

Urban ecosystem services: tree diversity and stability of tropospheric ozone removal

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Abstract. Urban forests provide important ecosystem services, such as urban air quality improvement by removing pollutants. While robust evidence exists that plant physiology, abundance, and distribution within cities are basic parameters affecting the magnitude and efficiency of air pollution removal, little is known about effects of plant diversity on the stability of this ecosystem service. Here, by means of a spatial analysis integrating system dynamic modeling and geostatistics, we assessed the effects of tree diversity on the removal of tropospheric ozone (O₃) in Rome, Italy, in two years (2003 and 2004) that were very different for climatic conditions and ozone levels. Different tree functional groups showed complementary uptake patterns, related to tree physiology and phenology, maintaining a stable community function across different climatic conditions. Our results, although depending on the city-specific conditions of the studied area, suggest a higher function stability at increasing diversity levels in urban ecosystems. In Rome, such ecosystem services, based on published unitary costs of externalities and of mortality associated with O₃, can be prudently valued to roughly US\$2 and \$3 million/year, respectively.

Key words: air quality; ecophysiology; ecosystem function; GIS; Rome, Italy; sanitary benefits; tropospheric ozone; urban forest.

INTRODUCTION

Human health and well-being is known to depend on “ecosystem goods and services” (Costanza et al. 1997, MEA 2005). The concept of ecosystem services was defined by Daily (1997) as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life.” Since then, different definitions have been proposed for “ecosystem services” (Boyd and Banzhaf 2007, Fisher and Turner 2008, Fisher et al. 2009). According to a recent review by Escobedo et al. (2011), which focused on air pollution mitigation by urban forests, ecosystem services are considered the components (including functions) of urban forests that are directly enjoyed, consumed, or used to produce specific and measurable human benefits. Ecosystem services are affected by the relationship between ecosystem functioning, stability, and biodiversity (Balvanera et al. 2006, Costanza et al. 2007, Gamfeldt et al. 2008). The understanding of these relationships is needed for devising the best management and policy tools for sustainable use of ecosystem services (Kremen 2005). Urban forests provide important ecosystem services (Bolund and Hunhammar 1999, Jim and Chen 2008, Young 2010), such as modification of urban

microclimate by lowering temperatures (Pataki et al. 2011a), changing wind patterns, reduction of building energy use (Akbari 2002), and improvement of local and regional air quality by removal of atmospheric pollutants (Nowak et al. 2006). These environmental benefits are becoming increasingly important, as today more than half of the world’s population (~3.3 billion people) live in urban areas. Additionally, according to UN projections, cities are growing to unprecedented sizes, absorbing nearly all of the growth in the human population over the next three decades (United Nations Population Fund 2007), with potential implications for biodiversity conservation issues (Dearborn and Kark 2009).

The role of urban forests in providing ecosystem services has been investigated in many papers, considering both basic ecosystem functions, like primary productivity (Kaye et al. 2006, Pataki et al. 2011b) and emerging services, such as the improvement of urban air quality (Yang et al. 2005, Nowak et al. 2006, McDonald et al. 2007, Escobedo and Nowak 2009). The reduction of air pollution by urban trees has been recognized as a cost-effective component of pollution reduction strategies in several urban areas, such as Washington, D.C., New York, Baltimore, Atlanta, and Chicago, in the United States (Nowak et al. 2000, 2006, Yang et al. 2008, Morani et al. 2011), Beijing (Yang et al. 2005), Santiago de Chile (Escobedo and Nowak 2009), London (Tiwarly et al. 2009), and Toronto (Millward and Sabir 2011). Among the air pollutants removed by urban forests, tropospheric ozone (O₃) is dominant in the

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photochemical air pollution mixture in urban areas during summer periods, particularly in Mediterranean areas (Millán et al. 2000), with negative effects on public health (Bell et al. 2006, Martuzzi et al. 2006). While robust evidence exists that the physiology of the main tree functional groups and their abundance and spatial distribution within cities are the basic parameters affecting the magnitude and efficiency of O₃ removal from the urban environment (Escobedo and Nowak 2009), little is known about the effects of tree species diversity on the magnitude and stability of this ecosystem service.

The aim of this paper was to quantify and value the effects of urban tree diversity on the O₃ removal in the city of Rome (Italy). The underlying hypothesis is that different tree functional groups exert a complementary role in stabilizing this emerging ecosystem service over time, and across different environmental conditions. A spatial analysis integrating system dynamic modeling and geostatistics was applied to estimate seasonal and annual ozone removal by three functional groups of urban trees, under two climatically different years: the extremely dry year 2003, and the year 2004, which was more representative of the average long-term climatic pattern of the city of Rome (Gerosa et al. 2009).

METHODS

Urban forests in the city of Rome

The city of Rome (41°54' N, 12°29' E) extends over an area of ~1270 km² and hosts roughly 2.8 million inhabitants. Overall, the city is characterized by high levels of urban traffic and urban expansion, which largely increased in the last decades. The urban landscape in Rome is very heterogeneous in terms of geology, soil, morphology, and land use (Fig. 1a). The climate is mediterranean, with an average annual temperature of 15.1°C, average annual rainfall of 839 mm, and a typical hot and dry summer period favoring high tropospheric O₃ concentrations (Manes et al. 2003). Notwithstanding the long-lasting human impact, spanning over >2700 years, and the recent increase of the urbanized surface, Rome is still considered as one of the “most green” Italian cities, with public green space covering >20% of the total municipality area and including a system of nine natural reserves for a total cover of ~16 000 ha, hosting roughly 1200 plant species (Celesti-Grapow et al. 2006). Residual fragments of ancient woodlands still occur within the city boundaries, hosting a wide set of different tree species ranging from typical Mediterranean evergreen broadleaf species (*Quercus ilex* and *Q. suber*; hereafter broadleaves) to deciduous *Quercus* woods (*Q. cerris*, *Q. frainetto*) and conifer plantations (*Pinus pinea*). These three groups of species show important functional differences in their ecophysiological and phenological traits (Manes et al. 1997, Anselmi et al. 2004), providing an optimal case study to test the diversity–stability relationship in urban ecosystems.

As a detailed inventory of the urban forests of the metropolitan area does not exist, a Landsat 5 TM image (from 21 July 1999), with a spatial resolution of 30 × 30 m, was used to assess the distribution of the main tree functional groups (evergreen broadleaves, deciduous broadleaves, and conifers) across the city of Rome. First, a supervised classification of the Landsat image into 18 land use classes was performed using the TM bands 3, 4, 5, and 7 by means of a maximum likelihood algorithm (Anselmi et al. 2003). The overall accuracy of the classification, calculated through an error matrix, was 96%. Large green areas can be observed in the suburban zones, and inside the city center (Fig. 1a). The most important forested areas, located in the southern coastal area, are the Castelporziano Presidential Estate, characterized by high plant community diversity (Seufert et al. 1997), and the Castel Fusano urban park. Moreover, patches of urban forests are present in the historical villas, such as Villa Ada, Villa Borghese, and Veio Park (Celesti-Grapow et al. 2006).

The area covered by each functional group was then estimated by assigning the urban forest classes in Fig. 1a to three leaf categories (Fig. 1b). The attribution of the forest classes to the corresponding functional groups, being scale dependent, was necessarily affected by some degree of approximation. Stands characterized by a dominant species were entirely attributed to the corresponding leaf type: for example, the land use classes “Holm oak prevailing” and “Cork oak prevailing” were assigned to the “evergreen broadleaf” functional group, deciduous woods (oak woods dominated by *Q. cerris*, *Q. frainetto*, *Tilia cordata*, *Platanus x acerifolia*, and *Robinia pseudoacacia* woods) were assigned to the “deciduous broadleaf” group, while Italian stone pine woods were assigned to the “conifer” group. Deciduous woods with sclerophyllous species were completely attributed to the “deciduous broadleaf” category, because in these stands the evergreen species are mostly located in the understory layers, thus giving a negligible contribution to pollutants uptake and deposition processes (Manes et al. 2007). The areas covered by mixed conifers and evergreen broadleaved species were partitioned at 50% between the two leaf categories, while the attribution of maquis with Holm oak prevailing areas to “evergreen broadleaf” was limited to 50% of the total coverage, considering only the area covered by trees and excluding shrubs and herbaceous species.

As a result, the area covered by the tree functional groups totaled 7198 ha, corresponding to 5.6% of the municipality area (Fig. 1b). Deciduous broadleaves represented the most abundant functional group (3474 ha), followed by evergreen broadleaves (2121 ha) and conifers (1605 ha). In particular, for the Castelporziano Presidential Estate, the Castel Fusano urban park, and Villa Ada, the vegetation cover was: 1080.3 ha, 230.8 ha, and 38.5 ha for evergreen broadleaves; 2348.9 ha, 215.1 ha, and 35.7 ha for deciduous broadleaves; and 735.2 ha, 429.7 ha, and 42.2 ha for conifers, respectively.

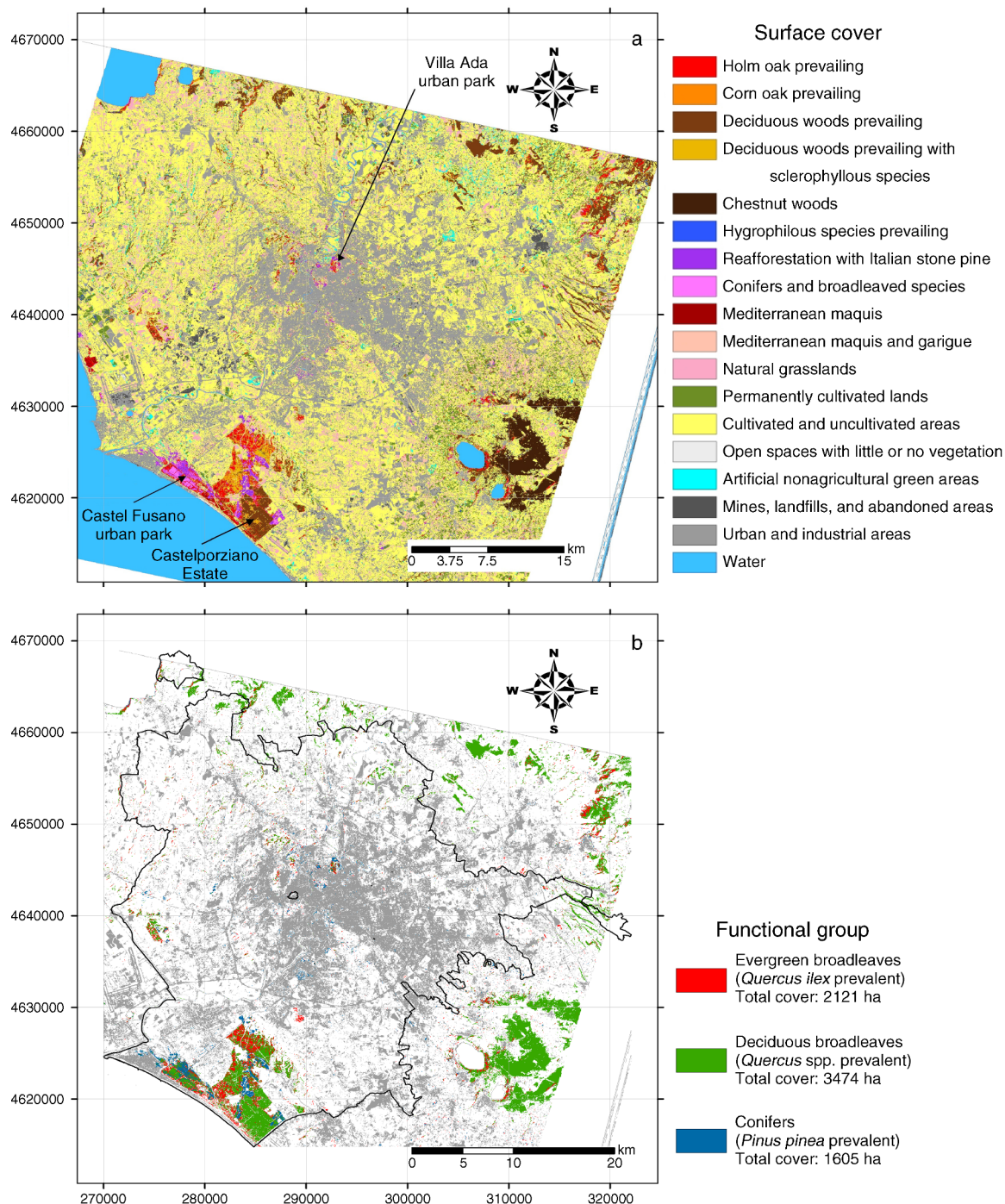


FIG. 1. Urban vegetation of the Municipality of Rome (city limits outlined in black): (a) Landsat 5 TM supervised classification (modified from Manes et al. [2008]) and (b) distribution and total surface cover of the three tree functional groups (evergreen broadleaf species, deciduous broadleaf species, and conifers) analyzed in this study within the metropolitan area. Axes report UTM grid coordinates, WGS84 zone 33 N.

Ozone data

Ozone (O₃) and nitrogen oxides (NO_x) concentrations in air, hourly recorded by 8 and 13 air quality monitoring stations, respectively, were considered for the years 2003 and 2004. All categories of air pollution

monitoring sites of the municipal network (urban traffic, urban background, suburban background, rural background) were included in the analysis. The spatial distribution of O₃ concentrations over the survey area was estimated by applying a spherical co-kriging model

(Isaaks and Srivastava 1989). Daily maps of O_3 concentrations were produced in a Geographic Information System (GIS) by using the geostatistical tool Spatial Analyst in ESRI ArcGIS v. 9.2 (ESRI 2006), where average daily values of O_3 and NO_x have been considered as input data. Further, NO_x was used as an external drift variable in the co-kriging interpolation technique.

All maps were produced by smooth interpolation at 30×30 m resolution to be interoperable with the urban vegetation distribution map (Fig. 1b). In this way, at each map cell (30×30 m pixel) where the target tree functional group was located, daily time series of O_3 concentrations were available for analyses. Finally, to provide a synoptic view of O_3 levels and spatial variability during 2003 and 2004, annual maps of "SOMO35" (i.e., sum of daily eight-hour running means of O_3 over 35 ppb; World Health Organization 2008) were also produced by summing up daily map values.

Modeling tree physiological parameters

The MOCA-Flux (Modeling of Carbon Assessment and Flux) model, implemented within the object-oriented software package STELLA II (Costanza and Gottlieb 1998, Isee Systems 2002), was used to simulate dynamics of physiological parameters for the three plant leaf types, for 2003 and 2004. MOCA-Flux is the newly implemented version of a system dynamic, semi-empirical model, previously applied to simulate functional responses to changes in air temperature (Vitale et al. 2003) and O_3 stomatal fluxes (Vitale et al. 2005) of *Q. ilex*. The MOCA-Flux model is based on the "big-leaf" approach, and it was conceived for estimation of plant physiological variables including stomatal conductance (g_s ; $\text{mol H}_2\text{O}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$), net photosynthesis (P_{NET} ; $\mu\text{mol CO}_2\cdot\text{m}^{-2}\cdot\text{s}^{-1}$), leaf transpiration (E ; $\text{mmol H}_2\text{O}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$), annual net primary productivity (NPP; $\text{g C}/\text{m}^2$), and leaf area index (LAI; m^2 of leaf/ m^2 of ground). All physiological variables are expressed as a diurnal average for the photoperiod.

The MOCA-flux model was applied in the current study due to its demonstrated ability to provide highly fitting prediction of several physiological parameters (including stomatal conductance) for different plant species (Manes et al. 1999, Vitale et al. 2005).

MOCA-Flux calculates stomatal conductance to water vapor (Eq. 1) by using the Ball et al. (1987) algorithm, and corrected by Harley et al. (1992), which is based on net photosynthesis (P_{NET}), relative humidity (RH), and air carbon dioxide concentration ($[\text{CO}_2]_{\text{air}}$), assumed to be constant at $370 \mu\text{mol}/\text{mol}$, as follows:

$$g_s(t) = g_{s0} + m \left[\frac{P_{\text{NET}}(t)RH(t)}{[\text{CO}_2]_{\text{air}}} \right] \quad (1)$$

where g_{s0} is the minimum stomatal conductance to H_2O vapor when $P_{\text{NET}} = 0$ and m is an empirical coefficient that represents the composite sensitivity of conductance to P_{NET} , $[\text{CO}_2]_{\text{air}}$, and RH. Net photosynthesis was

calculated as a function of species-specific quantum yield and solar irradiance by using a semi-empirical model reported in de Wit et al. (1978; see the Appendix). It is noteworthy that the leaf area index (LAI) is also related to net photosynthesis, affecting, in turn, solar irradiance (see the Appendix). The different modules constituting the MOCA-Flux model are highly integrated to each other, thus yielding stable functional interdependences, minimizing the number of input parameters.

Net primary productivity (NPP) is derived from the total of diurnal net photosynthesis values integrated in the phenological time span for each tree species. For further details on model equations, refer to Vitale et al. (2003) and (2005). The model has been parameterized using values of input physiological and structural variables (see the Appendix) derived from field measurements collected in different sampling sites of the survey area (Anselmi et al. 2003, Manes et al. 2007, Vitale et al. 2007), and simulations of daily average g_s were run for the years 2003 and 2004. The model validation was based on reference comparison of simulated O_3 fluxes with eddy covariance measurements, as reported in Vitale et al. (2005) for summer 2003.

Ozone removal by urban tree functional groups

Stomatal ozone fluxes (FO_3) were calculated on a daily time step based on estimated O_3 air concentration and simulated stomatal conductance to water vapor, corrected by the diffusibility ratio between O_3 and water vapor:

$$\text{FO}_3(i, p) = g_s(i) \times [\text{O}_3]_{i,p} \times 0.613 \quad (2)$$

where FO_3 is expressed in $\text{nmol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ and $[\text{O}_3]$ in parts per billion (ppb [nmol/mol]), and the indices i and p refer to the i th day of the reference period and to the p th location of each tree functional group, respectively.

Stomatal ozone fluxes were referred to unitary area of soil surface, thus allowing a geographical representation of the modeling outputs, based on the locations effectively covered by evergreen broadleaves, deciduous broadleaves, and conifers within the survey area, as reported in the vegetation map in Fig. 1b.

The annual time series of FO_3 was integrated over time at each site to estimate the cumulative amount of ozone yearly and seasonally taken up in 2003 and 2004 by each tree functional group:

$$\text{FO}_3\text{cum}(p) = \left(\sum_{i=1}^n \text{FO}_3(i, p) \times \text{Ph} \times 3600 \right) \times 10^6 \quad (3)$$

where n is the number of cumulative days, Ph is the photoperiod in hours, and 10^6 is a dimensional correction factor allowing to express the cumulated stomatal flux in $\text{mmol O}_3\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, when FO_3 is expressed in $\text{nmol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$.

To estimate the uncertainty of yearly cumulated ozone fluxes, the standard deviation of daily stomatal conductance (i.e., $\text{SD}_{g_s}(i)$ for each i th day of the year)

from six model runs was considered. Consequently, uncertainty for daily ozone flux was obtained from Eq. 2 for each vegetation type and for each day of the years 2003 and 2004

$$SD_{FO_3}(i,p) = SD_g(i) \times [O_3]_{i,p} \times 0.613. \quad (4)$$

Then, uncertainty for yearly ozone fluxes was obtained by summing up daily contributions.

Significant differences of ozone removal between tree functional groups were assessed by ANOVA (Duncan test). Significance was evaluated in all cases at $P < 0.05$.

Average values of the stomatal : total flux ratio for *Q. ilex* in the survey area were reported by Gerosa et al. (2005, 2009), both for 2003 and 2004 (0.29 and 0.43, respectively). Similar values, ranging from 0.21 and 0.33, were reported for conifers by Mikkelsen et al. (2004), though at higher latitude, thus allowing an estimation of the potential cumulated flux of O_3 removed from atmosphere by both stomatal uptake and non-stomatal processes (FO_{3t}), at each p th location, as follows:

$$FO_{3t}(p) = FO_{3cum}(p) \times \frac{1}{R} \quad (5)$$

where R values were 0.29 and 0.43 for 2003 and 2004, respectively.

To estimate the total amount of ozone removed from the atmosphere by evergreen broadleaves, deciduous broadleaves, and conifers in the Rome municipality, the cumulated fluxes calculated at each location were totaled.

Given that all fluxes were referred to 1 m² of soil covered by the target leaf types, and the resolution of the vegetation map was 30 × 30 m, then the fluxes in each pixel were calculated multiplying the flux by the pixel area, weighted by the relative coverage of each leaf type in each pixel:

$$FO_{3tot} = \sum_{p=1}^N FO_{3t}(p) \times 900 \quad (6)$$

where N is the number of 30 × 30 m pixels covered by the given tree functional type (Fig. 1b), and 900 is the area in m² of each pixel of the map.

The same method was used for estimating the O_3 removal that would have occurred in both years if all the trees belonged to one single functional group. Three configurations were considered, in each of which the total area covered by tree vegetation within the Municipality of Rome was attributed to one of the functional groups in Eq. 6.

RESULTS

The years 2003 and 2004 were characterized by different climatic conditions and ozone pollution levels. Mean temperatures recorded at the air quality monitoring stations in 2003 were higher than in 2004, for the months of April ($14.0^\circ \pm 1.8^\circ\text{C}$ vs. $12.9^\circ \pm 1.6^\circ\text{C}$), May ($20.7^\circ \pm 2.0^\circ\text{C}$ vs. $16.1^\circ \pm 1.8^\circ\text{C}$), June ($26.4^\circ \pm 2.1^\circ\text{C}$ vs.

$22.0^\circ \pm 2.2^\circ\text{C}$), July ($27.1^\circ \pm 2.0^\circ\text{C}$ vs. $24.1^\circ \pm 1.9^\circ\text{C}$), and August ($27.8^\circ \pm 2.1^\circ\text{C}$ vs. $24.1^\circ \pm 1.8^\circ\text{C}$), whereas total precipitation in 2003 was much lower than in 2004, especially for April (51 ± 12 mm vs. 104 ± 16 mm), May (6 ± 3 mm vs. 85 ± 4 mm), June (0 ± 0 mm vs. 22 ± 4 mm), and July (3 ± 1 mm vs. 41 ± 5 mm) (Fig. 2a, b).

Very different spatial (Fig. 3a, b) and temporal (Fig. 2c, d) patterns were observed in ozone concentrations across the city of Rome during the two years. Mean monthly values frequently exceeded the threshold of 70 $\mu\text{g}/\text{m}^3$ proposed by the World Health Organization to quantify O_3 impact on human health (WHO 2008). In particular, O_3 concentrations were equal to 74 ± 5 $\mu\text{g}/\text{m}^3$ in April, 79 ± 17 $\mu\text{g}/\text{m}^3$ in May, 95 ± 17 $\mu\text{g}/\text{m}^3$ in June, 97 ± 20 $\mu\text{g}/\text{m}^3$ in July, 94 ± 23 $\mu\text{g}/\text{m}^3$ in August, and 74 ± 42 $\mu\text{g}/\text{m}^3$ in September, 2003, with corresponding values for 2004 being fairly lower (51 ± 12 $\mu\text{g}/\text{m}^3$ in April, 67 ± 13 $\mu\text{g}/\text{m}^3$ in May, 74 ± 13 $\mu\text{g}/\text{m}^3$ in June, 81 ± 15 $\mu\text{g}/\text{m}^3$ in July, 75 ± 13 $\mu\text{g}/\text{m}^3$ in August, and 58 ± 13 $\mu\text{g}/\text{m}^3$ in September (Fig. 2c, d).

Potential stomatal ozone uptake in 2003 and 2004 was considerably different among evergreen broadleaves, deciduous broadleaves, and conifers, showing peculiar patterns both in time (Fig. 2e–j) and space (Fig. 3c–f). In spring of both years, deciduous broadleaves showed the highest, and conifers showed the lowest, potential stomatal O_3 fluxes (Fig. 2g, i). In summer 2003, deciduous broadleaves showed a reduced potential stomatal O_3 flux due to the reduction of stomatal conductance under limiting environmental conditions (Fig. 2g, h), while evergreen broadleaves were able to maintain high levels of potential stomatal O_3 fluxes (Fig. 2e, f), and conifers showed an increased O_3 uptake (Fig. 2i, j). In fall, the contribution of the three functional groups was again different, with higher values estimated for deciduous broadleaves and lower values for evergreen broadleaves and conifers (Fig. 2e–j).

The seasonal cumulated stomatal O_3 fluxes (g/m^2) maps depicted in Fig. 3c–f, highlighted that evergreen broadleaves showed cumulated stomatal flux values fairly constant from spring to summer 2003 (~ 0.8 g/m^2), as slightly increased in 2004 (from 0.4 to 0.5 g/m^2), as it can be observed in the Villa Ada urban park and in the Castelporziano Estate (Fig. 1b). The uptake values of deciduous broadleaves, shown for example in the large deciduous forest dominated by *Quercus cerris* and *Quercus frainetto*, located in the Castelporziano Estate (southern coastal area; see Fig. 1b), were on average similar in the spring–summer of 2004 (0.6 g/m^2) but, in 2003, were higher during spring (0.7 g/m^2) than during summer (0.4 g/m^2). In the same area, conifers showed ozone uptake values largely increasing from spring to summer in both years (from 0.2 to 0.8 g/m^2 in 2003, and from 0.4 to 0.8 g/m^2 in 2004).

Total (stomatal and non-stomatal) ozone uptake by urban trees was 311.1 Mg in 2003 and 306.9 Mg in 2004, with an interannual fluctuation between the two years of

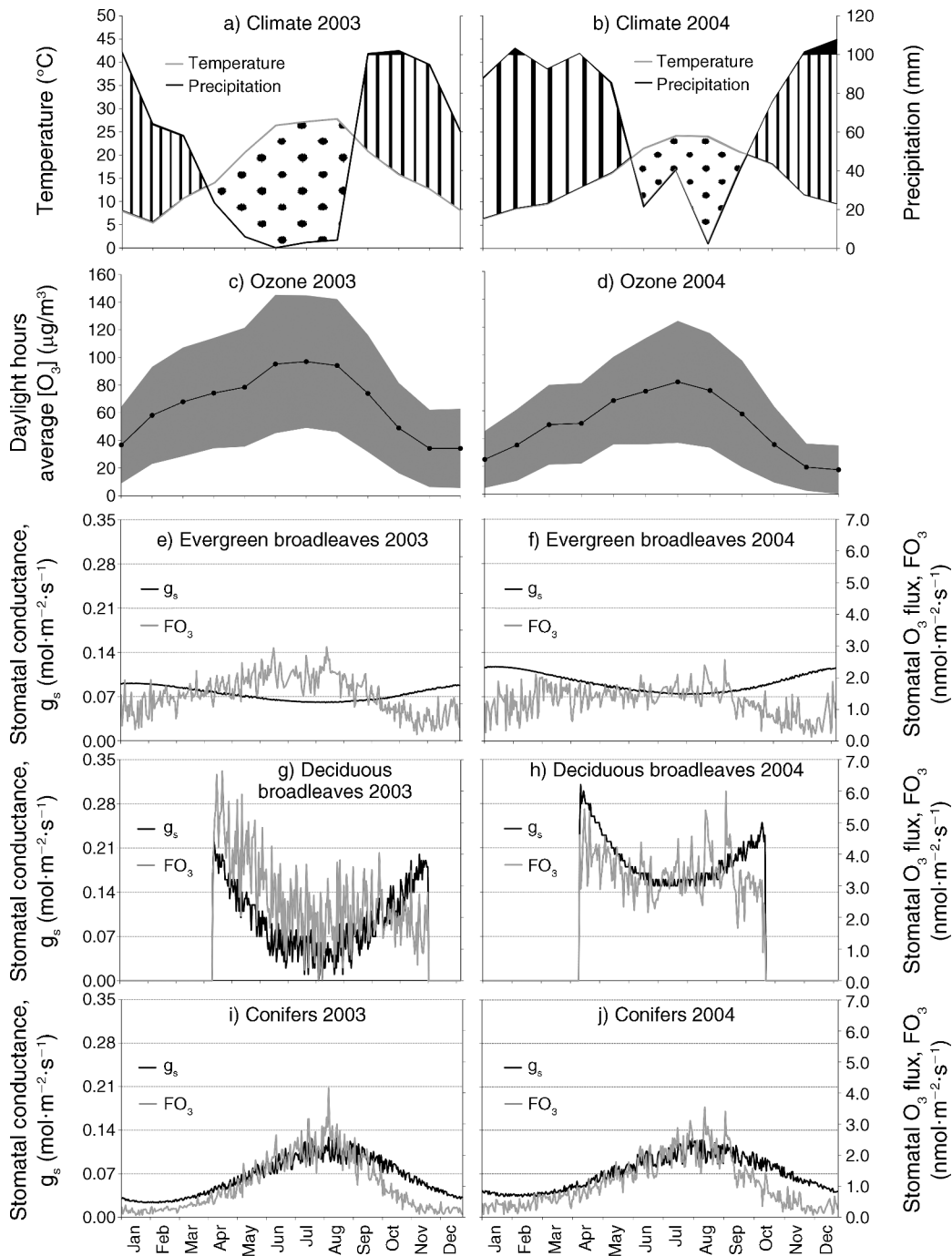


FIG. 2. (a, b) Climograms, (c, d) O_3 time series, and (e–j) time series of daily average stomatal conductance (g_s ; black line) and potential stomatal O_3 flux (FO_3 ; gray line) by the tree functional groups (evergreen broadleaf species, deciduous broadleaf species, and conifers) in the Municipality of Rome during the years 2003 and 2004.

<2% (Table 1). However, when the three functional groups were considered separately, a much higher variability was found. While annual ozone uptake decreased of 25% from 2003 to 2004 for evergreen broadleaves, an opposite pattern was found for deciduous broadleaves and conifers. For deciduous broad-

leaves, annual ozone uptake slightly increased by 4.5% from 2003 to 2004, while, for conifers, the annual ozone uptake increased considerably by 23% from 2003 to 2004.

The seasonal relative contribution of the three functional groups to total ozone removal is reported in

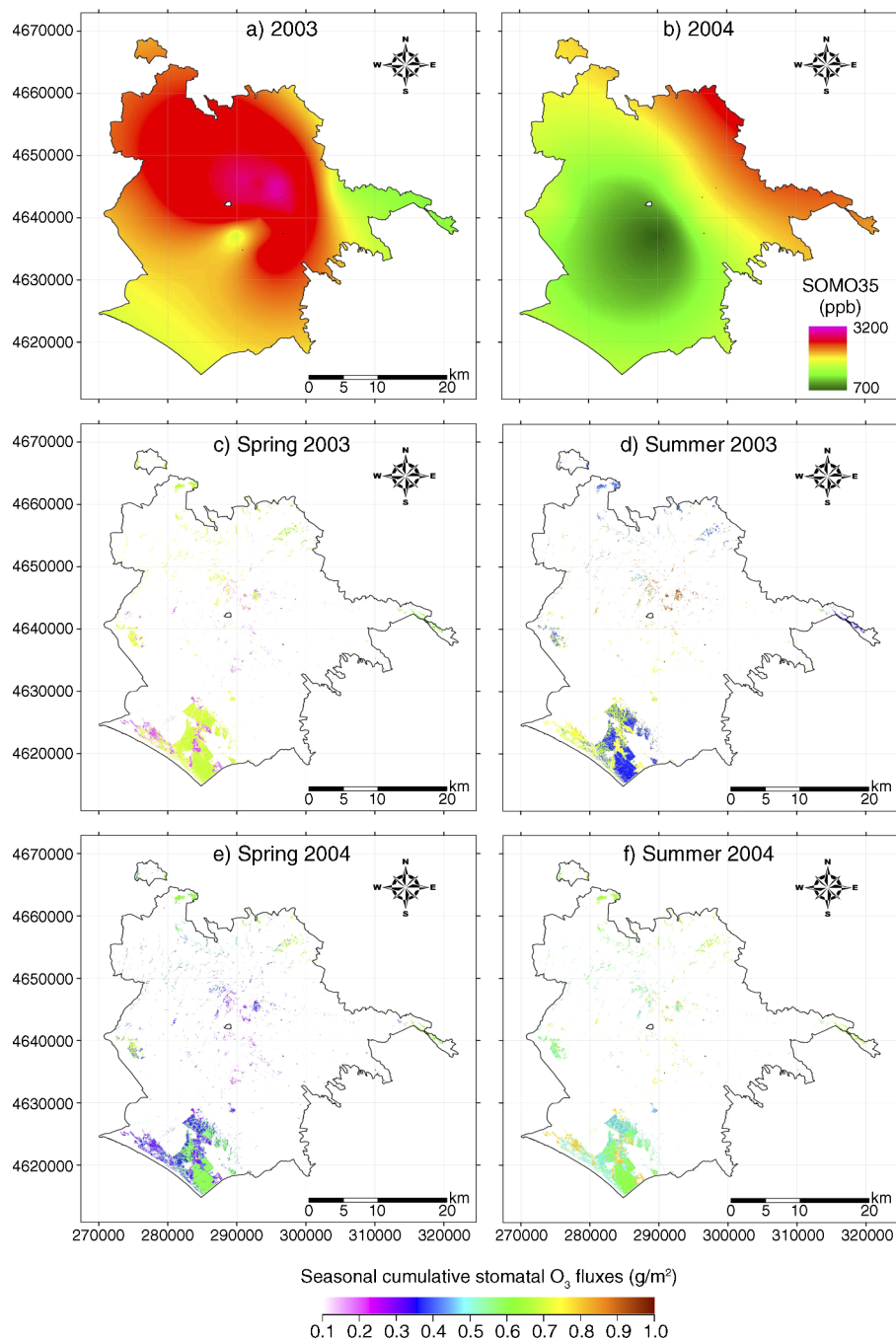


FIG. 3. Spatial patterns of O₃ concentrations and seasonal cumulated stomatal O₃ fluxes for urban trees in the years 2003 and 2004: (a, b) annual SOMO35 (i.e., sum of daily eight-hour running mean of O₃ over 35 ppb; Martuzzi et al. 2006); (c, d) seasonal cumulated stomatal O₃ uptake by urban trees in spring and summer 2003; and (e, f) seasonal cumulated stomatal O₃ uptake by urban trees in spring and summer 2004. Axes report UTM grid coordinates, WGS84 zone 33 N.

Table 1, where the removal data were weighted by tree cover for each group (Mg/ha).

The estimated annual O₃ uptake that would have occurred in both years if all the trees would belong to the same functional group is shown in Table 2. Although the three scenarios are not significantly different from the

actual data set as for the total O₃ removal (2003 + 2004), the inter-year differences (absolute values) in ozone uptake between 2003 and 2004 for the actual vegetation pattern (4.2 ± 39.7 Mg) are much lower than for all single-tree configurations (i.e., 15.1 ± 46.5 , 67.0 ± 55.2 , and 90.0 ± 25.4 Mg for the deciduous-only, conifers-

TABLE 1. Seasonal and yearly total ozone removal by the tree functional groups in Rome, Italy, in 2003 and 2004.

Season and year	Ozone removed, by functional group							
	Evergreen broadleaves		Deciduous broadleaves		Conifers		Total	
	Total (Mg)	Normalized (Mg/ha)	Total (Mg)	Normalized (Mg/ha)	Total (Mg)	Normalized (Mg/ha)	Total (Mg)	Normalized (Mg/ha)
Winter								
2003	9.9	0.0047	0.0	0.0000	2.6	0.0016	12.5	0.0017
2004	11.6	0.0055	0.0	0.0000	8.1	0.0050	19.7	0.0027
Spring								
2003	41.9	0.0198	81.9	0.0236	11.4	0.0071	135.2	0.0188
2004	25.7	0.0121	71.4	0.0205	17.2	0.0107	114.2	0.0159
Summer								
2003	41.6	0.0196	45.9	0.0132	43.5	0.0271	131.0	0.0182
2004	34.5	0.0163	72.9	0.0210	44.5	0.0277	151.9	0.0211
Fall								
2003	10.5	0.0050	16.6	0.0048	5.2	0.0032	32.3	0.0045
2004	5.5	0.0026	7.4	0.0021	8.1	0.0050	21.1	0.0029
All year								
2003	103.9	0.0490	144.4	0.0416	62.7	0.0391	311.1	0.0432
2004	77.4	0.0365	151.7	0.0437	77.8	0.0485	306.9	0.0426

Note: Values are expressed as total ozone removal and ozone removal normalized by tree cover.

only, and evergreen-only configurations, respectively; Table 2).

DISCUSSION

The values of annual ozone uptake normalized by tree cover estimated for the Municipality of Rome are within the range reported for other urban areas of different continents (Nowak et al. 2000, 2006, Yang et al. 2005, Escobedo and Nowak 2009). The observed differences in ozone removal ability among the main tree functional groups (evergreen broadleaves, deciduous broadleaves, and conifers) can be considered the results of four main factors: tree cover, plant physiology, leaf season length, and air ozone concentration. In particular, interannual and seasonal differences in ozone removal by evergreen broadleaves are primarily related to the dynamics of atmospheric ozone concentration, rather than to plant stomatal conductance, in good agreement with the well-known drought tolerance of this group (Manes et al. 1997). On the contrary, for deciduous broadleaves, stomatal conductance was highly affected by the extreme drought of summer 2003 (Vitale et al. 2007),

which produced lower ozone fluxes with respect to summer 2004. However, in fall, the recovery of stomatal conductance of this functional group determined higher ozone fluxes in 2003 than in 2004, in correspondence to higher ozone levels. For conifers, the slight increase in ozone uptake in 2004, as compared to 2003, was mainly related to the winter period. Overall, this group showed an interannual stability in ozone removal rate, associated to a rather constant trend of stomatal conductance (Fig. 2), suggesting a low sensitivity of conifers to the drought conditions occurring in 2003 (see e.g., Manes et al. 1997).

Spatial differences in seasonal ozone uptake are related to the complex interactions between the spatial distribution of the three functional groups across the city of Rome and the interannual spatial dynamics of ozone concentrations in 2003 and 2004. The majority of ozone removal by urban trees occurred in the southern coastal area, where the largest urban (Castel Fusano) and peri-urban (Castelporziano Presidential Estate) forests are located, and where tree diversity is highest. However, Fig. 3c–f shows that the urban forest patches

TABLE 2. Simulation of yearly ozone removal that would have occurred in Rome in 2003 and 2004 if all urban trees belonged to one single functional group.

Tree functional group	Ozone removal (Mg)			Inter-year difference (absolute value in Mg)
	2003	2004	Total	
Evergreen broadleaves	352.9 ^a ± 13.6	262.8 ^a ± 11.8	615.7 ^a ± 25.4	90.0 ^a ± 25.4
Deciduous broadleaves	299.7 ^{bc} ± 20.1	314.9 ^b ± 26.4	614.6 ^a ± 46.5	15.1 ^{bc} ± 46.5
Conifers	281.8 ^c ± 26.0	348.8 ^c ± 29.2	630.6 ^a ± 55.2	67.0 ^{ab} ± 55.2
Actual functional groups cover	311.1 ^b ± 20.7	306.9 ^b ± 19.0	618.1 ^a ± 39.7	4.2 ^c ± 39.7

Notes: Data are means ± SD, the latter calculated from six simulation runs of the MOCA-Flux (Modeling of Carbon Assessment and Flux) model. For each column, superscript letters indicate significant differences among tree functional groups (Duncan test, $P < 0.05$).

in the city center also played an important role by improving air quality in the most urbanized sites (Escobedo and Nowak 2009, Morani et al. 2011).

While showing only negligible effects on the total amount of O₃ removal in 2003–2004, at least for our specific case study, urban tree diversity significantly affected the stability of such ecosystem function. Indeed, a higher interannual difference in ozone uptake was shown by the simulated single-tree configurations as compared to the actual tree cover, especially for the evergreen-only and the conifers-only scenarios. Accordingly, the functional differences (such as response to drought and length of the leaf-growing seasons) among tree groups, together with their spatial distribution across the city of Rome, gave rise to a stable community function under very different climatic conditions, in spite of the seasonal fluctuations in ozone uptake of the different groups.

The observed results fit with a conceptual model of higher ecosystem function stability at increasing diversity levels. The diversity–stability debate (McCann 2000, Gamfeldt et al. 2008) is still controversial, with several long-term experimental studies providing evidence of both stable and unstable highly diverse ecosystems (Tilman et al. 2006), and only a few papers have analyzed the effects of biodiversity on urban ecosystem services (Bolund and Hunhammar 1999, Dearborn and Kark 2009). In this view, our study is the first reporting on the diversity–stability debate in urban environments, with particular focus on the function of air quality improvement by tree species. The peculiar dynamics of yearly ozone removal observed for each functional group (Table 1) shows that under changing climatic and air pollution conditions, the relative contributions of these groups to total ozone removal are likely to vary accordingly. From our analysis, the stabilizing effect of urban tree diversity on yearly O₃ removal appears as an emerging property of the urban ecosystem. A common opinion here is that the diversity–stability relationship is highly context dependent (Bezener and van der Putten 2007). However, while the impact of urban vegetation on O₃ removal strictly depends on site-specific conditions, cities usually host high levels of plant diversity (Celesti-Grapow et al. 2006), which could trigger such stabilizing effects irrespective of the environmental characters of the study area. Further studies are required to address the role of urban tree diversity on the stability of this ecosystem property in other urban areas, under different climatic conditions, pollution levels, and urban tree vegetation pattern.

In general, these results could have important implications for the development of future management strategies, such as targeted tree planting in selected locations or for evaluating the potential benefits to the stabilizing effect on ozone uptake that could derive from the replacement of native plant species with ornamental exotic ones. In particular, the flora of Rome is mainly composed by native species (Celesti-Grapow and Blasi

1998), and the main exotic tree species are the deciduous broadleaves *Platanus x acerifolia* and *Robinia pseudacacia*, and the conifer *Pinus pinea*, the introduction of which dates back to the Roman times.

It is worth noting that interannual difference of ozone removal estimated for the deciduous-only configuration, although higher, is statistically comparable to the value obtained for actual vegetation cover. Further, it is interesting to note the simulations for all single-functional group configurations, not biased by the relative tree cover of each functional group (Table 2). The interannual differences of ozone uptake by the three functional groups were statistically significant for all possible pairs, with the exception of the deciduous broadleaves as compared to conifers (Table 2). At first glance, such observations might suggest an interchangeability of these two tree functional groups with respect to the stability of the ecosystem function, considering their overall contributions in the city of Rome. However, the results of our simulations strongly depend on the trade-off between the variability of ozone trends and spatial distribution observed within the city in the study period. Consequently, it is not possible to state, for example, that replacing all the conifers with deciduous broadleaves in a specific urban park, would not affect the stability of overall ozone removal by urban trees in the long term. In general, under a perspective of urban green management, this implies that the cover of deciduous broadleaves should not be extended at the expense of the other functional groups. Further, as the response to drought of the studied groups represents a key aspect of their functional complementarity in removing ozone, it can be suggested that the replacement of drought-tolerant species with less tolerant ones may negatively affect the stabilizing effect of diversity.

On the other hand, urban trees may be also associated to “ecosystem disservices,” such as emissions of volatile organic compound (Escobedo et al. 2011). Varying amounts of biogenic volatile organic compounds (BVOCs) are emitted by different trees (Guenther 1997, Kesselmeier and Staudt 1999), which, in combination with NO_x, could have a negative impact on ozone formation (Owen et al. 2003, Noe et al. 2008). In Rome, evergreen broadleaves include both strong and medium monoterpene emitters (like *Quercus ilex* and *Q. suber*, respectively), deciduous broadleaves include both species with negligible VOC emissions (*Q. cerris*) and medium isoprene emitters (*Platanus x acerifolia*, *Robinia pseudoacacia*), while conifers are dominated by the medium monoterpene emitter *Pinus pinea* (Loreto 2002, Loreto et al. 2004, Calfapietra et al. 2009, Steinbrecher et al. 2009). Different approaches have been developed for the quantification of BVOCs emission and reaction potential at urban (Wang et al. 2003, Donovan et al. 2005), regional (Guenther et al. 2000, Leung et al. 2010), and global scales (Guenther et al. 2006). Nevertheless, evidence on the extent to

which BVOCs may contribute to increase O₃ concentrations is still contradictory. Assuming that all BVOCs emitted by trees react in air to form O₃, Benjamin and Winer (1998) estimated the O₃-forming potential of urban trees and shrubs. While useful for modelers, this approach is overly simplified because several compounds in the urban atmosphere can scavenge BVOCs, thus decreasing the amount of BVOCs available for ozone formation (Di Carlo et al. 2004). While a maximum O₃ increment of 10 µg/m³ due to BVOC emissions has been estimated for the Mediterranean area (Thunis and Cuvelier 2000), Nowak et al. (2000) reported that changing urban vegetation composition to low BVOC emitters would not produce significant effects on O₃ concentrations. Furthermore, Calfapietra et al. (2009) and Fares et al. (2010) have underlined that, given the scavenging operated by BVOC reactivity with ozone, trees which emit VOCs may have the capacity to increase their ozone uptake in comparison with non-emitting leaves at the same level of stomatal conductance.

From an economic viewpoint, ecosystem services may be divided into two general categories: market services and nonmarket services (Liu et al. 2010). While measuring market values simply requires monitoring market data for observable trades, nonmarket services are much more difficult to value. As we are not aware of any official data for Europe, the monetary value of the amount of O₃ removal was estimated using median externality values for the United States. For ozone, this value in U.S. dollars per metric ton is O₃ = \$6752/Mg, and is derived from the median monetized dollar per ton externality values used in energy decision making from various studies (Nowak et al. 2006). Based on this value, for the city of Rome (Table 1), we obtained an estimate of an annual saving of roughly \$2 million.

From a human health viewpoint, a meta-analysis estimation of the relationship between O₃ concentrations and the percentage of increase in mortality risk (Bell et al. 2006) indicated a 0.3% increase in mortality per 10 ppb increase in O₃ concentrations. For the population of Rome, the attributable fraction was calculated and applied to baseline cardiovascular and respiratory mortality rates (Martuzzi et al. 2006), which translates to an estimated mortality of roughly 80 deaths/year due to ozone. Using a value of a statistical life of \$1 million results in an annual cost of \$80 million. It has been shown that the urban forests leads to a decrease in mean O₃ concentrations of ~3% (Alonso et al. 2011). For the Rome population, this would correspond to an estimated decrease in mortality of roughly 3 deaths/year, with a yearly saving of roughly \$3 million. These are, of course, rough estimates and are conservative since they leave out morbidity and other health effects and use a conservative estimate of the value of a statistical life (Blomquist 2004). Nonetheless, if the value of ozone uptake by urban trees is calculated based on the reduction in human health damages, we

obtain a monetary estimate of the same order of magnitude of that based on externality values.

In conclusion, our results suggest the importance of urban tree diversity for stabilizing emerging ecosystem services, such as O₃ removal in urban environment, thus enhancing human health and well-being. Our findings, once extended to other urban environments, vegetation types, and climatic conditions, could bear important implications for environmental policy and green management plans oriented at increasing provision of services in large metropolitan areas, while contributing to a comprehensive valuing of urban forest diversity.

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SUPPLEMENTAL MATERIAL

Appendix

Input values of the MOCA-Flux (Modeling of Carbon Assessment and Flux) model (*Ecological Archives* A022-022-A1).