions over large areas. There is, however, a consensus that in urban corridors with a building height to street width ratio ≥ 0.5m – which is common in city centers and includes many of the world's great streets (e.g., Massengale & Dover, 2014) – trees can concentrate air pollution by reducing dispersion. Moreover, the pollen production of urban trees, synergistic interactions of pollen with air pollution, and the contribution of O_3 pollution from

BVOCs emitted by urban vegetation suggest that urban trees can in many circumstances have a negative effect on air quality and associated links to asthma.

Importantly, population-based empirical human health studies covered in this review often contradict or fail to support the purported asthma benefits of urban trees. With the exception of Alcock et al. (2017), those finding beneficial health links did not identify air pollution reduction as the mediating mechanism. Moreover, there are substantial differences in how epidemiology and natural science study the topic at hand. Epidemiological research generally starts with empirical observations of human morbidity and mortality, then examines etiological pathways to characterize chains and webs of causation that could contribute to observed patterns in the data. In this review, most epidemiological research linking trees to asthma highlights pollen production rather than pollution reduction. *This raises a fundamental question: if urban trees are a potentially meaningful strategy to decrease asthma and respiratory disease by reducing urban air pollution levels, why is there such a dearth of public health scholarship and empirical evidence demonstrating this relationship?*

Natural science, on the other hand, generally relies upon a combination of laboratory or field measurements and modeling studies to assess mechanistic links between trees and air pollution levels. The vast majority of this research is limited to the biophysical interaction of trees – through deposition, dispersion, and O_3 production/mitigation – and air pollution. In other words, most natural science research on links between urban trees and air quality focuses on biophysical processes (or ecosystem functions) and does not make assertions about human morbidity and mortality (ecosystem services). Those that do tend to rely upon simultaneous use of multiple models – each with its own uncertainty which taken together increases the total uncertainty of predictions – and methods that do not include all of the pathways linking urban trees with asthma. This type of epistemological reductionism and etiological oversimplification is misleading and renders public policy recommendations emerging from such research incomplete and unbalanced.

The substantial differences in how epidemiology and natural science study and draw conclusions on the health benefits of urban vegetation exhibit “disciplinary crosstalk” – poor communication, unconscious misunderstandings, and inconsistent use of terms and literature between disciplines (definition based upon synthesis of Inchausti, 2012; Kirk-Lawlor & Allred, 2017; Vogt, 2018). This is occurring in urban forestry scholarship (Vogt, 2018); and in the case at hand, disciplinary crosstalk is exacerbated by the ascendance of an ecosystem services construct that may not be well-suited for depicting and studying urban flora. More specifically, the biophysical processes – or ecosystem *functions* – of urban trees are often conflated with actual human health and well-being outcomes – or ecosystem *services/disservices*. The former is studied primarily in natural science, while the latter is largely the expertise of public health; and the terms are routinely misused in scholarship and practice (e.g., Boyd & Banzhaf, 2007; Fu et al., 2011; Lamarque, Quetier, & Lavorel, 2011). Our review suggests that assessment of ecosystem functions alone is insufficient to generate meaningful public health and associated urban planning guidance, and that epidemiological methods must be more thoroughly incorporated in urban ecosystem service scholarship (see figure 4).

Figure 4: Relationship between urban ecosystem functions, urban ecosystem services and disservices, and disciplinary expertise pertaining to air quality and human health.

To continue developing an evidence base that can accurately inform planning and public policy on urban air quality and asthma, we echo the recommendation of others for more interdisciplinary research on the human health and well-being benefits of urban vegetation (e.g., Sullivan et al., 2014; Vogt, 2018). This type of integrative scholarship must embrace an epistemological and etiological pluralism (Miller et al., 2008; Vandenbroucke, Broadbent, & Pearce, 2016) that is currently lacking in urban ecosystem services research and associated urban greening theory and practice. It will also require scholars and practitioners to reflect on how positionality – or personal and professional identity – informs epistemology (Takacs, 2003); and to critically examine beliefs that frame urban trees as an environmental panacea or urban sustainability fix (e.g., Ostrom, Janssen, and Anderies 2007; Pincetl 2018; Silvera Seamans 2013).

Building upon Sullivan et al. (2014), we issue a call for any research that draws conclusions on correlations between urban trees and asthma to cite epidemiological scholarship on this relationship, and to include public health expertise on the research team. It is also incumbent upon researchers to clearly state the limitations of findings and to be especially conscientious when making assertions about actual human illness and death. Anything less risks obfuscating more than illuminating science and policy on public health issues such as asthma that affect millions of people worldwide.

Importantly, this paper does not in any way imply that trees should not be planted in cities, or that urban greenery does not provide a cornucopia of wildlife and human health and well-being benefits (Beatley, 2016; Tan & Jim, 2017). While causal mechanisms linking greenspace to improved public health are unclear (Frumkin et al., 2017; Hartig et al., 2014; Markevych et al., 2017), having regular contact with urban flora may be essential to the well-being of future generations (Eisenman, 2016) – the vast majority of whom will live in cities. Moreover, it is conceivable that urban silva and other forms of nearby nature (Kaplan, 1985) may help to decrease asthma by reducing or promoting recovery from stress – a risk factor for asthma and one of the most consistently identified benefits that people derive from spending time in or near greenspaces (Bratman, Hamilton, & Daily, 2012; Kuo, 2015).

4.2 Implications for Urban Planning & Greening Practice

This interdisciplinary review finds no scientific consensus to support efforts to reduce asthma morbidity by mitigating air pollution via large-scale urban tree planting. Reciting this purported relationship in scientific and public media risks diverting limited resources from the most important strategy to reduce urban air pollution: decreasing pollutant emissions. Moreover, urban trees can in many circumstances reduce air quality and exacerbate asthma; and the pollen allergenicity and BVOC emissions of urban trees are important but frequently overlooked factors that deserve special attention when considering links between urban trees and asthma. This is of increasing relevance in light of growing interest in municipal tree planting programs. Due to space constraints, we cannot address pollen allergenicity, BVOC emission, and design guidelines in depth. But as outlined below, there is existing guidance on each of these topics, and urban tree planting actors are encouraged to evaluate and incorporate recommendations from cited sources. Urban tree planting and design should consider a range of factors including regional climate and land cover, existing urban tree composition and age, as well as aesthetic, psychosocial, wildlife, and ecosystem function considerations (Beatley, 2016; Nassauer, 1995; Tan & Jim, 2017). Beyond tree species selection and configuration, other types of urban design that incorporate vegetation such as green roofs and walls or ground-level plantings warrant further consideration (e.g., Pugh et al. 2012; Janhäll 2015), as do air pollution emission reduction strategies (Baró, Haase, Gómez-Baggethun, & Frantzeskaki, 2015). Discussion of these topics and their air quality and asthma implications is, however, beyond the scope of this review.

4.2.1 Urban Tree Planting for Low Pollen Allergenicity

The effect of urban flora on the development of pollinosis by city inhabitants is extensively documented (Cariñanos & Casares-Porcel, 2011), and recommendations to reduce impacts often place the burden on individuals to reduce pollen exposure by limiting outdoor activities during the pollen season, staying inside during peak pollen periods, using air filters and air conditioners, monitoring local aerobiological information, and taking medication (e.g., Asthma UK, 2018; Mayo Clinic, 2018). Infrequently, recommendations also include planting trees with low pollen allergenicity (e.g., Seitz & Escobedo, 2009).

Cariñanos and Casares-Porcel (2011) have outlined nine landscape planning and design guidelines for low-allergy impact: (a) Increase plant biodiversity; (b) Ensure moderate, controlled introduction of exotic flora; (c) Control invasive species; (d) Avoid extensive use of male individuals of dioecious species (avoid botanical sexism); (e) Choose species with low-to-moderate pollen production; (f) Adopt appropriate management, maintenance and gardening strategies to ensure removal of opportunist and spontaneous species; (g) Avoid forming large focal pollen sources and screens by respecting planting distances; (h) Obtain expert advice when selecting suitable species for each landscape, and avoid fostering cross-reactivity between panallergens; (i) Establish local authority by-laws ensuring that sufficient time is available for the design and planning of urban greenspaces.

For additional guidance, see Samson, Ningal et al. (2017), which identifies the pollen allergenicity of 150 common or potential urban tree species in Europe. See also *The Allergy-Fighting Garden: Stop Asthma and Allergies with Smart Landscaping* (T. L. Ogren, 2015; an update to the original version in 2000), which ranks on a 1–10 scale the allergenicity of over 3,000 plants.

4.2.2 Urban Tree Planting for Low BVOCs

The most important strategy to reduce ground-level O_3 is to reduce or eliminate NO_x pollution from anthropogenic sources. But in light of high NO_x pollution levels in cities today and the capacity of urban trees to contribute to O_3 formation through emission of BVOCs, municipalities that are pursuing tree planting initiatives are advised to select low BVOC-emitting species. This becomes all the more relevant in light of rising global temperatures, the exceptionally warm condition of cities resulting from urban heat island effect, and the increasing concentration of people in cities – all of which combine to increase potential O_3 exposure. But as noted by Churkina et al. (2015), municipal greening programs generally ignore or are unaware of this fact.

Plant species differ in their magnitude of BVOC emissions. Amongst trees commonly or likely planted in European cities, Samson, Ningal et al. (2017) have identified some 120 species with moderate or high BVOC emissions. In Beijing, Ren et al. (2017) noted that BVOC emissions are dominated by a few species including *Populus tomentosa*, *Sophora japonica* and *Salix babylonica*, *Populus canadensis,* and *Albizia julibrissin*. Churkina et al. (2015) have, in turn, developed a list of 24 common urban tree species with high, medium, and low BVOC emissions rates. *Nyssa sylvatica, Populus, Quercus robur* and *lobate, Robinia pseudocacia,* and *Platanus acerfolia* are among the highest BVOC emitters.

4.2.3 Open Road and Street Canyon Tree Planting

Lack of sufficient evidence makes it challenging to draw conclusive design recommendations for tree planting in urban street canyons and open road conditions, and there are numerous considerations beyond air pollution mitigation that inform how trees should be used in landscape planning and design. A synthesis of studies (Abhijith et al., 2017) suggests that for street canyon environments, consideration of trees and air pollution mitigation is heavily dominated by aspect ratio since buildings restrict wind circulation and dispersion of polluted air. Thus, trees in street canyons with aspect ratios ≥ 0.5 can further decrease air flow and dispersion of air pollutants. In these conditions, low-growing hedges have the potential to act as a vegetative barrier between air pollution source (vehicles) and receptor (people); yet, positive gains on local air quality may still be modest or negative (Jeanjean, Gallagher, Monks, & Leigh, 2017; Shaneyfelt et al., 2017; Vos, Maiheu, Vankerkom, & Janssen, 2013).

This does not in any way suggest that trees and flora should not be planted along urban streets. In addition to infiltrating stormwater and providing wildlife habitat, street trees provide important psychological and social benefits unrelated to air quality (e.g., Donovan and Prestemon 2012; Lin et al. 2014; Lindal and Hartig 2015). And by filtering sunlight, calming traffic, softening edges, enhancing human scale by providing a sense of enclosure, and offering the beauty of flora, well-maintained trees are essential elements of pedestrian-friendly streets that make for great, livable cities (Massengale & Dover, 2014).

Along open roads, research on the air pollution mitigation potential of trees is inconclusive and depends upon a complex range of factors including buffer height, width, length, and density, as well as species and leaf characteristics. Importantly, wind direction and speed – factors over which landscape planners and designers have little control – may have overriding effects. Moreover, roadside greenbelts often consist of conserved land, or land that has been allowed to naturalize on its own, making it difficult to control important factors such as density. The sheer scale of open road greenbelts can also make proactive planting prohibitive. With these caveats in mind, Baldauf (2017) offers some general guidelines addressing buffer height, thickness, density, and length, as well as vegetation characteristics including seasonality, leaf surface, and resistance to pollution. In general, the taller and wider the vegetative buffer, the better; and density should not be too low or too high. While this may be difficult to manage in the real world of living and constantly changing vegetation, Baldauf (2017) suggests that 50% to 90% of the buffer volume should consist of vegetative surfaces, as anything more or less could reduce air quality behind the buffer. The same considerations for pollen and BVOC emissions described earlier also apply here.

5. Conclusion

There is currently no scientific consensus to support ambitious canopy cover goals and large-scale urban tree planting as a meaningful strategy to reduce asthma by improving air quality. Models suggesting that this is the case are not consistently supported by epidemiological or field-based empirical evidence. Moreover, trees in urban street canyons can concentrate local air pollution by reducing air circulation; and urban trees can exacerbate asthma through pollen production, synergistic effects between pollen and air pollution, and O_3 formation through emission of BVOCs. Causal pathways between asthma and urban trees are very complex, and there are substantial differences in how disciplines approach this issue. Future research on this topic – as well as on urban ecosystem services and urban greening – should embrace epistemological and etiological pluralism and be conducted through interdisciplinary teamwork.

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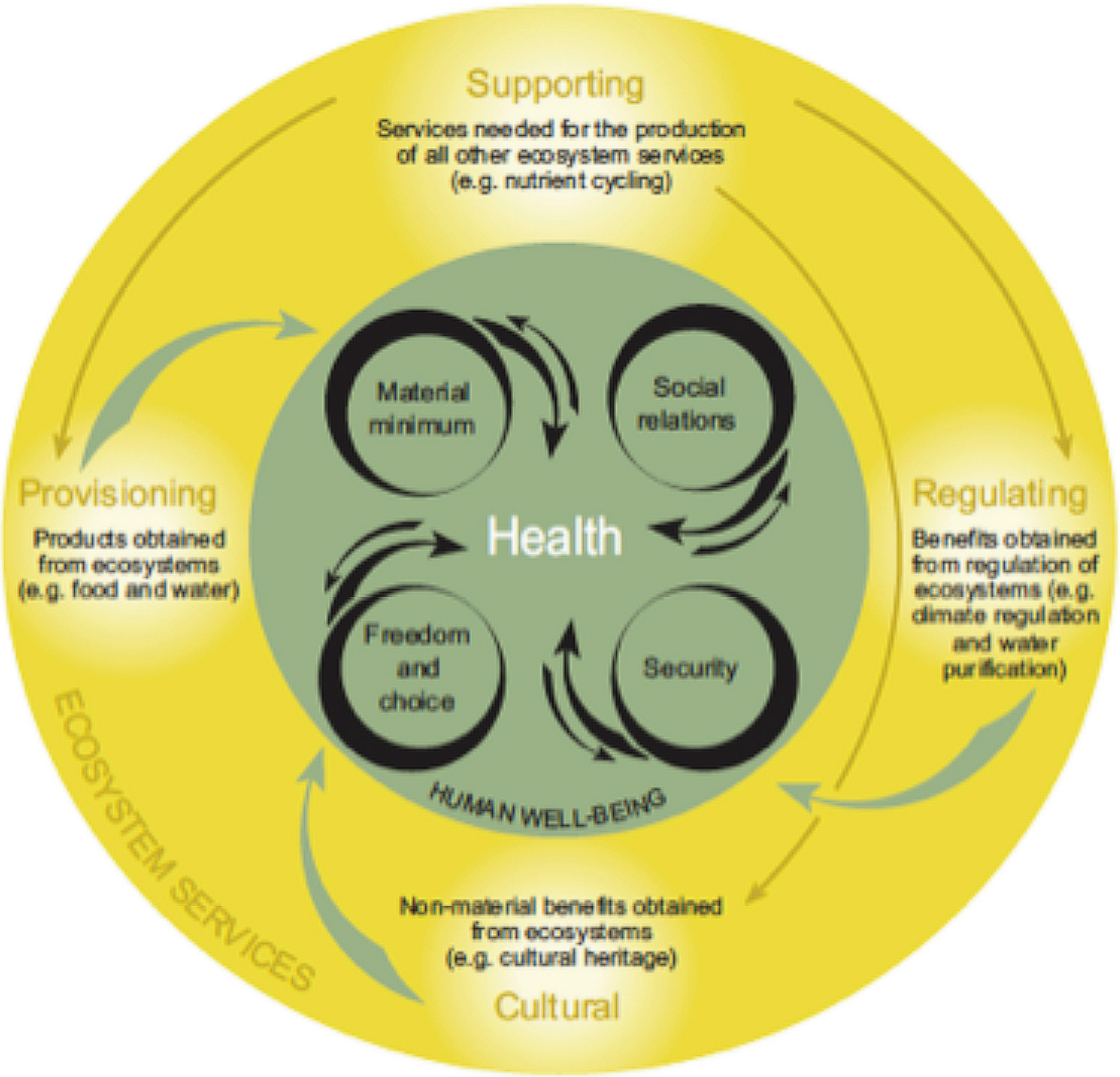
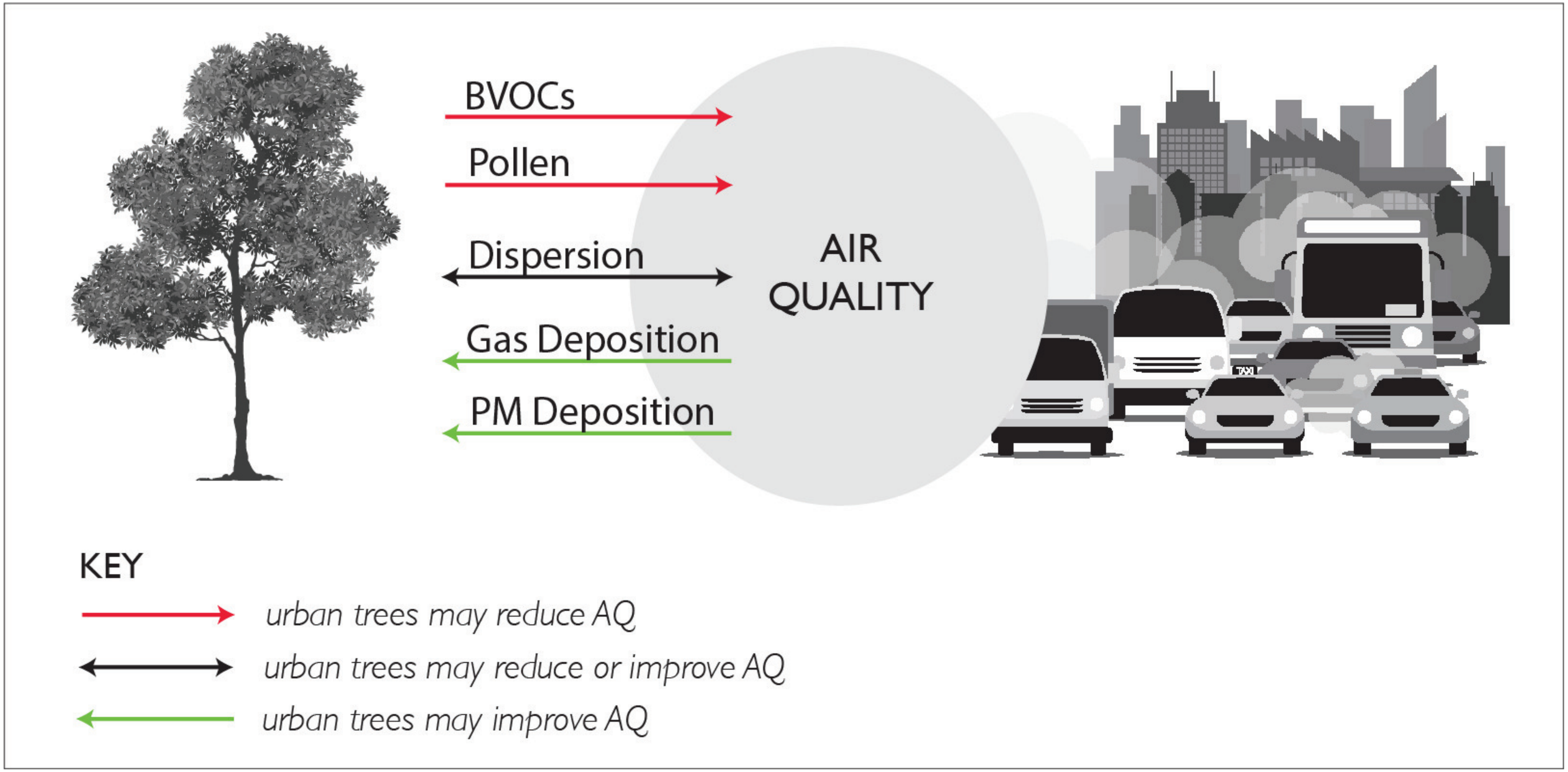

Supporting
Services needed for the production of all other ecosystem services (e.g. nutrient cycling)
Provisioning
Products obtained from ecosystems (e.g. food and water)
Regulating
Benefits obtained from regulation of ecosystems (e.g. climate regulation and water purification)
Cultural
Non-material benefits obtained from ecosystems (e.g. cultural heritage)
Material minimum
Social relations
Health
Freedom and choice
Security
HUMAN WELL-BEING
ECOSYSTEM SERVICES


BVOCs
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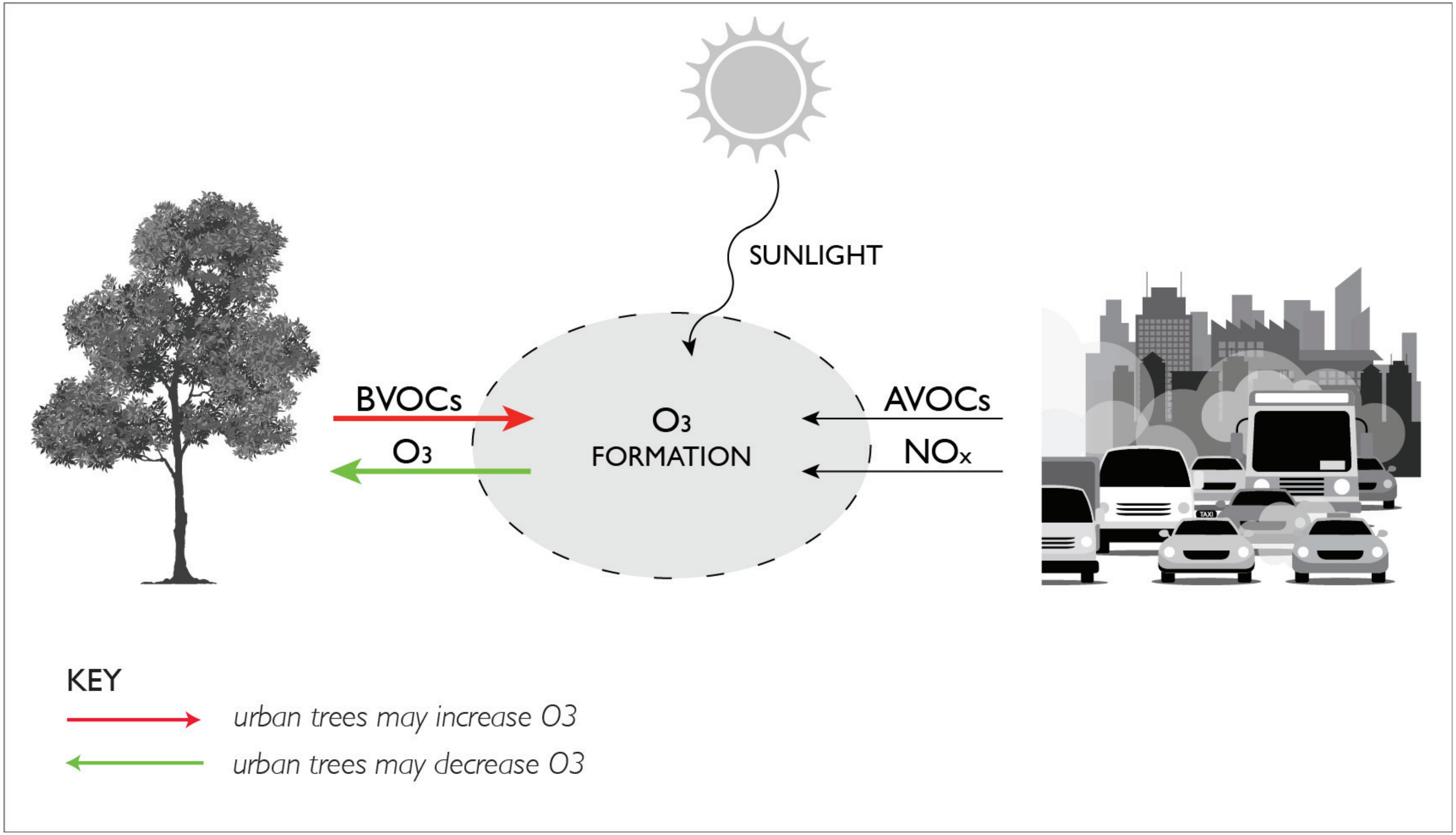
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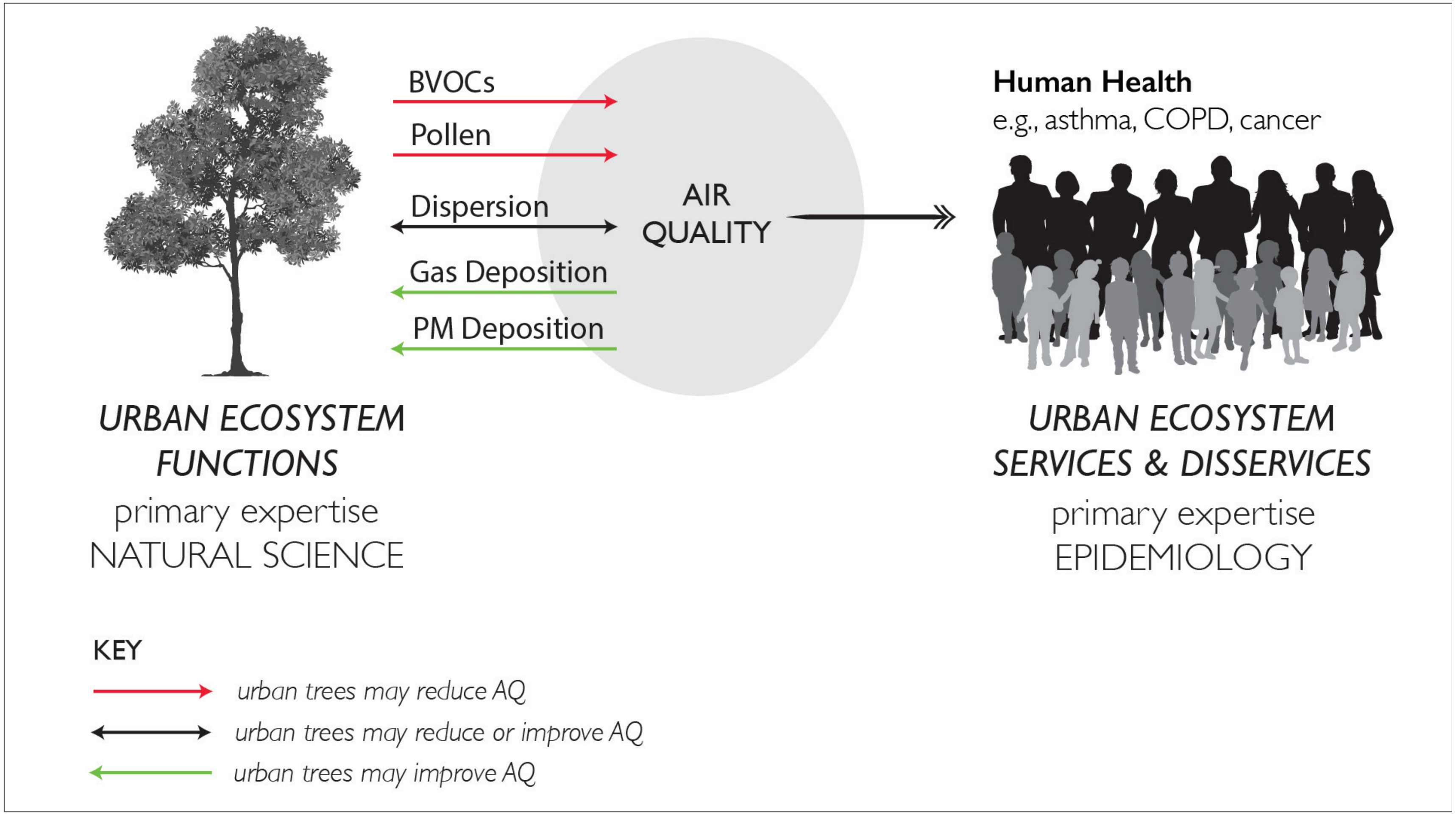
BVOCs
Pollen
Dispersion
Gas Deposition
PM Deposition
AIR QUALITY
Human Health
e.g., asthma, COPD, cancer
URBAN ECOSYSTEM FUNCTIONS
primary expertise
NATURAL SCIENCE
URBAN ECOSYSTEM SERVICES & DISSERVICES
primary expertise
EPIDEMIOLOGY
KEY
urban trees may reduce AQ
urban trees may reduce or improve AQ
urban trees may improve AQ