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Air pollution removal by urban trees and shrubs in the United States

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Abstract

A modeling study using hourly meteorological and pollution concentration data from across the coterminous United States demonstrates that urban trees remove large amounts of air pollution that consequently improve urban air quality. Pollution removal (O_3, PM_{10}, NO_2, SO_2, CO) varied among cities with total annual air pollution removal by US urban trees estimated at 711,000 metric tons (\$3.8 billion value). Pollution removal is only one of various ways that urban trees affect air quality. Integrated studies of tree effects on air pollution reveal that management of urban tree canopy cover could be a viable strategy to improve air quality and help meet clean air standards.
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Keywords: Air quality; Urban forests; Urban forestry; Environmental quality

1. Introduction

Air pollution is a major environmental concern in most major cities across the world. An important focus of research has been on the role of urban vegetation in the formation and degradation of air pollutants in cities. Through the emission of volatile organic compounds (VOC), urban trees can contribute to the formation of ozone (O_3) (Chameides et al., 1988). However, more integrative studies are revealing that urban trees, particularly low VOC emitting species, can be a viable strategy to help reduce urban ozone levels (Cardelino and Chameides, 1990; Taha, 1996; Nowak et al., 2000), particularly through tree functions that reduce air temperatures (transpiration), remove air pollutants (dry deposition to plant surfaces), and reduce building energy and consequent power plant emissions (e.g., temperature reductions; tree shade). One study (Nowak et al., 2000) has concluded that for the US northeast coast, the physical effects of urban trees were more important than the chemical effects in terms of affecting ozone concentrations.

Nationally, urban trees and shrubs (hereafter referred to collectively as "trees") offer the ability to remove significant amounts of air pollutants and consequently improve environmental quality and human health. Trees remove gaseous air pollution primarily by uptake via leaf stomata, though some gases are removed by the plant surface. Once inside the leaf, gases diffuse into intercellular spaces and may be absorbed by water films to form acids or react with inner-leaf surfaces (Smith, 1990). Trees also remove pollution by intercepting airborne particles. Some particles can be absorbed into the tree, though most particles that are intercepted are retained on the plant surface. The intercepted particle often is resuspended to the atmosphere, washed off by rain, or dropped to the ground with leaf and twig fall. Consequently, vegetation is only a temporary retention site for many atmospheric particles.

To investigate the magnitude of air pollution removal by urban trees throughout the lower 48 United States, computer modeling of air pollution removal of carbon

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monoxide (CO), nitrogen dioxide (NO_2), ozone, particulate matter less than 10 μm (PM_{10}) and sulfur dioxide (SO_2) was performed for 55 US cities and for the entire nation based on meteorological, pollution concentration, and urban tree cover data. Due to the need for various assumptions within the model, the model provides a first-order estimate of the magnitude of pollution removal by urban trees.

Methods

For each city, the downward pollutant flux (F; in $g\,m^{-2}\,s^{-1}$) was calculated as the product of the deposition velocity (V_d; in $m\,s^{-1}$) and the pollutant concentration (C; in $g\,m^{-3}$) ($F = V_dC$). Deposition velocity was calculated as the inverse of the sum of the aerodynamic (R_a), quasi-laminar boundary layer (R_b) and canopy (R_c) resistances (Baldocchi et al., 1987). Hourly estimates of R_a and R_b were calculated using standard resistance formulas (Killus et al., 1984; Pederson et al., 1995; Nowak et al., 1998) and hourly weather data from nearby airports for 1994. R_a and R_b effects were relatively small compared to R_c effects.

Hourly canopy resistance values for O_3, SO_2, and NO_2 were calculated based on a modified hybrid of big-leaf and multilayer canopy deposition models (Baldocchi et al., 1987; Baldocchi, 1988). Canopy resistance (R_c) has three components: stomatal resistance (r_s), mesophyll resistance (r_m), and cuticular resistance (r_t), such that: $1/R_c = 1/(r_s + r_m) + 1/r_t$. Mesophyll resistance was set to zero $s\,m^{-1}$ for SO_2 (Wesely, 1989) and $10\,s\,m^{-1}$ for O_3 (Hosker and Lindberg, 1982). Mesophyll resistance was set to $100\,s\,m^{-1}$ for NO_2 to account for the difference between transport of water and NO_2 in the leaf interior, and to bring the computed deposition velocities in the range typically exhibited for NO_2 (Lovett, 1994). Base cuticular resistances were set at $8000\,s\,m^{-1}$ for SO_2, $10{,}000\,s\,m^{-1}$ for O_3, and $20{,}000\,s\,m^{-1}$ for NO_2 to account for the typical variation in r_t exhibited among the pollutants (Lovett, 1994).

As removal of CO and particulate matter by vegetation are not directly related to photosynthesis/transpiration, R_c for CO was set to a constant for in-leaf season ($50{,}000\,s\,m^{-1}$) and leaf-off season ($1{,}000{,}000\,s\,m^{-1}$) (Bidwell and Fraser, 1972). For particles, the median deposition velocity (Lovett, 1994) was set to $0.064\,m\,s^{-1}$ based on 50-percent resuspension rate (Zinke, 1967). The base Vd was adjusted according to in-leaf vs. leaf-off season parameters. To limit deposition estimates to periods of dry deposition, deposition velocities were set to zero during periods of precipitation.

Each city was assumed to have a single-sided leaf area index within the canopy covered area of 6 and to be 10% coniferous (Nowak, 1994). Leaf area index value is total leaf area (m^2: trees and large shrubs [minimum 1 in stem diameter]) divided by total canopy cover in city (m^2) and includes layering of canopies. Regional leaf-on and leaf-off dates were used to account for seasonal leaf area variation. Total tree canopy cover in each city was based on aerial photograph sampling (Nowak et al., 1996) or advanced very high resolution radiometer data (Dwyer et al., 2000; Nowak et al., 2001).

Hourly pollution concentration data (1994) from each city were obtained from the US Environmental Protection Agency (EPA). Missing hourly meteorological or pollution-concentration data were estimated using the monthly average for the specific hour. In some locations, an entire month of pollution-concentration data may be missing and are estimated based on interpolations from existing data. For example, O_3 concentrations may not be measured during winter months and existing O_3 concentration data are extrapolated to missing months based on the average national O_3 concentration monthly pattern. Data from 1994 were used due to available data sets with cloud cover information. To estimate percent air quality improvement due to dry deposition (Nowak et al., 2000), hourly boundary heights were used in conjunction with local deposition velocities for select cities with boundary layer height data. Daily morning and afternoon mixing heights from nearby stations were interpolated to produce hourly values using the EPA's PCRAMMIT program (US EPA, 1995). Minimum boundary-layer heights were set to 150 m during the night and 250 m during the day based on estimated minimum boundary-layer heights in cities. Hourly mixing heights (m) were used in conjunction with pollution concentrations ($\mu g\,m^{-3}$) to calculate the amount of pollution within the mixing layer ($\mu g\,m^{-2}$). This extrapolation from ground-layer concentration to total pollution within the boundary layer assumes a well-mixed boundary layer, which is common in the daytime (unstable conditions) (Colbeck and Harrison, 1985). Hourly percent air quality improvement was calculated as grams removed/(grams removed + grams in atmosphere), where grams in atmosphere = measured concentration ($g\,m^{-3}$) × boundary layer height (m) × city area (m^2).

To estimate pollution removal by all urban trees in the United States, national pollution concentration data (all EPA monitors) were combined with standardized local or regional pollution removal rates. Pollution removal rates ($g\,m^{-2}$ of tree cover) standardized to the average pollutant concentration in the city ($g\,m^{-2}$ per ppm or per $\mu g\,m^{-3}$). As flux rates are directly proportional to pollutant concentrations, standardized removal rates are used to account for concentration differences among urban areas.

For all urban areas in the United States outside of the 55 analyzed cities, local pollution monitoring data were used to calculate the average pollution concentration in the urban area for each pollutant. Urban area boundaries are based on 1990 census definitions of urbanized areas (areas with population density $\geq$ 1000 people mi^{-2}) and urban places (incorporated or unincorporated (census-defined) places with a population $\geq$ 2500) outside of urbanized areas. If pollutant monitors did not exist within the urban area, minimum state pollution concentration data were assigned to the urban area. Likewise, standardized pollution removal rates were assigned to each urban area based on data from the closest analyzed city within the same climate zone. All urban areas within a state were assigned to the dominant climate zone (cool temperate, Desert, Mediterranean, steppe, tropical, tundra, warm temperate) in the state, except for California and Texas where urban areas were individually assigned to one of multiple state climate zones.

For each urban area exclusive of the 55 analyzed cities, standardized pollution removal rates were multiplied by average pollutant concentration and total amount of tree cover to calculate total pollution removal for each pollutant in every urban area. Urban area pollution removal totals were combined to estimate the national total. Pollution removal value was estimated using national median externality values (Murray et al., 1994). Values were based on the median monetized dollar per ton externality values used in energy-decision-making from various studies. These values, in dollars per metric ton (t) are: $NO_2 = \$6752\,t^{-1}$, $PM_{10} = \$4508\,t^{-1}$, $SO_2 = \$1653\,t^{-1}$, and $CO = \$959\,t^{-1}$. Externality values for O_3 were set to equal the value for NO_2. Externality values can be considered the estimated cost of pollution to society that is not accounted for in the market price of the goods or services that produced the pollution.

Results and discussion

Total pollution removal and value varied among the cities from 11,100 t a^{-1} (\$60.7 million a^{-1}) in Jacksonville, FL to 22 t a^{-1} (\$116,000 a^{-1}) in Bridgeport, CT (Table 1). Pollution removal values per unit canopy cover varied from 23.1 g m^{-2} a^{-1} in Los Angeles, CA to 6.2 g m^{-2} a^{-1} in Minneapolis, MN. The median pollution removal value per unit canopy cover was 10.8 g m^{-2} a^{-1}.

Pollution removal values for each pollutant will vary among cities based on the amount of tree cover (increased tree cover leading to greater total removal), pollution concentration (increased concentration leading to greater downward flux and total removal), length of in-leaf season (increased growing season length leading to greater total removal), amount of precipitation (increased precipitation leading to reduced total removal via dry deposition), and other meteorological variables that affect tree transpiration and deposition velocities (factors leading to increased deposition velocities would lead to greater downward flux and total removal). All of these factors combine to affect total pollution removal and the standard pollution removal rate per unit tree cover.

Jacksonville's urban forest had the largest total removal, but had below median value of pollution removal per unit tree cover. Jacksonville's high total pollution removal value was due to its large city size (1965 km^2) and relatively high estimated percent tree cover within the city (53%). Los Angeles had the highest pollution removal values per unit tree cover due to its relatively long in-leaf season, relatively low precipitation, and relatively high pollutant concentrations and deposition velocities. Minneapolis had the lowest pollution removal values per unit tree cover due, in part, to its relatively short in-leaf season.

Average leaf-on daytime dry deposition velocities varied among the cities ranging from 0.44 to 0.29 cm s^{-1} for NO_2, 0.40 to 0.71 cm s^{-1} for O_3, and 0.38 to 0.69 cm s^{-1} for SO_2. Deposition velocities did not vary for CO and PM_{10} as deposition rates for these pollutants were not related to transpiration rates, but rates did vary based on leaf-off and leaf-on seasons. The deposition velocities for CO and PM_{10} were based on literature averages and assumed to be constant. The highest deposition velocities occurred in San Jose, CA; the lowest in Phoenix, AZ.

Though urban trees remove tons of air pollutants annually, average percent air quality improvement in cities during the daytime of the vegetation in-leaf season were typically less than 1 percent (Table 2) and varied among pollutants based on local meteorological and pollution concentration conditions. Percent air quality improvement was typically greatest for particulate matter, ozone, and sulfur dioxide. Air quality improvement increases with increased percent tree cover and decreased mixing-layer heights. In urban areas with 100% tree cover (i.e., contiguous forest stands), average air quality improvements during the daytime of the in-leaf season were around two percent for particulate matter, ozone, and sulfur dioxide. In some cities, short-term air quality improvements (one hour) in areas with 100% tree cover are estimated to be as high as 16% for ozone and sulfur dioxide, 9% for nitrogen dioxide, 8% for particulate matter, and 0.03% for carbon monoxide (Table 2).

These estimates of air quality improvement due to pollution removal likely underestimate the total effect of the forest on reducing ground-level pollutants because they do not account for the effect of the forest canopy in preventing concentrations of upper air pollution from reaching ground-level air space. Measured differences in

Table 1. Annual pollution removal (1994) by trees and associated value in 55 US cities

City		O_3		PM_{10}		NO_2		SO_2		CO		Total		
		(t)	$(g\,m^{-2})$	(t)	$(g\,m^{-2})$	(t)	$(g\,m^{-2})$	(t)	$(g\,m^{-2})$	(t)	$(g\,m^{-2})$	(t)	$(g\,m^{-2})$	($\$ \times 1000$)
Albany, NY[a]	T	43	3.4	30	2.4	13	1.0	7	0.5	1	0.1	94	7.4	524
	R	12–54	1.0–4.3	12–47	0.9–3.7	6–15	0.5–1.2	4–10	0.3–0.8	na	na	35–128	2.7–10.1	183–698
Albuquerque, NM[b]	T	78	2.4	84	2.6	37	1.2	34	1.1	11	0.3	244	7.6	1220
	R	26–123	0.8–3.9	33–132	1.0–4.1	19–48	0.6–1.5	22–69	0.7–2.1	na	na	111–382	3.5–12.0	501–1870
Atlanta, GA	T	514	4.6	423	3.8	135	1.2	93	0.8	35	0.3	1200	10.7	6470
	R	106–610	0.9–5.4	165–661	1.5–5.9	61–156	0.5–1.4	40–137	0.4–1.2	na	na	407–1600	3.6–14.2	1970–8410
Austin, TX	T	1150	5.2	680	3.1	436	2.0	374	1.7	138	0.6	2770	12.6	14500
	R	295–1510	1.3–6.9	266–1060	1.2–4.8	180–543	0.8–2.5	154–596	0.7–2.7	na	na	1030–3850	4.7–17.5	4790–19800
Baltimore, MD	T	158	4.0	148	3.7	107	2.7	50	1.3	13	0.3	477	12.1	2550
	R	39–214	1.0–5.4	58–232	1.5–5.9	43–133	1.1–3.4	25–84	0.6–2.1	na	na	178–676	4.5–17.1	869–3540
Baton Rouge, LA	T	508	4.8	369	3.5	133	1.3	97	0.9	50	0.5	1160	11.0	6200
	R	108–634	1.0–6.0	144–576	1.4–5.5	58–165	0.6–1.6	42–156	0.4–1.5	na	na	400–1580	3.8–15.0	1880–8300
Boston, MA	T	101	3.8	75	2.8	57	2.1	30	1.1	9	0.3	272	10.2	1470
	R	24–121	0.9–4.6	29–118	1.1–4.4	24–68	0.9–2.5	15–45	0.6–1.7	na	na	102–361	3.8–13.6	492–1890
Bridgeport, CT	T	8	3.4	7	2.8	4	1.6	2	0.8	1	0.3	22	9.0	116
	R	2–9	0.8–3.9	3–11	1.1–4.4	2–4	0.7–1.8	1–3	0.4–1.1	na	na	8–28	3.3–11.5	39–145
Buffalo, NY	T	72	3.7	41	2.1	20	1.0	29	1.5	3	0.2	165	8.5	858
	R	18–85	0.9–4.3	16–64	0.8–3.3	9–23	0.5–1.2	12–42	0.6–2.1	na	na	58–216	3.0–11.1	274–1090
Charleston, WV	T	98	2.7	71	1.9	33	0.9	37	1.0	4	0.1	243	6.7	1270
	R	26–153	0.7–4.2	28–111	0.8–3.1	16–44	0.4–1.2	21–71	0.6–1.9	na	na	95–383	2.6–10.5	448–1950
Cincinnati, OH	T	213	3.0	241	3.3	126	1.8	85	1.2	19	0.3	683	9.5	3530
	R	51–278	0.7–3.9	94–376	1.3–5.2	52–155	0.7–2.2	37–131	0.5–1.8	na	na	253–959	3.5–13.3	1200–4860
Cleveland, OH	T	191	3.7	240	4.7	76	1.5	72	1.4	13	0.3	592	11.6	3010
	R	46–227	0.9–4.5	94–375	1.8–7.4	34–89	0.7–1.7	31–107	0.6–2.1	na	na	218–811	4.3–15.9	1020–4010
Columbia, SC	T	523	4.1	376	3.0	92	0.7	171	1.3	49	0.4	1210	9.6	6170
	R	143–728	1.1–5.7	147–587	1.2–4.6	47–117	0.4–0.9	78–291	0.6–2.3	na	na	464–1770	3.7–14.0	2120–8880
Columbus, OH[c]	T	448	3.7	357	3.0	211	1.8	75	0.6	22	0.2	1110	9.2	6210
	R	113–613	0.9–5.1	140–558	1.2–4.6	87–260	0.7–2.2	40–121	0.3–1.0	na	na	401–1570	3.3–13.0	2070–8640
Dallas, TX	T	1,160	4.7	905	3.6	296	1.2	110	0.4	124	0.5	2600	10.5	14200
	R	286–1420	1.2–5.7	354–1410	1.4–5.7	129–347	0.5–1.4	52–163	0.2–0.7	na	na	945–3470	3.8–14.0	4600–18700
Denver, CO	T	213	2.1	321	3.1	181	1.8	76	0.7	22	0.2	813	7.9	4250
	R	66–313	0.6–3.0	125–510	1.2–4.8	92–232	0.9–2.2	43–131	0.4–1.3	na	na	349–1210	3.4–11.6	1720–6180
Detroit, MI	T	234	3.1	257	3.4	85	1.1	69	0.9	12	0.2	657	8.7	3440
	R	60–280	0.8–3.7	100–402	1.3–5.3	43–99	0.6–1.3	31–98	0.4–1.3	na	na	246–891	3.2–11.8	1210–4550
El Paso, TX	T	140	3.1	176	4.0	19	0.4	47	1.1	19	0.4	401	9.0	1960
	R	47–247	1.1–5.6	69–275	1.5–6.2	10–26	0.2–0.6	31–95	0.7–2.1	na	na	176–663	4.0–14.9	768–3260
Fresno, CA	T	221	5.1	292	6.7	69	1.6	28	0.6	13	0.3	622	14.3	3330
	R	66–390	1.5–8.9	114–456	2.6–10.4	31–97	0.7–2.2	16–58	0.4–1.3	na	na	239–1010	5.5–23.2	1210–5450
Honolulu, HI	T	467	6.6	354	5.0	36	0.5	37	0.5	43	0.6	937	13.3	5090
	R	111–525	1.6–7.5	138–553	2.0–7.9	13–42	0.2–0.6	14–53	0.2–0.8	na	na	319–1220	4.5–17.3	1530–6460
Houston, TX	T	779	4.5	817	4.7	356	2.0	317	1.8	73	0.4	2340	13.4	11900
	R	164–968	0.9–5.5	319–1280	1.8–7.3	143–433	0.8–2.5	126–489	0.7–2.8	na	na	826–3240	4.7–18.5	3790–16100
Indianapolis, IN	T	1280	4.5	958	3.4	298	1.0	317	1.1	62	0.2	2910	10.3	15500
	R	313–1630	1.1–5.7	374–1500	1.3–5.3	147–364	0.5–1.3	133–471	0.5–1.7	na	na	1030–4020	3.6–14.2	5070–21000

Table 1. (*continued*)

City		O_3		PM_{10}		NO_2		SO_2		CO		Total		
		(t)	(g m^{-2})	(t)	(g m^{-2})	(t)	(g m^{-2})	(t)	(g m^{-2})	(t)	(g m^{-2})	(t)	(g m^{-2})	(\$ × 1000)
Jacksonville, FL	T	5210	5.0	3570	3.4	1137	1.1	931	0.9	295	0.3	11100	10.7	60800
	R	1170–6270	1.1–6.0	1390–5580	1.3–5.4	452–1380	0.4–1.3	355–1400	0.3–1.3	na	na	3670–14900	3.5–14.3	18100–79400
Jersey City, NJ[d]	T	21	3.6	24	4.1	16	2.7	9	1.6	4	0.7	74	12.7	375
	R	5–28	0.8–4.8	9–38	1.6–6.5	7–19	1.1–3.3	4–15	0.8–2.6	na	na	29–103	5.0–17.8	131–515
Kansas City, MO	T	1150	4.0	940	3.3	209	0.7	236	0.8	82	0.3	2620	9.2	13900
	R	285–1400	1.0–4.9	367–1470	1.3–5.2	97–248	0.3–0.9	106–348	0.4–1.2	na	na	937–3550	3.3–12.5	4490–18400
Los Angeles, CA	T	1260	6.9	1470	8.0	1150	6.3	105	0.6	228	1.2	4210	23.1	23300
	R	191–1290	1.0–7.1	573–2290	3.1–12.6	377–1350	2.1–7.4	42–161	0.2–0.9	na	na	1410–5320	7.7–29.2	6700–28600
Louisville, KY	T	189	4.3	153	3.5	60	1.4	84	1.9	12	0.3	498	11.3	2520
	R	46–250	1.0–5.6	60–239	1.4–5.4	27–75	0.6–1.7	35–136	0.8–3.1	na	na	179–712	4.0–16.1	828–3510
Memphis, TN	T	726	4.8	547	3.6	352	2.3	150	1.0	62	0.4	1840	12.1	10100
	R	171–955	1.1–6.3	214–855	1.4–5.6	145–435	0.9–2.9	65–239	0.4–1.6	na	na	655–2550	4.3–16.8	3260–13700
Miami, FL	T	104	5.5	88	4.6	33	1.7	7	0.4	10	0.5	243	12.7	1350
	R	24–121	1.3–6.4	34–137	1.8–7.2	12–40	0.6–2.1	3–11	0.2–0.6	na	na	84–320	4.4–10.8	414–1740
Milwaukee, WI	T	157	3.3	105	2.2	55	1.2	23	0.5	9	0.2	350	7.4	1950
	R	39–180	0.8–3.8	41–165	0.9–3.5	25–63	0.5–1.3	11–32	0.2–0.7	na	na	125–449	2.6–9.5	645–2440
Minneapolis, MN	T	87	3.1	30	1.1	44	1.5	7	0.2	8	0.3	176	6.2	1040
	R	25–107	0.9–3.7	12–47	0.4–1.7	20–51	0.7–1.8	3–10	0.1–0.3	na	na	68–222	2.4–7.8	370–1300
Nashville, TN	T	2190	3.4	2180	3.4	799	1.2	523	0.8	186	0.3	5870	9.1	31000
	R	577–3060	0.9–4.7	850–3400	1.3–5.3	372–976	0.6–1.5	237–851	0.4–1.3	na	na	2220–8470	3.5–13.1	10800–44200
New Orleans, LA	T	467	4.8	471	4.8	162	1.6	99	1.0	63	0.6	1260	12.8	6600
	R	107–587	1.1–6.0	184–736	1.9–7.5	70–207	0.7–2.1	43–159	0.4–1.6	na	na	470–1750	4.8–17.8	2160–9000
New York, NY	T	491	3.7	493	3.7	478	3.6	232	1.7	97	0.7	1790	13.5	9240
	R	117–623	0.9–4.7	193–771	1.5–5.8	188–568	1.4–4.3	112–355	0.8–2.7	na	na	706–2410	5.3–18.2	3020–12200
Newark, NJ	T	30	3.6	40	4.8	23	2.7	9	1.1	3	0.3	105	12.5	556
	R	7–40	0.8–4.7	16–63	1.9–7.5	10–28	1.1–3.3	5–15	0.6–1.8	na	na	39–148	4.7–17.6	193–767
Oklahoma City, OK	T	1170	5.7	575	2.8	161	0.8	157	0.8	61	0.3	2120	10.4	11900
	R	284–1380	1.4–6.8	224–898	1.1–4.4	73–185	0.4–0.9	65–229	0.3–1.1	na	na	708–2760	3.5–13.6	3590–15100
Omaha, NE[e]	T	192	2.6	244	3.4	53	0.7	48	0.7	13	0.2	551	7.6	2850
	R	55–241	0.8–3.3	95–382	1.3–5.3	25–63	0.3–0.9	20–71	0.3–1.0	na	na	208–770	2.9–10.6	1020–3900
Philadelphia, PA	T	289	3.8	417	5.5	152	2.0	152	2.0	26	0.3	1040	13.7	5130
	R	76–379	1.0–5.0	163–651	2.2–8.6	67–184	0.9–2.4	74–240	1.0–3.2	na	na	405–1480	5.4–19.6	1930–7160
Phoenix, AZ	T	404	4.3	670	7.2	219	2.3	21	0.2	86	0.9	1400	15.0	7350
	R	129–802	1.4–8.6	262–1050	2.8–11.2	102–336	1.1–3.6	14–49	0.1–0.5	na	na	593–2320	6.3–24.8	2850–12600
Pittsburgh, PA	T	147	3.6	153	3.8	65	1.6	96	2.4	10	0.2	470	11.6	2290
	R	34–184	0.8–4.5	60–238	1.5–5.9	29–76	0.7–1.9	45–141	1.1–3.5	na	na	177–650	4.4–16.0	778–3080
Portland, OR	T	406	3.0	449	3.1	203	1.5	132	1.0	87	0.6	1280	9.2	6440
	R	98–582	0.7–4.3	175–701	1.3–5.2	92–261	0.7–1.9	59–240	0.4–1.8	na	na	511–1870	3.8–13.8	2250–9330
Providence, RI	T	56	3.7	55	3.7	17	1.2	14	1.0	3	0.2	146	9.8	769
	R	13–63	0.9–4.2	22–86	1.5–5.8	8–20	0.6–1.3	7–20	0.5–1.4	na	na	53–193	3.6–13.0	255–987
Roanoke, VA	T	134	4.6	118	4.0	29	1.0	32	1.1	7	0.2	319	11.0	1690
	R	32–169	1.1–5.8	46–184	1.6–6.3	14–35	0.5–1.2	14–51	0.5–1.7	na	na	113–446	3.9–15.3	549–2300
Sacramento, CA	T	170	4.9	134	3.8	49	1.4	12	0.3	13	0.4	378	10.8	2110
	R	42–255	1.2–7.3	53–210	1.5–6.0	23–70	0.6–2.0	6–22	0.2–0.6	na	na	136–569	3.9–16.3	698–3190

Table 1. (*continued*)

City		O_3		PM_{10}		NO_2		SO_2		CO		Total		
		(t)	($g m^{-2}$)	(t)	($g m^{-2}$)	(t)	($g m^{-2}$)	(t)	($g m^{-2}$)	(t)	($g m^{-2}$)	(t)	($g m^{-2}$)	($ × 1000)
Salt Lake City, UT	T	219	3.0	381	5.2	116	1.6	37	0.5	22	0.3	774	10.5	4060
	R	68–347	0.9–4.7	149–595	2.0–8.1	60–152	0.8–2.1	20–71	0.3–1.0	na	na	318–1190	4.3–16.2	1590–6190
San Diego, CA	T	549	7.6	401	5.6	202	2.8	56	0.8	48	0.7	1260	17.4	7020
	R	105–603	1.5–8.4	157–627	2.2–8.7	72–243	1.0–3.4	21–84	0.3–1.2	na	na	403–1610	5.6–22.2	1980–8730
San Francisco, CA	T	80	3.5	107	4.7	63	2.7	12	0.5	15	0.7	276	12.1	1480
	R	19–90	0.8–3.9	42–166	1.8–7.3	22–75	1.0–3.3	4–17	0.2–0.8	na	na	103–364	4.5–15.9	491–1910
San Jose, CA[f]	T	305	4.6	243	3.7	188	2.8	28	0.4	34	0.5	798	12.0	4500
	R	55–312	0.8–4.7	95–380	1.4–5.7	67–207	1.0–3.1	10–38	0.1–0.6	na	na	261–972	3.9–14.6	1300–5320
Seattle, WA[g]	T	105	3.3	98	3.1	47	1.5	48	1.5	21	0.6	319	10.0	1570
	R	23–136	0.7–4.3	38–153	1.2–4.8	21–60	0.7–1.9	20–78	0.7–2.5	na	na	123–448	3.9–14.1	525–2160
St. Louis, MO	T	115	3.3	133	3.8	53	1.5	63	1.8	9	0.3	373	10.7	1840
	R	29–150	0.8–4.3	52–207	1.5–6.0	24–65	0.7–1.9	30–103	0.9–3.0	na	na	144–534	4.1–11.5	650–2560
Tampa, FL	T	156	5.8	123	4.5	29	1.1	64	2.4	13	0.5	385	14.2	1920
	R	34–184	1.3–6.8	48–192	1.8–7.1	11–35	0.4–1.3	23–94	0.9–3.5	na	na	129–518	4.8–19.2	569–2510
Tucson, AZ	T	252	4.6	213	3.8	70	1.3	15	0.3	21	0.4	572	10.3	3190
	R	75–398	1.3–7.2	83–333	1.5–6.0	37–95	0.7–1.7	8–30	0.1–0.5	na	na	224–877	4.0–15.8	1160–4900
Tulsa, OK	T	266	5.3	182	3.6	46	0.9	58	1.2	18	0.4	569	11.4	3040
	R	63–322	1.3–6.5	71–284	1.4–5.7	21–56	0.4–1.1	25–83	0.5–1.7	na	na	196–762	3.9–15.3	940–4000
Virginia Beach-Norfolk, VA	T	1660	5.8	794	2.8	428	1.5	330	1.2	71	0.2	3280	11.5	18300
	R	420–2030	1.5–7.1	310–1240	1.1–4.3	184–516	0.6–1.8	152–503	0.5–1.8	na	na	1140–4360	4.0–15.3	5790–23700
Washington, DC	T	192	3.9	161	3.3	100	2.0	81	1.6	24	0.5	558	11.3	2850
	R	46–251	0.9–5.1	63–252	1.3–5.1	42–119	0.9–2.4	37–125	0.7–2.5	na	na	212–771	4.3–15.6	963–3860

Total (T) and range (R) are given in metric tons (t) and in grams per square meter of canopy cover ($g m^{-2}$) for ozone (O_3), particulate matter less than 10 μm (PM_{10}), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and carbon monoxide (CO). The monetary value of pollution removal by trees is estimated using the median externality values for the United States for each pollutant (Murray et al., 1994). Bounds of total tree removal of O_3, NO_2, SO_2, and PM_{10} were estimated using the typical range of published in-leaf dry deposition velocities (Lovett, 1994).

[a] NO_2 data from Buffalo, NY
[b] SO_2 data from El Paso, TX
[c] NO_2 data from Cincinnati, OH
[d] Weather data from Newark, NJ
[e] NO_2 data from Kansas City, MO
[f] Weather data from San Francisco, CA
[g] NO_2 data from Portland, OR

Table 2. Estimated percent air quality improvement in selected US cities due to air pollution removal by urban trees

City	%tree cover	% air quality improvement				
		CO	NO_2	O_3	PM_{10}	SO_2
Atlanta, GA	32.9	0.002 (0.001–0.009)	0.5 (0.1–2.5)	0.7 (0.1–4.4)	0.7 (0.3–2.8)	0.7 (0.1–4.3)
Boston, MA	21.2	0.002 (0.000–0.006)	0.4 (0.0–1.8)	0.6 (0.1–3.4)	0.6 (0.1–1.8)	0.5 (0.1–3.4)
Dallas, TX	28.0	0.002 (0.001–0.008)	0.4 (0.1–2.2)	0.6 (0.1–3.9)	0.6 (0.2–2.4)	0.6 (0.1–3.8)
Denver, CO	26.0	0.001 (0.000–0.007)	0.2 (0.0–1.5)	0.3 (0.0–2.1)	0.4 (0.1–2.2)	0.3 (0.0–2.0)
Milwaukee, WI	19.1	0.001 (0.000–0.005)	0.3 (0.0–1.5)	0.4 (0.1–2.7)	0.4 (0.1–1.6)	0.4 (0.0–2.7)
New York, NY	16.6	0.001 (0.000–0.005)	0.3 (0.0–1.4)	0.4 (0.1–2.6)	0.5 (0.1–1.4)	0.4 (0.1–2.6)
Portland, OR	42.0	0.003 (0.001–0.012)	0.6 (0.1–2.7)	0.8 (0.1–3.7)	1.0 (0.3–3.5)	0.7 (0.1–4.0)
San Diego, CA	8.6	0.001 (0.000–0.002)	0.2 (0.0–0.7)	0.3 (0.0–1.4)	0.3 (0.1–0.7)	0.3 (0.0–1.4)
Tampa, FL	9.6	0.001 (0.000–0.003)	0.2 (0.0–0.8)	0.2 (0.0–1.4)	0.2 (0.1–0.8)	0.2 (0.0–1.4)
Tucson, AZ	13.7	0.001 (0.000–0.004)	0.1 (0.0–1.0)	0.1 (0.0–1.7)	0.2 (0.1–1.2)	0.1 (0.0–1.7)
Washington, DC	31.1	0.002 (0.001–0.009)	0.4 (0.2–2.3)	0.6 (0.1–3.9)	0.7 (0.2–2.6)	0.6 (0.1–3.9)

Estimates are given for actual tree cover conditions in city for ozone (O_3), particulate matter less than 10 μm (PM_{10}), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and carbon monoxide (CO) based on local boundary layer height and pollution removal estimates. Bounds of total tree removal of O_3, NO_2, SO_2, and PM_{10} were estimated using the typical range of published in-leaf dry deposition velocities (Lovett, 1994)

O_3 concentration between above- and below-forest canopies in California's San Bernardino Mountains have exceeded 50 ppb (40-percent improvement) (Bytnerowicz et al., 1999). Under normal daytime conditions, atmospheric turbulence mixes the atmosphere such that pollutant concentrations are relatively consistent with height (Colbeck and Harrison, 1985). Forest canopies can limit the mixing of upper air with ground-level air, leading to significant below-canopy air quality improvements. However, where there are numerous pollutant sources below the canopy (e.g., automobiles), the forest canopy could have the inverse effect by minimizing the dispersion of the pollutants away at ground level.

The greatest effect of urban trees on ozone, sulfur dioxide, and nitrogen dioxide is during the daytime of the in-leaf season when trees are transpiring water. Particulate matter removal occurs both day and night and throughout the year as particles are intercepted by leaf and bark surfaces. Carbon monoxide removal also occurs both day and night of the in-leaf season, but at much lower rates than for the other pollutants.

Urban areas are estimated to occupy 3.5% of lower 48 states with an average canopy cover of 27%. Urban tree cover varies by region within the United States with cities developed in forest areas averaging 34.4% tree cover, cities in grassland areas: 17.8%, and cities in deserts: 9.3% (Dwyer et al., 2000; Nowak et al., 2001). Total pollution air removal (5 pollutants) by urban trees in coterminous United States is estimated at 711,000 t, with an annual value of $3.8 billion (Table 3).

Though the estimates given in this paper are only for a 1-year period (1994), analysis of changes in meteorology and pollution concentration on pollution removal by urban trees over a 5-year period in Chicago (1991–1995) reveals that annual removal estimates were within 10% of the 5-year average removal rate. Estimates of pollution removal may be conservative as some of the deposition-modeling algorithms are based on homogenous canopies. As part of the urban tree canopy is heterogeneous with small patches or individual trees, this mixed canopy effect would tend to increase pollutant deposition. Also, aerodynamic resistance estimates may be conservative and lead to a slight underestimate of pollution deposition.

Though the average percent air quality improvement due to trees is relatively low (<1%), the improvement is for multiple pollutants and the actual magnitude of pollution removal can be significant (typically hundreds to thousands of metric tons of pollutants per city per

Table 3. Air pollution removal and value for all urban trees in the coterminous United States

Pollutant	Removal (t)	Value ($\$ \times 10^6$)
O_3	305,100	2,060
	(75,000–390,200)	(506–2635)
PM_{10}	214,900	969
	(84,000–335,800)	(378–1514)
NO_2	97,800	660
	(42,800–119,100)	(289–804)
SO_2	70,900	117
	(32,200–111,100)	(53–184)
CO	22,600	22
	na	Na
Total	711,300	3828
	(256,600–978,800)	(1,249–5158)

Estimates are given for ozone (O_3), particulate matter less than 10 μm (PM_{10}), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and carbon monoxide (CO). The monetary value of pollution removal by trees is estimated using the median externality values for the United States for each pollutant (Murray et al., 1994). Externality values for O_3 were set to equal the value for NO_2. Bounds of total tree removal of O_3, NO_2, SO_2, and PM_{10} were estimated using the typical range of published in-leaf dry deposition velocities (Lovett, 1994).

year). Percent air quality improvement estimates are likely conservative and can be increased through programs to increase canopy cover within cities. Air pollution removal is also only one aspect of how urban trees affect air quality. Ozone studies that integrate temperature, deposition and emission effects of trees are revealing that urban trees can have significant effects on reducing ozone concentrations (Cardelino and Chameides, 1990; Taha, 1996; Nowak et al., 2000). Based in part on these findings, the US Environmental Protection Agency has introduced urban tree cover as a potential emerging measure to help meet air quality standards (US EPA, 2004). So even though the percent air quality improvement from pollution removal by trees may be relatively small, the total effect of trees on air pollution can produce impacts that are significant enough to warrant consideration of tree cover management as a means to improve air quality.

Conclusion

Through pollution removal and other tree functions (e.g., air temperature reductions), urban trees can help improve air quality for many different air pollutants in cities, and consequently can help improve human health. While the existing percent air quality improvements due to pollution removal by urban trees are modest, they can be improved by increasing urban tree canopy cover. The combined total effects of trees on air pollutants are significant enough that urban tree management could provide a viable means to improve air quality and help meet clean air standards in the United States.

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