Environmental Science & Technology

pubs.acs.org/est

Article

Testing Removal of Carbon Dioxide, Ozone, and Atmospheric Particles by Urban Parks in Italy

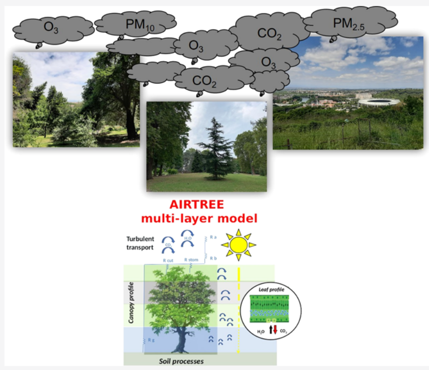
Silvano Fares,* Adriano Conte, Alessandro Alivernini, Francesco Chianucci, Mirko Grotti, Ilaria Zappitelli, Fabio Petrella, and Piermaria Corona

Cite This: Environ. Sci. Technol. 2020, 54, 14910−14922

Read Online

ACCESS | Metrics & More | Article Recommendations | Supporting Information

ABSTRACT: Cities are responsible for more than 80% of global greenhouse gas emissions. Sequestration of air pollutants is one of the main ecosystem services that urban forests provide to the citizens. The atmospheric concentration of several pollutants such as carbon dioxide (CO_2), tropospheric ozone (O_3), and particulate matter (PM) can be reduced by urban trees through processes of adsorption and deposition. We predict the quantity of CO_2, O_3, and PM removed by urban tree species with the multilayer canopy model AIRTREE in two representative urban parks in Italy: Park of Castel di Guido, a 3673 ha reforested area located northwest of Rome, and Park of Valentino, a 42 ha urban park in downtown Turin. We estimated a total annual removal of 1005 and 500 kg of carbon per hectare, 8.1 and 1.42 kg of ozone per hectare, and 8.4 and 8 kg of PM_{10} per hectare. We highlighted differences in pollutant sequestration between urban areas and between species, shedding light on the importance to perform extensive in situ measurements and modeling analysis of tree characteristics to provide realistic estimates of urban parks to deliver ecosystem services.



■ INTRODUCTION

Air quality in the Mediterranean cities is compromised by the expansion of urban population with serious implication for human health. Exposure to particles has been correlated with premature mortality worldwide.[1] In Europe, despite the air quality policies have produced improvements in the last few years, about 17% of the EU-28 urban population was exposed to PM_{10} above the EU daily limit value for the reference year 2017.[2] In Italy, exposure to fine particles and ozone is responsible for over 66,000 premature deaths every year and 19.5 times higher than road accidents in 2017.[3] Particles are generated mainly by residential energy use such as cooking and heating, but also by vehicular emissions, power generation, and agriculture.[4] By 2100, at midlatitudes of the Northern Hemisphere, the mean global background O_3 concentration is expected to increase from the current level of 35−50 to 85 ppb.[5,6] According to the analysis by Sicard et al. (2018),[7] the annual mean concentrations of tropospheric ozone (O_3) grew on average by 0.16 ppb per year in cities across the globe over the period 1995−2014. Future strategies of urban development will necessarily have to consider additional measures to ameliorate air quality to improve human health and well-being. Together with adoption of a new technology in the energy sector and mobility, it is now clear that urban and periurban parks can deliver a multitude of ecosystems services and, therefore, can provide a significant contribution to the air quality improvement.[8,9] Several studies demonstrated that the concentration of air pollutants primarily emitted from vehicular traffic and industrial combustion processes is lower in the middle of parks, compared with industrial areas or near heavy traffic roads.[10,11] Therefore, urban parks provide cleaner air to citizens, and in general air quality measurements in urban parks can be adopted as reference of local background levels.[12]

Dry deposition represents the main pathway in plant ecosystems, especially in the dry Mediterranean region, where vegetation has been described to remove pollutants from the atmosphere.[13] This "sink" capacity of plants results from interactions between meteorology, chemical, and physical characteristics of the pollutants and the properties of the canopy.[14] Carbon dioxide (CO_2) represents the main anthropogenic greenhouse gas emission, mainly coming from heating of buildings, vehicular emissions, and cooking. Urban trees can sequestrate CO_2 through photosynthesis, and store carbon as biomass in plants and soil. As previously demonstrated in urban-to-rural comparisons showing lower CO_2 concentration in the presence of vegetation,[15,16] urban parks in the city may slightly decrease concentrations of atmospheric CO_2 locally, especially when they are away from strong emitting sources.

Received: July 16, 2020
Revised: October 28, 2020
Accepted: October 28, 2020
Published: November 10, 2020

Table 1. NPP, Tropospheric Ozone (O_3), and Particle (PM_{10} and $PM_{2.5}$) Dry Deposition Simulated by the AIRTREE Model for the Year 2018 at Castel di Guido Natural Reserve[a]

species	dbh (cm)	NPP (g m^{-2})	NPP class	O_3 (g m^{-2})	O_3 class	PM_{10} (g m^{-2})	PM_{10} class	$PM_{2.5}$ (g m^{-2})	$PM_{2.5}$ class
A. campestre	35	354.23 ± 38.76	IV	2.97 ± 0.02	V	1.01 ± 0.0482	II	0.09 ± 0.0041	I
Acer negundo	15	46.6	I	2.75	V	0.77	I	0.06	I
A. cordata	35	438.59 ± 39.9	V	3.27 ± 0.05	VI	1 ± 0.0405	II	0.08 ± 0.0034	I
C. atlantica	35	938.24 ± 128.36	X	5.67 ± 0.33	X	7.39 ± 1.272	VIII	1.01 ± 0.1739	X
C. australis	35	392.79	IV	2.8	V	0.93	I	0.08	I
C. sempervirens	55	1084.6	X	7.4	X	16.23	X	2.27	X
Eucalyptus spp.	55	490.78	V	3.34	VI	1.44	II	0.12	II
Fraxinus angustifolia	15	253.71 ± 79.24	III	2.19 ± 0.09	IV	0.83 ± 0.1507	I	0.07 ± 0.0125	I
F. ornus	35	562.24 ± 95.6	VI	2.88 ± 0.26	V	1.42 ± 0.1802	II	0.12 ± 0.0155	II
Juglans nigra	15	140.48 ± 126.39	II	2.17 ± 0.09	IV	0.82 ± 0.1714	I	0.07 ± 0.0145	I
Juglans regia	35	370.26	IV	2.51	V	1.05	II	0.09	I
Malus sylvestris	35	225.31	III	2.02	IV	0.66	I	0.06	I
Ostrya carpinifolia	35	528.65 ± 91.68	VI	2.85 ± 0.25	V	1.32 ± 0.1603	II	0.11 ± 0.0138	II
Pinus eldarica	35	704.89 ± 97.32	VIII	6.19 ± 0.61	X	9.58 ± 2.4196	X	1.31 ± 0.333	X
Pinus halepensis	55	894.86	IX	6.67	X	13.73	X	1.88	X
Pinus pinaster	55	847.47	IX	6.46	X	12.01	X	1.65	X
Pinus pinea	55	794.22 ± 30.14	VIII	6.39 ± 0.12	X	10.86 ± 0.7685	X	1.49 ± 0.1055	X
Populus nigra	15	83.25	I	2.01	IV	0.69	I	0.06	I
Prunus avium	35	375.44	IV	2.76	V	1.12	II	0.1	I
Pyrus amygdaliformis	15	27.91	I	2.15	IV	0.66	I	0.06	I
Pyrus pyraster	35	319.2	IV	2.57	V	1.08	II	0.09	I
Q. cerris	35	412.27 ± 51.97	V	2.89 ± 0.21	V	1.21 ± 0.1192	II	0.1 ± 0.0103	I
Quercus frainetto	35	332.5 ± 16.7	IV	2.54 ± 0.09	V	1.12 ± 0.0405	II	0.1 ± 0.0035	I
Q. ilex	35	656 ± 162.43	VII	3.43 ± 0.48	VI	2.79 ± 0.7179	III	0.31 ± 0.0803	IV
Q. pubescens	35	486.99 ± 61.35	V	2.98 ± 0.22	V	1.21 ± 0.1192	II	0.1 ± 0.0103	I
Q. robur	15	250.37 ± 77.66	III	2.45 ± 0.12	IV	0.7 ± 0.1171	I	0.06 ± 0.01	I
Quercus suber	35	854.12 ± 87.2	IX	3.52 ± 0.2	VII	2.09 ± 0.1795	III	0.23 ± 0.0201	III
Quercus trojana	15	159.36 ± 80.58	II	2.24 ± 0.08	IV	0.56 ± 0.1356	I	0.05 ± 0.0115	I
R. pseudoacacia	35	474.66	V	2.65	V	1.16	II	0.1	I
Sorbus domestica	15	189.59	II	2.07	IV	1.01	II	0.09	I

[a]Model simulations were carried out for each species at different dbh. We grouped results according the highest dbh group. The groups were: 15 (dbh ranging from 5 to 15 cm), 35 (dbh ranging from 20 to 35 cm), and 55 (dbh ranging from 40 to 55 cm). SD is shown in cases where more dbh classes were present within each group. Evergreen species are marked in bold.

While the photosynthetic process and carbon assimilation have been widely investigated, plant leaves can additionally absorb pollutants when they penetrate through stomata.[17] Even particles can be intercepted by vegetation and retained on the surface of leaves, trunks bark, or can be absorbed into plant tissues.[18] Ozone deposition has been described in several agricultural and forest ecosystems, displaying two separate sinks: leaf stomata, and plant/soil surfaces.[19] Green surfaces have indeed an added value versus nonvegetation surfaces because the stomatal sink can represent on average 45% of total sequestration with peaks up to 70% thanks to the capacity of leaves to detoxify this pollutant once it penetrates inside intercellular spaces.[19] As leaves and crowns are the active interface between plant and atmosphere, canopy attributes like leaf area index (LAI) strongly influence plant ability to intercept atmospheric particles and gases. Previous studies clearly show that forest ecosystems can sequestrate particles,[20] but deposition is still poorly investigated especially in the Mediterranean region, also considering that the particle size and shape greatly influence deposition on plant surfaces.[21] It is clear that LAI, hairiness, and wax content affect deposition, but also meteorological variables (precipitation, solar radiation, humidity, wind speed, temperature, and turbulence) have an impact on the magnitude of deposition velocity and thus the capacity of plants to ameliorate air quality.[22,23] Urban planning and management (plant species or planting configuration) has also an impact on dry deposition at the stand/regional level.[23] Because most particles are deposited on leaves, higher deposition can be expected on evergreen species rather than on deciduous species.

Urban forest tree monitoring and assessment are rapidly evolving in a historic period in which new techniques and tools become rapidly available. However, there is a lack of evidence-based realistic estimates of the ecosystem services delivered by urban parks.[24] Models traditionally used to predict particulate matter (PM) removal are often based on empirical meteorological algorithms, which still deserve to be tested against a multitude of forest ecosystems and rely on a number of dendrometric attributes in order to provide realistic estimations.[8,18,25−27]

In this study, we performed two field campaigns in two relevant urban parks in Rome and Turin (Italy), and measured structural tree properties and ecophysiological indicators of a wide range and class distribution of typical tree species used in the Mediterranean regions in streets and parks. We adopted input parameters collected in situ into a canopy model (AIRTREE[26]) to estimate CO_2, O_3, and PM sequestration by thousands of trees. Our study was designed to improve knowledge of structural and ecophysiological characteristics of a wide range of urban trees to support realistic estimation of

https://dx.doi.org/10.1021/acs.est.0c04740
Environ. Sci. Technol. 2020, 54, 14910−14922

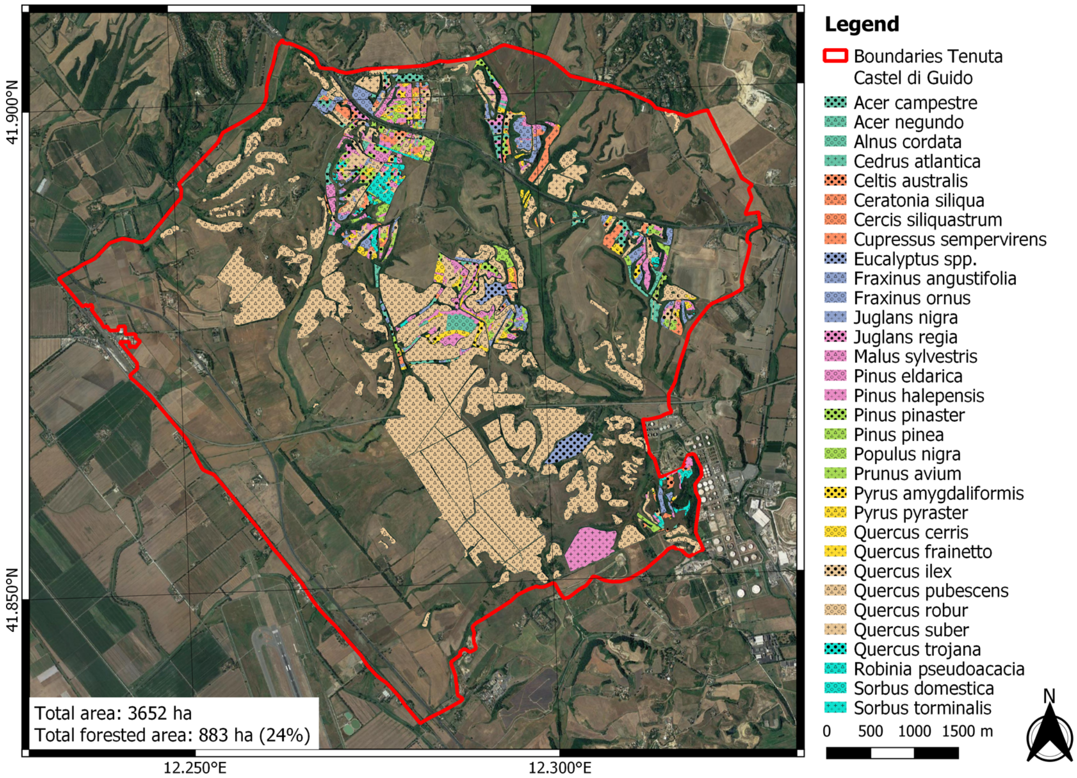


Figure 1. Map of the vegetation surveyed at the park of Castel di Guido, Rome. Map data 2020 Google.

ecosystem services provided by urban parks by resolving three critical working questions: (1) to what extent two urban parks with different exposure to anthropogenic pollutants and under different climatic conditions can sequestrate of CO_2, ozone, and PM from the atmosphere? (2) Are deposition rates of these pollutants significantly different between species? (3) Are deposition rates relevant for air quality amelioration?

MATERIALS AND METHODS

Study Sites. The first study site is the Castel di Guido estate (hereafter named "Castel di Guido") located within the Rome Municipality (41°54′ N, 12°17′ E) covering around 3673 ha along the Tyrrhenian coast. Castel di Guido is included in the "Litorale Romano" State Natural Reserve and incorporates the "Macchiagrande di Galeria" Natura 2000 Special Area of Conservation (SAC), and an Important Bird Area (IBA), the "Oasi Castel di Guido". The climate is mainly Mediterranean (i.e., Xerotheric Bioclimatic Region of Latium and Thermo-Mediterranean/Meso-Mediterranean Climate Subregion[28]). Such climate may potentially support various forests dominated by mesophytic and thermophilus broad-leaved tree species. The predominant soil types are calcic cambisols.[29] Castel di Guido is characterized by a forest planted in the 80s−90s by the Municipality, with the intent to limit urban expansion, using both native (mainly *Quercus* spp.) and non-native (mainly *Pinus* spp. and *Cedrus* spp.) species, which have been localized in monospecific stands to obtain fast soil coverage. Other species, such as *Acer campestre* L., *Celtis australis* L., and *Fraxinus ornus* L., have been planted on smaller areas (down to 60 m^2) in order to increase structural and compositional diversity of the plantation project. A full list of species is provided in Table 1. A map with each species is reported in Figures 1 and S1.

In the second part of the XIX century, the City of Turin focused its urban greening areas politics on development of new spaces that could respond to citizenship's needs in terms of health, urban hygiene, and comfortable walking, so the area around "Castello del Valentino" was selected to build a public park. With an area of 42 ha, the second study site is the Valentino Park (hereafter named "Valentino"), which is the second largest park of Turin. Located along the west bank of the Po river (45°3′N, 7°41′E), it was designed and built from 1633 to 1660 as a private garden of the "Castello del Valentino", one of the Savoy family's royal residences. The park is an "English landscape garden" with small hills, large meadows, flowerbeds, fountains, little streams, historic buildings, and with about 1800 tall trees mainly represented by beeches, elms, ginkgos, hornbeams, limes, maples, oaks, plane trees, and, poplars, wingnuts, sequoias, and willows. Tree species are listed in Table 2 and mapped in Figure 2. The climate is mainly temperate[28] and according to FAO World Soil Resources, the predominant soil types are leptic technic fluvisols.[30]

In both cities, climatic and air quality data are continuously monitored by a network of micrometeorological stations installed and maintained by the Regional Agency for Environmental Protection (ARPA). For the first study site, we used data collected by a monitoring station located inside the Castel di Guido estate (41°53′25.1″ N, 12°15′57.7″ E), while in the case of Valentino, data collected by the nearest monitoring station (45° 4′24.59″N, 7°41′34.43″E) was used.

Measurements of Structural Parameters. In both parks, field surveys collected tree species, diameter at breast height, total tree height, canopy cross-radii and height, LAI, and health state. Estimates of tree canopy attributes like tree crown cover and LAI were derived using digital cover photography (DCP). This is a restricted, upward-looking optical method described

Table 2. NPP, Tropospheric Ozone (O_3), and Particle (PM_{10} and $PM_{2.5}$) Dry Deposition Simulated by the AIRTREE Model for the Year 2018 at Valentino Urban Park[a]

species	dbh (cm)	NPP (g m^{-2})	NPP class	O_3 (g m^{-2})	O_3 class	PM_{10} (g m^{-2})	PM_{10} class	$PM_{2.5}$ (g m^{-2})	$PM_{2.5}$ class
A. alba	55	611.16	VII	0.73	I	13.82	X	2.21	X
Abies nordmanniana	55	686.03	VII	1.12	II	13.11	X	2.11	X
A. campestre	75	358.45	IV	1.19	II	2.97	III	0.39	IV
Acer negundo	55	327.59 ± 37.58	IV	0.8 ± 0.08	I	3.11 ± 0.311	IV	0.41 ± 0.041	V
Acer palmatum	55	248.59	III	0.56	I	2.79	III	0.37	IV
A. platanoides	55	308.2 ± 30.52	IV	0.9 ± 0.18	I	2.54 ± 0.03	III	0.34 ± 0.004	IV
Acer rubrum	35	132.07	II	0.43	I	1.98	II	0.26	III
Acer saccharum	95	260.01	III	0.86	I	1.88	II	0.25	III
Aesculus hippocastanum	115	327.4	IV	0.85	I	3.52	IV	0.46	V
Alnus glutinosa	55	275.9	III	0.67	I	3.57	IV	0.53	VI
Carpinus betulus	55	259.42 ± 12.19	III	0.67 ± 0.06	I	2.55 ± 0.032	III	0.34 ± 0.004	IV
C. atlantica	115	686.15	VII	1.06	II	27.51	X	4.61	X
Cedrus deodara	95	701.01 ± 41.95	VIII	1.26 ± 0.32	II	25.19 ± 0.135	X	4.23 ± 0.047	X
Cedrus glauca	75	572.77	VI	0.87	I	18.77	X	3.13	X
C. australis	155	214.19	III	1.19	I	1.46	II	0.19	II
Cercis siliquastrum	55	247.61	III	0.64	I	2.4	III	0.32	IV
Chamaecyparis lawsoniana	55	647.91	VI	0.84	I	18.4	X	3.12	X
Corylus avellana	55	384.37	IV	1.83	III	2.46	III	0.33	IV
Criptomeria japonica	55	699.28	VII	1.01	II	13.21	X	2.09	X
F. sylvatica	135	224.59	III	0.74	I	2.19	III	0.29	III
Fraxinus excelsior	75	215.64	III	0.93	I	1.77	II	0.24	III
Ginkgo biloba	95	208.17	III	1	II	2.12	III	0.28	III
Ilex aquifolium	35	269.42	III	0.9	I	6.97	VII	1.09	X
Juglans nigra	55	208.47 ± 41.08	III	0.77 ± 0.11	I	2.15 ± 0.449	III	0.29 ± 0.06	III
Lagerstroemia indica	15	53.36	I	1.52	IV	5.62	VI	0.88	IX
Libocedrus decurrens	55	337.97 ± 36.06	IV	1.06 ± 0.4	II	16.33 ± 0.201	X	2.7 ± 0.057	X
Liquidambar styraciflua	75	309.18	IV	1	II	2.45	III	0.32	IV
L. tulipifera	75	291.83 ± 12.59	III	1.16 ± 0.09	II	2.45 ± 0.005	III	0.32 ± 0.001	IV
Magnolia grandiflora	55	241.63	III	0.56	I	11.47	X	1.79	X
Magnolia obovata	55	259.3	III	0.58	I	11.47	X	1.79	X
Malus	15	58.43	I	0.57	I	0.75	I	0.1	I
Malus floribunda	15	54.99	I	0.44	I	0.75	I	0.1	I
Metasequoia glyptostroboides	95	679.01	VII	1	II	27.73	X	4.73	X
P. tomentosa	75	389.73	IV	0.9	I	14.68	X	2.3	X
P. abies	55	384.65	IV	0.78	I	16.64	X	2.74	X
Picea omorica	55	383.97	IV	0.79	I	16.59	X	2.73	X
Picea orientalis	35	255.95 ± 19.61	III	0.57 ± 0.02	I	9.96 ± 1.03	X	1.63 ± 0.17	X
Picea pungens	55	376.08 ± 19.12	IV	0.74 ± 0.03	I	15.69 ± 0.782	X	2.58 ± 0.13	X
Pinus excelsa	55	411.6	V	0.88	I	15.55	X	2.57	X
Pinus strobus	75	444.19	V	0.81	I	17.08	X	2.82	X
Pinus sylvestris	55	483.4	V	1.24	II	19.22	X	3.23	X
Platanus acerifolia	135	311.33 ± 30.96	IV	1.12 ± 0.08	II	2.16 ± 0.23	III	0.29 ± 0.03	III
Platanus hybrida	175	359.67	IV	1.17	II	2.95	III	0.39	IV
Platanus occidentalis	175	361.56	IV	0.92	I	2.94	III	0.39	IV
Platanus orientalis	115	337.19	IV	1.19	II	2.39	III	0.32	IV
Populus alba	115	226.16	III	0.96	I	1.91	II	0.25	III
Populus italica	115	195.6	II	0.7	I	1.9	II	0.25	III
Prunus	55	217.11	III	0.7	I	2.46	III	0.33	IV
P. avium	35	150.22	II	0.58	I	1.91	II	0.25	III
P. cerasifera	15	108.25	II	0.62	I	1.42	II	0.19	II
Prunus kanzan	15	128.43	II	1.06	II	1.45	II	0.19	II
Prunus pissardi	15	71.16	I	0.64	I	1.42	II	0.19	II
P. menziesii	55	640.81 ± 8.13	VII	0.83 ± 0.01	I	15.64 ± 0.655	X	2.57 ± 0.106	X
Quercus peduncolata	115	317.21	IV	1.06	II	2.58	III	0.34	IV
Q. pubescens	15	74.53	I	0.73	I	1.06	II	0.14	II
Q. robur	135	317.36	IV	0.94	I	2.53	III	0.34	IV
Q. rubra	115	324.42	IV	1.22	II	2.26	III	0.3	III
R. pseudoacacia	55	319.24 ± 6.86	IV	0.95 ± 0.04	I	2.67 ± 0.079	III	0.36 ± 0.011	IV
Salix babylonica	35	197.9	II	0.57	I	2.42	III	0.32	IV

https://dx.doi.org/10.1021/acs.est.0c04740
Environ. Sci. Technol. 2020, 54, 14910−14922

Table 2. continued

species	dbh (cm)	NPP (g m^{-2})	NPP class	O_3 (g m^{-2})	O_3 class	PM_{10} (g m^{-2})	PM_{10} class	$PM_{2.5}$ (g m^{-2})	$PM_{2.5}$ class
Sophora japonica	115	221.42	III	0.96	I	2.76	III	0.38	IV
Sterculia platanifolia	75	110.29	II	0.76	I	2.46	III	0.33	IV
T. distichum	95	761.59	VIII	1.07	II	27.62	X	4.71	X
Tilia argentea	75	281 ± 19.79	III	1 ± 0.15	II	2.56 ± 0.114	III	0.34 ± 0.015	IV
T. cordata	55	247.26	III	0.8	I	2.3	III	0.3	III
Tilia hybrida	75	307.98	IV	1.14	II	2.66	III	0.35	IV
Tilia platyphyllos	95	246.19	III	0.93	I	2.1	III	0.28	III
Ulmus pumila	35	264.99	III	0.73	I	2.23	III	0.3	III
Zelkova carpinifolia	95	260.05	III	0.72	I	2.16	III	0.29	III

[a]Model simulations were carried out for each species at different dbh. We grouped results according the highest dbh group. The groups were: 15 (dbh ranging from 5 to 15 cm), 35 (dbh ranging from 20 to 35 cm), 55 (dbh ranging from 40 to 55 cm), 75 (dbh ranging from 60 to 75 cm), 100 (dbh ranging from 80 to 100 cm), 115 (dbh ranging from 105 to 115 cm), 135 (dbh ranging from 120 to 135 cm), 155 (dbh ranging from 140 to 155 cm), and 175 (dbh ranging from 140 to 175 cm). SD is shown in cases where more dbh classes were present within each group. See Supporting Information for a complete list of values associated with dbh ranges. Evergreen species are marked in bold.

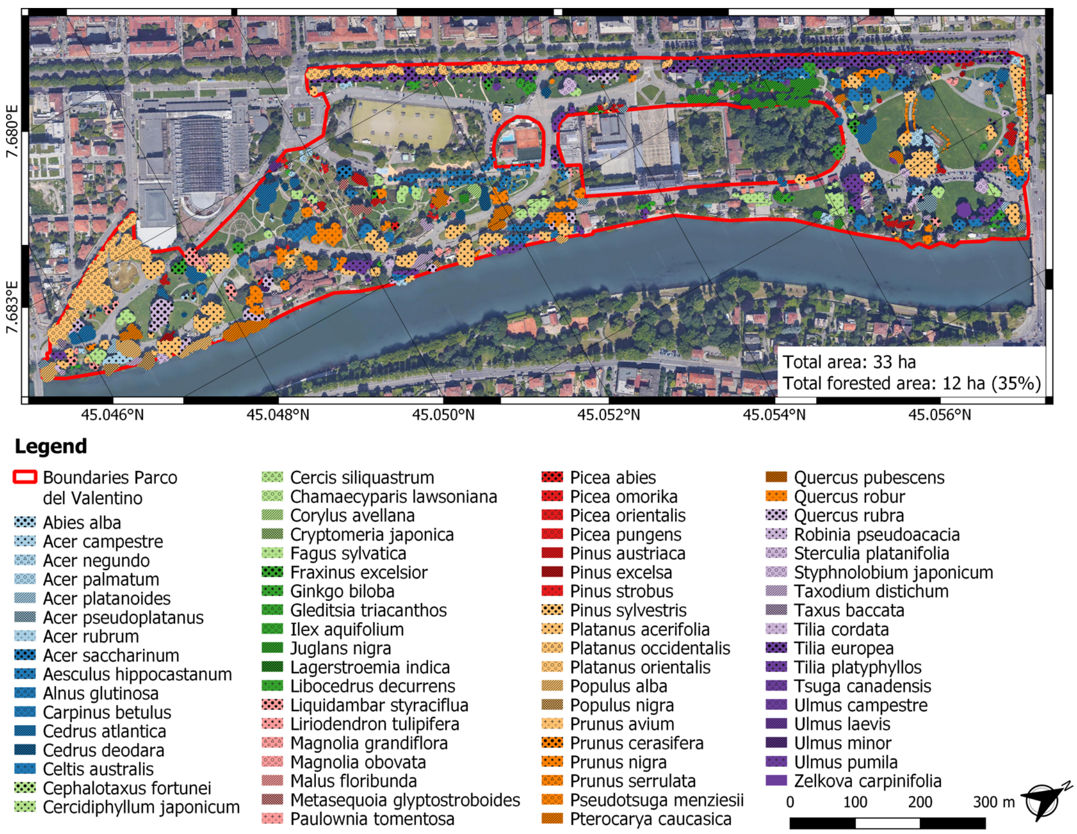


Figure 2. Map of the vegetation surveyed at the park of Valentino, Turin. Map data 2020 Google.

by Macfarlane et al. (2007)[31] and Chianucci (2020).[32] Images were acquired with a digital single-lens reflex camera (Canon EOS750D) equipped with a 35 mm lens, which yielded a restricted field of view (FOV 43° along the diagonal). Measurements were conducted at 198 trees representative of 16 different tree species at Valentino, while 2644 trees have been surveyed at Castel di Guido in February−March 2018 in a network of 70 permanent plots to sample most widespread combinations of species groups and environmental factors, according to a stratified random sampling design with strata defined by species groups and sample size in each stratum.[33] A complete formalism adopted to calculate LAI is reported in the Supporting Information.

Gas Exchange Measurements. Where not available in the literature (Table S1), gas exchange was measured in the field using a portable photosynthesis system (LI-6400, LI-COR, Inc., USA) with a 6 cm^2 leaf chamber. Measurements were made on fully expanded leaves from 09:00 to 11:00 a.m. For all measurements, photosynthetically active radiation was maintained at 1500 μmol m^{-2} s^{-1} with red and blue LEDs integrated into the leaf chamber fluorometer (LI6400-40). Leaf temperature was measured using a thermocouple touching the abaxial side of the leaf. CO_2 partial pressures were regulated using a 6400-01 CO_2 mixer. Five CO_2 response curves of net photosynthetic rates were measured for each species at the temperature of 25 °C following suggestions provided by Long and Bernacchi (2003).[34] The relative humidity in the chamber

https://dx.doi.org/10.1021/acs.est.0c04740
Environ. Sci. Technol. 2020, 54, 14910−14922

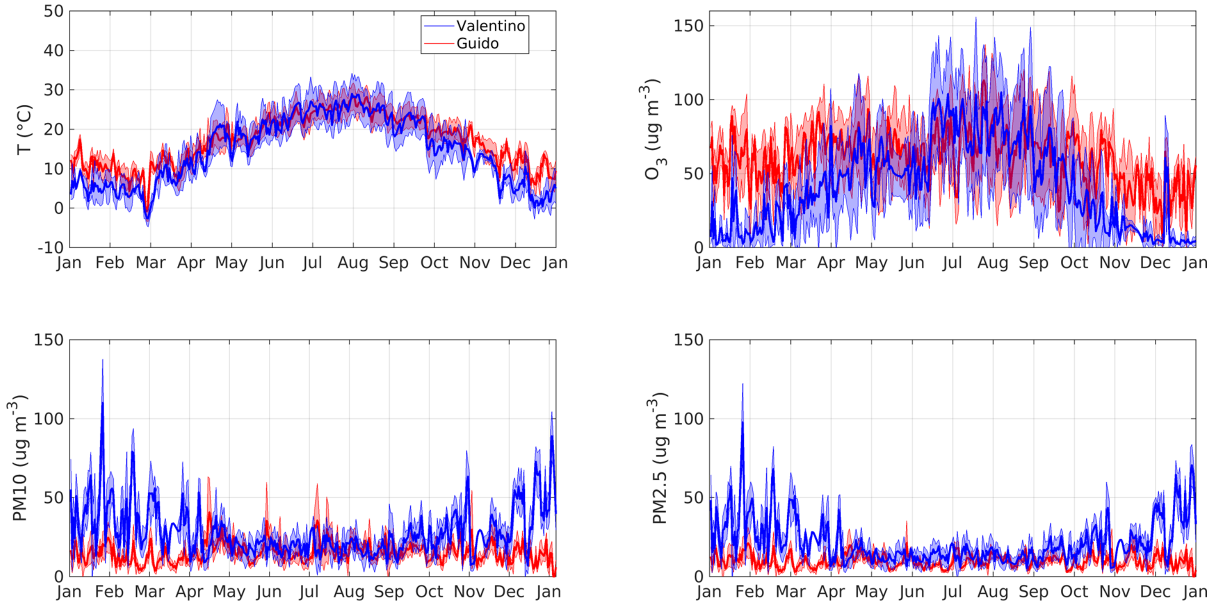


Figure 3. Daily average dynamics (±SD) of temperature, PM, and ozone concentrations at the urban parks of Valentino and Castel di Guido.

was maintained between 40 and 70%. From these data, $A-C_i$ response curves were fitted with the FvCB model by using the fitaci function from the Plantecophys R Package[35] to provide parameters for the biochemical model useful to estimate Vc_{max} (maximum rate of carboxylation) and J_{max} (maximum rate of electron transport) in the calculator provided by Sharkey (2016).[36]

AIRTREE Model Parameterization. Estimates of net ecosystem productivity, ozone, and particle sequestration were performed thanks to the AIRTREE model (aggregated interpretation of the energy balance and water dynamics for ecosystem services assessment[26]). AIRTREE is a one-dimensional multilayer model, which couples soil, plant, and atmospheric processes and minimizes the energy balance throughout these compartments. The solar radiation (PPFD + NIR, near infrared radiation), air temperature, relative humidity, wind speed, and CO_2, O_3, and PM concentrations retrieved from the meteorological stations were used as the model input.

Canopy photosynthesis was calculated according to the multilayer scheme described in detail by Fares et al. (2019)[26] and reported in detail in the Supporting Information. In particular, gross photosynthetic flux density of a single leaf including the contribution of photorespiration was calculated from the minimum value between the carboxylation rate when ribulose bisphosphate (RuBP) carboxylase/oxygenase is saturated and the carboxylation rate when RuBP regeneration is limited by electron transport. A reducing factor to account for photosynthetic limitation imposed by water stress was applied according to Keenan et al. (2010).[37] Net primary productivity (NPP) reported here was calculated as the difference between GPP (gross primary productivity, i.e., the integration of gross photosynthetic flux along the canopy profile) and the autotrophic respiration. The modules associated with AIRTREE are described in detail in the recently published article by Fares et al. (2019),[26] which also shows model performance and sensitivity analysis. Each species was studied for its distribution in the class diameter in order to provide a more realistic estimate of LAI and sequestration of pollutants driven by the total leaf area.

Deposition of ozone was estimated in analogy with an Ohm circuit according the formalism described by Zhang et al. (2002),[38,39] while the PM fluxes were calculated according the sum of deposition processes associated with interception, impaction, and gravitational settling as reported by Han et al. (2020)[18] and explained in detail in the Supporting Information. We emphasize here that PM collection by leaves can abate PM concentrations in the atmosphere, but in urban areas, additional processes caused by trees may limit PM deposition: under certain conditions, winds can promote PM resuspension, thereby slowing down collection processes onto plant surfaces, and friction offered by canopies can slow down dispersion. Unlike winds, rainfall events can cause PM to be washed off leaves and onto the ground, which represents a net removal of PM from the atmosphere.[40] In this study, the formalism we adopted does not specifically represent wash off events.

To calculate percent removal in the air column above parks (assuming an even distribution of pollutants under unstable atmospheric conditions), we estimated the height of the boundary layer by using a simple model, which estimates the lifting condensation level, which is the height of the cloud base at which relative humidity of an air parcel reaches 100% when it is cooled by adiabatic lifting of dry air as described by Karl et al. (2013)[41] and Stull (1997).[42]

In order to facilitate discussion of results and comparison between species in both parks, we identified a ranking procedure by selecting 10 classes (in ranges of 0.1 kg m^{-2} a^{-1}, 1 g m^{-2} a^{-1}, 1 g m^{-2} a^{-1}, and 0.1 g m^{-2} a^{-1} for NPP, ozone, PM_{10}, and $PM_{2.5}$, respectively) common to both parks to identify low (I) to high (X) fluxes of NPP, ozone, and PM. These classes are reported in both maps and tables.

■ RESULTS

Pollutant Concentrations at the Urban Parks. Concentration of pollutants varied during the year 2018 in both parks (Figure 3). Ozone was higher in Spring, Fall, and Winter at Castel di Guido compared to Valentino with daily average values often exceeding 50 $\mu g\ m^{-3}$ also during Winter. In particular, ozone followed a typical hourly trend (Figure 4)

https://dx.doi.org/10.1021/acs.est.0c04740
Environ. Sci. Technol. 2020, 54, 14910−14922

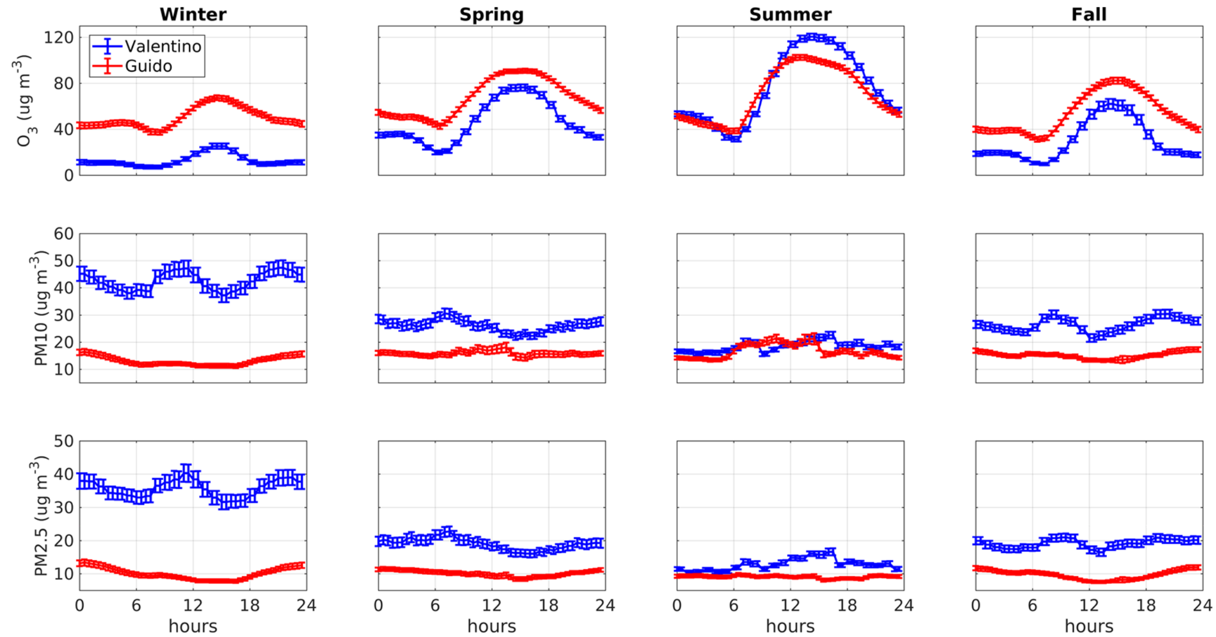


Figure 4. Hourly average dynamics (±SD) of PM and ozone concentrations at the urban parks of Castel di Guido and Valentino for each season of the year.

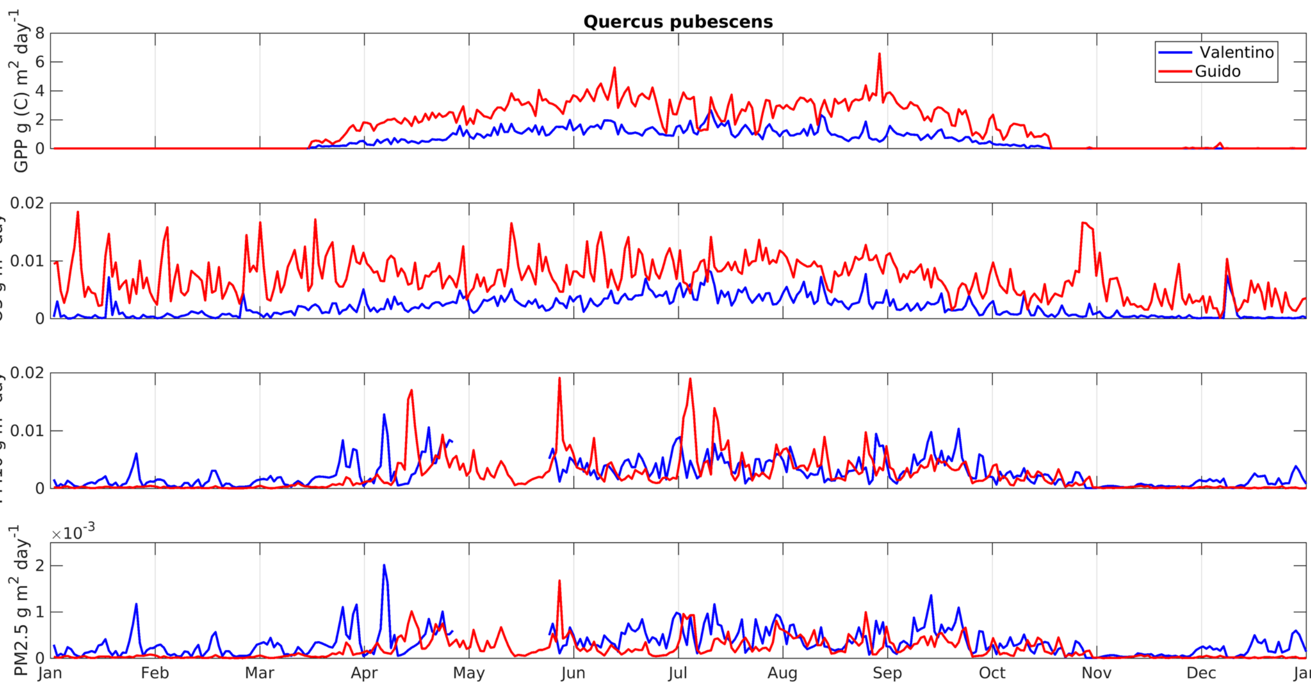


Figure 5. Example of daily dynamics of GPP, ozone, and PM fluxes for a tree species occurring at both Castel di Guido and Valentino urban parks. Positive values mean vertical fluxes from the atmosphere to the canopy.

with peak values in the central hours of the day as previously measured in the vicinity of Castel di Guido site.[43] In Summer, higher ozone concentrations at Valentino reflected the abundance of ozone precursors and the high temperatures recorded at the site (Figure 4). The periurban park of Castel di Guido displayed a lower concentration of PM relative to Valentino (PM_{10} rarely exceeded daily values of 50 $\mu g\ m^{-3}$, which is a daily limit according to the national regulation), where Winter peaks exceeding 100 $\mu g\ m^{-3}$ were recorded. These high PM concentrations are because of the close vicinity to some emission sources, considering that this park is in the center of the urban area of Turin (Figures S1 and S2). Seasonal averages (Figure 4) showed net differences in PM concentrations between the two parks: the $PM_{2.5}$ concentration was always higher at Valentino reflecting the close vicinity to the emission sources, while the PM_{10} concentration was generally higher at Valentino and of comparable magnitude with Castel di Guido only during Summer. PM concentration dynamics in both parks reflect vehicular traffic intensities in all seasons particularly in the morning (7 to 11 a.m.) and in the late afternoon (after 6 p.m.) with nocturnal levels still high because of boundary layer shallowing in the stable atmosphere.

https://dx.doi.org/10.1021/acs.est.0c04740
Environ. Sci. Technol. 2020, 54, 14910−14922

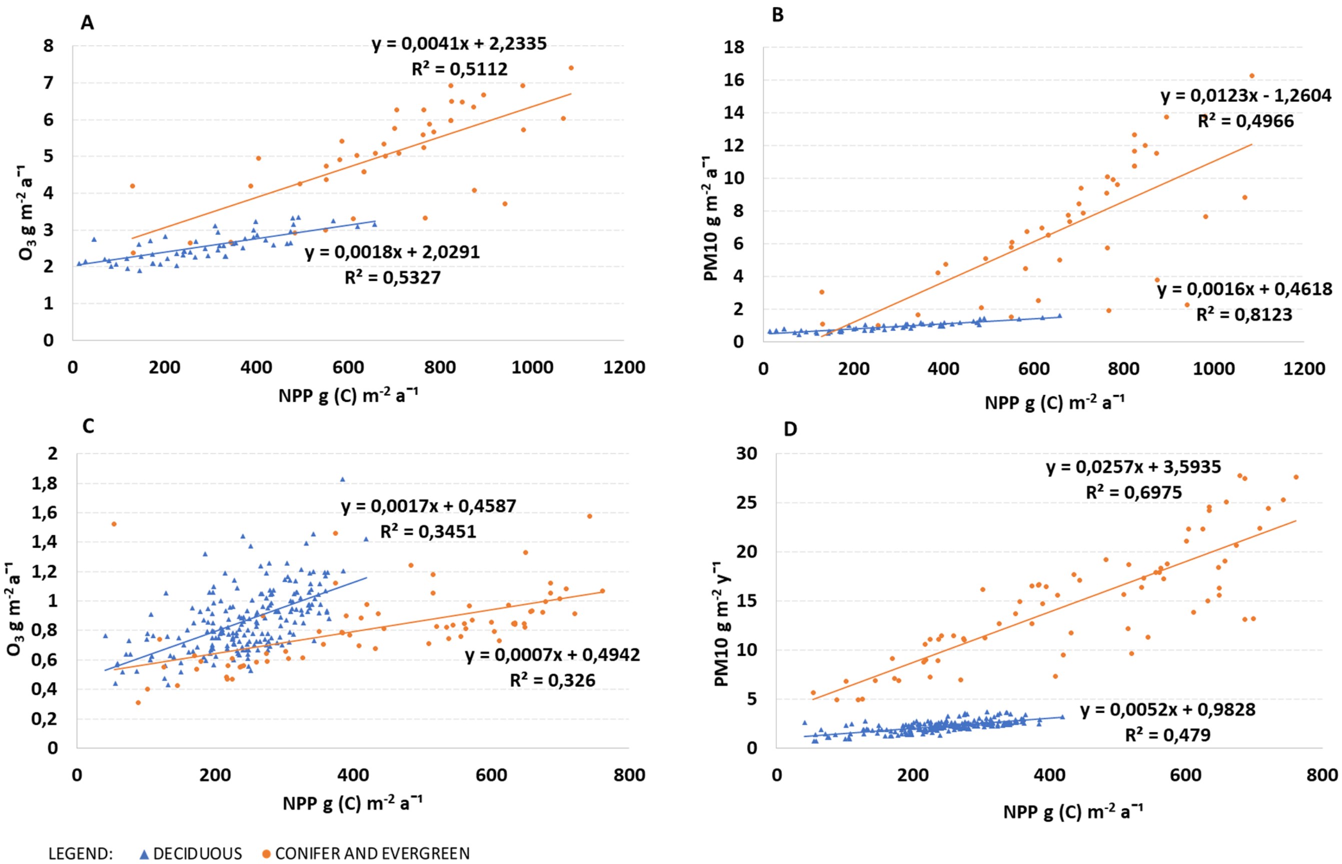


Figure 6. Correlation of NPP with ozone and PM for Castel di Guido (subplots A, B) and Valentino (subplot C, D) urban parks.

Pollutant Sequestration by Urban Trees. AIRTREE simulations revealed that tree species exhibit different capacity to sequestrate CO_2 based on their ecophysiological and structural characteristics. At Castel di Guido, evergreen species such as *Pinus* spp. and *Cedrus atlantica* showed the highest NPP with values close to class X (arbitrarily established >1 kg (C) m^{-2} a^{-1}) (Table 1, Figure S1). Broadleaves at Castel di Guido generally displayed higher values of NPP compared with those at Valentino, as shown by comparison of *Quercus pubescens* and *A. campestre*, two species with the same diameter class present in both parks (Figure 5). At Valentino, evergreen species such as *Abies alba*, *Pseudotsuga menziesii*, and *Taxodium distichum* exhibited the highest NPP values, exceeding 600 g (C) m^{-2} yr^{-1} ranking above class VI of our list of species (Table 2, Figure S2). AIRTREE allowed to estimate potential GPP under the condition of a well-watered environment (i.e., no drought stress). We found a 7% reduction in GPP and ozone deposition because of photosynthetic limitations especially during Summer under condition of low levels of the soil water content, as shown for *Quercus robur* (Figure S3). In total, the AIRTREE results showed that the parks are able to sequestrate CO_2 with values of 3875 and 22.6 t (C) a^{-1} for Castel di Guido and Valentino, which translates in 1.05 and 0.5 t (C) ha^{-1} for the two parks, respectively (Tables S3 and S8).

In both parks, AIRTREE simulations produced higher deposition of ozone and PM for evergreen species compared with broadleaf species (Figure 6), with NPP values generally higher for evergreen than broadleaf species. The capacity to sequestrate ozone was higher in Castel di Guido compared to Valentino. This is mainly because of higher ozone concentrations recorded in Castel di Guido, in particular during Winter, Spring, and Fall (Figure 4). At Castel di Guido, ozone fluxes for broadleaf species with high stomatal ozone fluxes such as *Quercus* spp., *Eucalyptus* spp., and *Alnus cordata* reached classes VI−VII (Figure S4), in agreement with ozone fluxes measured with the eddy covariance technique at a *Quercus ilex* forest nearby.[43] However, pines reached class X (>6 g (O_3) m^{-2} a^{-1}) most likely because of their high LAI, and longer vegetative period. At Valentino, values rarely exceeded class II (Figure S5) with broadleaf and fast-growing species such as *Tilia hybrida*, *Quercus rubra*, and *Platanus* spp. reaching the highest values above 1 g (O_3) m^{-2} a^{-1}. In order to better compare the two parks, we also show velocity of ozone deposition for each species and diameter class (Tables S5 and S10) grouped for daytime values in each season. Looking at two common species in both parks (*Q. pubescens* and *A. campestre*), it emerges that differences in magnitudes still occur even after removing the effect of ozone concentration, with values of ozone deposition velocity approximately twice as high as at Castel di Guido compared with Valentino (i.e., VdO_3 = 0.029 cm s^{-1} during Spring daytime vs 0.015 cm s^{-1} for *A. campestre*, and VdO_3 = 0.019 cm s^{-1} during Spring daytime vs 0.008 cm s^{-1} for *Q. pubescens*). In total, AIRTREE results showed that the parks are able to remove ozone with values of 30,275 and 66 kg (O_3) a^{-1} (8.1 and 1.42 kg (O_3) ha^{-1}) for Castel di Guido and Valentino, respectively (Tables S3 and S9), and both parks minimally contribute to ameliorate air quality (Figure S7) with around 1% especially during the central hours of the day in Spring and Fall (Figure S8).

Simulations showed that evergreen species have the highest levels of PM sequestration (Tables 1 and 2). At Castel di Guido, because of lower exposure to PM, only *Pinus* spp. and

Cupressus sempervirens reached class X concerning PM_{10} and $PM_{2.5}$ removal. The deposition simulations for Valentino showed high values above 10 and 1 g m^{-2} a^{-1} for PM_{10} and $PM_{2.5}$, respectively. Apart from the higher levels of PM (Figures 3 and 4), this results from the abundance of many conifers with high LAI all year long such as *Abies* spp., *Pinus* spp., and *Cedrus* spp., complemented by the deciduous species *Paulownia tomentosa*, a broadleaf species characterized by its large amount of foliage (LAI = 5.85, Table S1). Similar to ozone, deposition velocity could help disentangling the effect of PM concentrations on the PM deposition process. Restricting the comparisons to the same two species occurring in both sites, it emerged a different conclusion compared with ozone deposition for both *A. campestre* and *Q. pubescens* (PM_{10} values of 0.13 cm s^{-1} vs 0.20 cm s^{-1} during Spring daytime for both species for Castel di Guido and Valentino, respectively, Tables S5 and S11). This can be explained because AIRTREE does not take into account plants' ecophysiological status in response to PM exposure (i.e., there is no stomatal uptake for PM). Similar conclusions can be drawn for $PM_{2.5}$ (Tables S6 and S12). In total, the AIRTREE results showed that the parks are able to sequestrate PM_{10} with values of 31,788 kg and 342 kg a^{-1} and PM 2.5 with values of 4012 kg and 51.9 kg a^{-1} for Castel di Guido and Valentino, respectively (Tables S3 and S9). Our simplified simulation suggested that both parks could contribute to ameliorate air quality (Figure S6) by removing ozone and PM on average up to 10% percent. In particular, Valentino displayed the highest amount of removal during the night hours in all seasons, while Castel di Guido displayed a lower amount of removal in the light hours (Figure S8).

DISCUSSION

Differences between the Two Parks. Our hypothesis (1) was answered as the two parks display different features because of different climate conditions in the north and in central Italy, and their position relative to the urban area. In addition to high foliar biomass and LAI, high NPP values were often associated with high values of Vc_{max} (Tables S1 and S8), previously described as the main drivers of carbon assimilation through an AIRTREE sensitivity analysis performed in a *Q. ilex* forest.[26] In general, warmer temperatures and higher levels of insulation closer to photosynthetic optimum promoted higher growth rates at Castel di Guido, as shown for *Q. pubescens* and *A. campestre*, present in both parks (Figures 6, S6). Higher ozone concentrations at Castel di Guido are explained by the different local climatic conditions (Figure S1): Rome has a milder temperature than Turin especially in cold seasons (Figures 3, S2) and photochemistry driving ozone formation is more pronounced by the higher irradiation and temperatures, but also by abundance of precursors (VOC and nitrogen oxides). During Summer, the city of Turin experienced high temperature, this, together with abundance of precursors, may explain the higher ozone concentrations up to 120 μg m^{-3} observed during light hours. Ozone fluxes in Castel di Guido were generally higher than Valentino and are not only explained by the higher atmospheric concentration levels. Indeed, ozone deposition velocities for Castel di Guido were also higher (Tables S6 and S12). Reason for this difference lies in the turbulence levels promoting vertical gas exchange recorded at both sites, with turbulence levels recorded at Castel di Guido approximately twice as high as at Valentino because of the well-described land-sea breeze regime of northern Rome.[26] We also observed most favorable conditions (i.e., higher solar radiation and temperatures in Spring and Winter) for stomatal ozone uptake at Castel di Guido. AIRTREE evaluation showed that stomatal uptake represents an important ozone sink for both conifers and broadleaf species (Figure S9), with values close to 50% of total annual ozone fluxes, in agreement with field measurements, which showed a higher ratio between stomatal and nonstomatal fluxes.[19] High dependence of stomatal ozone fluxes (and stomatal conductance) on meteorological parameters that stimulate photosynthesis explains the higher fluxes at Castel di Guido. The simulated trends in ozone, CO_2, and PM removal rates suggested to correlate pollutants with NPP as a suitable metrics of pollutant sequestration activity. Figure 6, therefore, represents a way to interpret fluxes of pollutants with respect to assimilated carbon as a possible link between plant ecophysiology and dendrometric features of trees. Concerning ozone, there was a positive correlation between ozone and NPP explained by the amount of foliage, which in turn represents ozone sink by stomatal and nonstomatal deposition. The figure reveals no differences between plant types because the absence of leaves outside the vegetative period for broadleaves was compensated by high ozone fluxes during Spring−Summer months driven by a higher stomatal activity (overall average of 0.002 g O_3 per gram of carbon).

In agreement with previous findings,[18,40] the model simulations confirmed that conifers have in general a higher capacity to sequestrate PM in both Parks (approximately 0.02 g PM_{10} removed per gram of carbon) quantified in an order of magnitude higher than broadleaf species. The amount of foliage that in turn promotes photosynthesis and triggers PM deposition explains the positive correlation between PM and NPP. To some extent, NPP may be affected by exposure to high levels of PM as recently described by Łukowski et al. (2020)[44] with damage to the photosynthetic apparatus. However, in our study, we did not contemplate possible stress to the photosynthetic apparatus, in part because the load of PM exposure in our test sites may not be so high to shade leaves or impair stomatal aperture (at least at the periurban site) as it happens in highly polluted environments and in laboratory tests, in part because PM on leaf surfaces may not be harmful or even represent a source of nutrients.[45]

Performances of the AIRTREE Model. Few models in the world are able to calculate air purification services provided by urban parks with large financial benefits, which so far have been mostly estimated for north American urban parks.[46,47] The AIRTREE model adopted in this study was specifically developed for predicting CO_2 and pollutant deposition in Mediterranean forests;[26] therefore, our work supports evaluation of these benefits providing more empirical observations in order to perform future socioeconomic analysis for Mediterranean tree species. Thanks to this species-specific model parameterization, we tested our hypothesis (2) and demonstrated that the ability to capture atmospheric pollutants is species-specific and based on morphological and physiological (CO_2 assimilation and stomatal conductance) plant traits.

Other models such as the EMEP/MSC-W model[27] and UFORE (i-Tree)[8] model use semiempirical parameters such as LAI, canopy size, and meteorology in a resistance scheme analogy to predict carbon and pollutant deposition on urban trees. EMEP in particular incorporates the DO3SE Jarvis-type algorithm to estimate stomatal conductance in response to environmental parameters,[48] while AIRTREE is based on a more mechanistic representation of photosynthesis and stomatal conductance in response to meteorological conditions. EMEP and UFORE models were tested on an industrial-urban green area in Northern Italy.[49] These models estimated a removal capacity of O_3 of 0.42 g m^{-2} a^{-1} and a PM_{10} deposition of 0.79 and 0.76 g m^{-2} a^{-1}, respectively. These values are reported for *Tilia* spp. and *C. australis* and are in the same order of magnitude than those found in this study (around 1 and 2.5 g m^{-2} a^{-1} for *Tilia* spp. for ozone and PM_{10}, respectively, and 1.19 and 1.46 g m^{-2} a^{-1} for *C. australis* for ozone and PM_{10}, respectively). At a broader national scale, Manes et al. (2016)[9] calculated deposition rates using i-Tree[8] for a broad range of tree species. As regards PM_{10} deposition, forests with predominance of *Q. ilex* presented the highest rate, with a mean value of 1.93 g m^{-2} a^{-1}. This is in line with our results (2.79 g m^{-2} a^{-1}) and of a similar order of magnitude of fluxes directly measured via eddy covariance (Fares et al. 2016)[10] in a *Q. ilex* forest a few kilometers away from Castel di Guido of 0.3 g (PM_1) m^{-2} a^{-1} assuming that PM_{10} fluxes were one order of magnitude higher than PM_1 fluxes. Lower values were simulated by Manes and colleagues for coniferous (*Pinus* spp. and *Picea abies*, with 1.13 and 1.11 g m^{-2} a^{-1}, respectively) and deciduous broadleaved forests (mixed deciduous, *Fagus sylvatica,* and *Castanea sativa*, with 0.69, 0.75, and 0.94 g m^{-2} a^{-1}, respectively). Concerning ozone, deciduous broadleaved categories, such as *F. sylvatica* and *C. sativa*, displayed the highest ozone removal efficiency (a mean value of 5.62 and 4.71 g m^{-2} a^{-1}, respectively), which could be explained by their higher stomatal conductance (not shown). Lower values were instead obtained for *Q. ilex* stands, finally, by *Pinus* spp. and *P. abies* (a mean value of 2.30, 1.72 and 0.81 g m^{-2} a^{-1}, respectively). All these values reported by Manes et al. are in a similar range to that reported in this study.

More recently, Baraldi et al. (2019)[50] used i-tree to predict annual removal of a broad range of tree species. Results are in the same order of magnitude as those simulated in our study. In particular, O_3 and PM_{10} deposition range from about 58–140 g $plant^{-1}$ a^{-1} and from about 17–139 g $plant^{-1}$ a^{-1}, respectively, with total tree CO_2 storage in the range of 164–215 kg $plant^{-1}$ a^{-1}. Concerning ozone, a previous study by Sicard et al. (2018)[7] shows a range of ozone sequestration similar to what was found in our work: the average annual O_3 removal by urban trees was 5.4 g m^{-2} a^{-1} in 55 U.S. cities, 3.7 g m^{-2} a^{-1} in 86 Canadian cities, and 3.3 g m^{-2} a^{-1} in 10 Italian cities. The O_3 removal rate reported by Sicard and colleagues varied worldwide: Melbourne (0.9 g m^{-2} a^{-1}), Stockholm (1.3 g m^{-2} a^{-1}), Beijing (5.5 g m^{-2} a^{-1}), Florence (7.3 g m^{-2} a^{-1}), and San Diego (7.6 g m^{-2} a^{-1}).

Concerning PM in particular, modeling results showed that among different forest types, the coniferous forests often possess greater PM collection ability than broadleaf forests,[51] which is related to the coniferous properties with fine leaves and complex foliage structures.[52] Similarly, Bottalico et al. (2017)[53] through a spatially explicit method for the city of Florence, Italy, found higher sequestration of PM and ozone from mixed evergreen species (31.7 and 15.8 g m^{-2} a^{-1}, respectively) compared with broadleaf species (10 and 4.1 g m^{-2} a^{-1}, respectively). Furthermore, Qiu et al., (2015)[54] found that both $PM_{2.5}$ and PM_{10} deposition velocities are higher in dry periods than in wet periods and vegetation capacities for capturing coarse particles are stronger than those capturing fine particles.[55] Our results are in agreement with these previous studies, in particular, the majority of evergreen conifers showed the highest ranking (Tables 1 and 2), and fluxes of both PM_{10} and $PM_{2.5}$ were higher during late-Spring–Summer dry periods (Figures 4 and 5). Therefore, stand types, seasons, and even particle diameter all influence the PM dry deposition velocity. Similar findings were reported by Fusaro et al. (2017),[56] who showed higher rates of ozone sequestration by urban parks of Rome during warm seasons especially in the absence of drought stress.

Relevance of Pollutant Sequestration for Air Quality. Question (hypothesis 3) was answered positively and the results showed that two investigated urban Parks in Italy represent a moderate sink of pollutants. In a recent study, Sicard et al. (2018)[7] found that the rates of pollutant sequestration translate into a mean annual percent improvement usually less than 2% of O_3 levels as reported by using models such as UFORE/i-Tree. Following a similar exercise, our simulated percent improvement is in the same order of magnitude, with values close to 1% for ozone (Figures S7 and S8). In addition, our simulations did not consider ozone-forming potential from BVOC emitted in this study, considering that *Q. pubescens*, the most abundant tree species at Castel di Guido (Figure 1), has been described as a relevant isoprene emitter, isoprene being a high ozone-forming chemical species.[57] Also for PM, previous studies showed that the fraction removed at the level of the urban area depends on the type of pollutant and the structural attributes of the plant canopies. Such studies also showed that the proportion of PM removed is around 1%.[21,25,58−60] We found with our simplified modeling simulations a higher percent improvement for PM (up to 7% for PM_{10}) in particular at Valentino because of the higher concentration of PM, the close vicinity to the emission sources and the shallower boundary layer compared with the semirural and a more ventilated site of Castel di Guido. Similar to our hypothesis, such removal rates could compensate a small fraction of emissions arising from anthropogenic activities, in particular vehicular traffic. We estimate that the parks of Castel di Guido and Valentino could collect the PM emission of around 15,000 and 160 vehicles (classified as EURO-6) per year, and the CO_2 emission of around 2190 and 13 vehicles per year, respectively. These numbers may seem to be low compared to millions of vehicles circulating in large urban settlements; however, counting of the huge amount of parks present in many cities like Rome (43,000 ha of green areas out of a total extension of the municipality of 129,000 ha), the numbers are not negligible, also in light of future initiatives in the main Italian municipalities to further expand forested areas by planting millions of trees before 2030 as foreseen by a recent decree approved by the Italian Parliament.

Reducing significantly ozone and PM concentrations in cities would imply defining very ambitious urban forest management plans, including large surfaces and proper species selection, with a focus on the removal ability as highlighted in this study for a range of typical Mediterranean species. However, as recently reported for UK by the air quality expert group,[61] the magnitude of the reduction in the PM_{10} concentration by realistic planting schemes is small and in the range of 2–10%. Planting should also take into account other aspects such as biogenic emission rates (with ozone-forming potentials), allergenic effects, and maintenance requirements. Some of the species we identified as high PM collectors are in the same list proposed by Yang et al. (2015)[62] who developed a method to rank the relative suitability of common urban tree species for planting programs, which include the removal of $PM_{2.5}$ as a target. In particular, conifer species were ranked high in $PM_{2.5}$ removal efficiency mainly thanks to the high values of LAI, which were also measured in this study, but also some broadleaf urban tree species widely distributed and present at our urban parks were listed. Among these were *Robinia pseudoacacia*, *Acer platanoides*, *Platanus* spp. *Ailanthus altissima*, *Betula pendula*, and *Tilia cordata,* which we ranked as species with moderate capacity to sequestrate PM compared with conifers. We agree with Yang and colleagues with their finding that selection of a mixture of conifer and broadleaf species that have high PM removal efficiencies, good adaptability to urban environments, and fewer negative impacts on air quality (i.e., low BVOC emissions) can be used to reduce atmospheric concentrations of PM. In agreement with our study is also a recent work of Przybysz et al. (2019),[63] who identified conifer trees for their higher capacity to sequestrate PM especially during the Winter season. More recently, Baraldi et al. (2019)[50] identified the following species as particularly suitable for removing PM_{10} and O_3: *Liriodendron tulipifera*, *C. australis*, *A. campestre,* and *A. platanoides*. The authors also found species such as *Prunus cerasifera*, *Quercus cerris*, *C. australis*, *A. campestre,* and *A. platanoides* as favorable species to be used for carbon sequestration.

Vegetation has the feature to provide larger surface areas for pollutant deposition than the urban fabric;[21] therefore, it is plausible that tree bark and branches, outside the vegetative period, can also be seen as an important factor in removing air pollutants.[64] Our study may, therefore, underestimate sinks of PM and ozone particularly during the leafless period.

It must be taken into account that by applying the AIRTREE model, we indirectly assumed a homogeneous canopy distribution in the parks and a horizontally homogeneous distribution of pollutants with downward flux. As discussed by Xing and Brimblecombe (2020),[22] spatial scale is of high relevance for estimating pollutant deposition. Although we strongly believe that the mesoscale approach we adopt is more appropriate to describe pollutant deposition than laboratory measurements on a single leaf upscaled to a forest,[65] the weak point in such a mesoscale approach is that this may not be the best approximation for street trees or very small parks where vehicles and street canyon dynamics induce strong turbulence affecting both concentrations and deposition velocities close to the leaves.[66] While we believe our hypothesis is tenable for the large park of Castel di Guido, some possible issues may arise for Valentino because of its heterogeneity and vicinity to the emission sources. As also recently shown by Xing et al. (2019),[67] distribution of trees in parks can have a large influence on the airflow at the boundaries and inside the parks.

In conclusion, under realistic situations, the 1−7% percent reductions of pollutant concentrations from the atmosphere may appear a minor tribute to amelioration of air quality. Although this was not the object of this work, another relevant effect exerted by urban trees is on air flow and thus pollutant dispersion. A recent bibliometric analysis by Xing and Brimblecombe (2020)[22] suggests that urban parks can ameliorate air quality through two main pathways: on the one hand, they accelerate PM dispersion, and on the other hand, they reduce pollutant concentration by deposition. The authors argue that the balance between dispersion and deposition processes varies with spatial scales. In line with our findings, they conclude that in small parks, common in dense cities, PM removal by vegetation is unlikely to make the major contribution to improved air quality in their interiors. Moreover, dense tree canopies suppress dispersion so can increase localized pollutant concentrations. This phenomenon may be particularly relevant at Valentino, a small-sized park located in the city downtown compared to the Estate of Castel di Guido, thus suggesting that future studies on pollutant dispersion would be needed to better address the role of urban parks in ameliorating air quality.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.0c04740.

> Formalism adopted to estimate LAI from DCP; formalism adopted to estimate photosynthesis and deposition of particles; dendrometric and ecophysiological parameters used in this study to characterize the vegetation for Castel di Guido and Valentino, respectively; NPP, ozone, PM_{10}, and $PM_{2.5}$ dry deposition simulated by the AIRTREE model for Castel di Guido and Valentino, respectively; whole dbh class groups; dry deposition values for ozone, PM_{10}, and $PM_{2.5}$ for both parks; and magnitude and annual dynamics of NPP, GPP, ozone, PM_{10}, and $PM_{2.5}$ at the level of entire parks and for some relevant tree species discussed in the text (PDF)

■ AUTHOR INFORMATION

Corresponding Author

Silvano Fares − *National Research Council (CNR), Institute of BioEconomy (IBE), Rome 00100, Italy; Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy;* orcid.org/0000-0002-1990-0928; Email: silvano.fares@cnr.it

Authors

Adriano Conte − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy*

Alessandro Alivernini − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy*

Francesco Chianucci − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy*

Mirko Grotti − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy; Department of Architecture and Design, Sapienza University of Rome, Rome 00185, Italy*

Ilaria Zappitelli − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy*

Fabio Petrella − *Istituto per le Piante da Legno e l'Ambiente (IPLA), Turin 10132, Italy*

Piermaria Corona − *Council for Agricultural Research and Economics (CREA), Research Centre for Forestry and Wood, Rome 00166, Italy; Department for Innovation in Biological, Agro-food and Forest Systems (DIBAF), University of Tuscia, Viterbo 01100, Italy*

Complete contact information is available at: https://pubs.acs.org/10.1021/acs.est.0c04740

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

We are grateful to Dr. Craig Macfarlane, for kindly sharing routines for DCP image analysis and processing, and the CREA technicians Valerio Moretti and Tiziano Sorgi for supporting the field measurements. We want to thank research project: the Italian National project PRIN 2017-EUFORICC "Establishing Urban FORest based solutions In Changing Cities", the UE LIFE financial instrument (LIFE18 PRE IT 003) project VEG-GAP "Vegetation for Urban Green Air Quality Plans", the Regione Piemonte for funding a research collaboration between CREA and IPLA (Urban Forestry Project), and Agenzia Regionale per la Protezione Ambientale del Piemonte e del Lazio for providing us meteorological and pollution data.

■ REFERENCES

(1) Evans, J. P.; Ji, F.; Abramowitz, G.; Ekström, M. Optimally Choosing Small Ensemble Members to Produce Robust Climate Simulations. *Environ. Res. Lett.* **2013**, *8*, 044050.

(2) EEA. Air quality in Europe 2019—European Environment Agency https://www.eea.europa.eu/publications/air-quality-in-europe-2019 (accessed May 20, 2020).

(3) Istat.it Ambiente ed energia https://www.istat.it/it/ambiente-ed-energia (accessed May 20, 2020).

(4) Lelieveld, J.; Evans, J. S.; Fnais, M.; Giannadaki, D.; Pozzer, A. The Contribution of Outdoor Air Pollution Sources to Premature Mortality on a Global Scale. *Nature* **2015**, *525*, 367−371.

(5) Lefohn, A. S.; Emery, C.; Shadwick, D.; Wernli, H.; Jung, J.; Oltmans, S. J. Estimates of Background Surface Ozone Concentrations in the United States Based on Model-Derived Source Apportionment. *Atmos. Environ.* **2014**, *84*, 275−288.

(6) Pachauri, R. K.; Meyer, L.; Hallegatte France, S.; Bank, W.; Hegerl, G.; Brinkman, S.; van Kesteren, L.; Leprince-Ringuet, N.; van Boxmeer, F. *AR5 Synthesis Report: Climate Change 2014—IPCC*, 2014.

(7) Sicard, P.; Agathokleous, E.; Araminiene, V.; Carrari, E.; Hoshika, Y.; De Marco, A.; Paoletti, E. Should We See Urban Trees as Effective Solutions to Reduce Increasing Ozone Levels in Cities? *Environ. Pollut.* **2018**, *243*, 163−176.

(8) Nowak, D. J.; Hirabayashi, S.; Bodine, A.; Greenfield, E. Tree and Forest Effects on Air Quality and Human Health in the United States. *Environ. Pollut.* **2014**, *193*, 119−129.

(9) Manes, F.; Marando, F.; Capotorti, G.; Blasi, C.; Salvatori, E.; Fusaro, L.; Ciancarella, L.; Mircea, M.; Marchetti, M.; Chirici, G.; Munafò, M. Regulating Ecosystem Services of Forests in Ten Italian Metropolitan Cities: Air Quality Improvement by PM10 and O3 Removal. *Ecol. Indic.* **2016**, *67*, 425−440.

(10) Fares, S.; Savi, F.; Fusaro, L.; Conte, A.; Salvatori, E.; Aromolo, R.; Manes, F. Particle Deposition in a Peri-Urban Mediterranean Forest. *Environ. Pollut.* **2016**, *218*, 1278−1286.

(11) Dall'Osto, M.; Thorpe, A.; Beddows, D. C. S.; Harrison, R. M.; Barlow, J. F.; Dunbar, T.; Williams, P. I.; Coe, H. Remarkable Dynamics of Nanoparticles in the Urban Atmosphere. *Atmos. Chem. Phys.* **2011**, *11*, 6623−6637.

(12) Crompton, J. L. Evolution of the "Parks as Lungs" Metaphor: Is It Still Relevant? *World Leis. J.* **2017**, *59*, 105−123.

(13) Samson, R.; Grote, R.; Calfapietra, C.; Cariñanos, P.; Fares, S.; Paoletti, E.; Tiwary, A. *Urban Trees and Their Relation to Air Pollution*; Springer: Cham, 2017; pp 21−30.

(14) Pröhl, G.; Twining, J. R.; Crawford, J. Radiological Consequences Modelling. *Radioactivity in the Environment*; Elsevier Ltd., 2012; Vol. *18*, pp 281−342.

(15) Park, M.-S.; Joo, S. J.; Lee, C. S. Effects of an Urban Park and Residential Area on the Atmospheric CO2 Concentration and Flux in Seoul, Korea. *Adv. Atmos. Sci.* **2013**, *30*, 503−514.

(16) Helfter, C.; Famulari, D.; Phillips, G. J.; Barlow, J. F.; Wood, C. R.; Grimmond, C. S. B.; Nemitz, E. Controls of Carbon Dioxide Concentrations and Fluxes above Central London. *Atmos. Chem. Phys.* **2011**, *11*, 1913−1928.

(17) Dusart, N.; Gérard, J.; Le Thiec, D.; Collignon, C.; Jolivet, Y.; Vaultier, M.-N. Integrated Analysis of the Detoxification Responses of Two Euramerican Poplar Genotypes Exposed to Ozone and Water Deficit: Focus on the Ascorbate-Glutathione Cycle. *Sci. Total Environ.* **2019**, *651*, 2365−2379.

(18) Han, D.; Shen, H.; Duan, W.; Chen, L. A Review on Particulate Matter Removal Capacity by Urban Forests at Different Scales. *Urban For. Urban Green.* **2020**, *48*, 126565.

(19) Clifton, O. E.; Fiore, A. M.; Massman, W. J.; Baublitz, C. B.; Coyle, M.; Emberson, L.; Fares, S.; Farmer, D. K.; Gentine, P.; Gerosa, G.; Guenther, A. B.; Helmig, D.; Lombardozzi, D. L.; Munger, J. W.; Patton, E. G.; Pusede, S. E.; Schwede, D. B.; Silva, S. J.; Sörgel, M.; Steiner, A. L.; Tai, A. P. K. Dry Deposition of Ozone Over Land: Processes, Measurement, and Modeling. *Rev. Geophys.* **2020**, *58*, No. e2019RG000670.

(20) Pryor, S. C.; Gallagher, M.; Sievering, H.; Larsen, S. E.; Barthelmie, R. J.; Birsan, F.; Nemitz, E.; Rinne, J.; Kulmala, M.; Grönholm, T.; Taipale, R.; Vesala, T. A Review of Measurement and Modelling Results of Particle Atmosphere-Surface Exchange. *Tellus, Ser. B: Chem. Phys. Meteorol.* **2008**, *60*, 42−75.

(21) Janhäll, S. Review on Urban Vegetation and Particle Air Pollution - Deposition and Dispersion. *Atmos. Environ.* **2015**, *105*, 130−137.

(22) Xing, Y.; Brimblecombe, P. Trees and Parks as "the Lungs of Cities.". *Urban For. Urban Green.* **2020**, *48*, 126552.

(23) Barwise, Y.; Kumar, P. Designing Vegetation Barriers for Urban Air Pollution Abatement: A Practical Review for Appropriate Plant Species Selection. *npj Clim. Atmos. Sci.* **2020**, *3*, 1−19.

(24) Corona, P. Communicating Facts, Findings and Thinking to Support Evidence-Based Strategies and Decisions. *Ann. Silvic. Res.* **2018**, *42*, 1−2.

(25) Nowak, D. J.; Crane, D. E.; Stevens, J. C. Air Pollution Removal by Urban Trees and Shrubs in the United States. *Urban For. Urban Green.* **2006**, *4*, 115−123.

(26) Fares, S.; Alivernini, A.; Conte, A.; Maggi, F. Ozone and Particle Fluxes in a Mediterranean Forest Predicted by the AIRTREE Model. *Sci. Total Environ.* **2019**, *682*, 494−504.

(27) Simpson, D.; Benedictow, A.; Berge, H.; Bergström, R.; Emberson, L. D.; Fagerli, H.; Flechard, C. R.; Hayman, G. D.; Gauss, M.; Jonson, J. E.; Jenkin, M. E.; Nyíri, A.; Richter, C.; Semeena, V. S.; Tsyro, S.; Tuovinen, J.-P.; Valdebenito, Á.; Wind, P. The EMEP MSC-W Chemical Transport Model − Technical Description. *Atmos. Chem. Phys.* **2012**, *12*, 7825−7865.

(28) Blasi, C. Fitoclimatologia Del Lazio. *Fitosociologia* **1994**, *27*, 56.

(29) Barbati, A.; Marchetti, M.; Chirici, G.; Corona, P. European Forest Types and Forest Europe SFM Indicators: Tools for Monitoring Progress on Forest Biodiversity Conservation. *For. Ecol. Manage.* **2014**, *321*, 145−157.

(30) Food and Agriculture Organisation. *Status of the World's Soil Resources*, 2015. ISBN 978-92-5-109004-6.

(31) Macfarlane, C.; Hoffman, M.; Eamus, D.; Kerp, N.; Higginson, S.; McMurtrie, R.; Adams, M. Estimation of Leaf Area Index in Eucalypt Forest Using Digital Photography. *Agric. For. Meteorol.* **2007**, *143*, 176−188.

(32) Chianucci, F. An Overview of in Situ Digital Canopy Photography in Forestry. *Can. J. For. Res.* **2020**, *50*, 227−242.

(33) Grotti, M.; Mattioli, W.; Ferrari, B.; Tomao, A.; Merlini, P.; Corona, P. A Multi-Temporal Dataset of Forest Mensuration of Reforestations: A Case Study in Peri-Urban Rome, Italy. *Ann. Silvic. Res.* **2019**, *43*, 97−101.

(34) Long, S. P.; Bernacchi, C. J. Gas Exchange Measurements, What Can They Tell Us about the Underlying Limitations to

Photosynthesis? Procedures and Sources of Error. *J. Exp. Bot.* **2003**, *54*, 2393−2401.
(35) Duursma, R. A. Plantecophys - An R Package for Analysing and Modelling Leaf Gas Exchange Data. *PLoS One* **2015**, *10*, No. e0143346.
(36) Sharkey, T. D. What Gas Exchange Data Can Tell Us about Photosynthesis. *Plant Cell and Environment*; Blackwell Publishing Ltd., June 1, 2016; pp 1161−1163.
(37) Keenan, T.; Sabate, S.; Gracia, C. Soil Water Stress and Coupled Photosynthesis-Conductance Models: Bridging the Gap between Conflicting Reports on the Relative Roles of Stomatal, Mesophyll Conductance and Biochemical Limitations to Photosynthesis. *Agric. For. Meteorol.* **2010**, *150*, 443−453.
(38) Zhang, L.; Brook, J. R.; Vet, R. On Ozone Dry Deposition - With Emphasis on Non-Stomatal Uptake and Wet Canopies. *Atmos. Environ.* **2002**, *36*, 4787−4799.
(39) Zhang, L.; Brook, J. R.; Vet, R. On Ozone Dry Deposition—with Emphasis on Non-Stomatal Uptake and Wet Canopies. *Atmos. Environ.* **2002**, *36*, 4787−4799.
(40) Pace, R.; Grote, R. Deposition and Resuspension Mechanisms Into and From Tree Canopies: A Study Modeling Particle Removal of Conifers and Broadleaves in Different Cities. *Front. For. Glob. Change* **2020**, *3*, 26.
(41) Karl, T.; Misztal, P. K.; Jonsson, H. H.; Shertz, S.; Goldstein, A. H.; Guenther, A. B. Airborne Flux Measurements of Bvocs above Californian Oak Forests: Experimental Investigation of Surface and Entrainment Fluxes, OH Densities, and Damköhler Numbers. *J. Atmos. Sci.* **2013**, *70*, 3277−3287.
(42) Bergot, T. An Introduction to Boundary Layer Meteorology - Par Roland B. Stull. *Meteorol.* **1994**, *8*, 89.
(43) Fares, S.; Savi, F.; Muller, J.; Matteucci, G.; Paoletti, E. Simultaneous Measurements of above and below Canopy Ozone Fluxes Help Partitioning Ozone Deposition between Its Various Sinks in a Mediterranean Oak Forest. *Agric. For. Meteorol.* **2014**, *198−199*, 181−191.
(44) Łukowski, A.; Popek, R.; Karolewski, P. Particulate Matter on Foliage of Betula Pendula, Quercus Robur, and Tilia Cordata: Deposition and Ecophysiology. *Environ. Sci. Pollut. Res.* **2020**, *27*, 10296−10307.
(45) Cape, J. N. Interactions of Forests with Secondary Air Pollutants: Some Challenges for Future Research. *Environ. Pollut.* **2008**, *155*, 391−397.
(46) Liu, B.; Shen, Y.; Chen, X.; Chen, Y.; Wang, X. A Partial Binary Tree DEA-DA Cyclic Classification Model for Decision Makers in Complex Multi-Attribute Large-Group Interval-Valued Intuitionistic Fuzzy Decision-Making Problems. *Infusionstherapie* **2014**, *18*, 119−130.
(47) Kim, G.; Coseo, P. Urban Park Systems to Support Sustainability: The Role of Urban Park Systems in Hot Arid Urban Climates. *Forests* **2018**, *9*, 439.
(48) Emberson, L. D.; Ashmore, M. R.; Cambridge, H. M.; Simpson, D.; Tuovinen, J.-P. Modelling Stomatal Ozone Flux across Europe. *Environ. Pollut.* **2000**, *109*, 403−413.
(49) Guidolotti, G.; Calfapietra, C.; Pallozzi, E.; De Simoni, G.; Esposito, R.; Mattioni, M.; Nicolini, G.; Matteucci, G.; Brugnoli, E. Promoting the Potential of Flux-Measuring Stations in Urban Parks: An Innovative Case Study in Naples, Italy. *Agric. For. Meteorol.* **2017**, *233*, 153−162.
(50) Baraldi, R.; Chieco, C.; Neri, L.; Facini, O.; Rapparini, F.; Morrone, L.; Rotondi, A.; Carriero, G. An Integrated Study on Air Mitigation Potential of Urban Vegetation: From a Multi-Trait Approach to Modeling. *Urban For. Urban Green.* **2019**, *41*, 127−138.
(51) Zhang, X.; Du, J.; Huang, T.; Zhang, L.; Gao, H.; Zhao, Y.; Ma, J. Atmospheric Removal of PM2.5 by Man-Made Three Northern Regions Shelter Forest in Northern China Estimated Using Satellite Retrieved PM2.5 Concentration. *Sci. Total Environ.* **2017**, *593−594*, 713−721.
(52) Freer-Smith, P. H.; Beckett, K. P.; Taylor, G. Deposition Velocities to Sorbus Aria, Acer Campestre, Populus Deltoides × Trichocarpa "Beaupré", Pinus Nigra and × Cupressocyparis Leylandii for Coarse, Fine and Ultra-Fine Particles in the Urban Environment. *Environ. Pollut.* **2005**, *133*, 157−167.
(53) Bottalico, F.; Travaglini, D.; Chirici, G.; Garfì, V.; Giannetti, F.; De Marco, A.; Fares, S.; Marchetti, M.; Nocentini, S.; Paoletti, E.; Salbitano, F.; Sanesi, G. A Spatially-Explicit Method to Assess the Dry Deposition of Air Pollution by Urban Forests in the City of Florence, Italy. *Urban For. Urban Green.* **2017**, *27*, 221−234.
(54) Qiu, D.; Liu, J.; Zhu, L.; Mo, L.; Zhang, Z. Particulate Matter Assessment of a Wetland in Beijing. *J. Environ. Sci.* **2015**, *36*, 93−101.
(55) Liu, X.; Li, C.; Tu, H.; Wu, Y.; Ying, C.; Huang, Q.; Wu, S.; Xie, Q.; Yuan, Z.; Lu, Y. Analysis of the Effect of Meteorological Factors on PM2.5-Associated PAHs during Autumn-Winter in Urban Nanchang. *Aerosol Air Qual. Res.* **2016**, *16*, 3222−3229.
(56) Fusaro, L.; Marando, F.; Sebastiani, A.; Capotorti, G.; Blasi, C.; Copiz, R.; Congedo, L.; Munafò, M.; Ciancarella, L.; Manes, F. Mapping and Assessment of PM10 and O3 Removal by Woody Vegetation at Urban and Regional Level. *Remote Sens.* **2017**, *9*, 791.
(57) Genard-Zielinski, A.-C.; Boissard, C.; Fernandez, C.; Kalogridis, C.; Lathière, J.; Gros, V.; Bonnaire, N.; Ormeño, E. Variability of BVOC Emissions from a Mediterranean Mixed Forest in Southern France with a Focus on Quercus Pubescens. *Atmos. Chem. Phys.* **2015**, *15*, 431−446.
(58) Baró, F.; Chaparro, L.; Gómez-Baggethun, E.; Langemeyer, J.; Nowak, D. J.; Terradas, J. Contribution of Ecosystem Services to Air Quality and Climate Change Mitigation Policies: The Case of Urban Forests in Barcelona, Spain. *Ambio* **2014**, *43*, 466−479.
(59) Jeanjean, A. P. R.; Monks, P. S.; Leigh, R. J. Modelling the Effectiveness of Urban Trees and Grass on PM2.5 Reduction via Dispersion and Deposition at a City Scale. *Atmos. Environ.* **2016**, *147*, 1−10.
(60) Selmi, W.; Weber, C.; Rivière, E.; Blond, N.; Mehdi, L.; Nowak, D. Air Pollution Removal by Trees in Public Green Spaces in Strasbourg City, France. *Urban For. Urban Green.* **2016**, *17*, 192−201.
(61) Air Quality Expert Group. *Impacts of Vegetation on Urban Air Pollution*, 2018, 40.
(62) Yang, J.; Chang, Y.; Yan, P. Ranking the Suitability of Common Urban Tree Species for Controlling PM2.5pollution. *Atmos. Pollut. Res.* **2015**, *6*, 267−277.
(63) Przybysz, A.; Nersisyan, G.; Gawroński, S. W. Removal of Particulate Matter and Trace Elements from Ambient Air by Urban Greenery in the Winter Season. *Environ. Sci. Pollut. Res.* **2019**, *26*, 473−482.
(64) Klingberg, J.; Broberg, M.; Strandberg, B.; Thorsson, P.; Pleijel, H. Influence of Urban Vegetation on Air Pollution and Noise Exposure − A Case Study in Gothenburg, Sweden. *Sci. Total Environ.* **2017**, *599−600*, 1728−1739.
(65) Santiago, L. S.; Silvera, K.; Andrade, J. L.; Dawson, T. E. Functional Strategies of Tropical Dry Forest Plants in Relation to Growth Form and Isotopic Composition. *Environ. Res. Lett.* **2017**, *12*, 115006.
(66) Sheng, Y.; Chen, T.; Lu, Y.; Chang, R.-J.; Sinha, S.; Warner, J. H. High-Performance WS 2 Monolayer Light-Emitting Tunneling Devices Using 2D Materials Grown by Chemical Vapor Deposition. *ACS Nano* **2019**, *13*, 4530−4537.
(67) Xing, Y.; Brimblecombe, P.; Wang, S.; Zhang, H. Tree Distribution, Morphology and Modelled Air Pollution in Urban Parks of Hong Kong. *J. Environ. Manage.* **2019**, *248*, 109304.