4. Biodiversity, air quality and human health

1. Introduction

Air pollution is a significant problem in cities across the world. It affects human health and well-being, ecosystem health, crops, climate, visibility and human-made materials. Health effects related to air pollution include its impact on the pulmonary, cardiac, vascular and neurological systems (Section 2). Trees affect air quality through a number of means (Section 3) and can be used to improve air quality (Section 4). However, air pollution also affects tree health and plant diversity (Section 5). Bioindicators can be useful for monitoring air quality and indicating environmental health (Section 6). Understanding the impacts of vegetation biodiversity on air quality and air quality on vegetation biodiversity is essential to sustaining healthy and diverse ecosystems, and for improving air quality and consequently human health and well-being.

2. Air pollution and its effects on human health

Air pollution can significantly affect human and ecosystem health (US EPA 2010). Recent research indicates that global deaths directly or indirectly attributable to outdoor air pollution reached 7 million in 2012 (WHO 2014[1]). This was equivalent to 1 in every 8 deaths globally, making air pollution the most important environmental health risk worldwide (WHO 2014a). Other diseases affected by air pollution include cardiovascular disease, immune disorders, various cancers, and disorders of the eye, ear, nose and throat such as cataract and sinusitis. Epidemiological evidence suggests that prenatal exposure to certain forms of air pollution can harm the child, affecting birth outcomes and infant mortality. Childhood exposure to some pollutants also appears to increase the risk of developing health problems in later life, affecting the development of lung function and increasing the risk for development of chronic obstructive pulmonary disease (COPD) and asthma.

Several respiratory illnesses caused or otherwise affected by air pollution are on the rise. These include bronchial asthma, which affects between 100 and 150 million people worldwide, with another 65 million affected by some form of COPD. Other human health problems from air pollution include: aggravation of respiratory and cardiovascular disease, decreased lung function, increased frequency and severity of respiratory symptoms (e.g. difficulty in breathing and coughing, increased susceptibility to respiratory

[1] World Health Organization, 2015. Health and the Environment: Addressing the health impact of air pollution. Sixty-eighth World Health Assembly, Agenda item 14.6. A68/A/CONF./2 Rev.1 26 May 2015. http://apps.who.int/gb/ebwha/pdf_files/WHA68/A68_ACONF2Rev1-en.pdf (last accessed June 2015)

infections), effects on the nervous system (e.g. impacts on learning, memory and behaviour), cancer and premature death (e.g. Pope et al. 2002). People with pre-existing conditions (e.g. heart disease, asthma, emphysema), diabetes, and older adults and children are at greater risk for air pollution-related health effects. In the United States (US), approximately 130 000 particulate matter $(PM)_{2.5}$-related deaths and 4700 ozone (O_3)-related deaths in 2005 were attributed to air pollution (Fann et al. 2012).

Air pollution comes from numerous sources. Major causes of gaseous and particulate outdoor air pollution with a direct impact on public health include the combustion of fossil fuels associated with transport, heating and electricity generation, and industrial processes such as smelting, concrete manufacture and oil refining. Other important sources include ecosystem degradation (including deforestation and wetland drainage) and desertification.

Plants provide an important ecosystem service through the regulation of air quality. Although the effects of plants on air quality are generally positive, they can also to some degree be negative (as discussed in section 3 below). Likewise, air quality can have both positive and negative impacts on plant populations. These various impacts are partially dependent upon the diversity of the plant species, vegetation assemblages and size classes. This chapter explores the role of biodiversity in regulating air quality in positive and negative terms, including a discussion of current knowledge gaps and recommendations.

Air pollution also affects the environment. Ozone and other pollutants can damage plants and trees, and pollution can lead to acid rain. Acid rain can harm vegetation by damaging tree leaves and stressing trees through changing the chemical and physical composition of the soil. Particles in the atmosphere can also reduce visibility. The typical visual range in the eastern US parks is 15–25 miles, approximately one third of what it would be without human-induced air pollution. In western USA, the visual range has decreased from 140 miles to 35–90 miles (US EPA 2014). Air pollution also affects the earth's climate by either absorbing or reflecting energy, which can lead to climate warming or cooling, respectively.

Indoor air pollution is primarily associated with particulates from combustion of solid fuel (wood, coal, turf, dung, crop waste, etc.) and oil for heating and cooking, and gases from all fuels (including natural gas) in buildings with inadequate ventilation or smoke removal. The World Health Organization (WHO) reports that over 4 million people die prematurely from illness attributable to household air pollution from cooking with solid fuels. More than 50% of premature deaths among children under 5 years of age are due to pneumonia caused by particulate matter (soot) inhaled from household air pollution. It is estimated that 3.8 million premature deaths annually from noncommunicable diseases (including stroke, ischaemic heart disease, lung cancer and COPD) are attributable to exposure to household air pollution (WHO 2014b).

Some pollutants, both gaseous and particulate, are directly emitted into the atmosphere and include sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), particulate matter (PM) and volatile organic compounds (VOC). Other pollutants are not directly emitted; rather, they are formed through chemical reactions. For example, ground-level O_3 is often formed when emissions of NO_x and VOCs react in the presence of sunlight. Some particles are also formed from other directly emitted pollutants.

3. Impacts of vegetation on air quality

There are three main ways in which plants affect local air pollution levels: via effects on local microclimate and energy use, removal of air pollution, and emission of chemicals. Each of these are described below.

1) Effects of plants on local microclimate and energy use

Increased air temperature can lead to increased energy demand (and related emissions) in the summer (e.g. to cool buildings), increased air pollution and heat-related illness. Vegetation, particularly trees, alters microclimates and cools the air through evaporation from tree transpiration, blocking winds and shading various surfaces. Local environmental influences on air temperature include the amount of tree cover, amount of impervious surfaces in the area, time of day, thermal stability, antecedent moisture condition and topography (Heisler et al. 2007). Vegetated areas can cool the surroundings by several degrees Celsius, with higher tree and shrub cover resulting in cooler air temperatures (Chang et al. 2007). Trees can also have a significant impact on wind speed, with measured reductions in wind speed in high-canopy residential areas (77% tree cover) of the order of 65–75% (Heisler 1990).

2Temperature reduction and changes in wind speed in urban areas can have significant effects on air pollution. Lower air temperatures can lead to lower emission of pollutants, as pollutant emissions are often related to air temperatures (e.g. evaporation of VOCs). In addition, reduced urban air temperatures and shading of buildings can reduce the amount of energy used to cool buildings in the summer time, as buildings are cooler and air conditioning is used less. However, shading of buildings in winter can lead to increased building energy use (e.g. Heisler 1986).[2] In addition to temperature effects, trees affect wind speed and mixing of pollutants in the atmosphere, which in turn affect local pollutant concentrations. These changes in wind speed can lead to both positive and negative effects related to air pollution. On the positive side, reduced wind speed due to shelter from trees and forests will tend to reduce winter-time heating energy demand by tending to reduce cold air infiltration into buildings. On the negative side, reductions in wind speed can reduce the dispersion of pollutants, which will tend to increase local pollutant concentrations. In addition, with lower wind speeds, the height of the atmosphere within which the pollution mixes can be reduced. This reduction in the "mixing height" tends to increase pollutant concentrations, as the same amount of pollution is now mixed within a smaller volume of air.

2) Removal of air pollutants

Trees remove gaseous air pollution primarily by uptake through the leaves, though some gases are removed by the plant surface. For O_3, SO_2 and NO_2, most of the pollution is removed via leaf stomata.[3] Healthy trees in cities can remove significant amounts of air pollution. The amount of pollution removed is directly related to the amount of air pollution in the atmosphere (if there is no air pollution, the trees will remove no air pollution). Areas with a high proportion of vegetation cover will remove more pollution and have the potential to effect greater reductions in air pollution concentrations in and around these areas. However, pollution concentration can be increased under certain conditions (see Section 4). Pollution removal rates by vegetation differ among regions according to the amount of vegetative cover and leaf area, the amount of air pollution, length of in-leaf season, precipitation and other meteorological variables.

There are numerous studies that link air quality to the effects on human health. With relation to trees, most studies have investigated the

[2] This altered energy use consequently leads to altered pollutant emissions from power plants used to produce the energy used to cool or heat buildings. Air temperatures reduced by trees can not only lead to reduced emission of air pollutants from numerous sources (e.g. cars, power plants), but can also lead to reduced formation of O_3 ,as O_3 formation tends to increase with increasing air temperatures.

[3] Trees also directly affect particulate matter in the atmosphere by intercepting particles, emitting particles (e.g. pollen) and resuspending particles captured on the plant surface. Some particles can be absorbed into the tree, though most intercepted particles are retained on the plant surface. Many of the particles that are intercepted are eventually resuspended back to the atmosphere, washed off by rain, or dropped to the ground with leaf and twig fall. During dry periods, particles are constantly intercepted and resuspended, in part, dependent upon wind speed. During precipitation, particles can be washed off and either dissolved or transferred to the soil. Consequently, vegetation is only a temporary retention site for many atmospheric particles, though the removal of gaseous pollutants is more permanent as the gases are often absorbed and transformed within the leaf interior.

magnitude of the effect of trees on pollution removal or concentrations, while only a limited number of studies have looked at the estimated health effects of pollution removal by trees. In the United Kingdom, woodlands are estimated to save between 5 and 7 deaths, and between 4 and 6 hospital admissions per year due to reduced pollution by SO_2 and particulate matter less than 10 microns (PM_{10}) (Powe and Willis 2004). Modelling for London estimates that 25% tree cover removes 90.4 metric tons of PM_{10} pollution per year, which equates to a reduction of 2 deaths and 2 hospital stays per year (Tiwary et al. 2009). Nowak et al. (2013) reported that the total amount of particulate matter less than 2.5 microns ($PM_{2.5}$) removed annually by trees in 10 US cities in 2010 varied from 4.7 t in Syracuse to 64.5 t in Atlanta. Estimates of the annual monetary value of human health effects associated with $PM_{2.5}$ removal in these same cities (e.g. changes in mortality, hospital admissions, respiratory symptoms) ranged from $1.1 million in Syracuse to $60.1 million in New York City. Mortality avoided was typically around 1 person per year per city, but was as high as 7.6 people per year in New York City.

Trees and forests in the conterminous US removed 22.4 million t of air pollution in 2010 (range: 11.1–31.0 million t), with human health effects valued at US$ 8.5 billion (range: $2.2–15.6 billion). Most of the pollution removal occurred in rural areas, while most of the health impacts and values were within urban areas. Health impacts included the avoidance of more than 850 incidences of human mortality. Other substantial health benefits included the reduction of more than 670 000 incidences of acute respiratory symptoms (range: 221 000–1 035 000), 430 000 incidences of asthma exacerbation (range: 198 000–688 000) and 200 000 days of school loss (range: 78 000–266 000) (Nowak et al. 2014).

Though the amount of air pollution removed by trees may be substantial, the per cent air quality improvement in an area will depend upon on the amount of vegetation and meteorological conditions. Air quality improvement by trees in cities during daytime of the in-leaf season averages around 0.51% for particulate matter, 0.45% for O_3, 0.44% for SO_2, 0.33% for NO_2, and 0.002% for CO. However, in areas with 100% tree cover (i.e. contiguous forest stands), air pollution improvement is on an average around four times higher than city averages, with short-term improvements in air quality (1 hour) as high as 16% for O_3 and SO_2, 13% for particulate matter, 8% for NO_2, and 0.05% for CO (Nowak et al. 2006).

3) Emission of chemicals

Vegetation, including trees, can emit various chemicals that can contribute to air pollution. Because some vegetation, particularly urban vegetation, often requires relatively large inputs of energy for maintenance activities, the resulting emissions need to be considered. The use and combustion of fossil fuels to power this equipment leads to the emission of chemicals such as VOCs, CO, NO_2 and SO_2, and particulate matter (US EPA 1991).

Plants also emit VOCs (e.g. isoprene, monoterpenes) (Geron et al. 1994; Guenther 2002; Nowak et al. 2002; Lerdau and Slobodkin 2002). These compounds are natural chemicals that make up essential oils, resins and other plant products, and may be useful in attracting pollinators or repelling predators. Complete oxidation of VOCs ultimately produces carbon dioxide (CO_2), but CO is an intermediate compound in this process. Oxidation of VOCs is an important component of the global CO budget (Tingey et al. 1991); CO also can be released from chlorophyll degradation (Smith 1990). VOCs emitted by trees can also contribute to the formation of O_3. Because VOC emissions are temperature dependent and trees generally lower air temperatures, increased tree cover can lower overall VOC emissions and, consequently, O_3 levels in urban areas (e.g. Cardelino and Chameides 1990). Ozone inside leaves can also be reduced due to the reactivity with biogenic compounds (Calfapietra et al. 2009).

Trees generally are not considered as a source of atmospheric NO_x, though plants, particularly agricultural crops, are known to emit ammonia.

Emissions occur primarily under conditions of excess nitrogen (e.g. after fertilization) and during the reproductive growth phase (Schjoerring 1991). They can also make minor contributions to SO_2 concentration by emitting sulfur compounds such as hydrogen sulfide (H_2S) and SO_2 (Garsed 1985; Rennenberg 1991). H_2S, the predominant sulfur compound emitted, is oxidized in the atmosphere to SO_2. Higher rates of sulfur emission from plants are observed in the presence of excess atmospheric or soil sulfur. However, sulfur compounds also can be emitted with a moderate sulfur supply (Rennenberg 1991). In urban areas, trees can additionally contribute to particle concentrations by releasing pollen and emitting volatile organic and sulfur compounds that serve as precursors to particle formation. From a health perspective, pollen particles can lead to allergic reactions (e.g. Cariñanosa et al. 2014).

3.1 Overall effect of vegetation on air pollution

There are many factors that determine the ultimate effect of vegetation on pollution. Many plant effects are positive in terms of reducing pollution concentrations. For example, trees can reduce temperatures and thereby reduce emissions from various sources, and they can directly remove pollution from the air. However, the alteration of wind patterns and speeds can affect pollution concentrations in both positive and negative ways. In addition, plant compound emissions and emissions from vegetation maintenance can contribute to air pollution. Various studies on O_3, a chemical that is not directly emitted but rather formed through chemical reactions, have helped to illustrate the cumulative and interactive effects of trees.

One model simulation illustrated that a 20% loss in forest cover in the Atlanta area due to urbanization led to a 14% increase in O_3 concentrations for a day (Cardelino and Chameides 1990). Although there were fewer trees to emit VOCs, an increase in Atlanta's air temperatures due to the increased urban heat island, which occurred concomitantly with tree loss, increased VOC emissions from the remaining trees and other sources (e.g. automobiles), and altered O_3 chemistry such that concentrations of O_3 increased. Another model simulation of California's South Coast Air Basin suggests that the air quality impacts of increased urban tree cover may be locally positive or negative with respect to O_3. However, the net basinwide effect of increased urban vegetation is a decrease in O_3 concentrations if the additional trees are low VOC emitters (Taha 1996).

Modelling the effects of increased urban tree cover on O_3 concentrations from Washington, DC to central Massachusetts revealed that urban trees generally reduce O_3 concentrations in cities, but tend to slightly increase average O_3 concentrations regionally (Nowak et al. 2000). Modelling of the New York City metropolitan area also revealed that increasing tree cover by 10% within urban areas reduced maximum O_3 levels by about 4 ppb, which was about 37% of the amount needed for attainment (Luley and Bond 2002).

4. The role of plant biodiversity in regulating air quality

The impacts of vegetation on air quality depend in part on species and other aspects of plant biodiversity. Plant biodiversity in an area is influenced by a mix of natural and anthropogenic factors that interact to produce the vegetation structure. Natural influences include native vegetation types and abundance, natural biotic interactions (e.g. seed dispersers, pollinators, plant consumers), climate factors (e.g. temperature, precipitation), topographic moisture regimes, and soil types. Superimposed on these natural systems in varying degrees is an anthropogenic system that includes people, buildings, roads, energy use and management decisions. The management decisions made by multiple disciplines within an urban system can both directly (e.g. tree planting, removal, species introduction, mowing, paving, watering, use of herbicides and fertilizers) and indirectly (e.g. policies and funding related to vegetation and development) affect vegetation structure and biodiversity. In addition, the anthropogenic system alters the environment (e.g. changes in air temperature and solar radiation, air pollution, soil compaction) and can induce changes in vegetation structure (Nowak 2010).

Much is generally known about plant distribution globally, but less is known about factors that affect the distribution of plant diversity and human influences on plant biodiversity (Kreft and Jetz 2007). Variations in urban tree cover across regions and within cities give an indication of the types of factors that can affect urban tree structure and consequently biodiversity, with resulting impacts on human health. One of the dominant factors affecting tree cover in cities is the natural characteristics of the surrounding region. For example, in forested areas of the US, urban tree cover averages 34%. Cities within grassland areas average 18% tree cover, while cities in desert regions average only 9% tree cover (Nowak et al. 2001). Cities in areas conducive to tree growth naturally tend to have more tree cover, as non-managed spaces tend to naturally regenerate with trees. In forested areas, tree cover is often specifically excluded by design or management activities (e.g. impervious surfaces, mowing). In the US, while the per cent tree cover nationally in urban (35.0%) and rural areas (34.1%) are comparable, urbanization tends to decrease overall tree cover in naturally forested areas, but increase tree cover in grassland and desert regions (Nowak and Greenfield 2012).

In urban areas, land use, population density, management intensity, human preferences and socioeconomic factors can affect the amount of tree cover and plant diversity (Nowak et al. 1996; Hope et al. 2003; Kunzig et al. 2005). These factors are often interrelated and create a mosaic of tree cover and species across the city landscape. Land use is a dominant factor affecting tree cover (Table 1). However, land use can also affect species composition, as non-managed lands (e.g. vacant) tend to be dominated by natural regeneration of native and exotic species. Within areas of managed land use, the species composition tends to be dictated by a combination of human preferences for certain species (tree planting) and how much land is allowed to naturally regenerate (Nowak 2010).

Tree diversity, represented by the common biodiversity metrics of species richness (number of species) and the Shannon–Wiener diversity index (Barbour et al. 1980), varies among and within cities and through time. Based on field sampling of various cities in North America (Nowak et al. 2008; Nowak 2010), species richness varied from 37 species in Calgary, Alberta, Canada, to 109 species in Oakville, Ontario, Canada (Figure 1). Species diversity varied from 1.6 in Calgary to 3.8 in Washington, DC (Figure 2). The species richness in all cities is greater than the average species richness in eastern US forests by county (26.3) (Iverson and Prasad 2001). Species diversity

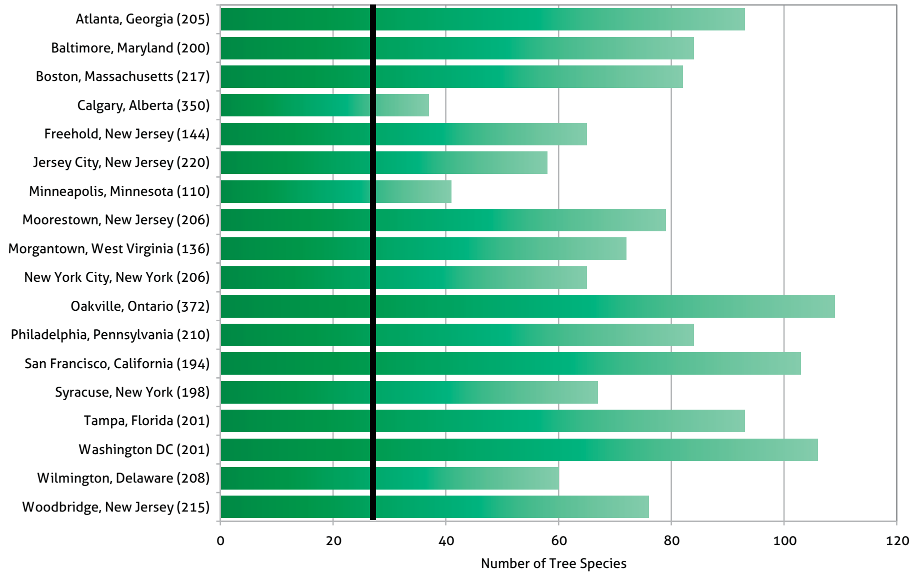
Table 1: Mean per cent tree cover and standard error (SE) for US cities with different potential natural vegetation types (forest, grassland, desert) by land use (from Nowak et al. 1996)

	Forest		Grassland		Desert	
Land use	Mean	SE	Mean	SE	Mean	SE
Park	47.6	5.9	27.4	2.1	11.3	3.5
Vacant/wildland	44.5	7.4	11.0	2.5	0.8	1.9
Residential	31.4	2.4	18.7	1.5	17.2	3.5
Institutional	19.9	1.9	9.1	1.2	6.7	2.0
Other[1]	7.7	1.2	7.1	1.9	3.0	1.3
Commercial/industrial	7.2	1.0	4.8	0.6	7.6	1.8

[1]Includes agriculture, orchards, transportation (e.g., freeways, airports, shipyards), and miscellaneous.

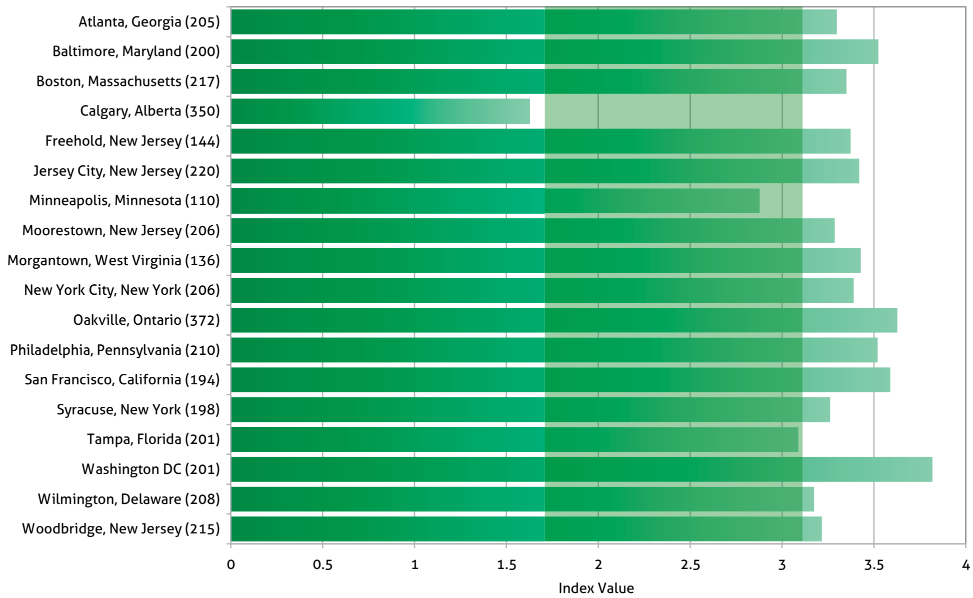
in these urban areas is also typically greater than found in eastern US forests (Barbour et al. 1980). Tree species diversity and richness is enhanced in urban areas compared with surrounding landscapes and/or typical forest stands, as native species richness is supplemented with species introduced by urban inhabitants or processes. People often plant trees in urban areas to improve aesthetics and/or the physical or social environment. Some non-native species can be introduced via transportation corridors or escape from cultivation (e.g. Muehlenbach 1969; Haigh 1980).

FIGURE 1: Species richness and values for tree populations in various cities. Numbers in parentheses are sample size based on 0.04 hectare plots. (A) Dark line indicates average species richness in eastern US forests by county (26.3).



Source: Nowak 2010

(B) Shannon–Wiener Diversity Index values. Shaded area indicates typical range of diversity values for forests in the eastern US (1.7–3.1).



Source: Nowak 2010

One of the most important vegetation attributes in relation to air quality is the amount of leaf area. Leaf area varies by plant form, with leaf area indices (m^2 leaf surface area per m^2 ground) of agricultural areas typically around 3–5 and leaf area indices of forests typically between 5 and 11 (Barbour et al. 1980). Thus, the magnitude and distribution of vegetation types (e.g. grasses, shrubs, trees) affect air quality. In general, plant types with more leaf area or leaf biomass have a greater impact, either positive or negative, on air quality.[4]

The second most important attribute related to air quality is vegetation configuration or design. Though reduction in wind speeds can increase local pollution concentrations due to reduced dispersion of pollutants and mixing height of the atmosphere, altering of wind patterns can also have a potential positive effect. Tree canopies can potentially prevent pollution in the upper atmosphere from reaching ground-level air space. Measured differences in O_3 concentration between above- and below-forest canopies in California's San Bernardino mountains have exceeded 50 ppb (40% improvement) (Byternowicz et al. 1999). Under normal daytime conditions, atmospheric turbulence mixes the atmosphere such that pollutant concentrations are relatively consistent with height. Forest canopies can limit the mixing of upper air with ground-level air, leading to below-canopy air quality improvements. However, where there are numerous pollutant sources below the canopy (e.g. automobiles), the forest canopy could increase concentrations by minimizing the dispersion of the pollutants away at ground level. This effect could be particularly important in heavily treed areas near roadways (Gromke and Ruck 2009; Wania et al. 2012; Salmond et al. 2013; Vos et al. 2013). However, standing in the interior of a forest stand can offer cleaner air if there are no local ground sources of emissions (e.g. from automobiles). Various studies have illustrated reduced pollutant concentrations in the interior of forest stands compared to the outside of the forest stands (e.g. Dasch 1987; Cavanagh et al. 2009).

The biodiversity of plant types within an area affects the total amount of leaf area and the vegetation design. Following biodiversity related to plant form, species diversity also affects air quality, as different species have different effects based on species characteristics. In general, species with larger growth forms and size at maturity have greater impacts, either positive or negative, on air quality. The following are the types of air quality impacts that can be affected by species and therefore species diversity:

Pollution removal: in addition to total leaf area of a species, species characteristics that affect pollution removal are tree transpiration and leaf characteristics. Removal of gaseous pollutants is affected by tree transpiration rates (gas exchange rates). As actual transpiration rates are highly variable, depending upon site or species characteristics, limited data exist on transpiration rates for various species under comparable conditions. However, relative transpiration factors for various species can be gauged from estimated monthly water use (Costello and Jones 1994). Particulate matter removal rates vary depending upon leaf surface characteristics. Species with dense and fine textured crowns and complex, small and rough leaves would capture and retain more particles than open and coarse crowns, and simple, large, smooth leaves (Little 1997; Smith 1990). Species ranking of trees in relation to pollution removal are estimated in i-Tree Species (www.itreetools.org). In addition, evergreen trees provide for year-round removal of particles.

VOC emissions: emission rates of VOCs vary by species (e.g. Geron et al. 1994; Nowak et al. 2002). Nine tree genera that have the highest standardized isoprene emission rate, and therefore the greatest relative effect on increasing O_3, are beefwood (*Casuarina* spp.), *Eucalyptus* spp., sweetgum (*Liquidambar* spp.), black gum (*Nyssa* spp.), sycamore (*Platanus* spp.), poplar (*Populus*

[4] Within forests, leaf area also varies with tree age/size, with large healthy trees greater than 30 inches in stem diameter in Chicago having approximately 60–70 times more leaf area than small healthy trees less than 3 inches in diameter (Nowak 1994).

spp.), oak (*Quercus* spp.), black locust (*Robinia* spp.) and willow (*Salix* spp.). However, due to the high degree of uncertainty in atmospheric modelling, results are currently inconclusive as to whether these genera contribute to an overall net formation of O_3 in cities (i.e. O_3 formation from VOC emissions is greater than O_3 removal).

Pollen: not only do pollen emissions and phenology of emissions vary by species, but pollen allergenicity also varies by species. Examples of some of the most allergenic species are *Acer negundo* (male), *Ambrosia* spp., *Cupressus* spp., *Daucus* spp., *Holcus* spp., *Juniperus* spp. (male), *Lolium* spp., *Mangifera indica*, *Planera aquatica*, *Ricinus communis*, *Salix alba* (male), *Schinus* spp. (male) and *Zelkova* spp. (Ogren 2000).

Air temperature reduction: similar to gaseous air pollution removal, species effects on air temperatures vary with leaf area and transpiration rates. Leaf area affects tree shading of ground surfaces and also overall transpiration. Transpiration from the leaves helps to provide evaporative cooling. Both the shade and evaporative cooling, along with effects on wind speed, affect local air temperature and therefore pollutant emission and formation.

Building energy conservation: although the effects of trees on building energy use is dependent upon a tree's position (distance and direction) relative to the building, tree size also plays a role on building energy effects (McPherson and Simpson 2000). Changes in building energy use affect pollutant emission from power plants.

Maintenance needs: like building energy conversation, species maintenance needs have a secondary effect on air quality. Plant species with greater maintenance needs typically require more human interventions (planting, pruning, removal) that utilize fossil fuel-based equipment (e.g. cars, lawn mowers, chain saws). The more fossil fuel-based equipment is used, the more pollutant emissions are produced. Plant attributes that affect maintenance needs include not only plant adaptation to site conditions but also plant life span (e.g. shorter lived species require more frequent planting and removal).

Pollution sensitivity: sensitivity to various pollutants vary by plant species. For example, *Populus tremuloides* and *Poa annua* are sensitive to O_3, but *Tilia americana* and *Dactylis glomerata* are resistant. Pollutant sensitivity to various species is given in Smith and Levenson (1980).

5. Impacts of air quality on plant communities

Air pollution can affect tree health. Some pollutants under high concentrations can damage leaves (e.g. SO_2, NO_2, O_3), particularly of pollutant-sensitive species. For NO_2, visible leaf injury would be expected at concentrations around 1.6–2.6 ppm for 48 hours, 0 ppm for 1 hour, or a concentration of 1 ppm for as many as 100 hours (Natl. Acad. of Sci. 1977a). Concentrations that would induce foliage symptoms would be expected only in the vicinity of an excessive industrial source (Smith 1990).

Eastern deciduous species are injured by exposure to O_3 at 0.20–0.30 ppm for 2–4 hours (Natl. Acad. of Sci. 1977b). The threshold for visible injury of eastern white pine is approximately 0.15 ppm for 5 hours (Costonis 1976). Sorption of O_3 by white birch seedlings shows a linear increase up to 0.8 ppm; for red maple seedlings the increase is up to 0.5 ppm (Townsend 1974). Severe O_3 levels in urban areas can exceed 0.3 ppm (Off. Technol. Assess. 1989). Injury effects can include altered photosynthesis, respiration, growth and stomatal function (Lefohn et al. 1988; Shafer and Heagle 1989; Smith 1990).

Toxic effects of SO_2 may be due to its acidifying influence and/or the sulfite (SO_3^{2-}) and sulfate (SO_4^{2-}) ions that are toxic to a variety of biochemical processes (Smith 1990). Stomata may exhibit increases in either stomatal opening or stomatal closure when exposed to SO_2 (Smith 1984; Black 1985). Acute SO_2 injury to native vegetation does not occur below 0.70 ppm for 1 hour or 0.18 ppm for 8 hours (Linzon 1978). A concentration of 0.25 ppm for several hours may injure some species

(Smith 1990). Indirect anthropogenic effects can alter species composition. For example, in a natural park in Tokyo, Japanese red pine (*Pinus densiflora*) was dying and being successionally replaced with broad-leaved evergreen species (Numata 1977). This shift in species composition has been attributed to SO_2 air pollution, with the broad-leaved species being more resistant to air pollution.

Particulate trace metals can be toxic to plant leaves. The accumulation of particles on leaves also can reduce photosynthesis by reducing the amount of light reaching the leaf. Damage to plant leaves can also occur from acid rain (pH <3.0). Acid rain and air pollution (NAPAP 1991) can be a source of the essential plant nutrients of sulfur and nitrogen, but also can reduce soil nutrient availability through leaching or toxic soil reactions. Particles can also affect tree pest/disease populations. Given the pollution concentration in most cities, these pollutants would not be expected to cause visible leaf injury, but could in cities or areas with high pollutant concentrations.

6. Bioindicators

A bioindicator is a quality of an organism, population, community or ecosystem used for indicating the health or status of the surrounding environment. Bioindicators, especially lichens and bryophytes, are widely used for monitoring air quality. The benefits of direct measurements of air quality include long-term integration of pollution levels over time and lower operational costs (often by orders of magnitude per study site). Biodiversity metrics, such as the number of sensitive species, relative abundance of functional groups, or genotypic frequencies, for example, are successfully employed for air quality biomonitoring in many nations (Markert et al. 1996; Aničić et al. 2009; Cao et al. 2009). Measuring pollutant concentrations in lichen and bryophyte tissues is another means of air quality mapping (Augusto et al. 2007; Augusto et al. 2010; Liu et al. 2011; Root et al. 2013). Most studies focus on environmental health (i.e. evaluating pollutant-mediated harms to the natural environment) to guide land management and air quality regulation (Hawksworth and Rose 1970; Cape et al. 2009; Geiser et al. 2010). Health and bioindicator experts often suggest utilizing bioindicators in public health assessments to overcome the lack of systematic air quality measurements from instrumented monitoring networks and for detecting chronic low levels of pollution below the detection limits of monitoring instruments (Brauer 2010; Augusto et al. 2012). Tissue-based bioindicators enable high spatial resolution mapping of toxic pollutants that are not frequently measured by instrumented networks. Nonetheless, it is rare for research to actually integrate bioindicator and public health data.

UNITED NATIONS PHOTO / FLICKR

Taking cues from the environment to assess air quality is a relatively old science. Lichens were first described as "health meters for the air" in 1866, when a Finnish botanist noted that certain species were restricted to a large city park in Paris (Nylander 1866). While many organisms exhibit a measurable response to pollution, lichen and bryophytes (i.e. mosses and liverworts) are the most widely utilized bioindicators in both environmental and human health studies. Lichen and bryophytes lack root structures and the capacity to store water, creating a dependence on moisture and nutrients scavenged from the atmosphere. By also lacking a protective cuticle, they absorb water and contaminants much like a sponge.

Biodiversity-based indices, including richness, relative abundance or dominance of sensitive lichen and bryophyte species are commonly used for mapping deposition of nitrogen (N)- and sulphur

(S)-containing pollutants. Species' sensitivities to H_2S, SO_2, acidic deposition, HNO_3, NH_3, NO_y, and the N- and S-containing aerosols have been well established through field studies and controlled fumigation experiments (Riddell et al. 2008; Riddell et al. 2012). Biodiversity indices usually correlate well with instrumented measurements of pollutant deposition (Gadsdon et al. 2010; Jovan et al. 2012), although some indices are intentionally non-specific, meaning they are not calibrated to track specific pollutants. In this case, biodiversity measures are interpreted as an integrated response to 'air quality' in general (Castro et al. 2014), which may provide a useful representation of human exposure as the human body integrates pollution from multiple sources.

Nitrogen, S, as well as metals (Wolterbeek 2002), radionuclides (Seaward 2002) and persistent organic pollutants (POPs) like polycyclic aromatic hydrocarbons (PAHs), dioxins and furans (PCDD/Fs), polychlorobiphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) (Augusto et al. 2013; Harmens et al. 2013) accumulate over time in lichen and bryophyte tissues, allowing their use as in-situ passive deposition monitors. Lichens and bryophytes tolerate exposure to many non-nutrient pollutants (e.g. heavy metals, radionuclides, POPs), and so typical biodiversity-based indices cannot be utilized for this group.

6.1 Air quality bioindicators: ways forward

There is clearly great potential for utilizing bioindicators in human health research; yet few scientists have done so. The use of bioindicator data in health studies has barely been explored, despite potential to overcome some of the most persistent data gaps in public health research on air quality. This potential can be explained by the fact that obtaining spatially and temporally representative air quality measurements is one of the most pervasive issues in health studies (Brauer 2010; Ribeiro et al. 2010) yet, for the most part, health research utilizes bioindicator maps tangentially or not at all. Bioindicators have the advantage of being living organisms and thus biologically reflecting the environment where they are growing. This information is not likely to be obtained through other monitoring methods, which solely represent physicochemical measures of pollutants. None of these research barriers are insurmountable. The main issue appears to be bringing together the right mix of skills. The proposed ways forward include the following:

Cross-sectoral collaboration is needed to foster information exchange and collaboration between bioindicator specialists and public health scientists. There is little crossover in the professional activities of these groups at present, and interdisciplinary workshops and meetings could further reduce this gap.

Future research should highlight the need to calibrate bioindicators with existing air monitoring stations or passive samplers, which are more flexible. While expensive to collect, investment in calibration data will facilitate the use of pollutant thresholds in bioindicator maps and also help define what time frame the bioindicator reflects, including how seasonal variations or sudden pollution episodes contribute to bioindicator values. Even if causality or mechanism cannot be established, an affordable bioindicator with the capacity to predict human health outcomes remains valuable for further research.

For large health research institutions, maintaining staff dedicated to data dissemination is critical for enabling access to detailed personal public health data. These intermediaries often help, for instance, by spatially joining bioindicator and health data, to keep confidential addresses for private residences.

Research that utilizes and cross-links resources that are already available, such as high-resolution maps from air quality and public health monitoring studies, should be encouraged. Also, lichens and bryophytes form the backbone of large-scale air quality monitoring programmes in both Europe (the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests operating under the UNECE Convention on Long-range Transboundary Air Pollution) and the US (the US Department of Agriculture's Forest Inventory and Analysis Program).

7. Knowledge gaps and ways forward

There are numerous gaps in knowledge related to biodiversity, including plant biodiversity (species-specific effects) and air quality. As there are numerous species globally, these gaps are felt across the world. However, leaf area is the dominant characteristic that affects many aspects of air quality. Thus, general magnitudes of impact can be assessed among plant communities based on leaf area. The individual species effects are most important in determining variations within plant communities, understanding the impacts of biodiversity and guiding vegetation management. There are gaps in all aspects of plant species effects on air quality, but some of the better-researched aspects are related to VOC emissions, which are species or genera dependent. Estimates and comparisons of pollen allergenicity among plant species also exist (e.g. Pettyjohn and Levetin 1997; Ogren 2000; Cariñanosa et al. 2014). One of the least understood aspects related to individual species characteristics and air quality effects relates to species-specific removal rates (deposition velocities) for various pollutants. In addition, while there are various studies relating air pollution to human health, there are few studies relating vegetation impacts to pollution concentrations and human health effects.

To facilitate air quality improvements through biodiversity and management of vegetation, there are various steps that managers and policy-makers could take. The first step could be to assess the local species composition and biodiversity as a basic foundation for understanding the local vegetation structure. The second could be to assess what impacts this current vegetation structure has on air quality (e.g. estimating pollution removal, VOC emissions, impacts on building energy conservation and emissions, etc). To aid in understanding the vegetation ecosystem services, various models exist (e.g. i-Tree). Policy-makers could also facilitate increased research to better understand the effects and impact of individual species on air quality.

Local vegetation management decisions can help improve air quality. Vegetation management strategies to help improve air quality include the following:

- Increase the amount of healthy vegetation (increases pollution removal).

- Sustain the existing vegetation cover (maintains pollution removal levels).

- Maximize the use of low VOC-emitting species (reduces O_3 and CO formation).

- Sustain large, healthy trees (large trees have greater per-tree effects).

- Use long-living tree species (reduces long-term pollutant emissions from planting and removal).

- Use low-maintenance species (reduces pollutant emissions from maintenance activities).

- Reduce fossil fuel use in maintaining vegetation (reduces pollutant emissions).

- Plant trees in energy-conserving locations (reduces pollutant emissions from power plants).

- Plant trees to shade parked cars (reduces vehicular VOC emissions).

- Supply ample water to vegetation (enhances pollution removal and temperature reduction).

- Plant vegetation in polluted or heavily populated areas (maximizes pollution removal and air quality benefits; however, specific vegetation designs need to be considered so that they do not increase local pollutant concentrations, such as near roadways).

- Avoid pollutant-sensitive species (improves plant health).

- Utilize evergreen species for particulate matter (year-round removal of particles).

Through proper design and management, plant systems and biodiversity can be utilized to enhance air quality and provide numerous other ecosystem services, and consequently improve the health and well-being of people and ecosystems across the globe.

Connecting Global Priorities: Biodiversity and Human Health

A State of Knowledge Review

UNEP

Convention on Biological Diversity

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