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Research Paper

Where the people are: Current trends and future potential targeted investments in urban trees for PM_{10} and temperature mitigation in 27 U.S. Cities

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ABSTRACT

Urban trees reduce respirable particulate matter (PM_{10}) concentrations and maximum daytime summer temperatures. While most cities are losing tree cover, some are considering ambitious planting efforts. Maximizing PM_{10} and heat mitigation for people from such efforts requires cost-effective targeting. We adapt published methods to estimate the impact of a decade (2004–2014) of tree cover change on city-level PM_{10} and heat mitigation in 27 U.S. cities and present a new methodology for estimating local-level PM_{10} and heat mitigation by street trees and tree patches. We map potential tree planting sites in the 27 cities and use our local-level PM_{10} and heat mitigation methods to assess the population-weighted return on investment (ROI) of each site for PM_{10} and heat abatement for nearby populations. Twenty-three of the 27 cities lost canopy cover during 2004–2014, reducing estimated city-level PM_{10} removal by 6% (168 tons) and increasing city-level average maximum daily summer temperature by 0.1 °C on average across cities. We find large potential for urban reforestation to increase PM_{10} and heat abatement. Intra-city variation in planting site ROI – driven primarily by differences in population density around planting sites – exceeds four orders of magnitude, indicating large scope for targeting to increase PM_{10} and heat abatement from reforestation. Reforesting each city's top 20% ROI sites could lower average annual PM_{10} concentrations by > 2 μg/m^3 for 3.4–11.4 million people and average maximum daily summer temperatures by > 2 °C for 1.7–12.7 million – effects large enough to provide meaningful health benefits – at a combined annual cost of $102 million.

1. Introduction

Large-scale increases in urban vegetation cover have been suggested as an important strategy for mitigating urban summer temperature extremes (Akbari, Pomerantz, & Taha, 2001; Taha, Konopacki, & Gabersek, 1999) and reducing premature heat-related mortality (Stone et al., 2014). Strategic deployment of urban vegetation additions also is considered highly effective for reducing local airborne particulate matter (PM) concentrations (Maher, Ahmed, Davison, Karloukovski, & Clarke, 2013; Pugh, Mackenzie, Whyatt, & Hewitt, 2012) and enhancing the PM-related mortality and morbidity reductions urban tree cover provides (Nowak, Hirabayashi, Bodine, & Hoehn, 2013). Indeed, urban planners increasingly consider vegetation management for air pollution and heat mitigation (Andersson-Sköld et al., 2015), though rarely in a systematic manner that maximizes population-weighted mitigation per dollar spent. Yet, despite the increasing recognition of the pollution and heat mitigating effects of urban trees, many cities in the U.S. are experiencing declines in urban tree canopy (Nowak & Greenfield, 2012), potentially impacting PM concentrations and air temperature.

In this paper, we quantify how current trends in urban tree canopy cover have impacted urban air quality and summer temperatures, and assess the potential for targeted investments in tree cover gains to improve these two parameters. In this section, we present basic background information on what is known about urban tree canopy mitigation of PM and temperature, and then present our specific research goals.

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1.1. Urban tree effects on particulate matter

Airborne particulate matter is widely considered the most pressing urban air quality problem globally and is a key factor in climate change (Fuzzi et al., 2015). In U.S. urban outdoor environments, the main contributors to ambient concentrations of respirable particles (< 10 µm in aerodynamic diameter, or PM_{10}) generally are agriculture, traffic (fuel combustion; road, tire and break wear), household fuel burning and powerplant and industrial emissions (Bauer, Tsigaridis, & Miller, 2016; Hopke et al., 2006; Karagulian et al., 2015; Sowlat, Hasheminassab, & Sioutas, 2016), from local, regional and long-range sources (Karnae & Kuruvilla, 2011; Kundu & Stone, 2014; Levy & Hanna, 2011). In the western U.S., wildfires can be a seasonally important source as well (Spracklen et al., 2009). Due to the high particle infiltration and indoor penetration factors of PM_{10} (Chen & Zhao, 2011), outdoor PM_{10} also is a major contributor to indoor PM_{10} concentrations (Almeida et al., 2011; Rovelli et al., 2014).

While coarse particles between 2.5 and 10 µm in aerodynamic diameter ($PM_{10\text{-}2.5}$) are suspected to have similar health effects, evidence of increased cardiovascular and respiratory morbidity and mortality from short-term and long-term exposure is strongest for fine particles ($PM_{2.5}$) (Correia et al., 2013; Hoek et al., 2013; Thurston et al., 2016; U.S. EPA, 2009) with specific pathways for cardiovascular effects recently demonstrated (Kaufman et al., 2016). In fact, $PM_{2.5}$ is considered the dominant contributor to the health burden from outdoor air pollution both in the U.S. (Fann et al., 2012) and globally, being responsible for an estimated 3.2 million (M) premature deaths per year worldwide (Lim & et al., 2012). $PM_{2.5}$ has been linked to neurodegenerative disease and Alzheimer's disease (Jung, Lin, & Hwang, 2015; Maher et al., 2016), and PM_{10} concentrations have a significant negative effect on worker productivity (Chang, Graff Zivin, Gross, & Neidell, 2016) and cognitive performance and development in children, resulting in lower academic test scores and reduced lifetime earnings potential (Clark-Reyna, Grineski, & Collins, 2016; Dadvand et al., 2015; Freire et al., 2010; Lavy, Ebenstein, & Roth, 2014; Suglia, Gryparis, Wright, Schwartz, & Wright, 2008).

Over the last 30 years, regulatory controls have achieved large reductions in urban PM concentrations (Darlington, Kahlbaum, Heuss, & Wolff, 1997; Hand, Schichtel, Malm, & Frank, 2014). However, the future trajectory of PM concentrations is uncertain (Dawson, Bloomer, Winner, & Weaver, 2014; Penrod, Zhang, Wang, Wu, & Leung, 2014; Trail et al., 2014; Val Martin et al., 2015) and is a function of expected increases in biogenic (Keywood et al., 2013; Spracklen et al., 2009) emissions, decreases in anthropogenic PM emissions (U.S. EPA, 2015) and climate change-driven changes in meteorology (Dawson et al., 2014). One study predicted moderate though significant increases in premature mortality from outdoor $PM_{2.5}$ pollution in U.S. urban areas (Lelieveld, Evans, Fnais, Giannadaki, & Pozzer, 2015).

Vegetation removes PM of all size classes through dry deposition to leaf, stem and branch surfaces (Litschke & Kuttler, 2008). Removal efficiencies of plants for different particle size fractions depend on leaf (and to a lesser extent, bark) characteristics and can differ by an order of magnitude among species (Freer-Smith, El-Khatib, & Taylor, 2004; Mitchell, Maher, & Kinnersley, 2010; Mo et al., 2015; Sæbø et al., 2012; Yang, Chang, & Yan, 2015), with species with higher leaf surface micro roughness better able to capture fine particles (El-Khatib, Abd El-Rahman, & Elsheikh, 2011; Wang, Liu, Gao, Hasi, & Wang, 2006). Much of the intercepted $PM_{2.5}$ becomes permanently incorporated into leaf wax or cuticle, while a portion of $PM_{10\text{-}2.5}$ is resuspended as a function of wind speed (Nicholson, 1988; Nowak et al., 2013) with the remainder eventually washed off to the ground by precipitation (Freer-Smith, Beckett, & Taylor, 2005; Matzka & Maher, 1999; Mitchell et al., 2010; Przybysz, Sæbø, Hanslin, & Gawronski, 2014).

The PM removal effect of plants has been documented in leaf and plant-level PM deposition studies (e.g. Freer-Smith et al., 2005; Liu et al., 2013; Mitchell et al., 2010; Mo et al., 2015), plant- and greenspace-level PM concentration reduction studies (e.g., Cavanagh, Zawar-Rezab, & Wilson, 2009; Cowherd, Muleski, & Gebhardt, 2006; Maher et al., 2013; Tiwary, Reff, & Colls, 2008; Zhu et al., 2011), and by land use regression and other spatial-statistical analyses examining the relationship between PM concentrations and urban forests or other green areas (Eeftens et al., 2012; Irga, Burchett, & Torpy, 2015; Ross, Jerrett, Ito, Tempalski, & Thurston, 2007; Wu, Xie, Li, & Li, 2015; Xu et al., 2016).

Some modeling studies have shown that dense canopies in high-emission street canyons under certain conditions can increase local PM concentrations if their aerodynamic effect (reduced ventilation and thus pollutant dispersion) exceeds their pollutant removal effect (Janhäll, 2015; Pugh et al., 2012; Vos, Maiheu, Vankerkom, & Janssen, 2013). However, which of the two effects prevails depends on the microscale (leaf-level) deposition velocity used in the modeling (Santiago, Martilli, & Martin, 2017), which in some studies may be set unrealistically low (Maher et al., 2013; Santiago et al., 2017). Empirical evidence shows that such pollution "trapping" by dense streetside canopies may (Chen et al., 2015) or may not (Hofman, Stokkaer, Snauwaert, & Samson, 2013) occur at heights where human exposure occurs. Roadside treed areas experiencing high pedestrian use therefore should be managed for less dense canopy crowns, which facilitate pollutant dispersion while still achieving high PM removal efficiency (Maher et al., 2013).

1.2. Urban tree effects on summer maximum daily air temperature

Near-surface temperatures in cities generally significantly exceed those in surrounding rural areas (Arnfield, 2003; Bounoua et al., 2015; Imhoff, Zhang, Wolfe, & Bounoua, 2010). This "urban heat island" (UHI) phenomenon is caused primarily by two factors: the replacement of vegetation with impervious surfaces that have higher heat absorption capacity and produce no evaporative cooling; and the release of waste heat from anthropogenic processes like motor vehicle operation and air conditioning (Arnfield, 2003). The intensity of the UHI effect is significantly positively related to both impervious surface fraction and city size (Bounoua et al., 2015; Imhoff et al., 2010).

While the UHI effect is observed in both summer and winter (Bounoua et al., 2015; Zhang, Imhoff, Wolfe, & Bounoua, 2010), it is of primary concern for human health in summer when it exacerbates already-high temperatures. Both heat waves and individual extreme and moderate heat days have significant short-term effects on all-cause and cardiovascular mortality and on morbidity (Andersen & Bell, 2011; Bobb, Peng, Bell, & Dominici, 2014; Gasparrini et al., 2015; Gosling, Lowe, McGregor, Pelling, & Malamud, 2009; Kingsley, Eliot, Gold, Vanderslice, & Wellenius, 2016; Ye et al., 2012).

In recent decades, the size of the heat-related mortality effect has been declining in the U.S. (Bobb et al., 2014) despite rising urban temperatures (Habeeb, Vargo, & Stone, 2015). This reduction is due to increased use of adaptive responses to extreme heat (e.g., heat advisories, behavior modification, increased penetration of air conditioning, natural acclimatization to higher temperatures; Mills et al., 2015; Wu et al., 2014) and reduced susceptibility to heat-related mortality (from advances in health care and access and reduced prevalence of cardiovascular mortality risk factors such as smoking). However, future climate change is expected to increase the frequency and severity of extreme urban heat events (Greene, Kalkstein, Mills, & Samenow, 2011; McCarthy, Best, & Betts, 2010; Rahmstorf & Coumou, 2011), which models suggest could significantly increase the incidence of premature heat-related urban mortality in the U.S. (Peng et al., 2011; Schwartz et al., 2015; Stone et al., 2014; Wu et al., 2014) even with further adaptation (Mills et al., 2015).

Vegetation affects urban microclimatic conditions mainly through transpiration (evaporative cooling of air), shading and its thermal properties, specifically, a low thermal storage capacity and resulting low re-radiation of heat compared with nonvegetated structures (Lin & Lin, 2010; Oke, 1989; Spronken-Smith & Oke, 1998; Spronken-Smith &

Oke, 1999; Taha, 1997). The intensity of the cooling effect of urban trees can vary substantially among tree species (Ballinas & Barradas, 2016; Legese Feyisa, Dons, & Meilby, 2014) and with tree size as a result of differences in foliage density and leaf area index, leaf thickness, orientation and light permeability, and photosynthetic and water use rates. Cooling intensity shows a strong positive relationship with canopy area and density (Barradas, 1991; Chang & Li, 2014; Chang, Li, & Chang, 2007; Chen & Wong, 2006; Giridharan, Lau, Ganesan, & Givoni, 2008; Hamada & Ohta, 2010; Hardin & Jensen, 2007; Jonsson, 2004; Legese Feyisa et al., 2014; Potchter, Cohen, & Bitan, 2006; Zoulia, Santamouris, & Dimoudi, 2009). The cooling effect extends into surrounding areas primarily through advection (Sugawara et al., 2016; Taha, Akbari, & Rosenfeld, 1991), its reach a positive function of canopy area but mediated by air flows and temperature gradients with surroundings (Legese Feyisa et al., 2014; Sugawara et al., 2016). Most studies find that the maximum cooling distance of urban forest patches or partially forested parks on sunny summer days extends to approximately one park width from the park (Jauregui, 1991; Spronken-Smith & Oke, 1998), though shorter (Hamada & Ohta, 2010) or longer (Ca, Asadea, & Abu, 1998; Shashua-Bar & Hoffman, 2000) distances are possible.

In addition to experimental studies, a large number of remote-sensing studies at different spatial resolutions also demonstrate the significant control the vegetative fraction (i.e., percent area in vegetative cover) exerts over temperatures in urban environments (Li et al., 2011). For example, Pincetl, Gillespie, Pataki, Saatchi, and Saphores (2013) found that the percentage of shade tree cover explained more than 60% of land surface temperature (LST) variations among Los Angeles, California city blocks. Imhoff et al. (2010) and Bounoua et al. (2015) found that the share of impervious surface area (ISA) was significantly and linearly related to the intensity of the UHI effect in the U.S., with percent ISA explaining approximately 70% of the variation in LST on average across biomes (Imhoff et al., 2010).

Importantly, trees outperform lower vegetative covers in mitigating PM and maximum summer daytime temperature because their larger canopy surface area achieves higher PM deposition (Fowler et al., 2004; Pryor et al., 2008) and cooling (Giridharan et al., 2008; Hamada & Ohta, 2010; Huang, Li, Zhao, & Zhu, 2008; Jonsson, 2004; Ketterer & Matzarakis, 2014; Potchter et al., 2006; Shashua-Bar, Erell, & Pearlmutter, 2009; Spronken-Smith & Oke, 1998).

1.3. *Goals of this paper*

A systematic approach to using urban tree cover for PM and heat abatement requires information on how much mitigation particular tree cover additions and canopy designs can achieve in given locations; how many people receive those mitigation effects; and how much those tree cover additions cost. Integrating these data would provide spatially-explicit information about the return on investment (ROI) that specific tree cover additions in particular locations could achieve, in terms of PM and heat mitigation for people – information cities need to identify cost-effective tree planting portfolios.

In this paper, we draw on both the city-level and site-level literatures on PM and summer maximum daytime temperature mitigation by trees to estimate how recent urban tree cover change in 27 U.S. cities has affected provision of these services, as well as how much additional PM and heat mitigation could be achieved by ambitious urban reforestation. We use high-resolution land cover and population data to estimate for each potential tree planting site in each city the size of the expected mitigation effects and the number of people affected. Using tree planting and maintenance costs, we estimate the population-weighted ROI of each site in mitigating PM and heat and construct aggregate ROI curves for each city that indicate how much additional PM and heat abatement can be achieved for given annual tree planting and maintenance budgets.

2. Methods

Below we present the main steps in our analysis. Additional details and data can be found in the Supplementary Information (SI).

We selected cities for inclusion in the analysis primarily based on availability of common-methodology-based, existing estimates of both average city-level PM_{10} removal rates by urban forests (Nowak, Crane, & Stevens, 2006) and recent urban forest cover change (Nowak & Greenfield, 2012). We included several additional cities that lacked Nowak and Greenfield (2012) forest cover change estimates but in which substantial tree planting efforts are being considered or implemented.

Where available, we estimated 2004–2014 change in percent urban area in forest cover (%FC) based on reported recent average annual change (Nowak & Greenfield, 2012). For other cities, we first estimated recent average annual %FC change from 30 m resolution University of Maryland (UMd) continuous %FC (Sexton et al., 2013) in years 2000 and 2010. Assuming that decadal pixel-level FC change < 25% is mainly due to noise in the remotely sensed imagery, we first calculated the mean %FC difference between 2000 and 2010 in pixels with < 25% FC change, and then normalize the two distributions by adding this mean difference to the 2010 product for each pixel. To statistically account for street trees which often may not be detected by 30 m imagery, we calibrate the 2010 UMd %FC estimates to 2 m resolution National Agricultural Inventory Program (NAIP) percent tree cover (%TC) data using the empirical relationship between the two datasets in the study cities (See SI A).

We identify the potential street tree planting area in each street segment in each of the 27 cities as the difference between current NAIP %TC in a 16 m-wide buffer around the segment centerline and the city's 95%ile NAIP segment %TC, multiplied by buffer area. In each city, potential patch planting sites outside of street segments are identified by first excluding impervious, agriculture, water and wetland areas (Homer et al., 2015) and current tree cover (NAIP) within 2010 U.S. Census city boundaries. To exclude non-impervious areas likely not available for tree planting, we then extracted Normalized Difference Vegetation Index (NDVI) data, adjusted NDVI thresholds for each city to account for image variation across the U.S., and used a combination of NDVI and a city-specific entropy-based texture analysis to remove smooth-texture NDVI areas, which correlate with golf-courses, sports fields and lawns. We exclude patches < 100 m^2 in size, and, to avoid potential biodiversity conflicts, in non-forest biomes (identified using Olson et al., 2001) patches ≥ 20 ha or > 50% in natural land cover (SI B).

2.1. *City-level analysis*

Our estimates of the city-level effect of tree cover change on PM_{10} removal in each city are based on the estimated average annual PM_{10} removal rate per square meter forest canopy (g PM_{10} yr^{-1} m^{-2}) in 1994 in each city (Nowak et al., 2006). These removal rates were estimated using the dry deposition module of the Urban Forest Effects (UFORE) model that quantifies pollutant removal based on vegetation canopy characteristics and hourly pollutant concentration, climatic and meteorological data (Hirabayashi, Kroll, & Nowak, 2012; Nowak & Crane, 2000; Nowak et al., 2000). The rates represent the net (i.e., resuspension-adjusted) PM_{10} fluxes (F_p, in μg s^{-1} m^{-2}) towards leaf surfaces, calculated as the product of PM_{10} deposition velocity (v_d, in cm s^{-1}) and atmospheric PM_{10} concentration (C_p, in μg m^{-3}),

$$F_p = v_d \cdot C_p \tag{1}$$

Because Nowak et al.'s (2006) PM_{10} deposition velocities are based on studies of homogenous canopies in non-urban areas (Davidson & Wu, 1991; Lindberg, Lovett, Richter, & Johnson, 1986), they may underestimate – perhaps by one order of magnitude (Maher et al., 2013) – deposition on generally more heterogeneous urban canopies that promote more turbulent air flows (Baldauf et al., 2009; Britter & Hanna, 2003; Nowak et al., 2006). To remove this bias and account for changes in PM_{10} concentrations between 1994 and 2014, we multiply Nowak et al.'s (2006) removal rates for each city by the ratio of the deposition velocities used in their study and those reported in recent urban canopy microscale PM_{10} deposition studies, and by each city's ratio of average annual PM_{10} concentrations in 2014 and 1994 (SI C). We estimate total annual change in PM_{10} removal from urban tree cover change by multiplying the estimated city-level average PM_{10} removal rates by historic and prospective canopy change, respectively, in each city.

To estimate tree cover change impacts on city-level average daily maximum summer (June, July, August; JJA) air temperature (MaxAirT), we first calculate average annual change in percent city forest cover to ISA ($\Delta\%FC_{ISA}$) from Nowak and Greenfield (2012) as:

$$\Delta\%FC_{ISA} = \Delta\%FC_{Impervious} + \Delta\%FC_{Soil}\cdot\frac{\Delta\%SC_{Impervious}}{\Delta\%SC}, \tag{2}$$

where $\Delta\%FC_{Impervious}$ and $\Delta\%FC_{Soil}$ are the average annual % city area changing from tree cover to impervious or soil cover (*SC*), respectively; and the right hand side fraction represents the share of total *SC* change that changes to impervious, which we add to account for the fact that a large portion of change from tree cover to soil is temporary and followed by further conversion to impervious (Nowak & Greenfield, 2012). We assign cities to biomes based on Olson et al. (2001), and use our estimated city-specific rates of $\Delta\%FC_{ISA}$ and Imhoff et al.'s (2010) biome-specific city-level relationships between ISA fraction and summer daytime LST to estimate the effect of forest conversion to ISA on city-level average summer LST at 13:30 local time in each city.

Applying the methodology described in Zhang, Bounoua, Imhoff, Wolfe, and Thome (2014), we estimate biome-specific relationships between average JJA LST at 13:30 local time (MODIS Aqua 1:30 pm) and average JJA MaxAirT for the over 890 Global Historical Climatology Network [GHCN] (Menne, Durre, Vose, Gleason, & Houston, 2012) weather stations in forest, grassland, desert or Mediterranean-biome areas with ISA ≥ 25% (SI D; Table SI-D). We then calculate for each city the estimated absolute change in average JJA daytime MaxAirT (in °C) resulting from forest conversion to impervious as

$$\Delta MaxAirT = \Delta\%FC_{ISA}\cdot\Delta LST/\Delta\%ISA\cdot\Delta MaxAirT/\Delta LST. \tag{3}$$

2.2. Local-level analysis

We estimate PM_{10} removal by tree patches based on the removal efficiencies reported in the, to the best of our knowledge, only four empirical tree patch PM_{10} concentration fate studies. The observed near-source PM_{10} mass concentration decrease in tree patches can be fitted into an exponential function of patch depth,

$$C_x = C_0 e^{-kx}, \tag{4}$$

where C_x and C_0 are the particle mass concentrations (μg m^{-3}) at horizontal distance x (in m) from the source and at the source, respectively, and k is the depletion coefficient that incorporates vertical diffusion and deposition (VanCuren, Pederson, Lashgari, Dolislager, & McCauley, 2012). Assuming conservatively that patch trees would replace low vegetation, we use the reported high and low k values for trees and low vegetation, respectively, to calculate the incremental mean (high, low) PM_{10} concentration reduction at the downwind tree canopy edge vs low vegetation, as a function of patch depth. We derive average annual incremental PM_{10} concentration reductions at the downwind canopy edge by multiplying the calculated incremental PM_{10} concentration reductions by the percent time a given city's trees are in-leaf, calculated from in-leaf and percent evergreen share in each city. We use published urban PM_{10} concentration decay functions to define high (30 m) and low (31–65 m) PM_{10} impact buffers around tree patches and to calculate how much of the incremental concentration reduction achieved at the downwind canopy edge remains within each of these buffers on average. We multiply that mean remaining % reduction by the average annual 2014 PM_{10} concentration in each buffer to estimate the absolute reduction in ambient PM_{10} concentrations a patch achieves in its buffers (SI E). Because a patch lowers PM_{10} concentrations only downwind from it, we multiply the calculated absolute concentration reductions by half the population in each buffer (calculated from the Wildland Urban Interface database 2010 U.S. Census block-level population density layer; Radeloff et al., 2010). Finally, we sum ΔPM_{10} *concentration* * *persons* over both buffers.

To estimate street tree PM_{10} removal, we use the five empirical studies of street tree PM_{10} capture efficiency (i.e., % difference in PM_{10} in air upwind and downwind, respectively, of the trees) and generate two additional estimates by applying the high and low exponential tree patch PM_{10} concentration decay functions in the literature to an assumed 5 m-wide canopy. Using the same approach used for tree patches, we estimate the average $\%PM_{10}$ concentration reduction street trees achieve within a 30 m buffer from the street edge. We adjust for average annual in-leaf and evergreen share, multiply the % reductions by the average annual 2014 PM_{10} concentration in the buffer and calculate the population affected as the product of % canopy addition in a street segment and total population within the segment buffer (SI E).

We estimate local-level cooling impacts based on cooling intensity and distance reported in the empirical studies that examine the impact of urban parks or forests (n = 20) or street or individual trees (n = 5) on summer daytime MaxAirT. Because both park cooling intensity (PCI; a term applied to both parks and street trees) – the temperature difference between a greenspace and reference sites – and cooling distance (PCD) are strongly related to park size, we group patch sites in the studies into four size classes based on calculated park width distance (PWD), defined as the square root of green space area (Jauregui, 1991; Spronken-Smith, 1994) and highly correlated ($R^2 = 0.88$) with PCD in our data set (SI F). We construct size class-specific high, medium and low PCI buffers based on two PCI distance decay functions covering a combined 71 urban greenspaces (Chang & Li, 2014; Shashua-Bar & Hoffman, 2000). We calculate mean (high, low) PCI at greenspace edge for each size class as the mean (± 1SD) of the empirical PCIs reported for the greenspaces in that class. To estimate mean cooling in each (exclusive) buffer of each patch size class, we construct a composite of the daytime summer MaxAirT PCI distance decay function from the two studies that to our knowledge estimate such functions for urban parks from more than two measurement points (Hamada, 2011; Shashua-Bar & Hoffman, 2000). We multiply estimated PCI, mean %PCI remaining in each buffer, and buffer population, and sum over all buffers to obtain for each patch the total high, central and low estimate of summer MaxAirT abatement (Δ°*C·population*; SI F).

We calculate mean (high, low) street tree PCI as the mean (± 1SD) of the eight reported empirical average street tree MaxAirT PCI values, and estimate a best-fit ($R^2 = 0.98$) PCI distance decay function from the empirical PCI values reported in the only study that provides empirical observations of street tree MaxAirT PCI distance decay (Shashua-Bar & Hoffman, 2000). We calculate absolute (low, medium, high) MaxAirT reduction in the 30 m street tree buffer as the product of street tree PCI and mean percent residual PCI in the buffer, and assume the population affected by cooling from plantings is proportional to the absolute % segment street tree canopy increase (SI F).

2.3. Tree planting and maintenance cost

We use city-specific or regional mean per-tree planting and maintenance costs for urban street, park and private yard trees reported in contractor reports, peer-reviewed studies and U.S. Forest Service Community Tree Reports. In many cases, the maintenance costs include most of the post-establishment costs of trees, including pest, disease and weed control, pruning, fertilization, watering, mortality replacement, storm debris removal, leaf collection, sewer, sidewalk and curb repair and trip-and-fall claims (McPherson et al., 2005; Vogt, Hauer, &

Fischer, 2015). We assume that small (> 100 m^2 to ≤300 m^2) patches are privately while medium (> 300 m^2 to < 1 ha) and large (≥1 ha) patches are publicly owned. For large patches, we use Kroeger et al.'s (2014) estimated mean cost ha^{-1} for a peri-urban reforestation project but add annual forest health management costs. Except for large patches, we calculate the number of trees planted at a given site by dividing site canopy area by estimated mean crown area per tree (77.1 m^2) at age 40. We adjust all literature costs to 2015$ using the U.S. Consumer Price Index and calculate annualized 40-yr total (planting plus maintenance) cost for each planting site as the product of trees or ha planted and annualized cost per tree or hectare, respectively (SI G).

2.4. Return on investment

For each city, we calculate estimated total annual PM_{10} and average summer MaxAirT reductions for all potential planting sites and rank them by their respective ROI, defined as abatement (mitigation * people impacted) per dollar (Δμg PM_{10} m^{-3}·people \$$^{-1}$; Δ°C·people \$$^{-1}$). We also calculate for each city the total number of people in all prospective impact buffers who are expected to receive reductions exceeding 2 μg PM_{10}·m^{-3} or 2 °C in summer MaxAirT.

2.5. Comparison of city- and local-level estimates

Because our city- and local-level analyses are based on very different approaches and in the case of PM_{10} even use different metrics (mass removal vs ambient concentration reductions), we develop a methodology that permits comparing their results. For PM_{10}, this comparison is very approximate (SI K).

3. Results

3.1. Change in urban tree canopy and PM_{10} and heat mitigation

Twenty-three of the 27 cities studied experienced a decline in forest cover during 2004–2014. Overall, cities lost an average of 1355 ha/city (range: loss of 9410 ha to gain of 948 ha), equivalent to a loss of forest cover of 1.5% (95% CI: 0.9–2.2) of city area (Fig. 1). Houston and New Orleans experienced the largest absolute declines in forest cover, and Houston and Baltimore the largest percent declines. Forest cover increased only in Dallas, Seattle and Sacramento.

The overall decline in forest cover has reduced particulate matter removal. PM_{10} removal by trees declined on average by 168 t yr^{-1} under our medium removal rate estimate (low estimate: 58 t yr^{-1}; high estimate: 278 t yr^{-1}). In three cities (Houston, New Orleans, Los Angeles) the decline exceeded 650 t yr^{-1} in our medium removal rate estimate (low: 220 t yr^{-1}; high: 1100 t yr^{-1}). To put this decline in context, one could compare it to the total amount of PM_{10} within the urban boundary layer (UBL) above the city. Under the simplifying assumption of a well-mixed UBL with an average height of 200 m (Nowak et al., 2006), this loss in pollutant removal within the tree canopy layer (i.e., generally below 20 m above ground) on average is equivalent to 0.6% (low removal rate estimate: 0.2%; high: 1.0%) of the total UBL PM_{10} mass, although in some cities the decline is as high as 2% (SI H).

The change in forest cover also impacted air temperatures. The average increase in summer MaxAirT was 0.1 °C. Temperature increases exceeded 0.2 °C in Baltimore, Detroit, Atlanta, Boston and Louisville (Fig. 1). Note that Houston had the largest forest cover decline but only an average increase in temperature. This is due to the fact that in Houston, only a relatively small fraction of forest lost is converted to ISA (0.3, vs 0.6 average in other cities).

3.2. Reforestation potential and gains in ecosystem service provision

Tree cover in the 27 cities could be increased on average by nearly 18% of city land area (range across cities: 3%–41%). This increase would be spread among many sites: we identified on average 69,000 individual sites per city (range: 6000–240,000). Street trees on average accounted for 46% of total potential reforestation area. However, this percentage varied widely among cities, from 9% to 80% (SI I).

If this full reforestation potential were realized, the newly planted trees could remove an additional 1317 t (491–2141 t) PM_{10} per year on average 'per city'; as 1317 t PM_{10} is the average increase in estimated

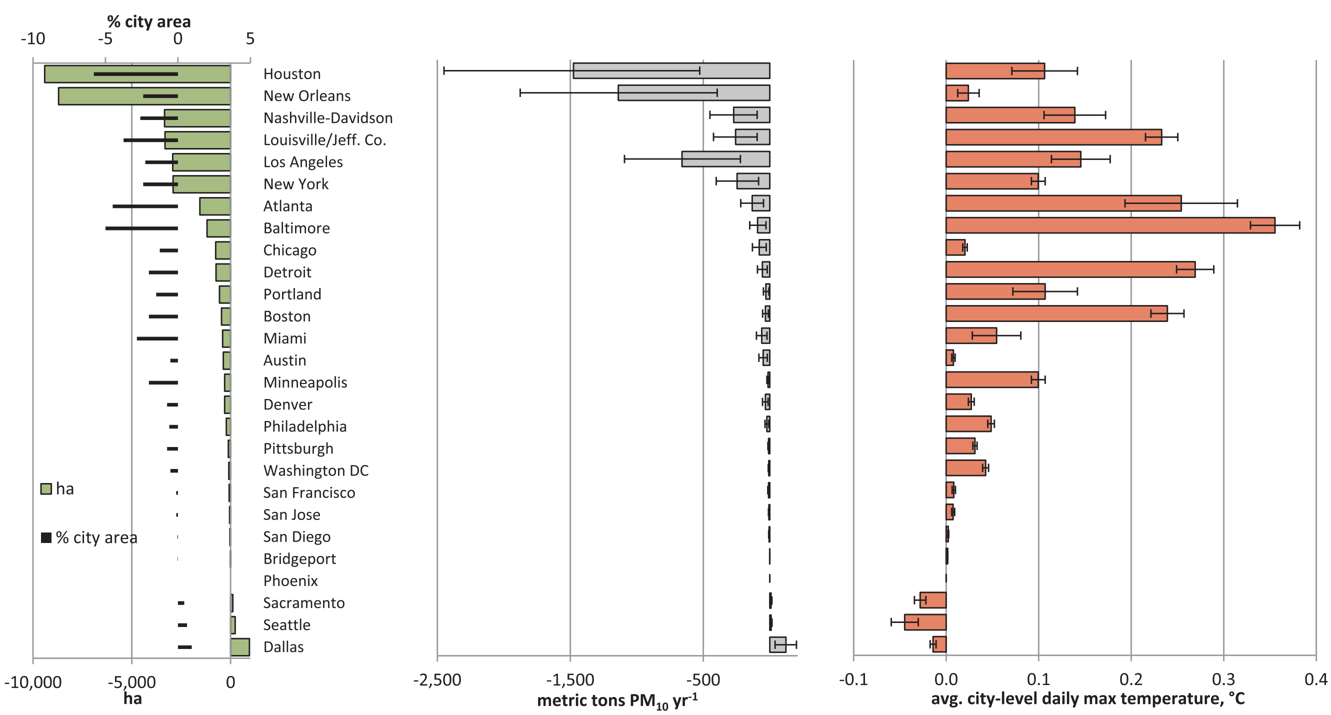

Fig. 1. Percent and absolute change in forest cover (2014 vs 2004; left panel), and absolute change in city-level annual PM_{10} removal (center panel) and average summer daytime maximum temperature (right panel) in 2014 vs no forest change.

annual PM_{10} removal across the city sample based on our local-level analysis. Potential incremental removal varies widely among cities, ranging from 51 t (10–84 t) in Bridgeport to 4856 t (1790–7839 t) in Houston. Similarly, average daily maximum summer air temperatures would decline by an estimated average 1.7 °C (1.4–2.0 °C) across cities (local-level analysis). As with PM_{10}, there is substantial variation among cities, ranging from 0.3 °C in San Francisco (0.3–0.4 °C) to 5.1 °C in Louisville (4.7–5.5 °C; SI I).

3.3. Variation in return on investment across sites

To understand how return on investment (ROI) of tree planting varies across sites, we developed a unique methodology that calculates for each site the total abatement, expressed as the number of people who would receive PM_{10} or heat reductions from planting multiplied by the magnitude of the reductions (i.e., mitigation * people). We then divided total abatement by planting costs to estimate ROI (i.e., mitigation * people/$). We calculated ROI separately for PM_{10} and heat mitigation, to understand how patterns of site-level ROI compare.

Within each city, the ROI of potential planting sites varies by four orders of magnitude for both PM_{10} and heat mitigation. Generally, sites that have a high ROI for PM_{10} mitigation also have a high ROI for temperature mitigation (Fig. 2). Using Washington DC as an example, Fig. 3 shows that population density is the key driver of intra-city variability in site ROI for PM_{10}. This is caused by the much wider range in population density than in either PM_{10} concentrations or reforestation costs per square meter, so density variations tend to drive variations in site ROI. The same is true for the ROI of tree planting for temperature mitigation.

Population density is also a key driver of inter-city differences in ROI. Additionally, inter-city differences are caused by often considerable differences in average reforestation costs among cities. Fig. 4 indicates clear differences among cities in their median cost-effectiveness of reforestation for PM_{10} and summer heat mitigation. Cities towards the top of the figure have higher median ROI for PM_{10} and heat mitigation than cities towards the lower end of the figure, meaning they can deploy relatively larger (as % of their maximum potential) reforestation efforts at similar cost-effectiveness in PM_{10} and heat mitigation than cities towards the lower end. However, the variation in median ROI among cities is much smaller than the variation within cities, generally less than one order of magnitude for PM_{10} and temperature (Fig. 4). This means that even cities with comparatively lower median ROI still have many high-ROI sites.

3.4. Optimal investment curves

We constructed optimal investment curves for each city separately for PM_{10} and temperature, assuming reforestation would begin on the site with the highest ROI, followed by the site with the next-highest ROI and so forth. We recognize that in reality, most cities would prioritize sites based on a variety of factors, not just PM_{10} and temperature mitigation.

A relatively small proportion of sites account for a large portion of overall benefits from reforestation. Therefore, focusing investments on high-ROI sites can yield substantial abatement at moderate cost. For example, reforestation of the top 20% ROI sites in our 27 cities would cost $102 M annually. Under our low or high impact assumptions, respectively, this would provide 3.4 M or 11.4 M people a greater than 2 $\mu g{\cdot}m^{-3}$ reduction (equivalent to 10 percent of the 2014 mean annual concentration for our cities, 19.46 $\mu g{\cdot}m^{-3}$) in average annual PM_{10} concentrations. Similarly, investing in the top 20% ROI sites would decrease average maximum daily summer temperatures by more than 2 °C for 1.7 M (low impact assumption) to 12.7 M (high impact assumptions) people (SI J).

Total abatement in each city initially rises sharply with investment in tree planting but the slope of the abatement curve continually declines and eventually becomes zero at sites with no people in their impact buffers (Fig. 5). On average across the 27 cities, 80% of the maximum potential PM_{10} and heat abatement from tree planting can be achieved at approximately 1/2 of the cost of full reforestation (PM_{10}: 49%; heat: 53%). Similarly, one half of maximum potential abatement can be achieved at approximately 1/10 the cost of full reforestation cost (PM_{10}: 14%; heat: 12%).

At one half maximum abatement, the average ROI of planted sites is 0.34 $\mu g\ PM_{10}{\cdot}m^{-3}{\cdot}person{\cdot}\$^{-1}$ in the medium impact scenario for PM_{10} (low: 0.19, high: 0.50) and 0.22 $°C{\cdot}person{\cdot}\$^{-1}$ for heat (low: 0.13, high: 0.30). Thus, on average across these cities and under medium impact assumptions about the PM_{10} and heat mitigation effects of trees, each dollar spent per year on urban reforestation lowers average annual PM_{10} concentrations by 0.34 $\mu g{\cdot}m^{-3}$ for one person, and lowers maximum summer air temperature by 0.22 °C for one person. Crucially, these ROI values are calculated by dividing either total PM_{10} or total heat abatement, respectively, by total reforestation costs, thus assigning the full costs to either one of the impacts. Reforestation achieves both objectives, so for cities concerned about both the total ROI would be higher (twice as high under equal importance), since both temperature and PM_{10} mitigation are achieved by a given planting.

Investment in all possible reforestation sites would yield maximum possible additional abatement. Under such full potential reforestation, total additional PM_{10} abatement in each city from reforestation would range from 265,000 $\mu g\ PM_{10}{\cdot}m^{-3}{\cdot}person{\cdot}yr^{-1}$ (Bridgeport) to 12.5 M $\mu g\ PM_{10}{\cdot}m^{-3}{\cdot}person{\cdot}yr^{-1}$ (Los Angeles) in the medium impact scenario (approximately ± 45% in the low and high impact scenarios). Similarly, total additional temperature abatement from full potential reforestation would reach 146,000 °C·person (Bridgeport) to 10.5 M °C·person (New York) (approximately ± 39% in the low and high impacts scenarios) (SI J, Figs. SI-Q and SI-R). This means that under full planting, in Los Angeles, average annual PM_{10} concentrations would decrease by the equivalent of 1 $\mu g\ PM_{10}{\cdot}m^{-3}$ for 12.5 M people, and average summer maximum temperature in New York by the equivalent of 1 °C for 10.5 M people. In reality of course, reforestation will yield wide ranges of PM_{10} and heat reductions for different people.

3.5. Comparison of site- and city-level methodologies

Our city-level estimates of PM_{10} removal and temperature mitigation use previously published methodologies, whereas our local-level estimates utilize a unique methodology developed for this paper (see Methods). Given that the two approaches differ substantially – including in the case of PM_{10} in the metrics used – the question is to what extent their results diverge. Estimated annual PM_{10} removal obtained via our city-level approach, based on PM_{10} mass removal rates per square meter canopy cover, is on average 75% (range: 30%–520%) that of our site-level, PM_{10} concentration reduction-based removal estimates converted to city-wide equivalent mass removal (see SI K for calculations). For all but seven of the cities, the mean removal estimate derived with one approach lies within the low-high range estimate of that derived by the other, and for all but two cities (Minneapolis and New Orleans), the low-high ranges overlap. Given the different methodologies used in generating the two sets of PM_{10} mitigation estimates and the assumptions needed to make them comparable, this constitutes an acceptable match, indicating that the two approaches generally yield broadly similar results (Fig. SI-T).

Similarly, estimated heat reduction based on the city-level approach is on average 130% (range: 41%–234%) that derived via the local-level approach (Fig. SI-S). Given that the summed local-level heat reduction estimates cover only a portion of total city area (and thus, population), they would generally be expected to be lower than the city-level estimates, unless areas surrounding identified tree planting sites are characterized by above-average population density, in which case the reverse would be true. As in the case of PM_{10}, the comparison indicates broad alignment of the results generated via the city- and local-level

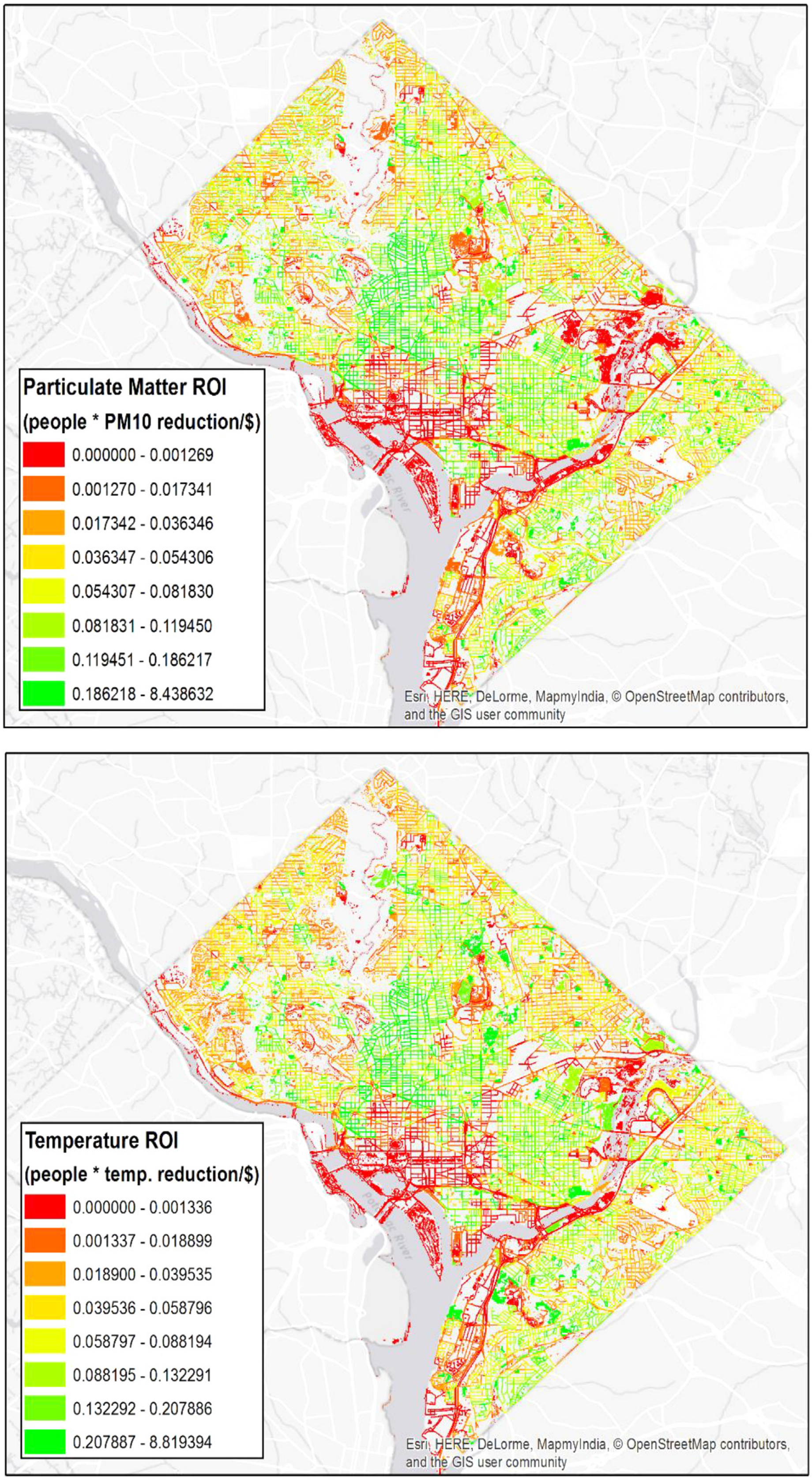

Fig. 2. Potential planting site ROIs for PM_{10} and heat abatement in Washington, DC.

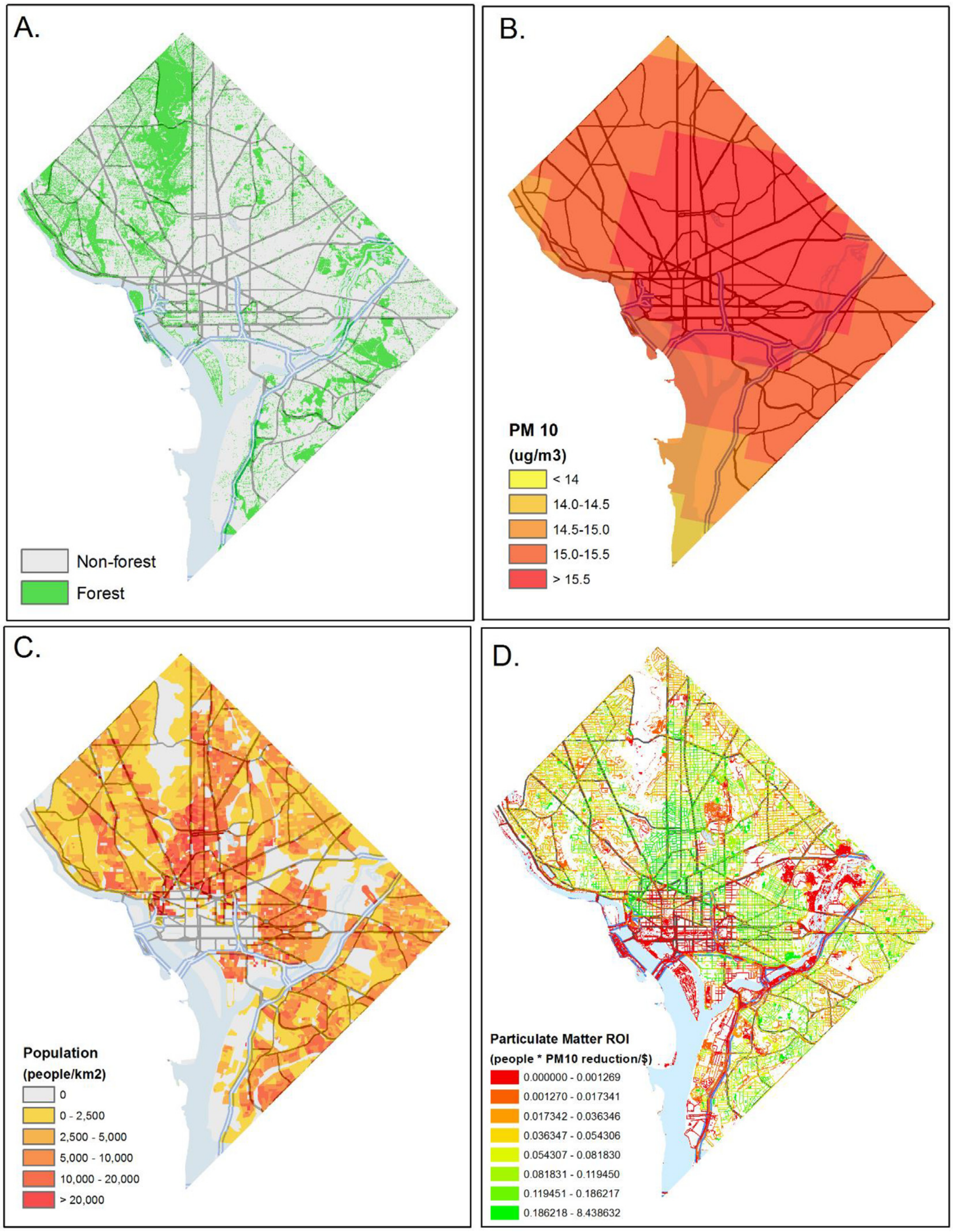

Fig. 3. Current forest cover (A), average annual PM_{10} concentrations (B), population density (C) and ROI of potential reforestation sites (D) in Washington, DC.

methods.

4. Discussion

4.1. Current decline in ecosystem service provision

In 23 of the 27 cities we studied, tree cover declined by an estimated 0.0–5.8% of city area during 2004–2014, leading to modest estimated increases in city-wide average PM_{10} concentrations in the urban boundary layer of 0.0–3.9% and city-wide average temperatures of 0.0–0.4 °C in these cities in 2014, compared to a situation of 2004 tree cover. Note that these changes in ecosystem service provision are modest because the changes in forest cover were relatively modest. *Total* PM_{10} and temperature mitigation by trees in cities is of course significantly larger than the *changes* in mitigation over the past decade. Total forest cover in the cities varied from 5.7% to 37.6% of city area, roughly one order of magnitude greater than forest cover change.

More importantly, our results suggest that any estimates of the city-level average effect of trees on PM and heat can be misleading. Our local-level analysis shows that the loss of canopy cover can significantly affect PM concentrations and temperature in the immediate surroundings, where people may experience changes in PM and heat mitigation that far exceed city-level average change. Accounting for the location of people relative to planting sites and for the number of beneficiaries thus is crucial when analyzing air pollutant and heat mitigation effects of prospective plantings. Given the thresholdless, linear dose-response relationships established in the literature for all-cause (cardiovascular, cerebrovascular and respiratory) mortality (Dominici, Daniels, Zeger, &

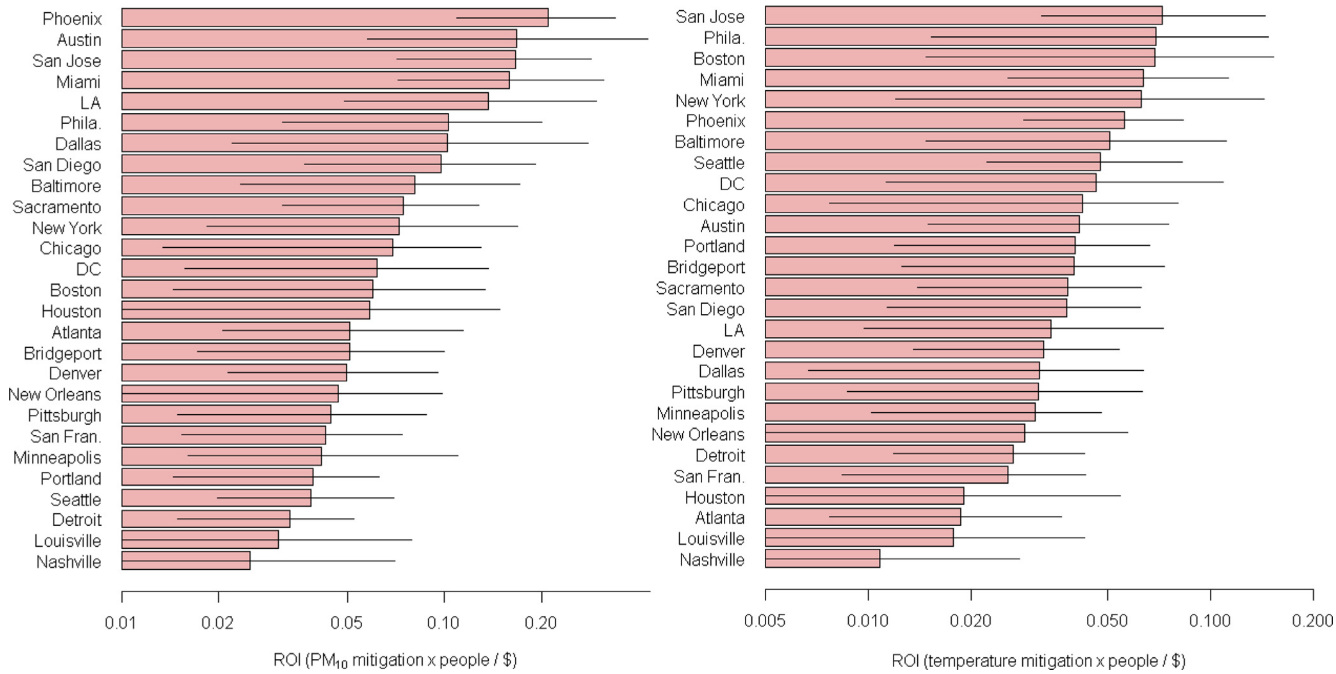

Fig. 4. ROI of the median and interquartile (indicated by error bars) planting sites for PM_{10} and heat abatement for medium PM_{10} removal and heat reduction scenarios.

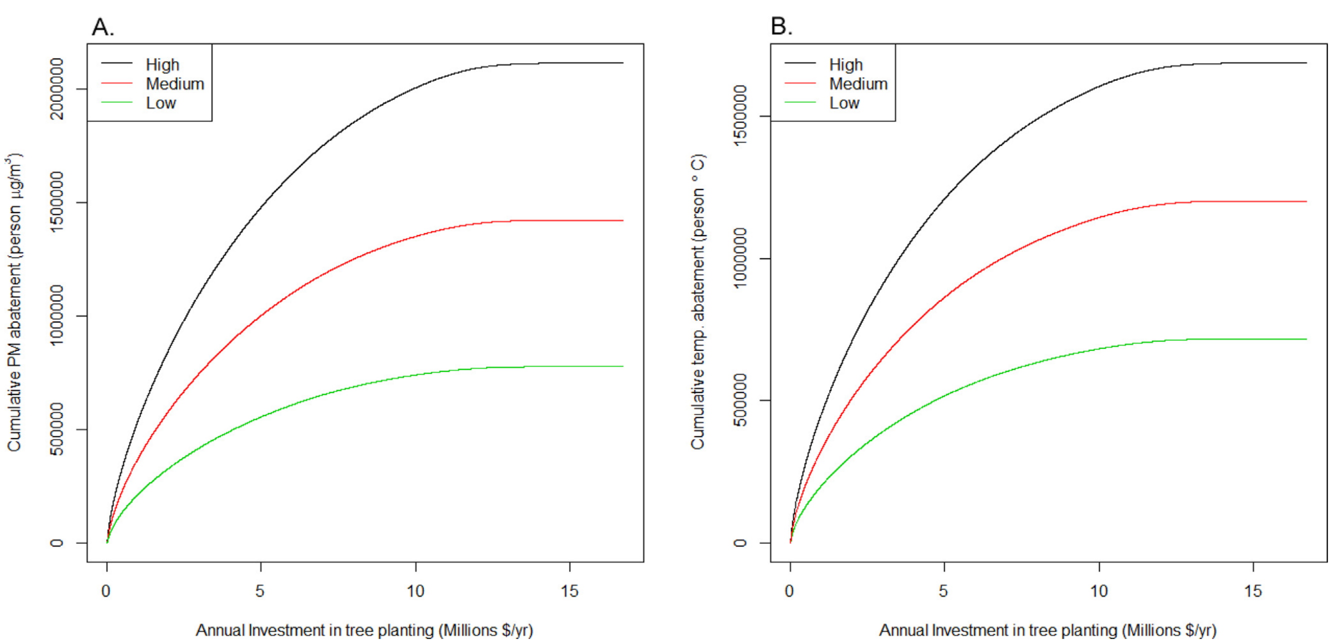

Fig. 5. Cumulative abatement curves for tree planting in Baltimore for PM_{10} and average maximum daily summer temperature, for low, medium and high mitigation assumptions.

Samet, 2002; Samoli et al., 2005) and respiratory (Liu et al., 2014) and cardiovascular morbidity (Brook et al., 2010) for levels of PM_{10} concentrations typically found in U.S. cities, and given the exponential exposure-response relationship between heat and mortality above the apparent (U.S. average) threshold of 90th percentile (95% CI lower bound: 50th percentile) summer daily maximum temperatures (Fig. S5 in Gasparrini et al. (2015)), locally-concentrated mitigation effects from tree cover additions can provide meaningful health benefits for nearby people.

4.2. Large potential for benefits

If tree planting were conducted at all possible sites in the 27 cities, newly planted trees could remove an estimated additional 1317 t (491–2141 t) PM_{10} per year on average, and average daily summer maximum temperatures would decline on average by an estimated 1.7 °C (1.4–2.0 °C).

Our city-level PM_{10} mitigation estimates mean that PM_{10} removal by trees may be an order of magnitude larger than reported in an earlier, UFORE-based analysis (Nowak et al., 2006). This difference results from our adjustment of that study's PM_{10} removal rates using more recently reported, urban-specific, micro-scale canopy PM_{10}

deposition velocities expected to better represent urban processes than the rural forest stand-based values used by Nowak et al. (2006). The results suggest that trees may play a much larger role in pollution control than suggested by UFORE modeling, a finding supported by other recent studies (Bottalico et al., 2017; Maher et al., 2013; Rao, George, Rosenstiel, Shandas, & Dinno, 2014). The fact that this adjustment yields city-level mitigation estimates that are generally similar to the ones obtained by scaling up our local-level, empirically-based PM_{10} mitigation estimates that are constructed using a markedly different methodology provides further evidence for UFORE PM_{10} deposition velocities being too low for urban settings.

Our estimates of potential city-level heat mitigation through urban tree cover increases are in line with findings reported in other studies using different methodologies. For example, using the Colorado State University Mesoscale Model (CSUMM), Sailor (1995) estimated that a 14% increase in the vegetative fraction in L.A. from tree planting would reduce peak summertime temperatures by more than 1.3 °C, compared to our medium estimate of 1.7 °C decrease for a 15% tree cover increase. Similarly, using the mesoscale meteorological model (MM5), Civerolo, Sistla, Rao, and Nowak (2000) estimated that an approximately 40% increase in urban tree cover would lower maximum midday temperatures in individual urban grid cells (16 km^2) by up to 3.5 °C (Baltimore), 5 °C (Philadelphia) and 5.5 °C (New York) – 2–3 times our tree cover increase and 1.5–2.75 times our medium heat reduction estimates for these cities. Modeling the effect of large-scale tree cover increases on maximum temperatures in ten U.S. metro areas, Taha et al. (1999) estimated the spatially-averaged reduction in ambient air temperatures in each metro area. While their additional tree numbers per unit area on average are less than 1/10 of those in our maximum planting scenario, their average estimated heat reduction is 6/10 of ours.

Future analysis of PM_{10} and temperature mitigation by urban trees could improve upon our estimates in four key areas:

1) *Spatial variability in removal:* Our analysis of close to 1.9 M sites for tree planting did not permit detailed consideration of all the factors that affect local heat mitigation. Rather, our local-level mitigation estimates are based on functional relationships representing the average atmospheric, morphological or climatic conditions found at the literature study sites rather than the specific conditions of individual prospective planting sites in our 27 cities. The literature shows that the local heat mitigation effect of trees depends on a number of site-specific atmospheric (cloud cover; speed and direction of air flow), morphological (street height-to-width ratio, building density and surface materials in surrounding areas), climatic (temperature, humidity, precipitation) and other (orientation of trees relative to the sun's zenith and airflow; tree species) conditions (Ali-Toudert & Mayer, 2007; de Abreu-Harbich, Labaki, & Matzarakis, 2015; Dimoudi & Nikolopoulou, 2003; Gromke et al., 2015; Legese Feyisa et al., 2014; Norton et al., 2015; Pearlmutter, Berliner, & Shaviv, 2007; Sanusi, Johnstone, May, & Livesley, 2016; Shashua-Bar, Potchter, Bitan, Boltansky, & Yaakov, 2010). Similarly, fine-scale spatial variability in PM_{10} concentrations, wind speed, and other factors can significantly alter the rate and spatial pattern of site-level PM_{10} removal. Importantly, our site-specific ROI results can be combined with both fine-scale analyses that incorporate some of the mitigation-relevant conditions omitted in our analyses and with local-scale guidance on tree planting, to identify tree planting portfolios that achieve maximum abatement ROI by including cooling-relevant characteristics omitted in our analysis (SI L). Similarly, additional objectives such as greater equality in greenspace access can readily be incorporated, although our ROI metric is likely to promote the latter (SI N).
2) *Population exposure modeling*: Our ROI estimates are based on residential population data and thus identify business districts or heavily commercial areas as having extremely low ROIs. In reality, locations in such areas that experience high daytime population density may also be sites of high population exposure, and thus may have high ROIs for tree planting.
3) *Choice of abatement metric*: Our ROI metrics implicitly assign equal weight to all sizes of PM_{10} and heat reductions, respectively. For PM, this is clearly justified given that avoided health effects are the primary benefit of PM reductions and given the thresholdless, linear dose-response relationships for the levels of PM_{10} concentrations found in U.S. cities. For heat, equal weighting is justified for the benefit of reduced energy use given the linear relationship between maximum summer temperature and peak urban electricity demand above thresholds much lower than typical summer maximum temperatures (Akbari et al., 2001; Santamouris, Cartalis, Synnefa, & Kolokotsa, 2015); it arguably is justified also for health benefits, given the exponential mortality exposure-response relationship at high summer temperatures.
4) *PM_{10} versus $PM_{2.5}$*: While the scientific evidence identifies the fine fraction as the principal driver of human health effects from PM_{10}, our analysis focuses on PM_{10} due to availability of removal estimates by trees for a larger number of cities. However, the available evidence indicates that trees on average are similarly efficient at removing PM_{10} and $PM_{2.5}$ (Cowherd et al., 2006; Freer-Smith et al., 2005; Liu et al., 2015; Maher et al., 2013). Our mitigation and ROI results therefore can be approximately scaled to $PM_{2.5}$ using the ambient concentration ratios of the two size fractions.

4.3 The importance of targeting

Our local-level analysis reveals a > 10,000-fold difference in the ROI among prospective tree planting sites. Moreover, the variability in ROI among sites within a city exceeds variability of ROI among cities by over 1000-fold (Fig. 4). Our findings suggest large scope for maximizing PM_{10} and summer heat abatement for given budgets through ROI-based targeting of tree planting. The location of people relative to trees is crucial because in the turbulent atmospheric environment of the urban surface layer, both PM_{10} and heat mitigation effects generally experience rapid decay over fairly short distances. In all 27 cities the ROI of PM_{10} and heat abatement declines exponentially with total planted area. For example, in Phoenix (New York, Los Angeles), the ROI of a (ROI-maximizing) \$2M yr^{-1} reforestation portfolio for PM_{10} abatement is 1.0 (0.9, 1.1) $\mu g\, PM_{10} \cdot m^{-3}$·people per \$, > 3 (5, 4) times that of the 75%ile ROI site (SI M; Fig. 4). Thus, at anything less than full potential reforestation, ROI-based targeting is crucial to achieve maximum abatement for given budgets. Reforesting only high-ROI sites still can achieve high levels of abatement: the top 20% ROI sites in each city could provide 3.4–11.4 M people a > 2 $\mu g/m^3$ reduction in average annual PM_{10} concentrations and 1.7–12.7 M people a > 2 °C reduction in maximum summer temperatures, at a combined annual investment of \$102 M for all cities. Thus, even cities with relatively low median ROI will have many sites whose ROI exceeds that of the median site of other cities. Consequently, all the studied cities offer opportunities for tree cover increases with a high ROI for PM_{10} and heat abatement. These are generally sites in more densely populated areas where more people would benefit from the increased ecosystem service provision.

4.4. Cost-competitiveness with conventional solutions

Given municipal budget constraints, urban reforestation competes with conventional, "grey" solutions to PM and urban heat abatement such as point and non-point source PM emission controls and high-albedo "cool" surfaces, with cost-effectiveness (i.e., cost per unit of target outcome produced – the inverse of ROI) often a key selection criterion. While some studies suggest that trees can be cost-competitive with grey strategies for PM (Escobedo et al., 2008) and summer heat mitigation (McPherson & Simpson, 2003), cost-effectiveness comparisons between trees and grey alternatives along any single objective such as PM or heat mitigation are inherently biased against trees because they ignore the multi-benefit nature of trees: trees provide both PM and temperature mitigation while grey solutions provide one or the other. Moreover, trees and greenspace provide many co-benefits, including reduced flooding and water pollution from stormwater runoff (Bartens, Day, Harris, Dove, & Wynn, 2008; McPherson, Simpson, Xiao, & Wu, 2011; Wang, Endreny, & Nowak, 2008), psychological well-being (Krekel, Kolbe, & Wüstemann, 2016; Maas et al., 2009), cognitive development in children (Dadvand et al., 2015) and promotion of physical activity (Bell, Wilson, & Liu, 2008; West, Shores, & Mudd, 2012; Wolch et al., 2011).

Co-benefits can be accounted for in cost-effectiveness or ROI analyses by assigning a share of a tree project's costs to each of the various target objectives trees achieve (based for example on the relative importance weights of the objectives), and including only the relevant cost share when calculating the cost-effectiveness of trees for each target objective. To take a simple example, a city that has PM abatement and temperature mitigation as coequal goals would assign half the cost of tree planting to each objective. This would double the ROI of trees for each objective. Given the range of benefits urban trees provide (Escobedo, Kroeger, & Wagner, 2011; Livesley, McPherson, & Calfapietra, 2016; Roy, Byrne, & Pickering, 2012) and the priority urban challenges they mitigate in addition to air pollution and heat, such an approach would dramatically increase and more accurately reflect the true cost-competitiveness of trees. This underlines the need for integrated assessments of strategies that aim to address multiple urban challenges (Andersson-Sköld et al., 2015). Such assessments should consider both the desirable and undesirable impacts (e.g., production of allergenics or storm debris) of urban trees (Escobedo et al., 2011).

Comparisons of the cost-effectiveness of trees and grey PM and heat mitigation alternatives are complicated by several additional factors. First, conventional controls may use different cost-effectiveness metrics. For example, grey PM control measures are commonly assessed in terms of cost per unit of mass emissions avoided at the source rather than per unit of ambient concentration reduction or ambient PM mass removal (National Research Council, 2002). If a portion of the PM emissions from controlled sources leaves the target airshed or the air within the boundary layer is not perfectly mixed, a unit of ambient removal is equivalent to more than one unit of avoided emissions, and cost-effectiveness metrics must be adjusted accordingly.

Similarly, while increases in tree cover and high-albedo surfaces (light-colored, "cool" roofs and pavements) both lower surface and ambient air temperatures and form part of comprehensive urban heat mitigation strategies (Akbari et al., 2001; Rosenzweig et al., 2009; Taha et al., 1999), high-albedo surfaces achieve their cooling largely by transferring radiative loads from building façades or streets and sidewalks to surrounding areas. Thus, while trees lower the mean radiant temperature to which pedestrians are exposed (Andreou, 2014; Cohen, Potchter, & Matzarakis, 2012; Erell, Pearlmutter, Boneh, & Kutiel, 2014; Sanusi et al., 2016; Taleghani, Sailor, & Ban-Weiss, 2016) – the principal determinant of human thermal comfort in summer (Ketterer & Matzarakis, 2014; Mayer, Holst, Dostal, Imbery, & Schindler, 2008) – high-albedo surfaces can significantly increase it (Erell et al., 2014; Taleghani, Sailor, Tenpierik, & van den Dobbelsteen, 2014; Yang, Lau, & Qian, 2011). In areas with heavy pedestrian use high-albedo surfaces may best be suited for rooftops (Lynn et al., 2009). As in the PM example, the mitigation effects therefore are not fully functionally equivalent.

5. Conclusion

Our results show that while currently cities are losing tree cover, leading to greater PM concentrations and higher summer temperatures, there exists great scope for additional tree planting to reverse this trend. Moreover, in many cities trees can form part of a cost-effective multi-objective strategy that includes urban air pollution and heat mitigation. Crucially, site-level prioritization of tree planting for PM and summer heat abatement requires accounting for the number of people found in the proximity of prospective planting sites.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.landurbplan.2018.05.014.

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