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STUDY OF URBAN GREEN AND ATMOSPHERIC POLLUTION MITIGATION CAPACITIES

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To Failure, of which I have made a Teacher and a Friend

"I don't like people who have never fallen or stumbled.

Their virtue is lifeless and it isn't of much value.

Life hasn't revealed its beauty to them."

Il Dottor Zivago

SUMMARY

Short Abstract

The PhD thesis project foresees studies of urban parks in different cities in Italy and France. As a first study, the candidate focused on the city of Rome and its peri-urban park of the Presidential Estate of Castelporziano, a study site accessible to CREA and of great impact in the mitigation of pollution in the capital. In particular, Castelporziano was also used as a sampling site for the validation of the Leaf Area Index (LAI) parameter. Thematic maps and field observations in this peri-urban park together with additional parks in the city of Turin were used to parameterize the AIRTREE model, a multi-layer canopy model parameterized with biometric and ecophysiological data. Finally, studies and statistical analyses were conducted on urban green areas in three target cities (Milan and Bologna) in the frame of the EU LIFE VEG-GAP programme. Specific questionnaires were submitted to the municipalities, and a statistical analysis based on tree inventories was carried out. For each city, thematic maps of green areas were elaborated in a GIS environment, with maps of tree species, Phenology and LAI. Subsequently, AIRTREE was applied to these target cities to predict the sequestration of CO2 and pollutants by urban trees.
Indeed, the AIRTREE model, applied in this way, becomes a very important tool to help municipalities in their urban green planning, contributing to an effective choice of the type and quantity of green to be planted.
This is useful not only locally, but as a contribution to the new European regulations, which push more and more towards the fight against climate change, and the greening of the urban environment. The project was realized by combining, for the first time, the characteristics of air pollution in the target city and the pollution mitigation capacity of the urban plant ecosystem. The results of this study highlight the importance of developing an integrated approach that combines modelling and satellite data to link air quality and the functionality of green plants as key elements in improving the effectiveness of useful Ecosystem Services in cities.

Keywords

Ecosystem Services; Street Trees; Municipality; Urban Green; Urban Park; Peri-urban Park; AIRTREE; Pollutant

Extended abstract

Cities are responsible for more than 80% of global greenhouse gas emissions. To maximize Ecosystem Services (ES), proper planning of urban green areas is necessary. The sequestration of air pollutants is one of the main ES that urban forests provide to citizens. The atmospheric concentration of various pollutants such as carbon dioxide (CO_2), tropospheric ozone (O_3) and particulate matter (PM) can be reduced by urban trees through adsorption and deposition processes. It was predicted the amount of CO_2, O_3 and PM removed by urban tree species with the AIRTREE multilayer canopy model in two urban parks (Castel di Guido Park and Valentino Park), in a peri-urban park (Presidential Estate of Castelporziano) and in street trees in the cities of Milan and Bologna.

The work on the two urban parks consisted of extensive in situ measurements and modelling analyses of tree characteristics to provide realistic estimates. In the Presidential Estate of Castelporziano, were integrated ground-level and satellite measurements to estimate the sequestering capacity of atmospheric pollutants from the atmosphere, transported to the study site in particular from the city of Rome. On the other hand, vegetation maps were used for the street trees in Milan and Bologna, which were developed through a supervised classification algorithm, trained on remote sensing images, supplemented by the inventory of local trees and used as input for the AIRTREE model.

Among many results, the two urban parks showed 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. In the Castelporziano Estate, on the other hand, due to a significant loss of forest cover, estimated by satellite data at -6.8% between 2014 and 2020, it was found that carbon sink capacity decreased by 34% during the period considered. In addition, the deposition of pollutants on tree crowns decreased by 39%, 46% and 35% for PM, NO_2 and O_3, respectively.

Going deeper into the city and examining the urban greenery of the street trees in Milan and Bologna, both cities found a large presence of deciduous broadleaf trees, which showed a greater capacity to sequester CO_2 (3,953. 280 g m^2 y^{-1}), O_3 (5677.76 g m^2 y^{-1}) and NO_2 (2358.30 g m^2 y^{-1}) than evergreen needle-leaves, which, on the other hand, showed higher performance in removing particulate matter (14711.29 g m^2 y^{-1} and 1964.91 g m^2 y^{-1} for PM_{10} and $PM_{2.5}$, respectively). It has been identified the tree species with the highest carbon uptake capacity, with values of up to 1025.47 g CO_2 m^2 y^{-1} for *Celtis australis, Platanus x acerifolia, Ulmus pumila* and *Quercus rubra*. Furthermore, it was estimated thatthat trees in Milan are able to sequester the CO_2 emissions of 460 vehicles per year (0.06% of the car fleet circulating in Milan in one year). For PM, on the other hand, it was estimated

that trees in Milan sequester the emissions of 16000 vehicles per year (2.28% of the car fleet circulating in Milan in one year).

In light of impending and unprecedented policy measures to plant millions of trees in urban areas, this study highlights the importance of developing an integrated approach that combines field surveys, modelling, and satellite data to link air quality and green plant functionality as key elements in improving ES provision in cities. Thanks to these studies, it is possible to understand which species, or better still which species associations, should be chosen according to the specific city context and the type of pollution in that city in order to achieve the best performance. Since this study can be replicated in any context, it becomes a valuable aid tool for green planners in municipalities.

1. Introduction

1.1 The role of greenery and the importance of Ecosystem Services in the urban contest

The concept of urban green is constantly evolving in society, in social relations, thanks also to advances in science and technology. There are numerous types of urban green areas, which have their own characteristics and whose recognition is important not only for scientific, but also for cultural and management aspects. Proposals for classifying types of urban green areas are reported, both on an international level by the European Commission (European Commission, 2016), and on a national level by ISPRA (2009).

In general, urban and peri-urban green areas are included in the so-called trees outside the forests (TOF). And the discipline that deals with studying them is called Urban and Peri-urban Forestry (UPF) (Konijnendijk et al., 2006). In particular, urban forest can be defined as networks or system comprising all woodlands, groups of trees, and individual trees located in urban and peri-urban areas, they include, therefore, street trees, trees in parks and gardens.

Delving into the above-mentioned documents (European Commission and ISPRA) three types of urban green areas are classified, two referring to public green areas and one to private green areas. Public green, understood as the patrimony of green areas present in the territory and managed by public bodies, according to the context and the purpose for which it is set up, can be recognized as furnishing green (historical, urban and public gardens, urban and peri-urban parks, roof gardens, street trees, flowerbeds, zoological and botanical gardens) and functional green (sports, school, health, cemetery, nursery). Public green is therefore a fundamental element of the urban landscape. It is of great importance for improving the quality of life in municipalities with multiple functions depending on the type, such as aesthetic, social and recreational (equipped green) and historical-artistic (historical green).

In particular, street trees, urban and peri-urban parks represent a real multifunctional resource for the city and its inhabitants. Indeed, they can be a tool for redevelopment, continuity and integration between urban buildings and the surrounding natural environments (AA. VV. 2010). In this sense, for example, street trees can be considered 'green corridors' connecting the green lungs (urban and peri-urban parks) spread throughout the territory, becoming an integral part of it. Urban forests are the backbone of the green infrastructure bridging rural and urban areas and ameliorating a city’s environmental footprint (Sheets and Manzer 1991; Lovasi et al., 2008). Public green is therefore an integrated system in the urban environment and the choice of tree species becomes decisive in characterizing it (Micheli, 2008). Among its many functions, urban greenery regulates the microclimate, mitigating the temperature through the process of evapotranspiration (Minelli, 2007). Moreover, trees also perform filter functions by retaining particulate matter (PM), produced by vehicle exhausts, and finally they play an important role in mitigating noise pollution, creating a natural dissipative barrier (Minelli, 2007). In addition to these, there are other typical functions of trees: shading, reducing wind speed and mitigating surface runoff of rainwater (Mimo, 2012). In general, therefore, urban greenery has a decisive influence on the appearance, ecology and economic value of the environment (Figure 1 and Figure 2). Several publications and reports have been produced in recent years with the aim of highlighting the economic and environmental importance of TOF in the context of international conventions (Rigueiro-Rodríguez et al., 2008; Barbati et al., 2014; Schnell et al., 2015).

Figure 1: Row of Holm oaks in Corso Giuseppe Mazzini, Campobasso. *Photo by Ilaria Zappitelli.*

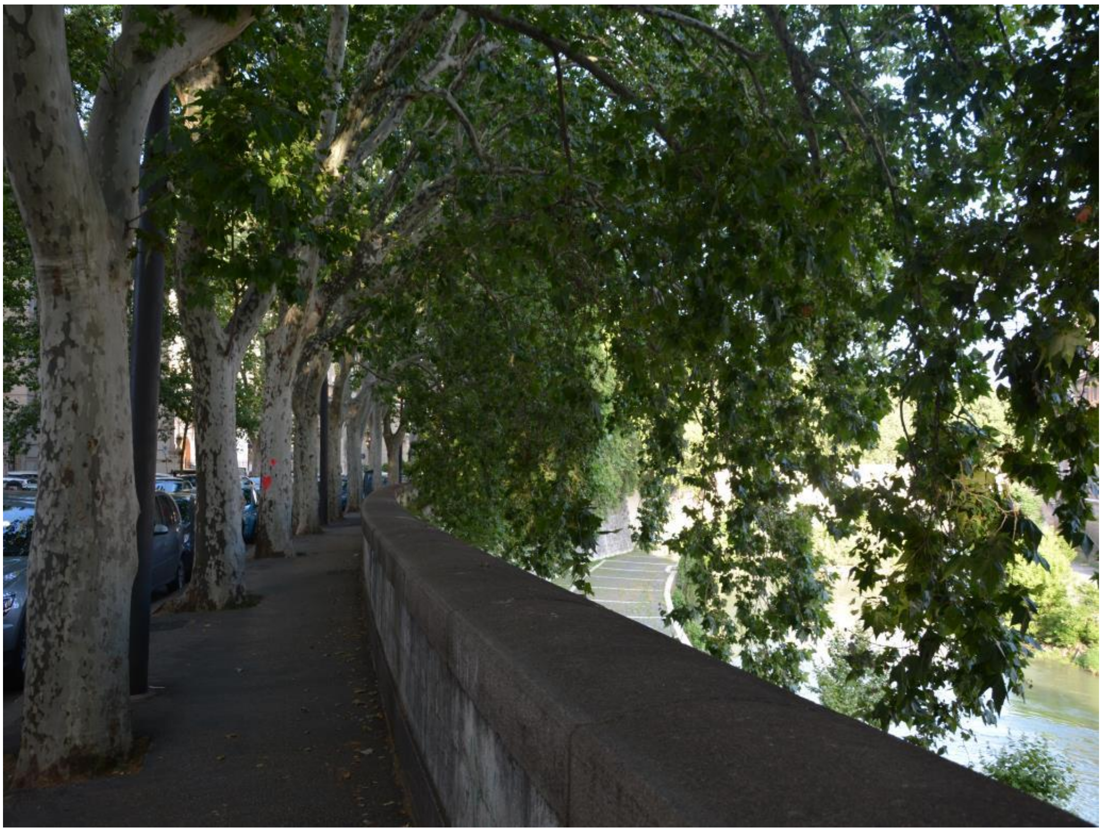


Figure 2: Street Tree of *Platanus* of the Lungotevere Avenue, Rome. *Photo by Ilaria Zappitelli.*

The evaluation of the species to be planted in street trees and in urban and peri-urban parks is important because it is necessary, first of all, to consider the functions they will have to perform and prevent them from creating problems for structures or inconveniences for citizens. This implies that urban greenery must be treated as a system, i.e., a set of elements whose sum but also the integration of individual functions must be considered (Santolini, 2011). When an ecosystem function becomes an element from which benefits can be derived, it is called a service. This is the background to the concept of Ecosystem Service (ES), which has gained increasing acceptance since the late 1990s, with the study by Costanza (1997). Ecosystem Service (ES) refers to both the goods (such as food, water, raw materials, building materials, genetic resources) and the functions and processes of ecosystems, many of which are emergent properties resulting from the functional integration between the elements of an ecosystem that contribute to produce a process or perform a function: absorption of pollutants, protection from erosion and flooding, regulation of surface water run-off and drought, maintenance of water quality, soil formation, climate micro-regulation, noise reduction, conservation of biodiversity etc. (Bolund, 1999; MA, 2005; Elmqvist et al., 2015; Grote et al., 2016). These processes and functions provide irreplaceable benefits, either directly or indirectly, to the inhabitants of a territory, contributing to maintaining the functionality and ecological quality of the local ecosystem, but not only: some services are of global interest (e.g., maintaining the chemical composition of the atmosphere), others depend on the proximity of inhabited areas (e.g. protection function from destructive events), and others are only locally expressed (e.g. recreational function) (Costanza, 2008).

Since Costanza's study (1997), numerous national and international projects on ecosystem services have been promoted in various fields: TEEB (The Economics of Ecosystems and Biodiversity), EEA/MA 2015 (European Environmental Agency/Millennium Ecosystem Assessment 2015), DIVERSITAS (International Scientific Programme on Biodiversity). These various studies have identified four general categories of Ecosystem Services that can also be considered as possible evaluation criteria that can be used in the selection of plants in tree plantations:

- availability and provisioning of resources (provisioning);
- regulation or mitigation of processes and events (regulating);
- availability of living environments and conditions (supporting);
- cognitive and cultural function (cultural).

(For an extensive list of Ecosystem Services provision see Samson et al. (2017)).

The availability of SE is recognized as an important contributor to human well-being and poverty reduction (MA, 2005). In the Millennium Ecosystem Assessment (MA, 2005), in particular, it is noted that most ES are already in decline or are indicated with possible negative trends in the future. For example, biodiversity is considered the basis of the ES, as it is important for the maintenance of many of them, and the COPI study ("Cost of Policy Inaction - The case of not meeting the 2010 biodiversity target") highlights and evaluates the economic consequences of biodiversity loss on a planetary scale as very significant (Braat & Ten Brink, 2008).

Another example is the Regulating Ecosystem Services (RES), which represent the benefits to mankind deriving from the regulation of ecosystem processes (Reid et al., 2005), i.e., the sequestration of pollutants, which are at the root of the current disruptive climate change. Air pollution is the main environmental health risk in Europe and is associated with heart disease, stroke, lung disease and lung cancer (Lovasi et al, 2008). It is estimated that exposure to air pollution causes more than 400000 premature deaths per year in the EU (Lelieveld et al., 2015; EEA, 2019; Psistaki et al., 2022). The role of forests as nature-based solutions to mitigate the effects of climate change has been largely discussed (Bäckstrand et al., 2006), becoming a fundamental part of the proposal formulated at international level during important summits as the G20 (Rome) and COP26 (Glasgow) that fix the ambitious goals of planting one trillion trees (Rome G20) and stop de-forestation (COP26 Glasgow) by 2030.

The provisioning of ES by urban and peri-urban forests has direct effect on the social, economic and environmental benefits. There is, in fact, a growing demand for ES due to the increase in world

population but above all due to the desire to maintain a high lifestyle on the part of a portion of the population of the rich and economically more advanced countries (Santolini, 2011). On the other hand, ecosystems and their functionality are subject to a series of pressures, arising from factors related to economic policies, technological development and also dependent on welfare expectations and consumption choices.
In contrast to ESs, there are the Ecosystem Disservices (ED). There is still no unanimous definition of Disservices (Lyytimaki, 2009), however, it has been pointed out that some ecosystem functions can also have detrimental effects on human well-being, and this can be understood as a disservice (von Döhren, 2014). For example, the allergenicity of certain products of certain species, or the greater or lesser emission of BVOCs by certain species. Ecosystem disturbances have to be carefully evaluated because an analysis of the functionality provided by a street tree, urban and peri urban parks must consider not only its benefits but also the possible problems associated with the various species selected. Street trees and urban and peri-urban parks, as integrated systems in the urban ecosystem, can be considered subjects of analyses concerning Ecosystem Services and Disservices, in order to establish what are the criteria for choosing a species that can become part of urban green. Applying the approach used in the assessment of ES to street trees is a complex operation as it must respond to numerous factors simultaneously.

1.2 Regulatory Ecosystem Services and their estimation through the use of modelling

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 (Evans et al., 2016). 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 very often to PM_{10} above the EU daily limit value (EEA, 2019). In Italy, exposure to fine particles and ozone is responsible for over 66000 premature deaths every year and 19.5 times higher than road accidents (www.Istat.it). Particles are generated mainly by residential energy use such as cooking and heating, but also by vehicular emissions, power generation, and agriculture (Lelieveld et al., 2015). 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 85ppb (Lefohn et al., 2014; Pachauri, 2014). According to the analysis by Sicard et al. (2018), 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 wellbeing. Together with adoption of a new technology in the energy sector and mobility, it is now clear that street trees, urban and peri-urban parks can deliver a multitude of ecosystems services and, therefore, can provide a significant contribution to the air quality improvement (Nowak et al., 2014; Manes et al., 2016). 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 (Fares et al., 2016; Crompton et al., 2017). Therefore, street trees, urban and peri-urban parks provide clean air to citizens, and in general air quality measurements in urban parks can be adopted as reference of local background levels (Dall'Osto et al., 2015; Crompton et al., 2017). 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 (Samson et al., 2017). This "sink" capacity of plants results from interactions between meteorology, chemical, and physical characteristics of the pollutants and the properties of the canopy (Pröhl et al., 2012). 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 demonstrated in urban-to-rural comparisons showing lower CO_2 concentration in the presence of vegetation in urban parks and street trees in the city may slightly decrease concentrations of atmospheric CO_2 locally, especially when they are away from strong emitting sources (Park et al., 2013; Helfter et al., 2011). While the photosynthetic process and carbon assimilation have been widely investigated, plant leaves can additionally absorb pollutants when they penetrate through stomata (Dusart et al., 2019). Even particles can be intercepted by vegetation and retained on the surface of leaves, trunks bark, or can be absorbed into plant tissues (Han et al., 2020). Ozone deposition has been described in several agricultural and forest ecosystems, displaying two separate sinks: leaf stomata, and plant/soil surfaces (Clifton et al., 2020). Green surfaces have indeed an added value versus non vegetation 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 (Clifton et al., 2020). As leaves and canopies 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 (Pryor et al., 2008), 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 (Janhall et al., 2015). 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 (Xing et al., 2020; Barwise et al., 2020). Urban planning and management (plant species or planting configuration) has also an impact on dry deposition at the stand/regional level (Barwise et al., 2020). 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 green (Corona et al., 2018). 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 several dendrometric attributes in order to provide realistic estimations (Nowak et al., 2014; Han et al., 2020; Nowak et al., 2006; Fares et al., 2019; Simpson et al., 2012).

Estimating the contribution of “process-driven” RES such as air quality improvement, water purification, and climate regulation to human well-being is typically made by models (Mengist et al., 2020), often challenged by a lack of input data, especially for estimations at broad scales (Villamagna et al., 2013).

Canopy models help unravel the factors that influence the response of the forest ecosystems in different environmental conditions by testing scenarios (Bernstein et al, 2008) or evaluating the impact of different stressors (Otu-Larbi, 2020). Such models need climatic and ecophysiological inputs that are often unavailable, and direct measurements needs to be performed to obtain reliable estimates (Wang et al., 2022). Meteorological and air quality data for big cities are often available at hourly to daily resolution (ARPA, 2016; Deserti, 2001; Fares et al., 2020), and when missing (rural areas), high-resolution gridded datasets can be used when proper spatial and temporal resolutions are available (Haylock et al, 2008; Holloway et al, 2021). Better estimation of the biosphere–atmosphere interactions has been observed by multilayer canopy models (Bonan et al., 2021). These models are more complex than single-layers (or big-leaf models) and often require species-specific ecophysiological parameters to properly estimate plant’s capacity to provide RES (Fares et al., 2019). Stomatal conductance is a key parameter not only for carbon assimilation but also to estimate the uptake of pollutants as tropospheric ozone (O_3), nitrogen dioxide (NO_2) and Sulphur dioxide (SO_2) (https://apps.who.int; Delaria et al., 2020). Therefore, stomatal regulation processes need to be estimated accurately. This is particularly relevant for the Mediterranean region, where dry deposition is the dominant pathway (Paoletti et al., 2006) and stomatal conductance is limited by stressors such as drought and ozone exposure (Fusaro et al., 2018; Hoshika et al., 2022). On one hand, coupled photosynthesis–stomatal conductance (A gs) models such the semi-empirical model proposed by Ball, Woodrow & Berry (BWB) (Ball et al., 1987), or the theoretical optimal stomatal behavior models proposed by Medlyn et al. (2011) and by Katul et al. (2009) are best indicated for simulating leaf-level gas exchanges (Fares et al, 2018); on the other hand, these models require physiological parameters such as carboxylation velocity (Vcmax) and light use efficiency (Jmax) not always available in literature. To overcome this issue, direct field measurements are often required (Fares et al., 2020; Fares et al., 2019). In addition, such complex models need bio-metric parameters to characterize the trees' structure or the total leaf area, which is a key parameter to upscaling fluxes at the canopy level (Fares et al., 2020). Structural features determination requires an extensive campaigns effort that are difficult to realize at metropolitan scale. However, plant-trait databases and satellite-derived vegetation indices are effective means to derive at large spatial scale those essential information (Fuster et al., 2020; Kamenova et al., 2020). Indeed, vegetation indicators of key structural parameters such as the leaf area index (LAI, m^2 m^{-2}) and canopy cover (%) were available from 2014, thanks to the monitoring activity of the Proba-V satellite (Fuster et al., 2020). The provisioning of RES is not costless for forests since air pollution represents a threat for vegetation as well as. For instance, ozone damage can cause up to 10% of carbon loss in Mediterranean forest ecosystems (Conte et al., 2021; Ot-Larbi et al., 2021).

A few other tools are dedicated to tree inventory in urban settings. These include the i-Tree suite (so far only adapted for application in the USA) (Morani et al. 2014), Greece's Urban Tree Management applications (Tasoulas et al., 2013) and the canopy model AIRTREE (Fares et al., 2019).

AIRTREE is the main model of this Doctoral work, so in this study, it was used for the first time in an urban environment on street trees, and parameterised on urban and peri-urban parks, so that an estimate of RES could be obtained.

1.3 The satellite approach and the use of vegetation mapping

The urban forest ecosystem is complex and multifaceted; understanding how it works requires new types of data, at appropriate spatial and temporal scales, to explain forest dynamics, and remote sensing can fill this gap (Recanatesi et al., 2018; Kamenova et al., 2021). In an urban environment, Remote Sensing can support evaluation of land use and tree cover (Huang et al, 2013; Xu et al, 2020; Shahtahmassebi et al., 2021; Kowe et al., 2021; Bai et al., 2022). Detailed information on tree species characteristics can be produced with the support of remotely sensed products. For instance, vegetation indexes such as Leaf Area Index (LAI), and phenological indicators may strongly support management and planning of urban green infrastructures and/or support parameterization of models used to evaluate Ecosystem Services provided by urban vegetation (Fares et al., 2019; Fares et al., 2020). Nowadays, forest management problems are multiscale and intricately linked to society's need to measure and preserve multiple forest values (Boyd, 2004). Considering this, it is necessary to build green inventories and maps to quantify such benefits and at the same time to better manage the planning of green areas inside and outside the cities.

Significant progress in remote sensing technology and data collection and analysis has been made to facilitate land assessment at different levels (local, national, regional and global).

A thorough assessment requires fine spatial resolution imagery to detect the presence of individual or small groups of trees. With this in mind, the tool allows users to visualize and interpret very high-resolution images (from the Digital Globe archive in Google Earth and Bing Maps), in conjunction with a number of bio-geophysical indices and automatically generated time-series data (e.g., derived from Landsat , MODIS and Sentinel 2 data in the Google Earth Engine) (Bey et al., 2016).

For example, the city surveys of the AIRFRESH project (on part of which the PhD foreign internship was carried out) were done on maps acquired via very high-resolution satellites. This enabled very good data management.

There are numerous study examples of the use of satellite-acquired green mapping. Some studies are done on the time series of the Landstat and MODIS satellites. Most studies, however, require a not inconsiderable economic investment for the purchase of very high-resolution maps, and do not include the application of modelling to estimate ES on this type of data. Examples of studies are: Li et al., 2015; Talukdar et al., 2020; Fang et al., 2021.

Among other things, the fully automated counting of trees in urban and non-urban locations has never been done before, so we still need a manual phase for calibration or validation of the satellite data. In this study, satellite data was merged with ground truths and the species list provided by the municipalities in question, or with field sampling.

1.4 Thesis objective and structure

The capacity to provide Ecosystem Services is a species-specific trait, generally attributed based on the size and shape of the canopy and other specific plant traits (Grote et al., 2016). Indeed, as reported by Bodnaruk (2017) and Conte et al. (2022), there are plants that are better suited to be used in environments with strong anthropic pressure. Bearing this in mind, it is therefore necessary to assess the best combination of species according to their eco-physiological characteristics (Fares et al., 2019; Conte et al., 2022), in light of the environment in which they are to be placed (Calfapietra et al., 2013; Anderegg et al., 2022).

In this study, the AIRTREE model (Fares et al., 2019) was applied for the first time to the urban context of Milan and Bologna, two cities located in the Po River basin, one of the areas exposed to the highest air pollution levels in Europe, and characterized by different size, population density and geographic location. The model was applied with modelled atmospheric pollutants and species-specific biometric data combined with georeferenced maps. By using microclimatic, physiological, and morphological characteristics of each species inventoried by the Municipalities, it was adopted a spatially explicit method to generate vegetation maps with the goal to identify the tree species with the higher performances in sequestrating carbon and pollutants from the atmosphere and estimate the capacity of the overall greenery within each municipality to provide such relevant ES.

During the first two years of the PhD program, AIRTREE model was parameterized in two urban parks (Valentino Park in Turin; Castel di Guido Park in Rome) and one peri-urban park (Presidential Estate of Castelporziano). This type of parameterization was necessary because the ultimate aim was to use the model on the street trees of two target cities (Milan and Bologna), which had never been done before. The parameterization therefore made it possible to approach the objective by taking care to assign the right parameters and variables to the species present, but also subsequently to be able to make a comparison on the sequestration of pollutants between the various types of urban greenery.

In the case of the two urban parks, two field campaigns were conducted (by CREA teams), where the structural tree properties and ecophysiological indicators of a wide range and distribution class of typical tree species used in Mediterranean regions in streets and parks were measured. This study was designed to improve the knowledge of the structural and ecophysiological characteristics of a wide range of urban trees in order to support a realistic estimate of the ES provided by urban parks, but also to begin to familiarize with green mapping and the AIRTREE model at the beginning of the PhD study.

As for the study on Castelporziano, it was an excellent field and mapping exercise considering the extraordinary access that CREA has to this estate. This important vulnerable natural forest area was therefore designated a test site. In this case, the reduction and the gaps in the percentage of canopy coverage between three study years (2014, 2019 and 2020) were first highlighted. The three years of the study are characterized by different rainfall regimes, ranging from a known wet year (www.nimbus.it; Ratna et al., 2017; Conte et al., 2019) to increasingly dry years in 2019 and 2020. In terms of exposure to pollutants, 2020 differed from previous years since the lockdown status of Sars-CoV-2 (Donzelli et al., 2020; Collivignarelli et al., 2020; Cameletti et al., 2020; Gualtieri et al., 2020; Balasubramaniam et al., 2020) has produced a drastic decrease in emissions from road traffic. As regards biotic stress, probably to limit the expansion of *Tomicus destruens* and *Toumayella parvicornis* infestation, several thinning campaigns were carried out on the estate between 2017 and 2019 (Recanatesi et al., 2019; Cucca et al., 2020). Thus, a comparison between these years, in this area, aimed to reveal the impact of altered anthropogenic emissions on ES supply capacity, and the impact of both pathogenic infestation and management practices. In a second phase, the sequestration of carbon and air pollutants was estimated using the multilayer model AIRTREE (Fares et al., 2019) parameterized with measurements at leaf level, structural data derived from satellite and georeferenced vegetation maps. Applying in this way the model as it was then applied on the green maps of Milan and Bologna.

The project on Milan and Bologna was carried out by combining, for the first time, the characteristics of air pollution in the target city and the pollution mitigation capacity of the urban plant ecosystem. This study highlights the importance of developing an integrated approach that combines modelling and satellite data to link air quality and green plant functionality as key elements in improving the effectiveness of useful ecosystem services in cities.

As it is crucial to know which and how many trees are needed to optimize ES, the question was whether AIRTREE can be understood as a tool to help municipalities in their urban green planning, contributing to an effective choice of the type and quantity of green to be planted.

This would become useful not only at the local level, but as a contribution to new European regulations, which increasingly push towards combating climate change and greening the urban environment.

These studies were also conducted in cooperation with the LIFE VEG-GAP project, which produced the VEG-GAP portal (https://www.lifeveggap.eu/), an information platform that evaluates the multiple services provided by plant ecosystems in an integrated way, through multi-scale and multi-pollutant approaches that resemble the real world as closely as possible. It further contributes to a real-world application in cities by using these means for reasoned urban planning.

Regarding the structure of the thesis:

- **Case Study 1** (images and texts) have been published in the article: **Fares, S., Conte, A., Alivernini, A., Chianucci, F., Grotti, M., Zappitelli, I., ... & Corona, P. (2020). Testing removal of carbon dioxide, ozone, and atmospheric particles by urban parks in Italy. *Environmental Science & Technology*, *54*(23), 14910-14922.**, of which Doctoral student Ilaria Zappitelli is coauthor.
- **Case Study 2** (images and texts) have been published in the article: **Conte, A., Zappitelli, I., Fusaro, L., Alivernini, A., Moretti, V., Sorgi, T., ... & Fares, S. (2022). Significant Loss of Ecosystem Services by Environmental Changes in the Mediterranean Coastal Area. *Forests*, *13*(5), 689.,** of which Doctoral student Ilaria Zappitelli is coauthor and on a par with the first author as stated on the paper.

- **Case Study 3** (images and texts) have been published in the article**: Zappitelli, I., Conte, A., Alivernini, A., Finardi, S., & Fares, S. (2023). Species-Specific Contribution to Atmospheric Carbon and Pollutant Removal: Case Studies in Two Italian Municipalities. *Atmosphere*, *14*(2), 285.,** of which Doctoral student Ilaria Zappitelli the only first author.

2. State-of-the-art

2.1 Urban and Peri-urban areas in Mediterranean basin

Throughout Europe, the Mediterranean region is facing many challenges, some of which can be addressed by nature-based solutions such as urban forests and green space. However, at best, urban forest research from Mediterranean countries has been only briefly addressed in review papers up to date (Krajter Ostoić et al., 2018).

Its variety of marine and coastal ecosystems and wide diversity of fauna and flora make the Mediterranean one of the world's richest natural habitats. The region is characterized by extraordinary biodiversity, including a large number of endemic species. At the same time, it is threatened by critical levels of habitat loss (Ernoul et al., 2012).

A Mediterranean climate is one characterized by mild winters and hot and dry summers, with precipitation concentrated in autumn, winter and early spring. Total rainfall varies strongly from year to year and violent precipitation events, or dry winds can occur. The winter temperature occasionally falls below 0 °C at sea level, and snow and below-zero temperatures are common at high altitudes.

The unusual geographical and topographical variability and a pronounced climatic bi-seasonality have strongly influenced species richness and distribution in the Mediterranean region, which is a biodiversity hotspot with high endemism (Médail and Quézel 1997; Myers et al., 2000) (Figure 3).

Especially as regards the ecosystem/vegetation state of the art, data on trees outside forests (TOF) remains scarce and incomplete (Schnell et al., 2015). Most institutional assessments only give a partial depiction of the state of TOF, the focus being primarily on a particular typology. It is also difficult to reach a consistent and accurate assessment of TOF between countries, particularly as there is not yet a clear, agreed definition of the term.

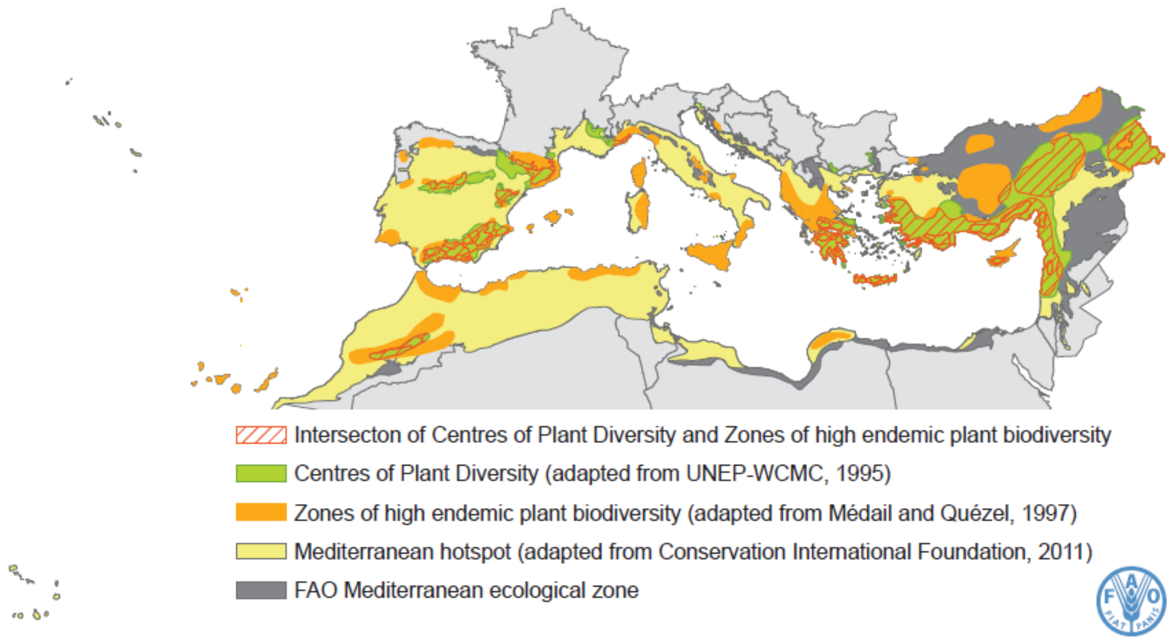


Figure 3: Mediterranean hotspots and centers of plant diversity *(from Médail and Quézel, 1997; FAO and Plan Bleu, 2013).*

Anyway, it is known that the state of the art describes are various approaches to Urban and Peri-urban Forestry which allow defining the ecosystemic and vegetational general state of the art of the moment. The approach taken by north-eastern Mediterranean countries such as Slovenia, Turkey, Romania Croatia, Israel and Morocco, is linked to the planning, design and management of woodlands in and around towns, in line with the central–northern European approach (FAO and Plan Bleu, 2013).

A strong forestry tradition is shared between these countries, this guarantee that urban forestry is considered part of the wider forestry technical environment. For this reason, the planning, the designing and the management of urban and peri-urban forests and tree systems is responsibility of central governmental institutions too. A second approach focusing on urban parks and gardens is strong in Algeria, Bulgaria, Lebanon, Cyprus, Malta, France, Spain and Tunisia, Greece, Morocco and Egypt. These countries share the common consideration that urban forests only as one of the possible types of urban and peri-urban green open spaces where you can find tree. According to this approach, UPF is less management-oriented and concerns more territorial and forest planning than urban planning (FAO and Plan Bleu, 2013).

In a third approach, urban and peri-urban forests and tree systems are key elements in an ecological network of protected areas. They provide a series of multifunctional set of benefits, such as biodiversity, climate change adaptation, carbon sequestration and environmental quality. The European Mediterranean countries of France, Croatia, Italy, Greece, Spain, Portugal and Slovenia are very familiar with this approach (FAO and Plan Bleu, 2013).

A fourth approach, which could be called "integrated approach", involves the wide-ranging application of UPF techniques. This approach promotes the integration at multiple levels, of individual or street trees and cultural landscapes. For this reason, UPF has an open and advantageous dialogue with related disciplines, particularly landscape ecology, landscape architecture and urban and ornamental arboriculture. Countries in which this approach is pursued are France, Italy and Portugal and, to some extent, Croatia, Turkey, Greece and Slovenia (FAO and Plan Bleu, 2013).

Quantitative data on urban and peri-urban forests in the Mediterranean are generally very poor. On the basis of European Environment Agency data, Fuller and Gaston (2009) produced a map of green spaces in European countries (Figure 4) and found that the area of green space per person (the "per capita green space provision") varied significantly. The lowest values were found in the south and east and the highest in the north and northwest. In Italy, Corona et al. (2011) used national surveys to calculate the total number and surface area of national urban and peri-urban forests and tree systems and their average size, availability and macro species composition (conifer or broadleaved) (Table 1). However, while the World Health Organization has set minimum standards of 9 m^2 of green space per inhabitant, there is no official definition of "green space" itself, and as the meaning of the term changes significantly depending on local context and institutional framework, the interpretation of the data is problematic. A survey of 75 urban forests in major cities in Turkey reported an average size of 377 hectares with a very wide range (from 1 hectare to more than 11000 hectares). The survey applied a simple equation to estimate the carbon stored in those forests, which was more than 42000 tonnes. In France, data from the National Forest Inventory (IFN) and a population census were used to calculate the rate of forests influenced by urban agglomeration. The survey found that a significant part (21%) of the urban area was covered by forests and there was about 200 m^2 of forest per inhabitant (Inventaire Forestier National, 2010). Few data are available for other Mediterranean countries, especially in the southeast region. The Guide of the Urban and peri-urban Forests of Morocco (Haut Commissariat aux Eaux et Forêts et à la Lutte Contre la Désertification, 2010) reported an area of 2.5 m^2 of green space per inhabitant in Moroccan cities. This guide defined urban forest as a "forest inside the urban texture", and peri-urban forest as a "forest area influenced by an urban context" (according to a range of requested services, especially recreation and tourism) located less than 30 km from the urban area.

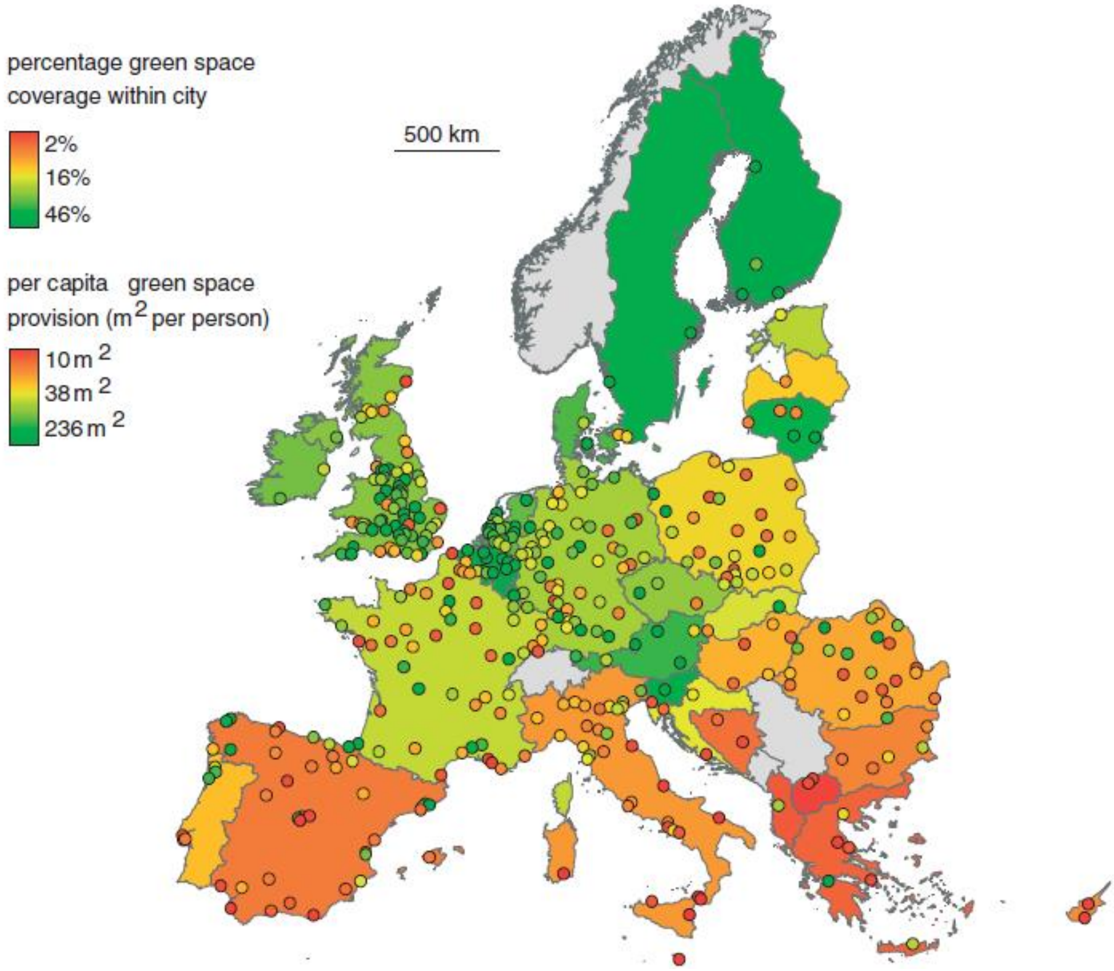


Figure 4: Urban green space coverage in Europe. Points representing cities are coloured according to proportional coverage by urban green space within the city. Country polygons are coloured according to per capita green space provision for its urban inhabitants *(from Fuller and Gaston, 2009)*.

Indicator		Size	Unit
Total number of urban forest		19 806	n
Total surface of urban forest		43 000	ha
Average size of urban forest		2.2	ha
Availability of UF per capita		7	m2
Tree species	Conifers	58	%
	Broadleaves	42	%

Table 1: Data concerning Urban Forests according to the National Forest and Carbon of Italy *(from Corona et al., 2011)*.

As for the qualitative data (choice of plants), typical Mediterranean forests are composed of broadleaved species, both evergreen and deciduous, such as *Quercus ilex*, *Quercus suber*, *Quercus Pubescens, Quercus coccifera, Quercus Ithaburensis, Quercus cerris, Quercus Toza, Quercus pyrenaica, Quercus calliprinos, and conifers such as Pinus halepensis, Pinus Pinea, Pinus brutia, Pinus pinaster* and *Juniperus* species. The degradation of such forests has produced low-density woody vegetation known as the macchia and the garrigue. Where there is no water limitation, forests of *Quercus robur, Quercus petraea, Fraxinus species, Populus alba* and *Populus nigra* can prosper. Typical of the Iberian Peninsula, the dehesa is characterized by pastures with scattered oaks, both evergreen and deciduous, sometimes mixed with *Pinus pinea*. The vegetation in the Xerothermo-Mediterranean zone comprises mainly shrub communities. Non-endemic species introduced over the course of the centuries may also be found in Mediterranean forests.

Obviously, as regards the trees in the city, the choices are dictated by the garden services, which often, especially in our peninsula, tend to choose the species based on low prices, or what they have in their hands at that time. This has favoured the introduction of numerous non-native plants, often harmful to native species.

A first national, comprehensive assessment of urban forest resources was carried out by the United States Forest Service (Dwyer et al., 2000). It applied a combination of methods, including satellite imagery, national statistical data and assessments for particular cities or metropolitan areas. Tree canopy cover was used as a more reliable indicator than land-use types. The assessment showed that 74.4 billion trees cover 33.4% of the metropolitan areas in the 48 adjacent states. In urban areas, 3.8 billion trees cover 27.1% of the land. No comprehensive, comparative assessments of urban forest resources in Europe seem to be available at the time. Table 2 includes information about some (partial or less reliable) assessments of urban green space cover. The European Environment Agency has provided statistics on urban green area cover in selected European cities but mentioned that statistics are unreliable and not easy to compare (EEA, 1995). It is uncertain, for example, what types of green areas have been included.

REGION/COUNTRY	INFORMATION ON URBAN GREEN SPACE RESOURCE	REFERENCE
Europe	Green space cover of selected cities varied between 5% (Madrid) and 60% (Bratislava). Tree (canopy) cover in cities in 8 European countries ranging from 1.5 to 62%. Green area cover for 14 European cities surveyed varied between 5% (Thessaloniki) and 56% (Ljubljana); average of approximately 30% for all cities. Green space per inhabitant from 6 to approximately 7000 m^2.	EEA (1995); Pauliet et al. (2002); Ottitsch (2002)
Belgium	Flanders region: 9% of municipalities covered by green areas (1991 survey). Brussels region: 14% (2300 ha) of land area covered by green space.	Basiaux et al. (1999)
Great Britain	Green areas cover approximately 14% of urban areas. Parks and green spaces estimated to account for 120 000–150 000 ha.	DTLR (2002)
The Netherlands	Average municipal green space cover of 19% for 22 of the largest Dutch cities (i.e., average of 228 m^2/inhabitant).	CBS (1998)

Table 2: Data on urban green space cover in Europe *(from Konijnendijk, 2003)*.

The major difficulty is to determine what woodlands are to be classified as ‘urban’. Table 3 provides the results of some assessments made. In many cases, urban woodland area cover is assessed by only including the areas defined as ‘forest’ within the municipal boundaries. Table 3 suggests that Nordic and Central European countries assess their urban woodland resource to one or several percents of their overall forest resource. (Konijnendijk, 2003).

REGION/COUNTRY	INFROMATION ON SIZE OF URBAN WOODLAND RESOURCE	REFERENCE
Europe	Average woodland cover of 18.5% within municipal boundaries of 26 larger European cities (104 m2/inhabitant)	Konijnendijk (2001)
Belgium	Flanders region: approximately 4.5% of municipalities included in 1999 survey covered by woodlands. Walloon region: 25 000 ha of forests located in suburban areas and managed for community uses. No data on municipal forests available. Brussels region: 1950 ha (12%) of land area covered by woodlands.	Bassiaux et al. (1999)
Czech Republic	Fifteen percent of all forests owned by municipalities and cooperations 100 larger cities own between 500 and 8000 ha of forests.	Zaruba (1998)
Finland	Municipalities in Finland own 430 000 ha of forests.	Lofstrom (1999) cited by Tyrvainen (1999)
France	270 000 ha of forests in the Greater Paris region; 80 m^2 of forest per inhabitant (compared to 800 m2 for France as a whole).	Moigneu (2001)
Latvia	0.8% of all Latvian forests considered urban forests (owned by cities and towns). Twenty percent of urban areas covered by forests.	Donis (2001)
The Netherlands	Average municipal woodland cover of approximately 7% for 22 of the largest Dutch cities. Larger cities usually have municipal forest cover of between 0 and 5% (1993 data).	CBS (1998)
Slovakia	Ten percent (186 000 ha) of Slovakian forests owned by municipalities.	Graus (1998)
Sweden	300 000 ha considered ‘urban fringe forests’, i.e., more than 1% of the overall Swedish forest cover.	Carlborg (1991) cited by Rydberg (1998)
United Kingdom	Community forests programme aimed at achieving a 30% woodland cover (≈119 000 ha) around 12 large agglomerations over next decades. Actual cover in 1999 was 6.5%.	Ball et al. (1999)

Table 3: Data on urban woodland area and cover in Europe *(from Konijnendijk, 2003)*.

3. Materials and Methods

3.1 Case Studies

Urban parks, peri-urban parks and street trees in cities are the three main categories of spaces, which provide Ecosystem Services for municipalities and their inhabitants thanks to the trees present.
As the work on street trees in cities is more fine-grained, the urban and peri-urban parks were also used to parameterize the AIRTREE model, as well as to compare species and pollutant sequestration.

3.1.1 Case Study 1: Urban Parks

3.1.1.1 Study area: Park of Castel di Guido and Park of Valentino

Regarding the study of urban parks, two parks were considered, one in Rome and one in Turin, representative of two different cities reflecting different conditions. These study sites were chosen because very timely measurements had been made by the CREA team in 2018 on the two parks. Those data were useful for the parameterization of AIRTREE.
The first study site is the Castel di Guido Park (hereafter named "Castel di Guido") located within the Rome Municipality covering around 3673 ha along the Tyrrhenian coast. Castel di Guido (41°54′ N, 12°17′ E) 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 (Cs climate in the classification of Koppen, 1936 and Kottek, 2006) (i.e., Xerotheric Bioclimatic Region of Latium and Thermo Mediterranean/Meso-Mediterranean Climate Subregion) (Blasi, 1994). Such climate may potentially support various forests dominated by mesophytic and thermophilus broadleaved tree species. The predominant soil types are calcic cambisols (Barbati et al., 2014). 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, Celtis australis, and Fraxinus ornus*, have been planted on smaller areas (down to 60 m^2) in order to increase structural and compositional diversity of the plantation project. A map with each species is reported in Figure 5. A full list of species is provided in Table 4.

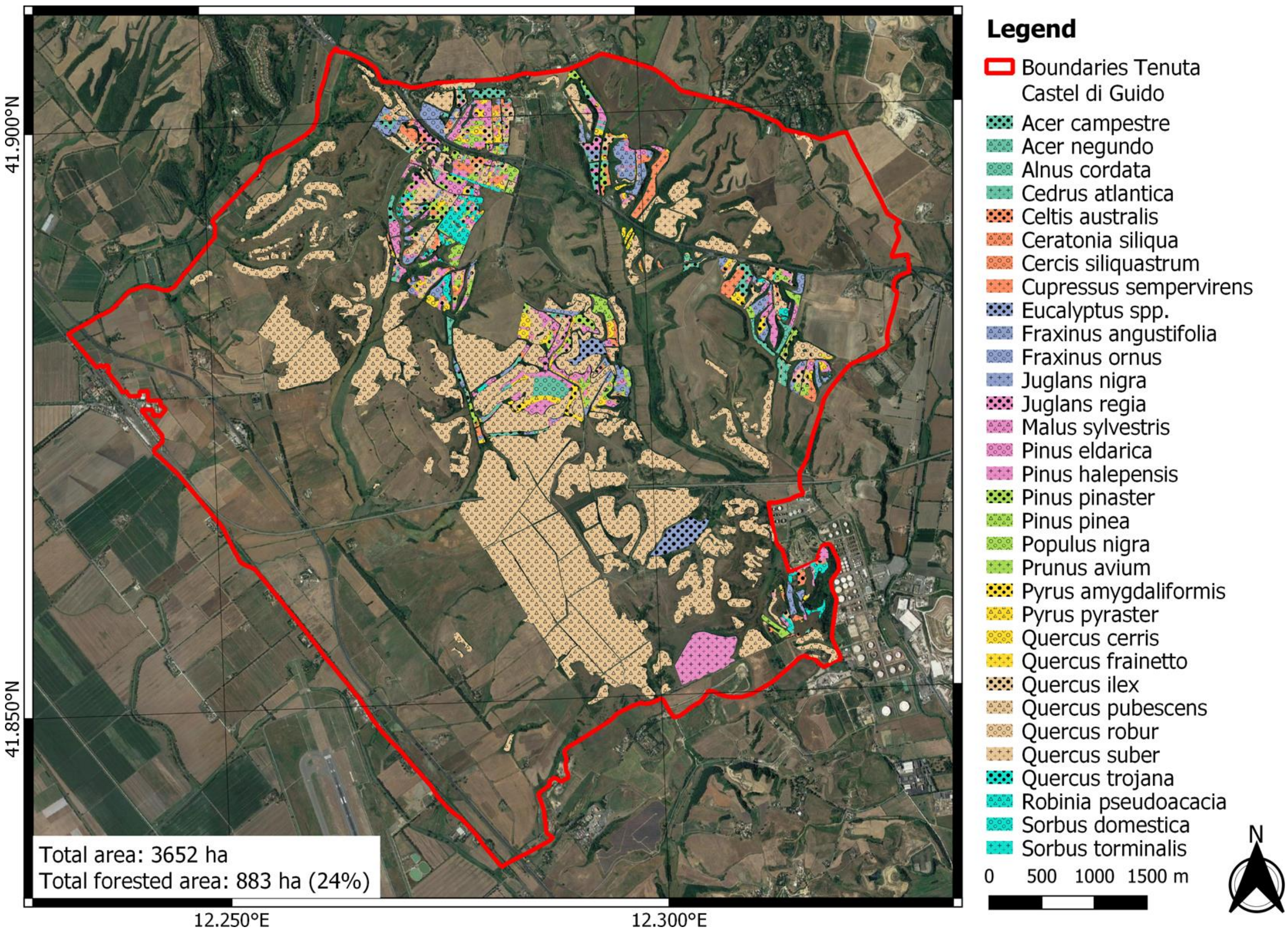


Figure 5: Map of the vegetation surveyed at the park of Castel di Guido, Rome. Map data 2020 Google *(from Fares et al., 2020).*

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

Table 4: 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. Model simulations were carried out for each species at different dbh. The results were grouped 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. *(From Fares et al., 2020).*

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 5 and mapped in Figure 6. The climate is mainly temperate (Blasi, 2014) (Cfa climate in the classification of Koppen, 1936 and Kottek, 2006) and according to FAO World Soil Resources, the predominant soil types are leptic technic fluvisols (FAO, 2015).

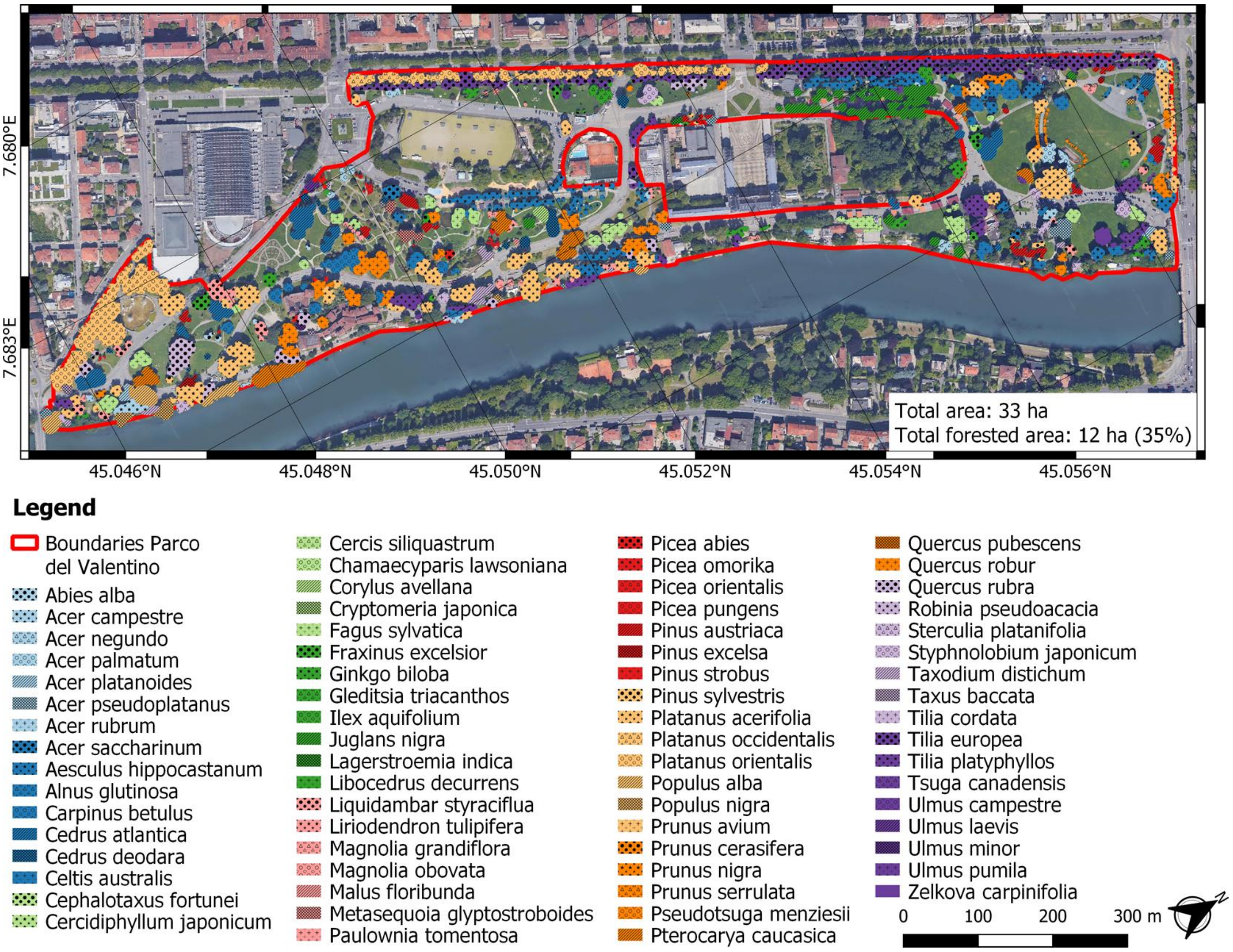


Figure 6: Map of the vegetation surveyed at the park of Valentino, Turin. Map data 2020 Google *(from Fares et al., 2020).*

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

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

Table 5: 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. Model simulations were carried out for each species at different dbh. The results were grouped 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. Evergreen species are marked in bold. *(From Fares et al., 2020).*

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).

3.1.1.2 Measurement 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 by Macfarlane et al. (2007) and Chianucci (2020). 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 from CREA's work team, 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 (Grotti et al., 2019).

3.1.1.3 Gas Exchange Measurements

Where not available in the literature, gas exchange was measured in the field by CREA team, using a portable photosynthesis system (LI-6400, LICOR, 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 $\mu mol\ m^{-2}\ s^{-1}$ 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). The relative humidity in the chamber was maintained between 40 and 70%. From these data, A-Ci response curves were fitted with the FvCB model by using the fitaci function from the Plantecophys R Package (Duursma and Plantecophys, 2015) to provide parameters for the biochemical model useful to estimate Vcmax (maximum rate of carboxylation) and Jmax (maximum rate of electron transport) in the calculator provided by Sharkey (2016).

3.1.2 Case Study 2: Peri-urban Park

3.1.2.1 Study area: Presidential Estate of Castelporziano

With regard to the study of a peri-urban forest, the Presidential Estate of Casteporziano, the site of numerous experiments of CREA and ambitious projects, was considered.

The Presidential Estate of Castelporziano (41°43′35″N, 12°24′14″E) is a natural reserve of about 60 km^2 (of which 85% are natural forests), 25 km away from the south-southwest area of Rome on the Tyrrhenian Sea coast. The Estate represents a hotspot for biodiversity in the Mediterranean area, which hosts more than 1000 plant species (della Rocca et al., 2001; Davison et al., 2009). Oak forests and mixed deciduous broad-leaved woods predominate, occupying 23.6 km^2 (40% of the woods), a surface higher than the evergreen oaks, and are represented by holm oaks (4.3 km^2) and cork oaks (2.4 km^2), by pine forests (9.1 km^2) and by Mediterranean maquis favored by anthropic land use changes in past centuries (Giordano et al., 2017). The most significant aspect that makes Castelporziano a territory now unique in the Mediterranean Basin is given by the presence of a specific and therefore very high genetic biodiversity, with 5037 species recorded, representing an exceptional wealth of flora and fauna on an area of just 60 square kilometers (Manes et al., 1997) (Figure 7).

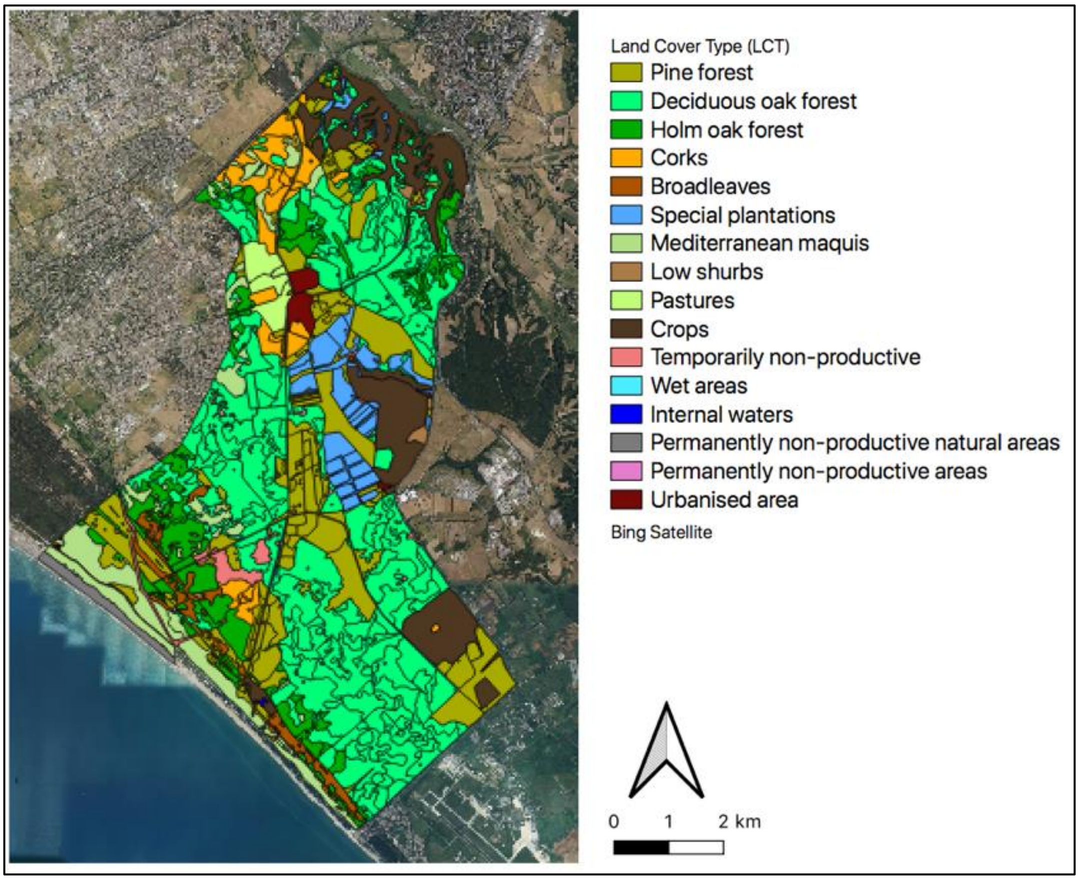


Figure 7: Vegetation map of the Presidential Estate of Castelporziano, colored contours represent homogeneous vegetation, each color represent one Land Cover Type (LCT). *(From Conte et al., 2022).*

However, this valuable Green Infrastructure being located in the Metropolitan area of Rome is subjected to wide types of stressors, both natural and anthropogenic (Fares et al., 2009; Fusaro et al., 2015). The intense urban sprawls nearby the Estate have caused an overexploitation of the water table and enhanced the seawater intrusion that exacerbate the natural drought that typically occurs during the summer. Moreover, high concentrations of pollutants that are transported over Castelporziano from the city center of Rome have been recorded. Indeed, the Estate receives plumes of air masses all day long from the sea and from the city of Rome because of the wind circulation, that follows a seal and breeze regime, the dominant wind direction is S–SW during the morning and N–NE during the afternoon. The site can experience high summer levels of tropospheric ozone up to 90 ppb (Fares et al., 2014). Soil presents a flat topography, and it can be divided in two main soil type: the coastal area, characterized by a sandy texture (77% sand, 14% silt, 9% clay) and low water-holding capacity, and the inland area, with a loamy-sandy soil type (47% sand, 29% silt, 24% clay) (Fischer et al., 2015). The study area is characterized by the typical Mediterranean climate, with pronounced seasonality. Summers are hot and dry, and winters are moderately cold (Df climate in the classification of Koppen, 1936 and Kottek, 2006). Precipitations occur prevalently during winter, spring and autumn. Annual precipitation up to 1019 +o- 105.5, 854 +o- 130.8 and 507 +o- 95.6 mmy^{-}

[1] was observed in the 3 years (reference years for this part of the study) in 2014, 2019 and 2020, respectively, considering averaged values for all the meteorological stations in the Estate. During the three years of study, mean annual temperature was of 16 °C with seasonal peaks in summer 2019 (Figure 8).

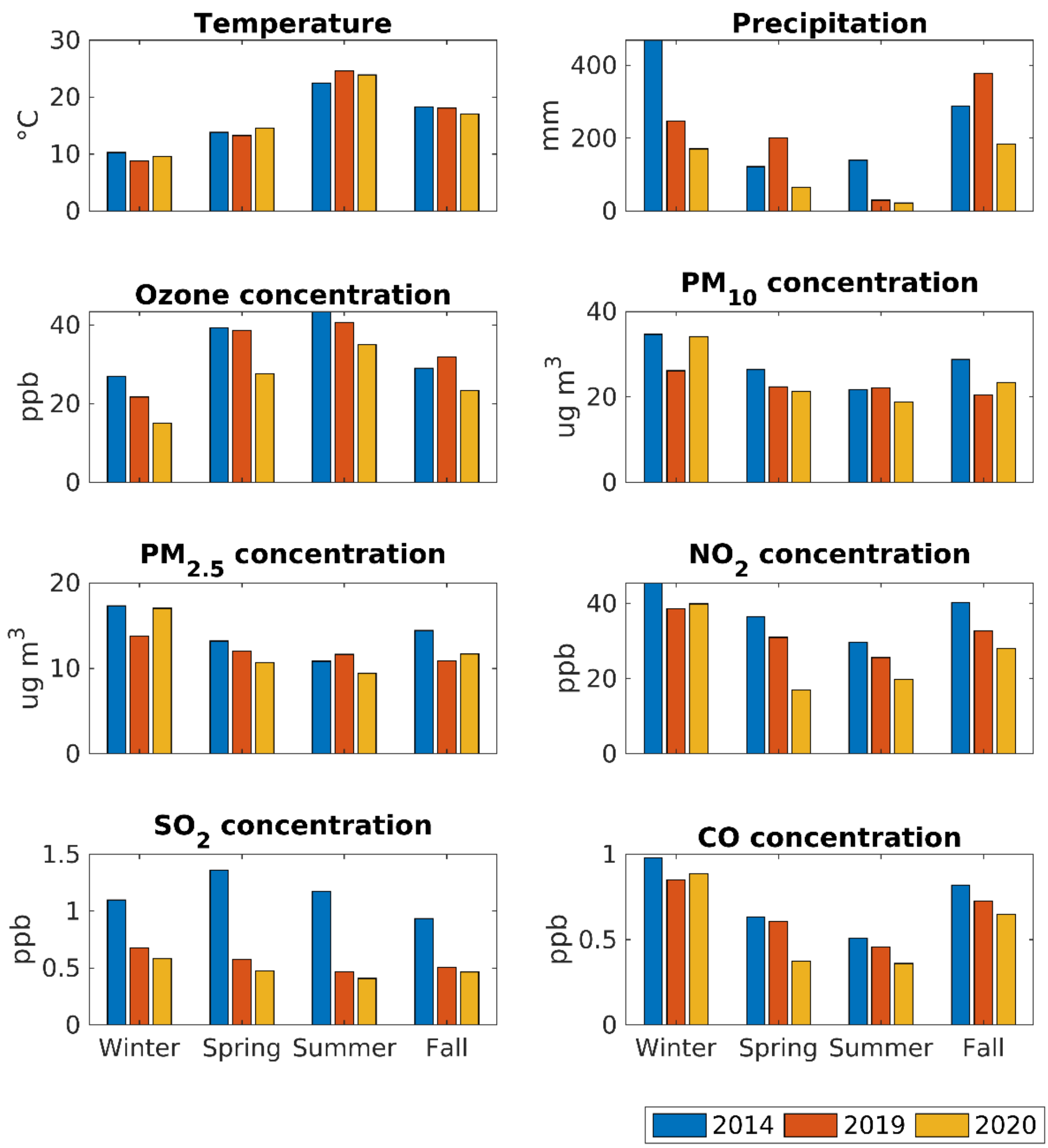


Figure 8: Meteorological inter-comparison between the three years of study seasonal values of mean temperatures, and cumulated precipitation (top left and top right, respectively) are shown together with average concentrations of PM, NO2, SO2 and CO. *(From Conte et al., 2022).*

3.1.2.2 Meteorological variables

For the entire Estate, it was performed a Voronoi tessellation using data from a network of 6 meteorological stations where continuous monitoring of local climatic conditions have been carried out (Figure 9).

Figure 9: Spatial association of meteorological stations to homogeneous portions of the natural reserve. Yellow dots indicate the position of each one of the meteorological stations. Climatic conditions of the portions of the Estate (red shapes) were associated to each meteorological station. *(From Conte et al., 2022).*

Each station is equipped with sensors measuring air temperature and relative humidity, wind speed and direction, precipitation and solar radiation; for details on instrumentation, see Aromolo et al. (2021). In addition, the Estate hosts a flux tower for continuously monitoring gas exchanges (i.e., CO_2, O_3, NO_2, PM) between the vegetation and the atmosphere in the coastal area, 1.5 km away from the Thyrrenean sea (see Fares et al., 2014 and Conte et al., 2019 for details). Just outside the Estate, within 1.5 km, there are 4 air quality monitoring stations maintained by the regional agency for the environmental protection (ARPA Lazio) that continuously monitor concentration of PM, NO_2, SO_2, CO, O_3 and other pollutants. Those air quality data were used as input for the AIRTREE model. The data were assumed to be exemplificative of Castelporziano site since a comparison between $PM_{2.5}$ concentration measured in 2014 and 2015 at the ICOS It-Cp2 flux site and average values measured at the 5 ARPA stations showed a good correlation (Pearson's $r > 0.7$, data not shown). The Voronoi tessellation method allowed us to generate a spatial-metric decomposition determined by distances to discrete set of elements in space (meteorological stations), and therefore, 6 Voronoi polygons were created for the whole Estate.

3.1.2.3 Vegetation maps

The vegetation map of the Castelporziano Estate (Figure 7) used in this study (Recanatesi et al., 2015) is characterized by 7 groups of forested land cover type (LCT) (Other broadleaves; Low shrubs; Holm oak forests; Mediterranean maquis; Pine forests; Deciduous oak forests; Corks) and 9 non-forested LCT (Internal waters; Permanently non-productive areas; Permanently non-productive natural areas). To associate species-specific ecophysiological traits to each of the forested LCT group, it was used the state-of-the-art of literature about the vegetation in Castelporziano to identify the most representative species included in each group (della Rocca et al., 2001). Therefore, the values of carbon, PM and other pollutants deposition for each pixel (associated to a specific LCT) are the results of a single model simulation. For each model run, the AIRTREE model was parametrized for an ideal tree-type by synthetizing the ecophysiological characteristics of the dominant trees representative of the LCT (Table 6).

LCTs	Association	Dominant species	Vcmax	O3 Tolerance
Mediterranean maquis	*Quercetum ilicis galloprovenzale*	*Pistacia lentiscus L. Arbutus unedo L., Juniperus oxycedrus L., Phillyrea angustifolia L.*	51.8	moderate
Corks	*Quercetum ilicis galloprovenzale suberetosum*	*Quercus suber L*	61	low
Holm oak forests	-	*Quercus ilex L., Quercus frainetto Ten., Quercus cerris L., Quercus pubescens Willd. Fraxinus ornus L.;*	63.5	low
Oak forests	Quercetum ilicis var. farnetto	*Quercus cerris L., Quercus frainetto Ten., Carpinus orientalis Mill.*	63.5	low
Pine forest	-	*Pinus pinea L.*	18.7	low
Broadleaves	Hygrophilous woods	*Fraxinus oxycarpa Willd., Alnus glutinosa L., Ulmus minor Mill., Fraxinus ornus L., Ficus carica L.*	71	low
Low shrubs	-	*Rubus ulmifolius Schott., Prunus spinosa L., Crateagus monogyna Jacq.;*	38.6	high

Table 6: For each LCT, the corresponding association is reported. For each association, average ecophysiological parameters (e.g., Vcmax and ozone tolerance) of the dominant species are used as model input to characterize the LCT ideal tree-type. *(From Conte et al., 2022).*

3.1.2.4 Retrieval of Biometric Vegetation Data

A raster map of the species associations was superimposed and aligned with the maps of Leaf Area Index (LAI), canopy cover and heights (Figure 10).

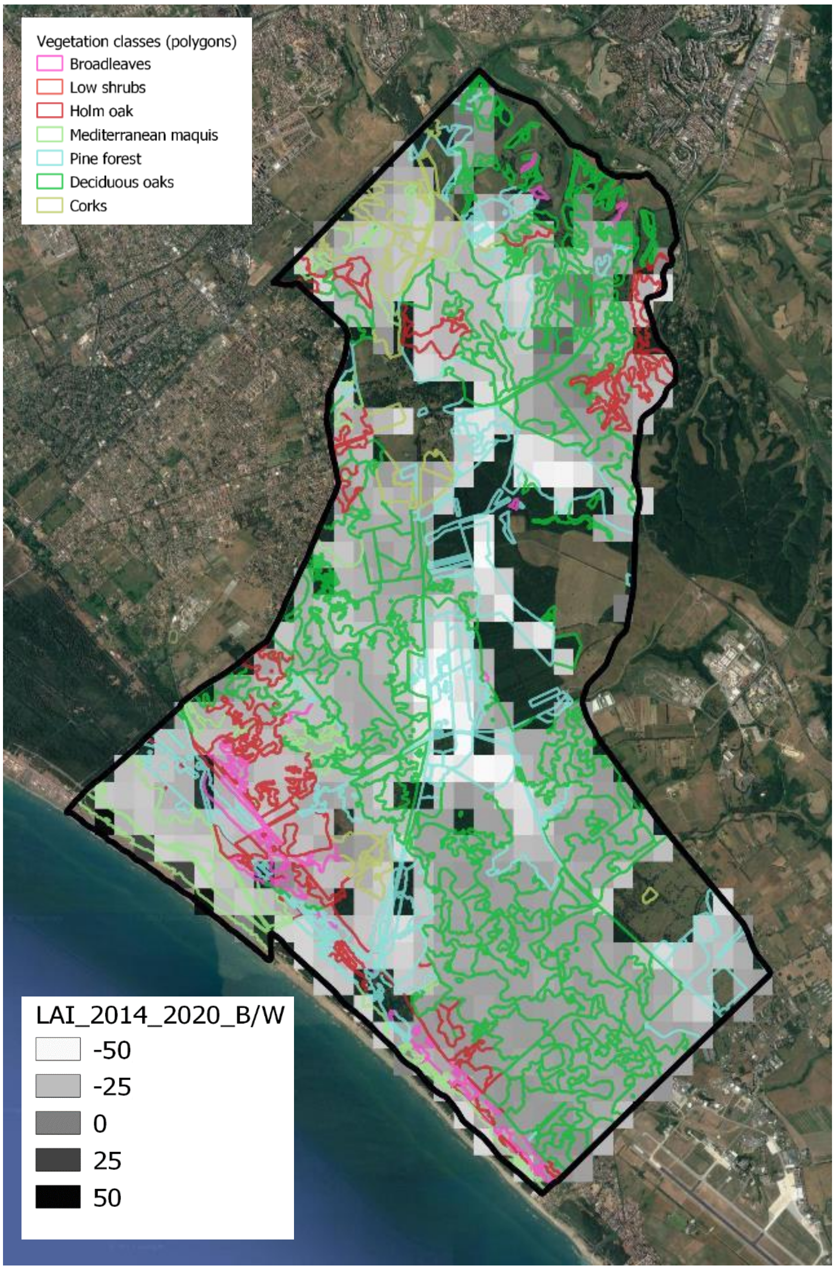


Figure 10: Superimposed maps of vegetation (colored shapes) to LAI (black and white pixels). Colored shapes represent homogeneous groups of vegetation (LCTs). Pixels represents percent change of satellite values of LAI between 2014 and 2020. The vividness of color represent loss or increase of LAI. *(From Conte et al., 2022).*

Concerning LAI and canopy cover, we used 300 m resolution satellite images from Copernicus Global Land Service, the Earth Observation programme of the European Commission (http://land.copernicus.vgt.vito.be). For each year of study (i.e., 2014, 2019, 2020), the maximum values observed during the growing season were used. AIRTREE simulations were conducted for each pixel of the study area, and for each year of study, the LAI values were extracted filtering out pixels classified as buildings, urban, wetlands, meadows and artificial settlements. LAI is a critical value for the proper evaluation of ecosystem services via AIRTREE model (Fares et al., 2019). Therefore, the satellite data were compared with field data observations collected with multiple simultaneous measurements of transmittance using the LAI-2200C Plant Canopy Analyzer (Li-Cor, Lincoln, NE, USA) and with a method based on the use of hemispherical photos collected in the frame of ICOS activities (Gielen et al., 2017). The sampling points were selected through a stratified sampling procedure (Figure 11) over the main four LCTs: Mediterranean maquis, Deciduous broadleaves, Pine Forest and Holm oak forest.

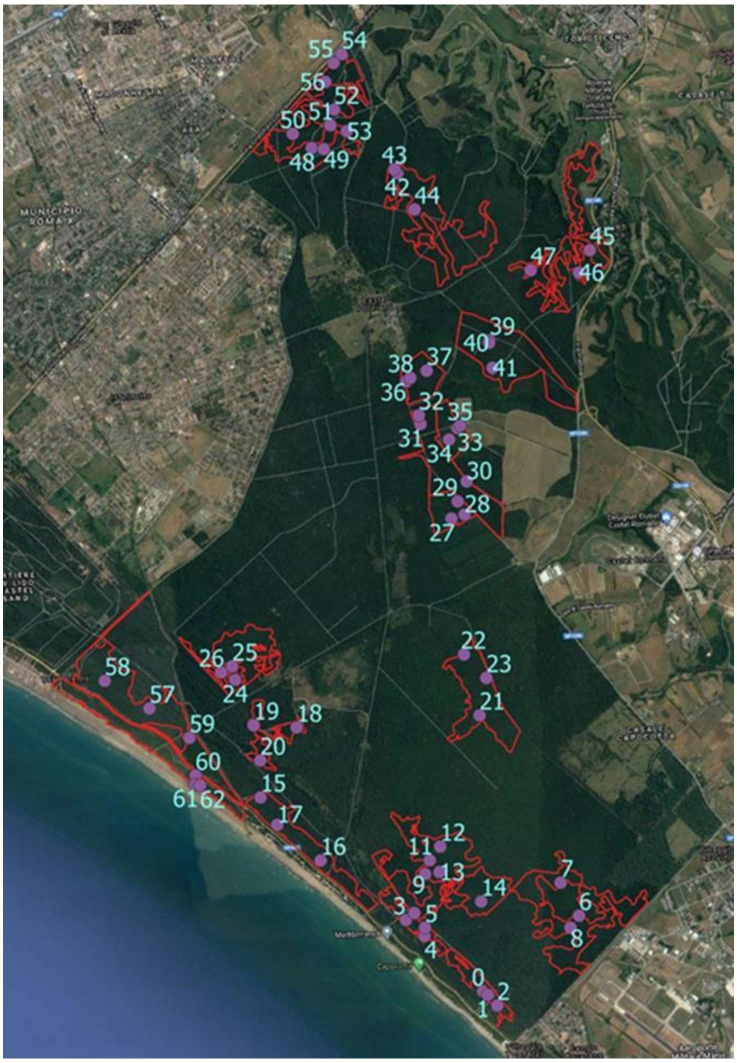


Figure 11: Castelporziano map, showing the sampling points chosen randomly using QGis. *(From Conte et al., 2022).*

To derive a function suitable to provide a realistic estimate of the annual dynamics of LAI for each LCT group, monthly values of LAI were fit to a set of parametric non-linear models (Gaussian, Exponential, Rational, Power, Sin, Weibull and Fourier). The function that best fitted data (lowest RMSE) was implemented in the model. The Fourier model was identified as the most appropriate model to reproduce LAI dynamics through the three years of study for the LCTs functional groups:

$$\text{LAInorm} = a + b \cdot \cos(x \cdot d) + c \cdot \sin(x \cdot w) \qquad \text{(eq. 1)}$$

where a, b, c and d were LCT-specific fixed values (i.e., the coefficients of the fitted model), and x is the day of the year (Figure 12).

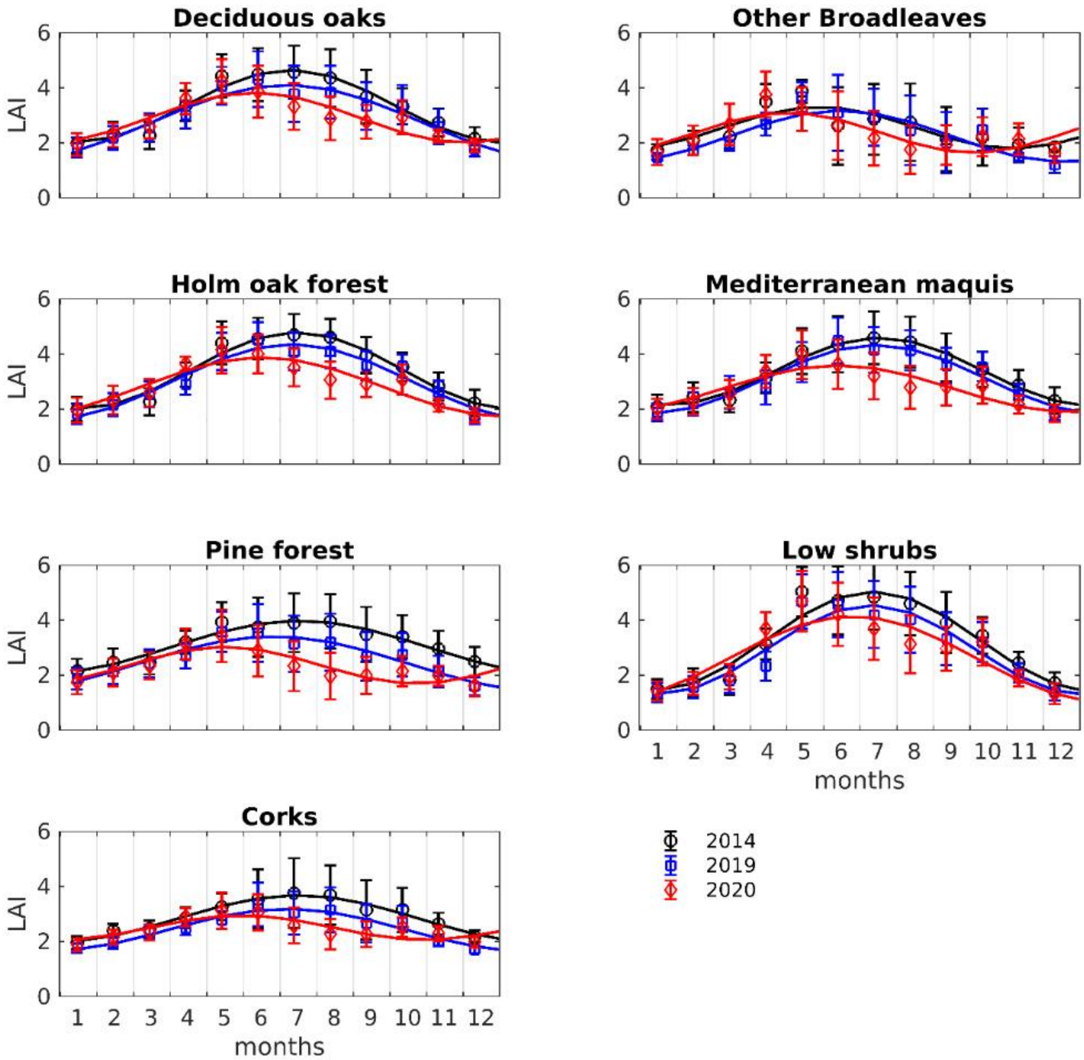


Figure 12: Fourier's model (lines) fitted to monthly LAI ($m^2\ m^{-2}$) satellite data (circles) for each LCT, for the three years of study. Each function was implemented into the AIRTREE model and used to estimate LAI dynamics during the year. *(From Conte et al., 2022).*

Such a function was finally implemented into the AIRTREE model for each reference year in order to derive the daily LAI at each model time step. Canopy heights were only available for the year 2019 at a spatial resolution of 25 m (Potapov et al., 2020). These were resampled to 300 m in order to align the maps with those of LAI and canopy cover. All maps out on the boundaries of the Estate were cut, and pixels falling into the non-forested LCT were removed, so as to work only on the actual forested areas.

3.1.3 Case Study 3: Urban Street Trees

3.1.3.1 Study area: Municipality of Milan and Municipality of Bologna

As for the cities chosen for this study (Milan and Bologna), these are two Municipalities located in the Po River basin, one of the areas exposed to the highest air pollution levels in Europe, and characterized by different size, population density and geographic location. These cities had also been selected by the above-mentioned LIFE VEG-GAP Project and for this official census of the trees present were available.

In Italy, the average temperature value in 2015 (the reference year for this part of the study) was the highest of the entire series since 1961, just above that of 2014 (Desiato et al., 2005). The average annual anomaly was +1.58 °C and was attributable to all four seasons, with the most marked anomaly in summer (+2.53 °C). In particular, the average annual temperature anomaly was +2.07 °C in the North of Italy with the warmest month compared to the mean on July (+4.31 °C). (https://www.isprambiente.gov.it/it).

The Municipality of Milan (45°27′51″ N, 9°11′22″ E) occupies an area of 367.67 km^2 in the west of Lombardy, located 25 km east of the Ticino River, 25 km west of the Adda river, 35 km north of the Po River and 50 km south of Lake Como. It is therefore located to the west of the Po Valley basin and is characterized by a Cfa type climate (humid subtropical) (Koppen, 1930; Kottek; 2006), that is the typically temperate climate of the middle latitudes, with a significant annual temperature range (hot summer and cold winter).

Like all big cities, temperatures are higher than in the surrounding countryside. In the center of Milan, annual average temperatures is around 3.5 °C in winter, 14 °C in spring, and 25 °C in summer. The average temperature range on a sunny day is about 10 to 13 degrees. Winters are colder than in the coastal cities, although the more southerly latitude and the protection of the Alpine chain prevent the extremes of central Europe. Summers are hot and muggy, with very frequent thunderstorms mitigating the heat. (Reference period 1981–2010).

In January 2015, average temperatures and solar radiation were just above the norm for the period. The same was true for February and April. In contrast, the summer (June, July, and August in particular) was characterized by much higher-than-normal temperatures and irradiation, up to 34 °C. Rainfall was only slightly lower than the normal average of 40 mm (annual average value). Towards September, the temperature returned towards the normal average of 25 °C and rainfall increased more than normal of this month. Between October and December, the temperature fell back to the average and went from 17 °C to 9 °C (monthly averages). Precipitation increased in October but was not manifested between November and December (http://www.arpalombardia.it).

The Municipality of Bologna (44°29′38.00″N, 11°20′34.00″ E) covers an area of 277.22 km^2, and is in the southern Po Valley, close to the foothills of the Tuscan-Emilian Apennines, between the mouths of the valleys of the Reno River and the Savena stream, which longitudinally bathe it to the west and east respectively. The altimetry of the municipal territory ranges from 29 to about 390 m above sea level.

Bologna has a Cfa type climate (same classification as Milan) (Koppen, 1930; Kottek; 2006), with humid temperate climate with very hot and humid summers and rather cold and wet winters.

Generally, the average temperature in winter is around 4 °C, in spring around 13 °C, and in summer around 26 °C. Rainfall is moderate, amounting to 670 mm per year. Rainfall is well distributed throughout the year, although there are two highs in spring and autumn, and two relative lows in winter and summer. (Reference period 1981–2010).

During winter 2015, temperatures far above the general climate (+3.2 degrees) were recorded (this is one of the highest anomalies recorded). In particular, the winter was particularly mild in December (+3.1 degrees) and especially in January (a good +5.7 degrees above the climatic average); February was more average, with maximum temperatures of +0.9 °C above the climate. Average minimum temperatures were also higher than the long-term average: the seasonal minimum temperatures were 2 °C degrees above the climatic average.

The abundance of precipitation confirmed the anomalous nature of the winter of 2015 (with 197 mm reached in the month of February alone).

3.1.3.2 Statistical analysis

An analysis was also made based on the list of trees and shrubs present in the urban cities of Milan and Bologna, provided by the Municipalities themselves. These lists also include the heights of each plant and the class to which they belong in relation to their height. The domain of the dataset is relative to the limits of the above-mentioned Municipalities.
A list was extracted from the tree survey databases, with the aim of grouping together species with the same characteristics as much as possible; plant varieties were merged to make the data more homogeneous, without losing information. In the first part, the most relevant species have been grouped for each city. The second section goes into the details of the species with a botanical analysis.

3.1.3.3 Meteorological and air pollutant concentration modelling

The air quality modelling system used for simulations in Milano and Bologna is AMS-MINNI (Mircea et al., 2014; Mircea et al., 2016), which includes a meteorological prognostic model (WRF v3.9.1.1- Weather Research and Forecasting) (Skamarock and Klemp, 2008) and a chemical transport model (FARM—Flexible Air Quality Regional Model) (Silibello et al., 2008). Three nested domains with (approximately) 10 km, 4 km, and 1 km spatial resolution, respectively, were used to perform the simulations needed in this project, from European to city scale, to account not only the pollutant emissions located within the city but also for the effect of advection from outside the target domains. Meteorology and air quality has been reconstructed for Bologna and Milan areas with a 1 km spatial resolution and hourly temporal resolution.
Details on model configurations, urban vegetation treatment, and scenarios simulated are described in the VEG-GAP project document available from the project web site (https://www.lifeveggap.eu/filesharer/documents/VEG-GAP%20Guides) and in de la Paz et al. (2022). Meteorological and air quality model simulation results are available for browsing and download from the VEG-GAP Information Platform (https://veggap.adamplatform.eu/) supported by the ENEA CRESCO computational facility (http://www.cresco.enea.it/). This meant that all vegetation maps were resampled to 1 km resolution in order to work consistently.
To obtain a primary understanding of the effect of vegetation on urban heat and air quality patterns, a simulation with no urban vegetation has been carried out as part of the activities foreseen in the VEG-GAP project. This scenario has been produced assuming bare soil conditions in all the presently vegetated areas located inside the administrative boundaries of the city. No change has been imposed over agricultural areas located inside the municipalities. Model simulations covered the whole year

2015 coherently with anthropogenic emission inventories and urban vegetation inventories that all refer to that particular year.

3.1.3.4 Tree cover maps

A land cover classification was carried out to identify the tree cover, using the cloud computing platform of Google Earth Engine. The land cover classification was required because the public tree inventory (provided by the Municipalities for the construction of these maps) does not account for private gardens, thus underestimating the overall urban green.

Municipal administrations of many European cities developed detailed digital vegetation inventories to support urban green maintenance. They contain geographic location, specie and the main features of each single tree located along the roads and within public gardens and parks. This information is available through institutional web geoportal as, e.g., for Milan Municipality (https://geoportale.comune.milano.it/sit/patrimonio-del-verde/). Inventories have been provided as vector type data by Bologna and Milan municipalities to support the VEG-GAP project activity. Since tree canopy diameter was not available for Bologna, it has been estimated following Hemery et al. (2005), providing "crown diameter–stem diameter ratios" estimates for different species of broadleaved trees. Since the list is referred to the year 2015, for the sake of completeness of the data, this year was chosen as reference year to carry out this study.

The classification legend adopted (Table 7) is based on the Modis Land Cover Type 2 scheme (Sulla-Menashe et al., 2012); the classes that were not present in the study areas were filtered out from the scheme.

Digital Number	Name
1	Barren or sparsely vegetated
2	Croplands
3	Deciduous Broadleaf treed areas
4	Evergreen Broadleaf treed areas
5	Evergreen Needleleaf treed areas
6	Grasslands
7	Shrubs
8	Urban and built_up
9	Water

Table 7: Land use map legend based on the Modis Land Cover Type 2. *(From Zappitelli et al., 2023).*

Support vector machine was applied for land-cover characterization to a Sentinel 2 time-series (McHugh et al., 2012). A total of 485 ground truth areas were delimited by on-screen digitalization based on Google and Bing imagery, and ancillary data, i.e., Corine Landcover, the city tree database and Copernicus Street tree layer. The ground truths were randomly divided as follows: 30% to train the algorithm and 70% to validate the classification, for each legend class.
Sentinel-2 imagery was filtered to remove the clouds, using the Sentinel-2 quality layer. The imagery was selected in the leaf-off period of winter 2018/2019 and in the leaf-on period of 2019. Since previous satellite imagery was not available, we assumed no relevant changes between 2015 and 2018/2019 in the spatial pattern of vegetation within each Municipality. For each period, the multi-temporal dataset was reduced to a single image using a median filter, thus minimizing the noise from remaining reflectance artifacts, e.g., shadows. Each image was composed by 8 bands shown in Table 8.

Band	Resolution	Central Wavelength	Description
B2	10 m	490 nm	Blue
B3	10 m	560 nm	Green
B4	10 m	665 nm	Red
B5	20 m	705 nm	Visible and Near Infrared (VNIR)
B6	20 m	740 nm	Visible and Near Infrared (VNIR)
B7	20 m	783 nm	Visible and Near Infrared (VNIR)
B8	10 m	842 nm	Visible and Near Infrared (VNIR)
B8a	20 m	865 nm	Visible and Near Infrared (VNIR)

Table 8: Sentinel 2 bands used in classification. *(From Zappitelli et al., 2023).*

For classification purposes, the infrared bands were upscaled to the resolution of 10 m using the nearest neighbor method. This resolution was required to catch as much as possible the isolated tree crowns in the urban context. The classification was finally carried out merging the bands of the two

images, one for leaf-on and one for the leaf-off period, to better gauge the difference between evergreen and deciduous tree.
The classification accuracy was tested using Cohen's Kappa score (Petropoulos et al., 2012) because it is more representative of the classification accuracy for the study areas that presented an unbalanced data distribution among the classes. The classification validation resulted in a Kappa of 0.77 and 0.71 for Bologna and Milano, respectively.
Tree cover masks were generated separately for deciduous broadleaf, evergreen broadleaf and needleleaf treed areas. A visual assessment of these masks showed that the tree cover for small trees was not identified, especially when isolated trees were not centered on a pixel. To minimize this issue, these masks were merged with others, obtained rasterizing the public tree geodatabase of the city at same resolution (10 m) for the three classes. The resulting masks had value 1 when either or both the classification and the tree database masks had value 1.
The computation of the tree cover ratio for the three classes (dataset A) was carried out using a mobile square window with a side of 500 m over the mask. For each iteration the ratio between the number of pixels of each land cover class and the total number of pixels in the mobile window was computed. The results were assigned to a pixel having the same location and dimension of the mobile window.
To assess the ecosystem services through the AIRTREE model, it was further required to have information of the tree cover for each tree species. It was assumed that within a squared area having a side 500 m, the tree species percent composition was the same for private and for public properties. Using the public tree geodatabase, it was first rasterized the tree cover mask for each species at the resolution 10 m. As a second step it was rasterized the tree cover masks for the three land cover classes at the resolution of 10 m, and downscaled the masks obtained at a resolution of 500 m. Such method allowed to compute the species percent composition (dataset B) with a moving window using the methodology explained above.
Finally, the tree cover for each tree species was computed multiplying the dataset A by dataset B. The resolution of 500 m was later downscaled to 1 km to match the resolution of climate and pollutant data.

3.2 The AIRTREE Model

The AIRTREE (Aggregated Interpretation of the Energy Balance and Water Dynamics for Ecosystem Services Assessment) model (Fares et al., 2019) is a one-dimensional multilayer model that couples soil, plants, and atmospheric processes to predict exchanges of carbon dioxide (CO_2), water vapor

(H_2O), tropospheric ozone (O_3), particulate matter (PM_{10} and $PM_{2.5}$), and nitrogen dioxide (NO_2) between leaves and the atmosphere and integrates them across five layers to obtain fluxes at the canopy level.

The model requires a limited number of input variables (PPFD + Near Infrared Radiation; Air Temperature; Relative humidity; Wind speed; CO_2 concentration; Ozone concentration; Air pressure; PM concentration; Precipitations; Friction velocity) to estimate individual leaf gas exchanges and integrate fluxes through these five layers to obtain fluxes (e.g., GPP, λE, PM, O_3) at the canopy level. The model incorporates a canopy environmental module to determine the leaf temperature, stomatal conductance, and radiative transfers at different levels from the top to the bottom of the canopy.

The model allows the estimation of stomatal conductance (gs) and photosynthesis (An) by coupling the Farquhar-Von Caemmerer-Berry photosynthesis model (FvCB model) and the Ball, Woodrow, and Berry stomatal conductance (BWB) (Ball et al, 1987) model at different levels from the canopy top-down model (Roy et al., 2012). Oxidative limitations to An and gs were accounted accordingly to Keenan (2010) and Lombardozzi et al. (2012, 2013, 2015) for drought and O_3 stress, respectively.

The AIRTREE model is constituted by four different modules:

1) The radiative transfer model, which calculates leaf temperature and the radiation through canopy.
2) The photosynthesis module, that calculates An and gs.
3) The deposition module, which calculates resistances to gas diffusion and deposition of gases and pollutants.
4) The hydrological model, which predicts the ecosystem hydrological status.

A detailed description of the formulation above mentioned was extensively described in Fares et al. (2019) and in Conte et al. (2021).

Gross primary productivity (GPP) was calculated by integrating the photosynthetic flux of each layer. Finally, Net Primary Productivity (NPP) was calculated as the difference between GPP and the autotrophic respiration (see Fares et al., 2019 for details). Particulate Matter and NO_2 fluxes calculation were extensively described in Conte et al. (2022) and in Fares et al. (2019, 2020).

The AIRTREE model was developed to be a site-specific model calibrated on site observations, to provide the most accurate simulations of ecosystem gas exchanges, so specific considerations must be made for each site considered in these case studies.

As regard the Park of Valentino and the Park of Castel di Guido, canopy photosynthesis was calculated according to the multilayer scheme described in detail by Fares et al. (2019). 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). 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 published article by Fares et al. (2019), 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 to the formalism described by Zhang et al. (2002), while the PM fluxes were calculated according to the sum of deposition processes associated with interception, impaction, and gravitational settling as reported by Han et al. (2020) and explained in detail in the Supporting Information. It was emphasized 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 (Pace and Grote, 2020). In this study, the formalism that was 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), it was 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) and Stull (1997).

In order to facilitate discussion of results and comparison between species in both parks, it was 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.

In the study for Castelporziano Estate, NO_2, SO_2 and CO fluxes were calculated as:

$$F = Vd \cdot C \cdot 1800 \qquad \text{(eq. 2)}$$

where F is the pollutant flux (µmol m^2 s^{-1}), Vd is the deposition velocity (m s^{-1}) and air pollutant concentration (µmol m^{-3}). Vd was estimated through a resistance scheme (Baldocchi et al., 1987; Zhang et al., 2002):

$$V_d = \frac{1}{R_a + R_b + R_c} \quad \text{(eq. 3)}$$

where Ra is the atmospheric resistance (see Fares et al., 2019 for details), Rb is the boundary layer resistances (s m^{-1}) calculated according to Pederson et al. (1995) and Rc is the canopy resistance calculated according to Bidwell & Fraser (2011) as:

$$R_b = 2(Sc)^{\frac{2}{3}} \cdot (Pr)^{-\frac{2}{3}} \cdot (ku)^{-1} \quad \text{(eq. 4)}$$

$$\frac{1}{R_c} = \frac{1}{r_s + r_m} + \frac{1}{r_{soil}} + \frac{1}{r_t} \quad \text{(eq. 5)}$$

where Sc is the Schmidt number, Pr is the Prandtl number (0.72), k is the von Karman constant (0.41), u is wind speed (m s^{-1}), rs (s m^{-1}) is the inverse of gs, the formulation of which is extensively described in Fares et al. (2019). Soil resistance $rsoil$ (s m^{-1}) was set equal to 2941 and 2000 (m s^{-1}) during vegetative and non-vegetative periods, respectively. Mesophyll resistance rm (s m^{-1}) was set to zero for SO_2 (Wesely, 2007) and 600 for NO_2 (Lovett, 1994). Cuticular resistance rt (s m^{-1}) was set to 8 × 103 and 2 × 104 for SO_2 and NO_2, respectively (Lovett, 1994). Rc for CO was set equal to 5 × 104 and 1 × 106 during vegetative and non-vegetative periods, respectively (Bidwell and Fraser, 2011).
Concerning PM deposition, the model incorporates traditional atmospheric models to predict particle deposition extensively described in Fares et al. (2019; 2020).

As regarding the urban street trees in Milano e Bologna, the AIRTREE model was applied on the vegetation maps of those two cities. A model simulation was performed for each species in a pixel that was identified by the supervised classification. Biometric parameters, such as tree height (m), diameter at breast height (dbh), and canopy diameter (m) from the tree inventories, were used to define the tree structures while data regarding ecophysiological parameters, such as leaf area index (LAI) and carboxylation velocity (Vcmax), were retrieved from the literature and available from past field studies (Fares et al, 2019; Conte et al, 2022). Phenology was calculated using the Phenological Maps extracted from NASA Visible Infrared Imaging Radiometer Suite (VIIRS) Global Land Surface Phenology (GLSP), with a daily temporal resolution, which was used to identify the beginning and end of the vegetation period. The meteorological and air quality data for each pixel were extracted from the modelled data (1 km spatial resolution maps in netCDF as stored on CRESCO and made available through VEG-GAP Information Platform and integrated in hourly datasets (1,004,679 and 21,390 pixels for Milan and Bologna, respectively).

The Radar Plots (Figure 38) were obtained from the AIRTREE outputs (expressed in g m^2 y^{-1}). In this case, the values obtained for the sequestration of each pollutant were cumulated according to Macrotype. Subsequently, the data were normalized for the maximum value.

The percent presence of each species in the pixel data was also cumulated to obtain occupancy maps (Figure 39). To get the total sequestration of CO_2, O_3, PM, and NO_2 of each single pixel, results of each species were multiplied for their relative occupancy (squared kilometers in each pixel, Figure 40) and cumulated. Results were finally used to develop maps showing the spatial sequestration of CO_2 and the deposition of PM_{10} pollutants.

4. Results and Discussion

4.1 Case Study 1: Urban Parks

4.1.1 Pollutant concentration at the urban parks

Concentration of pollutants varied during the year 2018 in both parks. Ozone was higher in Spring, Fall, and Winter at Castel di Guido compared to Valentino with daily average values often exceeding 50 µg m^{-3} also during Winter. In particular, ozone followed a typical hourly trend with peak values in the central hours of the day as previously measured in the vicinity of Castel di Guido site (Fares et al., 2014). In Summer, higher ozone concentrations at Valentino reflected the abundance of ozone precursors and the high temperatures recorded at the site. The peri-urban park of Castel di Guido displayed a lower concentration of PM relative to Valentino (PM_{10} rarely exceeded daily values of 50 µg m^{-3}, which is a daily limit according to the national regulation), where Winter peaks exceeding 100 µg 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 13 and 14). Seasonal averages 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.

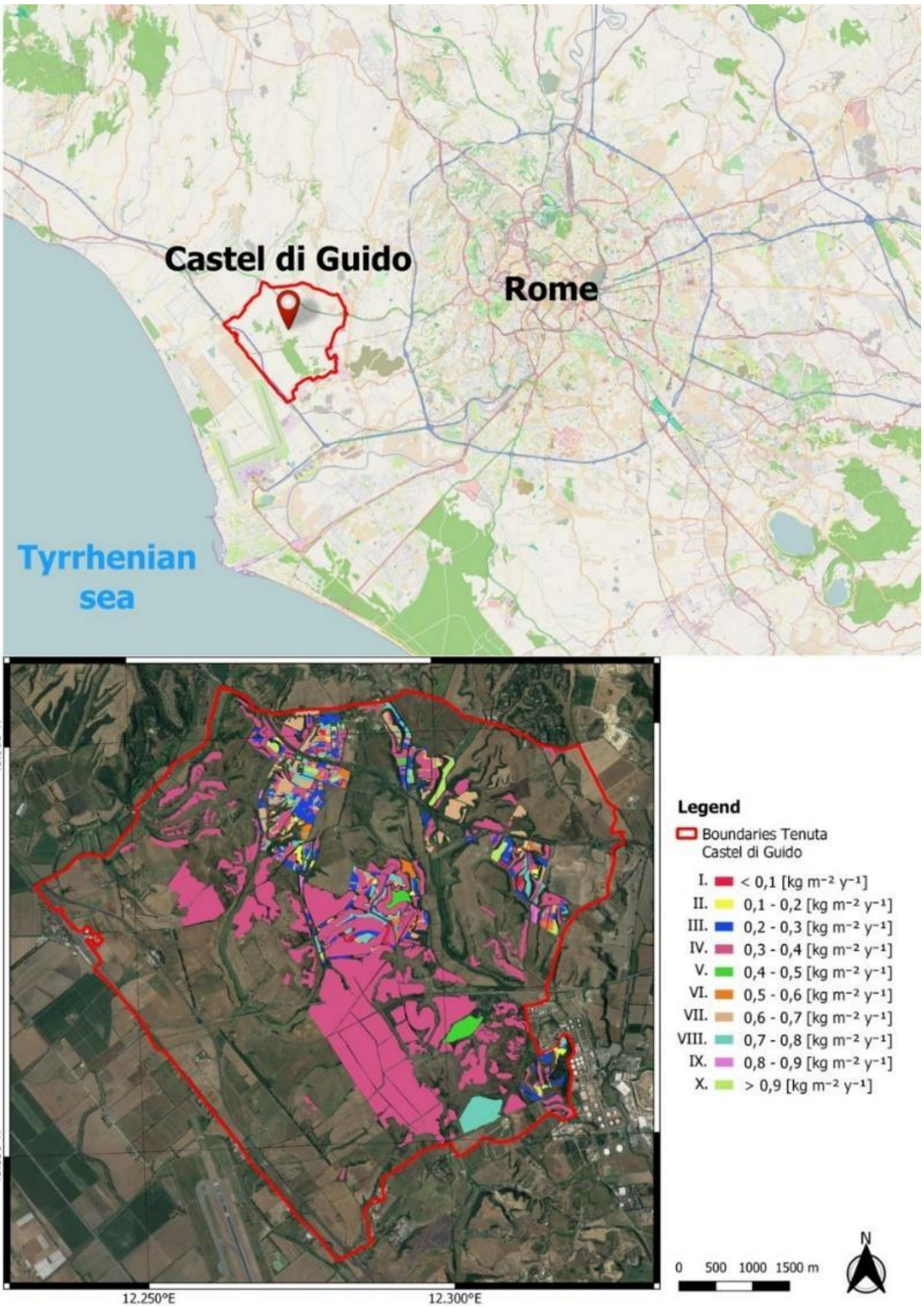


Figure 13: Location of Castel di Guido reserve, with a map showing NPP values in ten ranks. Map data © 2015 OpenStreetMap (top) and © 2020 Google (bottom). *(From Fares et al., 2020).*

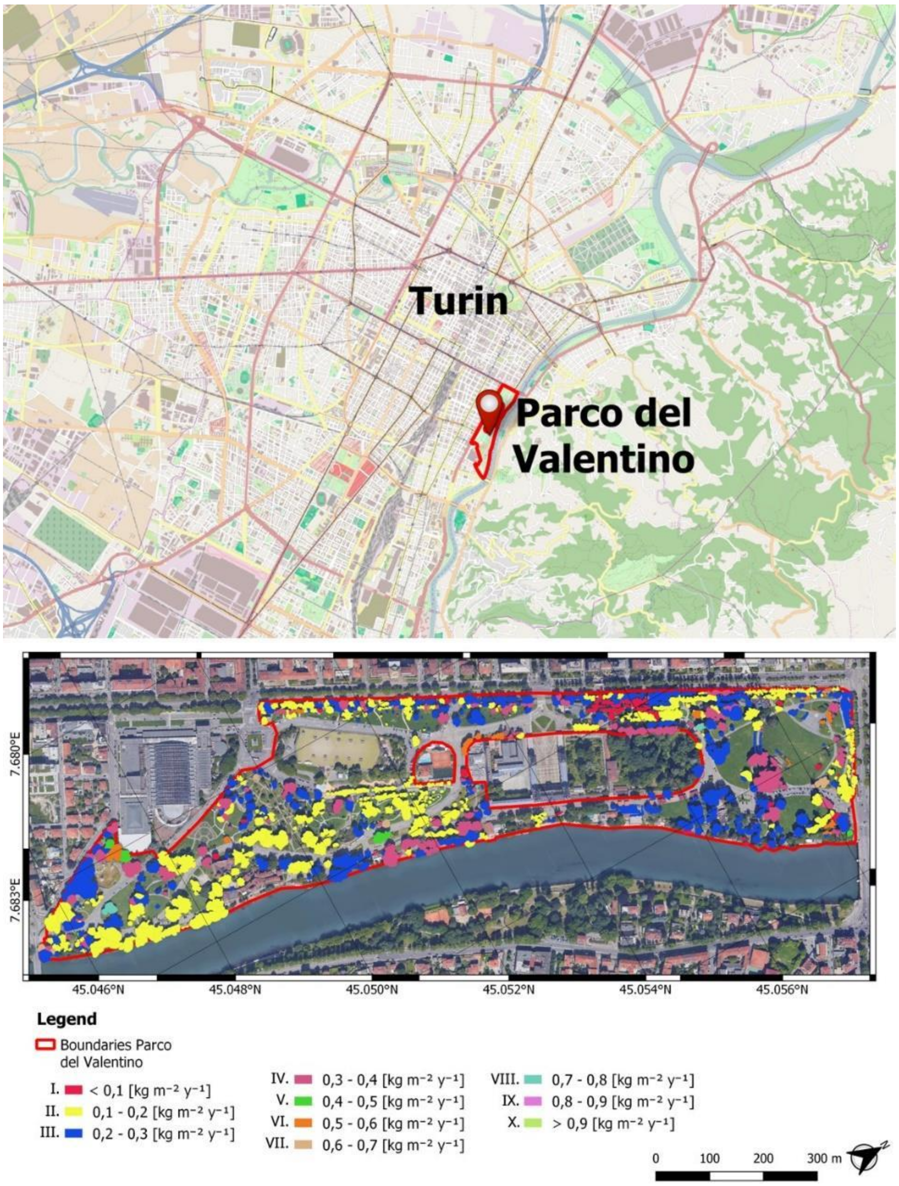


Figure 14: Location of Park of Valentino, with a map showing NPP values in ten ranks. Map data © 2015 OpenStreetMap (top) and © 2020 Google (bottom). *(From Fares et al., 2020).*

4.1.2 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 4 and Figure 13). Broadleaves at Castel di Guido generally displayed higher values of NPP compared with those at Valentino, as shown by comparison of *Quercus pubescens* and *Acer campestre*, two species with the same diameter class present in both parks. 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 the list of species (Table 5 and Figure 14). AIRTREE allowed to estimate potential GPP under the condition of a well-watered environment (i.e., no drought stress). It was 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 it was for *Quercus robur*. 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. In both parks, AIRTREE simulations produced higher deposition of ozone and PM for evergreen species compared with broadleaf species (Figure 15), 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.

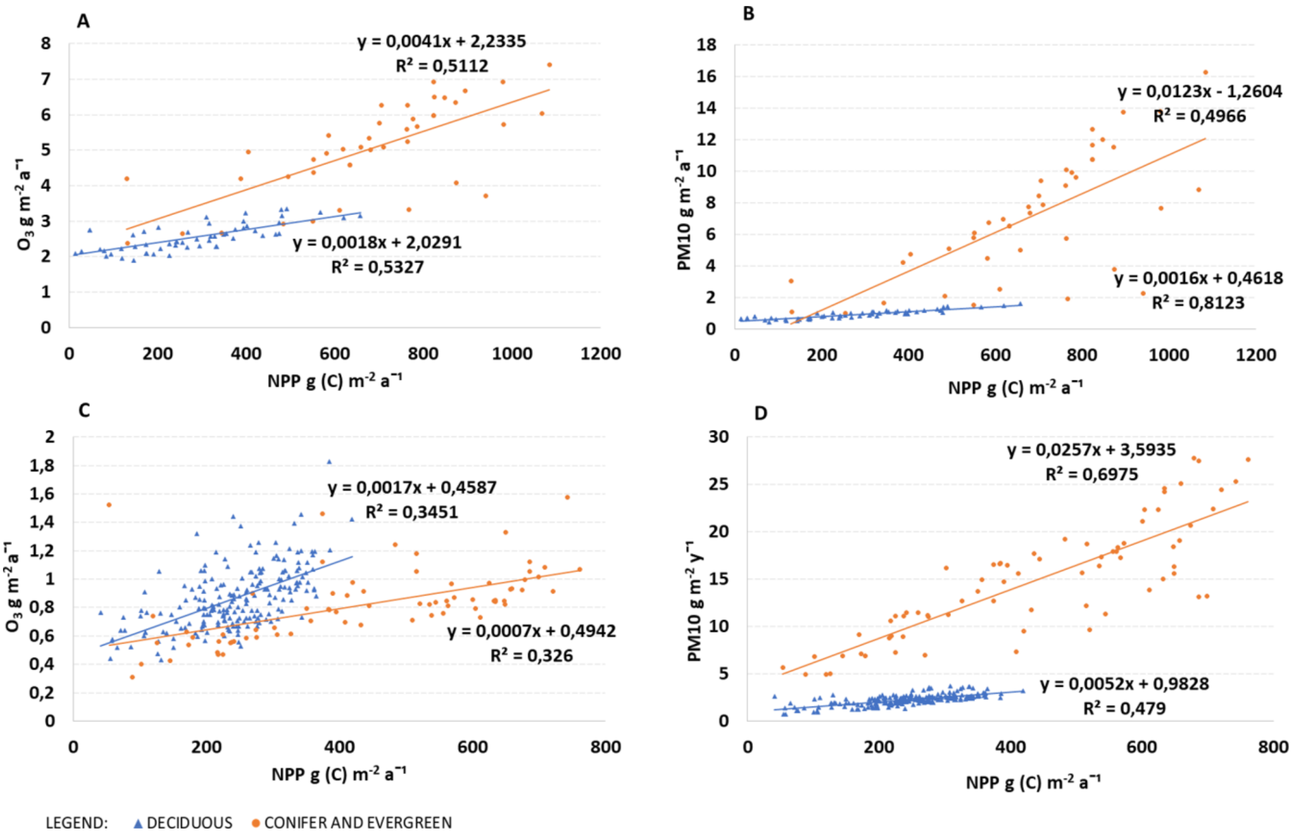
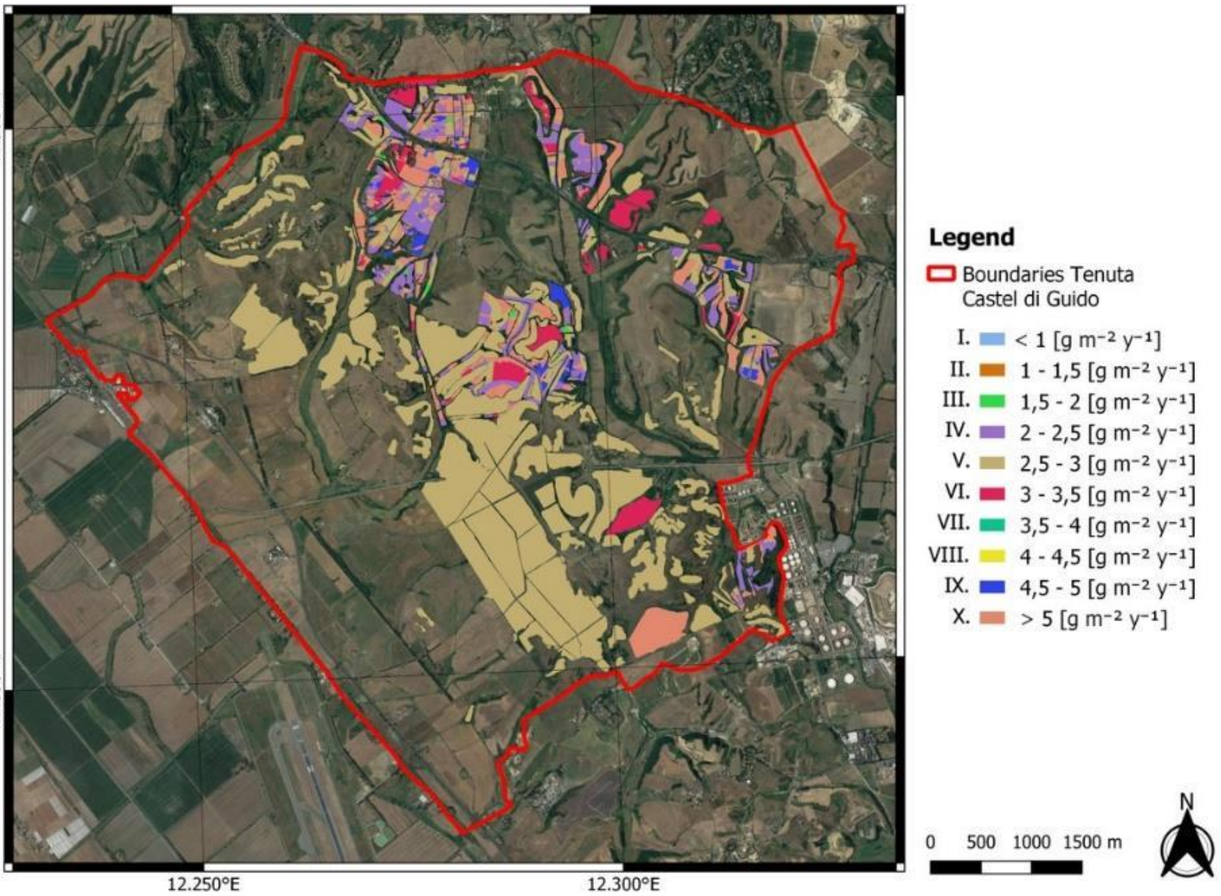


Figure 15: Correlation of NPP with ozone and PM for Castel di Guido (subplots A, B) and Valentino (subplot C, D) urban parks. *(From Fares et al., 2020).*

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 16), in agreement with ozone fluxes measured with the Eddy Covariance technique at a *Quercus ilex* forest nearby (Fares et al., 2014). However, pines reached class X (>6 g (O3) m^{-2} a^{-1}) most likely because of their high LAI, and longer vegetative period. At Valentino, values rarely exceeded class II (Figure 17) with broadleaf and fast-growing species such as *Tilia hybrida*, *Quercus rubra*, and *Platanus spp*. reaching the highest values above 1 g (O3) m^{-2} a^{-1}.



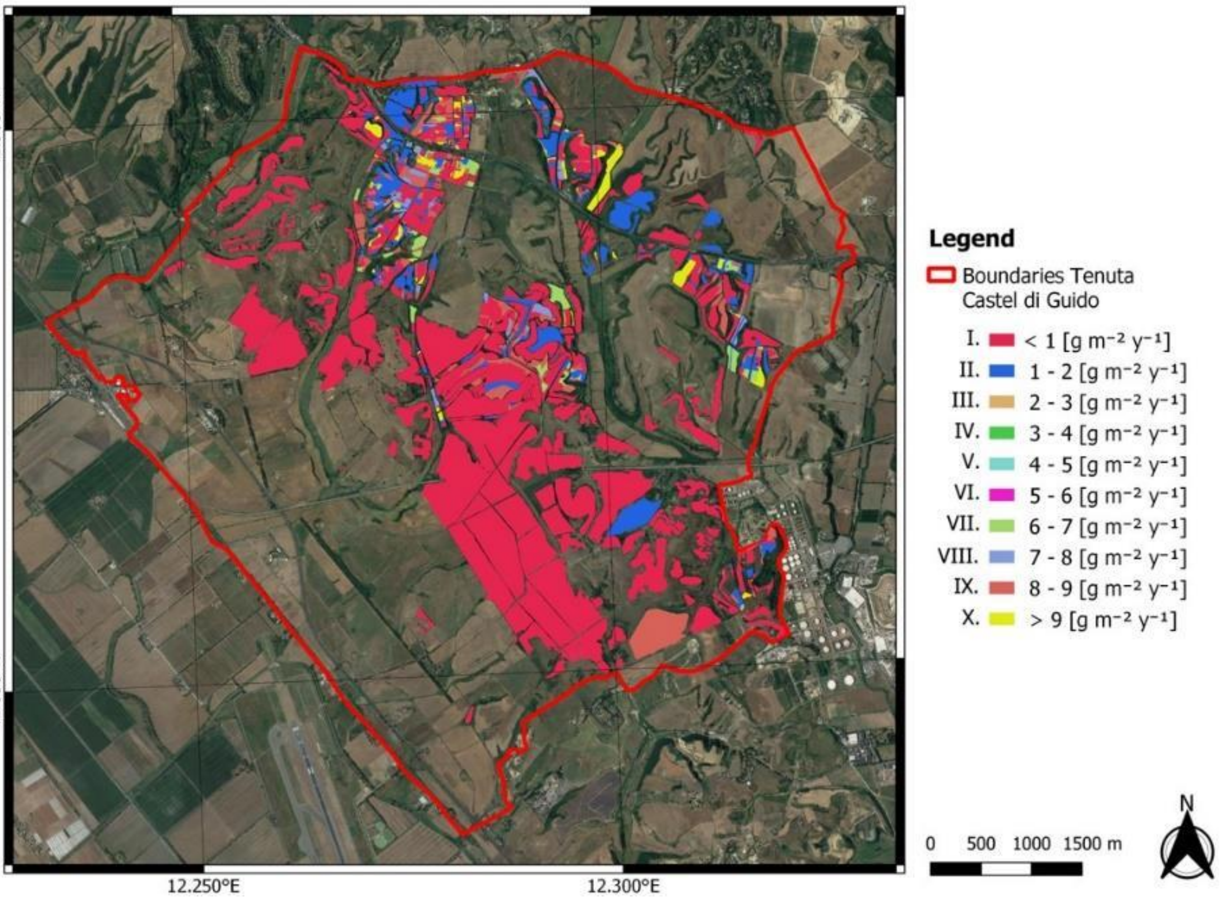


Figure 16: Maps showing ozone removal (top) and PM10 removal rates (bottom) for Castel di Guido Park. Map data ©2020 Google. *(From Fares et al., 2020).*

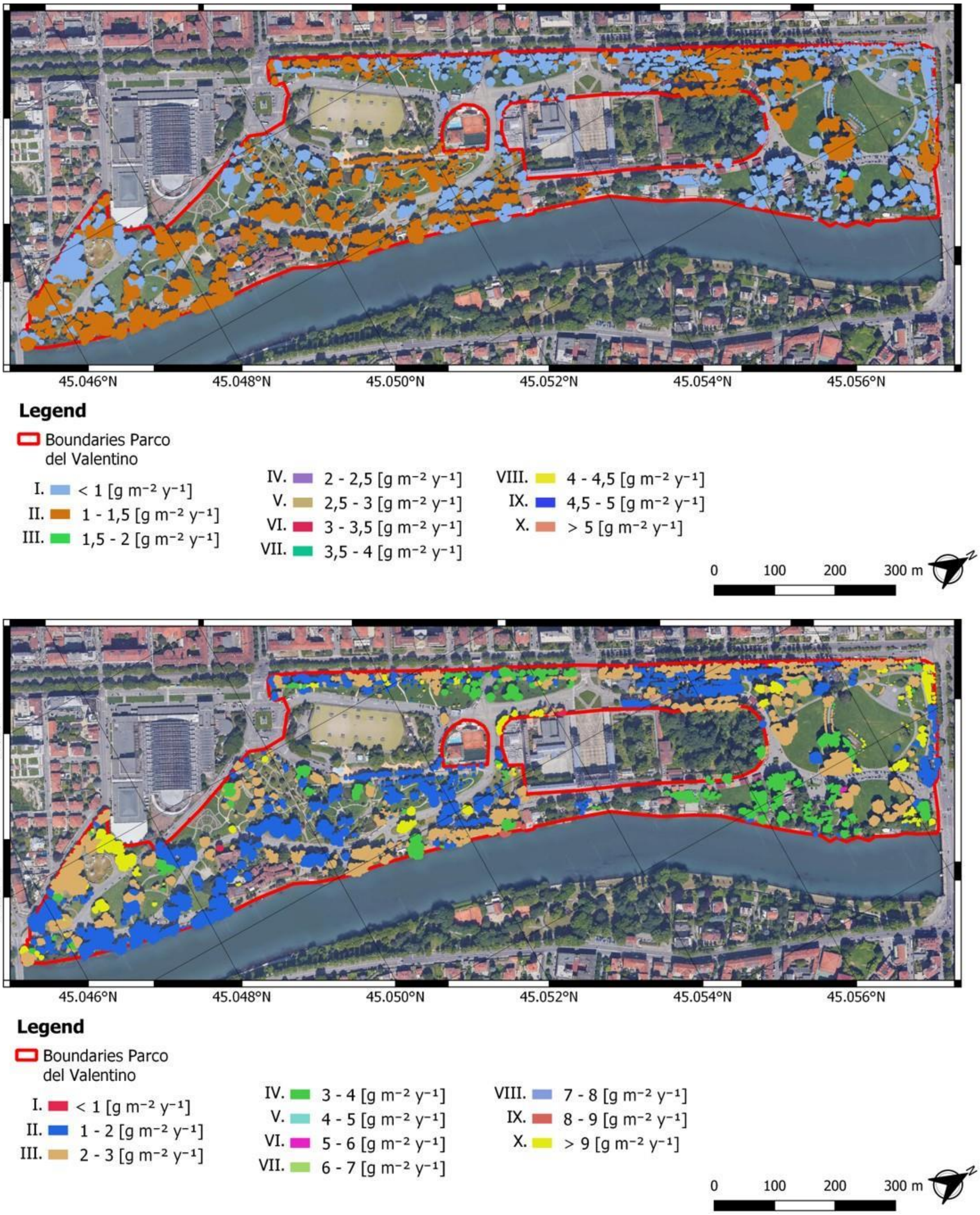


Figure 17: Maps showing ozone removal (top) and PM10 removal rates (bottom) for Valentino Park. Map data © 2020 Google. *(From Fares et al., 2020).*

Looking at two common species in both parks (*Quercus pubescens* and *Acer 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 *Acer campestre*, and VdO_3 = 0.019 cm s-1 during Spring daytime vs 0.008 cm s^{-1} for *Quercus 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, and both parks minimally contribute to ameliorate air quality with around 1% especially during the central hours of the day in Spring and Fall.

Simulations showed that evergreen species have the highest levels of PM sequestration (Tables 4 and 5). 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, 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). 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 *Acer campestre* and *Quercus 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). 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 $PM_{2.5}$. In total, the AIRTREE results showed that the parks are able to sequestrate PM_{10} with values of 31788 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. This simplified simulation suggested that both parks could contribute to ameliorate air quality 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.

4.1.3 Differences between the two parks

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 Vcmax, previously described as the main drivers of carbon assimilation through an AIRTREE sensitivity analysis performed in a *Quercus ilex* forest (Fares et al., 2019). In general, warmer temperatures and higher levels of insulation closer to photosynthetic optimum promoted higher growth rates at Castel di Guido, as shown for *Quercus pubescens* and *Acer campestre*, present in both parks (Figure 15). Higher ozone concentrations at Castel di Guido are explained by the different local climatic conditions (Figure 13): Rome has a milder temperature than Turin especially in cold seasons (Figure 14) 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. 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 (Fares et al., 2019). It was 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 (Clifton et al., 2020). 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 15, 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 (Han et al., 2020; Pace and Grote, 2020), 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) with damage to the photosynthetic apparatus. However, in this study, it is 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 peri-urban 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 (Cape, 2008).

4.1.4 Relevance of pollutant sequestration for air quality

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 (Liu et al., 2014; Kim and Coseo, 2018). The AIRTREE model adopted in these studies was specifically developed for predicting CO_2 and pollutant deposition in Mediterranean forests (Fares et al., 2019); therefore, this 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, is now 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.

In a study, Sicard et al. (2018) 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, this simulated percent improvement is in the same order of magnitude, with values close to 1% for ozone. In addition, these simulations did not consider ozone forming potential from BVOC emitted in this study, considering that *Quercus pubescens*, the most abundant tree species at Castel di Guido (Figure 5), has been described as a relevant isoprene emitter (isoprene being a high ozone-forming chemical species) (Genard-Zielinsk et al, 2015). 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% (Nowak et al, 2006; Baro`et al., 2014; Janhall et al., 2015; Jeanjean et al., 2016; Selmi et al., 2016). It was found, with this 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 the hypothesis of this study, such removal rates could compensate a small fraction of emissions arising from anthropogenic activities, in particular vehicular traffic. It was estimate that the parks of Castel di Guido and Valentino could collect the PM emission of around 15000 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 (43000 ha of green areas out of a total extension of the municipality of 129000 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 (Air Quality Expert Group, 2018), 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 that has been identified as high PM collectors are in the same list proposed by Yang et al. (2015) 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 in this study ranked as species with moderate capacity to sequestrate PM compared with conifers. This study is agreed 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 this study is also a recent work of Przybysz et al. (2019), who identified conifer trees for their higher capacity to sequestrate PM especially during the Winter season. More recently, Baraldi et al. (2019) identified the following species as particularly suitable for removing PM_{10} and O_3: *Liriodendron tulipifera, Celtis australis, Acer campestre,* and *Acer platanoides.* The authors also found species such as *Prunus cerasifera, Quercus cerris, Celtis australis, Acer campestre*, and *Acer 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 (Janhall et al., 2015); 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 (Klingberg et al., 2017).

This 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, it was 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), spatial scale is of high relevance for estimating pollutant deposition. Although it is strongly believed that the mesoscale approach adopted in this study is more appropriate to describe pollutant deposition than laboratory measurements on a single leaf upscaled to a forest (Santiago et al., 2017), 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 (Sheng et al., 2019). While this 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), 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) 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 this studies' 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.

4.2 Case Study 2: Peri-urban Park

4.2.1 LAI and Canopy Cover dynamics

Overall, average LAI for the entire Estate was 4.88, 4.31 and 3.90 in the 3 years of study, respectively (Figures 10), indicating a decrease of 11.83% and 20.05% in 2019 and 2020 compared to 2014, respectively. LAI showed a decreasing trend for all LCTs (Figure 17), in particular for Pine Forest (for which the representability of the total LAI changes from 14% in 2014 to 12% in 2020, Figure 11). In 2020, LAI increased for low shrubs by only 1% when compared to 2019. This is also confirmed by canopy cover data showing an average loss of 6.83% between 2014 and 2020. In particular, the Pine Forest showed a loss of cover of 13.33%, of which 9.83% occurred in a year, between 2019 and 2020. Indeed, a positive linear correlation between changes in LAI and changes in canopy cover was observed (Pearson's r = 0.63). Although an increase in forest cover of 2% has been observed between 2010 and 2015 in the Mediterranean basin (https://news.un.org/en/story/2018/11/1026761), the loss of Cover (7%) and LAI (up to 20%) experienced by the vegetation of the Estate between 2014 and 2020 confirms the increasing vulnerability of forests due to the climatic stressors and aridity the vegetation is exposed to (Penuelas et al., 2010). Indeed, from 2014 (previously described wet year in Conte et al., 2019), warmer and drier years succeeded. This may have caused stress and loss of productivity for Mediterranean forests, even if they are adapted to the typical dry and hot summers (Misson et al., 2011; Sardans and Peñuelas, 2013; Bongers et al, 2017; Castagneri et al., 2017). All ecosystems except shrubs showed a reduction in their LAI up to 14% (Figure 17).

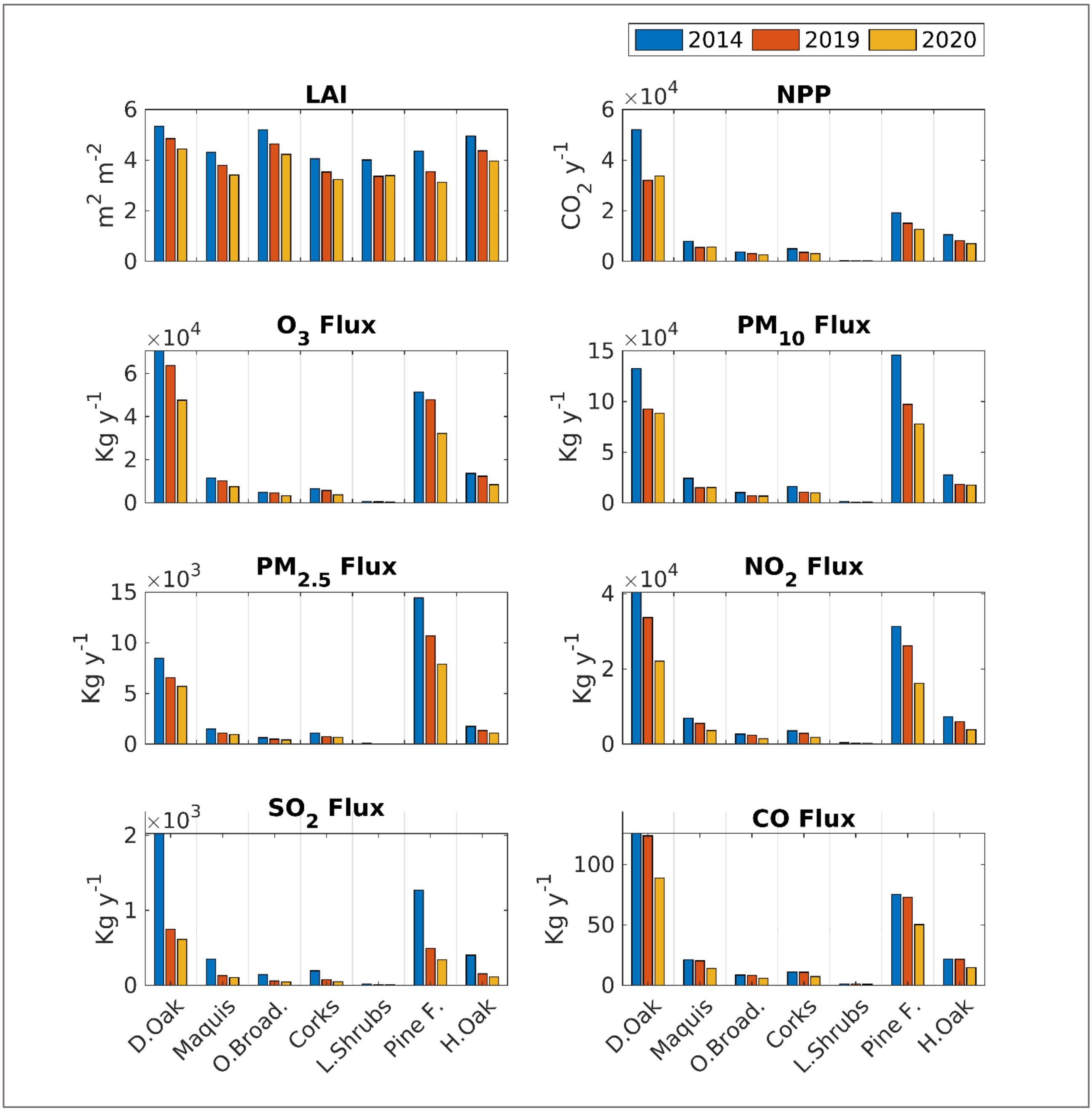


Figure 18: Intercomparison of the three years of study for each of the six Land Cover Types (Deciduous oaks, Mediterranean maquis, other broadleaves, Corks, Low shrubs, Pine forest, Holm oak forest) used in this study. LAI (leaf area/ground area, $m^2\ m^{-2}$), Net Primary Productivity (NPP, kg of $CO_2\ y^{-1}$), deposition of ozone (kg $O_3\ y^{-1}$), particulate matter (kg $PM_{2.5}\ y^{-1}$ and kg $PM_{10}\ y^{-1}$), Nitrogen dioxide (kg $NO_2\ y^{-1}$), Sulphur dioxide (kg $SO_2\ y^{-1}$) and Carbon monoxide (kg CO y^{-1}) estimated by the AIRTREE model are shown in the bar charts. Values represents cumulated values of uptake and deposition of pollutants for each LCT. *(From Conte et al., 2022).*

The environmental monitoring data concerning the health status of the Castelporziano forests, detected from 2015 to today through the diachronic interpretation of the NDVI values provided by Sentinel-2 data, showed widespread suffering in pine and deciduous oaks forests (Recanatesi et al., 2018). Regarding the pine forests, two insect infestations occurred in 2015 and 2017, causing a widespread decline in the photosynthetic capacity (-34% in the average NDVI values for the period

2015–2021) of the forest, which caused the death of plants over large area (Recanatesi et al., 2018). In Deciduous oak forests, the limiting factor seems to be strongly correlated with summer drought and springs characterized by low amount of rain, causing a mean decrease in NDVI values of -27% for the observation period (Recanatesi et al., 2018; Recanatesi et al., 2019).

Interestingly, there is high resilience from several species of Evergreen shrubs, capable of adapting to drought stress even more than *Quercus ilex* has been previously observed (Ogaya and Peñuelas, 2006; Barbeta and Peñuelas, 2016), and therefore, a transformation of forest structure into a shrubland under increasing heatwaves and drought is the most likely scenario (Sardans and Peñuelas, 2013; Barbeta et al., 2015). Indeed, Mediterranean plants have developed various morphological and physiological strategies to adapt to drought (Medrano et al., 2009), but the largest trees are frequently more sensitive and less resistant and resilient to increases in aridity (Medrano et al., 2009; Sperlich et al., 2015). In line with these observations, Lloret et al. (2004) showed that shorter trees are more resistant and resilient to increases in drought than taller trees. Tall shrubs such as *Phillyrea spp.*, which are abundant in the Estate of Castelporziano, have been shown to have a higher capacity than trees to adapt and resist intensive droughts (Rosas et al., 2013; Sarris and Koutsias, 2014) and have a higher capacity than sympatric trees to maintain their foliage and concentrations of non-structural carbohydrates after droughts (Sarris and Koutsias, 2014).

On the 21 fits performed (i.e., 3 years for 7 LCTs), the Fourier function (Figure 18) was found to be the best model (highest R^2_{adj} and lowest RMSE) 12 times, although it produced some minor biases (i.e., increase in LAI for “other broadleaves” and “Corks” at the end of the year).

A general change in phenology was observed, in 2020 compared to 2014, with an anticipated peak in LAI that shifted from June to July in 2014 to May to June 2020 (Figure 18). This is in line with previous findings showing that key Mediterranean species such as *Quercus ilex* display a longer period of growth by approximately 10 days by advancing the onset of spring by winter warming, with an early cessation of growth in spring and summer (Lempereur et al., 2017).

Satellite LAI values used in this study were compared with field measurements with an overestimation of 20% considering all LCTs (Figure 19). Deciduous broadleaves and Holm oak forests showed the highest LAI values (4.12 and 4.41, respectively), in accordance with what was previously found for Deciduous oaks stands (Cutini et al., 1998). It was measured lower LAI values for Pine forests and Mediterranean maquis (2.08 and 3.85, respectively), in line with previous findings (Gratani and Crescente, 2000; Manes et al., 2001).

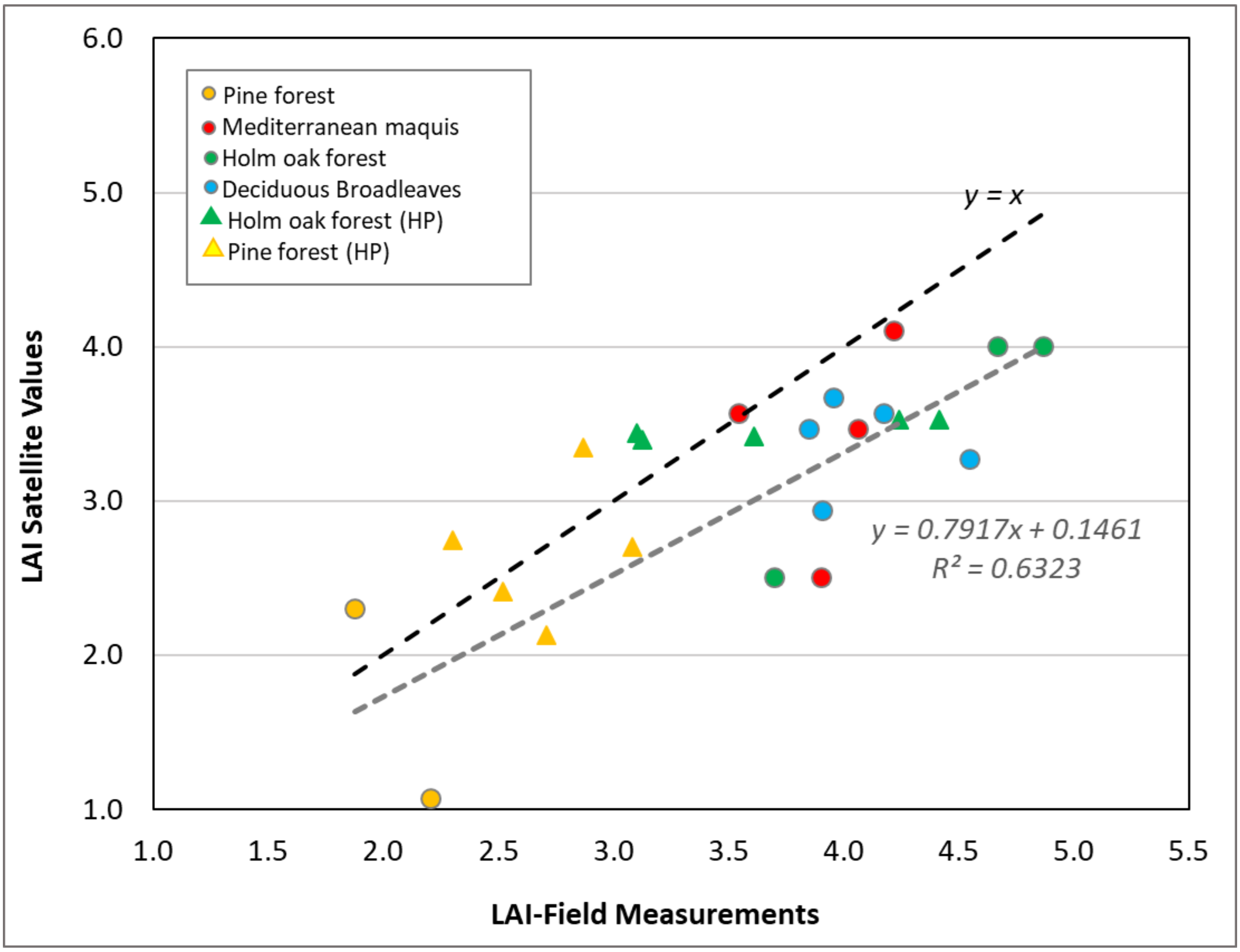


Figure 19: Correlation between ground-level measurement and satellite-level values of LAI (m^2 m^{-2}); different colors represent different LCT sampled inside the Estate. The circles indicate field measurements carried out with LAI-2200C Plant Canopy Analyzer; triangles indicate field measurements carried out with hemispherical photography (HP). *(From Conte et al., 2022).*

4.2.2 Sequestration of Carbon Dioxide

Total yearly estimates of Net Primary Productivity (NPP) indicate that the Estate sequestrated 0.0981, 0.0675 and 0.0648 mt CO_2 y^{-1} for the years 2014, 2019 and 2020, respectively (Figure 20).

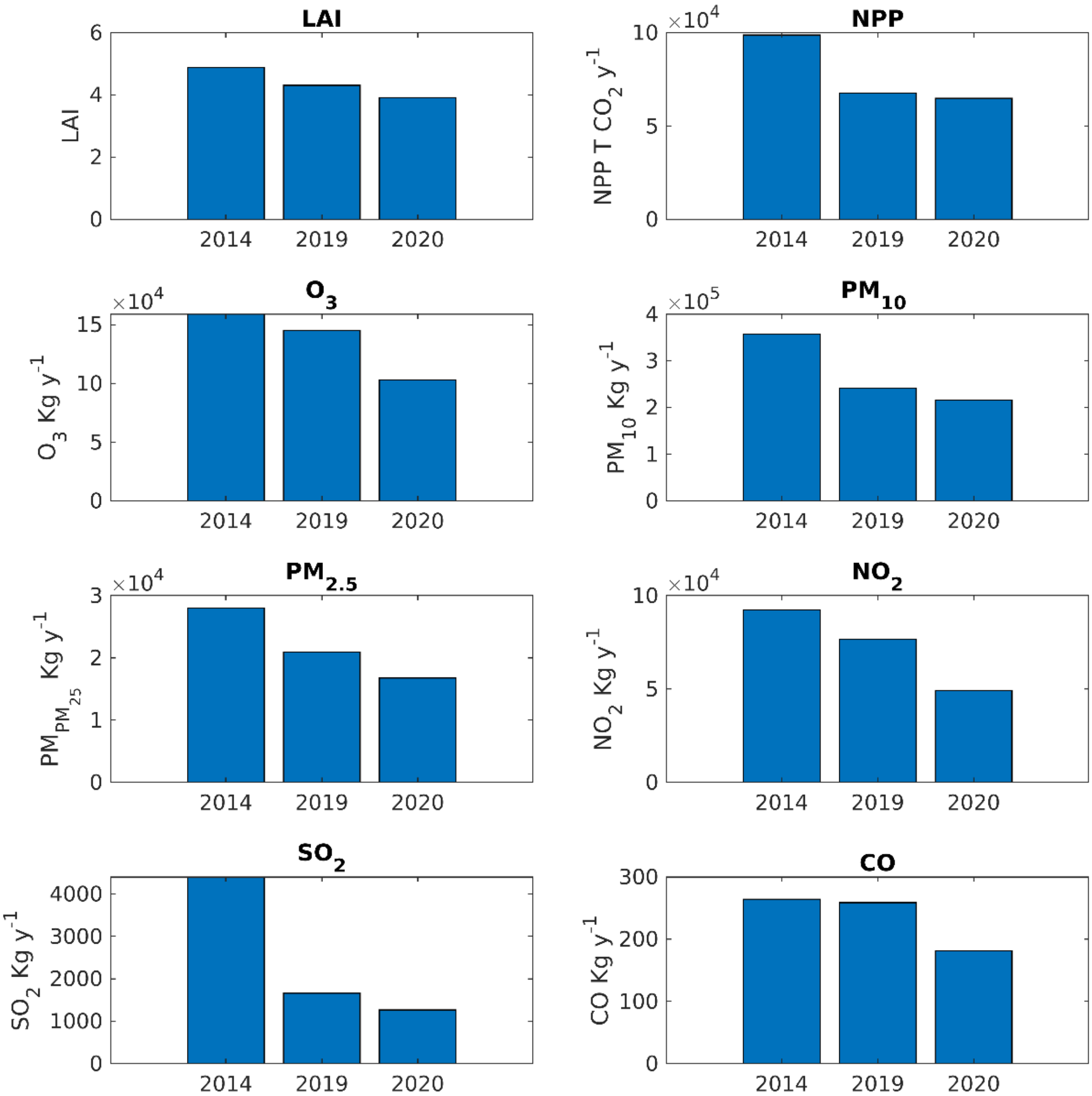


Figure 20: Total estimates of LAI, NPP, and pollutants deposition for the whole Estate in the three years of study. *(From Conte et al., 2022).*

In particular, Deciduous Oaks, Pine forests and Holm oak forests were the LCTs that contributed most: Deciduous Oaks contributed up to 53% (0.052 mt y^{-1}), Pine Forest up to 22% (0.019 mt y^{-1}) and Holm oak forests up to 12% (0.01 mt y^{-1}). The contribution to total carbon uptake by the other ecosystems (Other broadleaves 5%, corks 4%, Mediterranean maquis 9% and low shrubs <1%) was below 20%.

Compared to 2014, the Estate showed a decrease in productivity by 31.6% and 34.3% in 2019 and 2020, respectively (Figures 21a and Figure 22). Between 2019 and 2020, Deciduous Oak and

Mediterranean maquis and Low shrubs showed an increase in carbon uptake by 5.7%, 2.6% and 22.5%, respectively (in line with our previous statements about highest tolerance to drought stress).

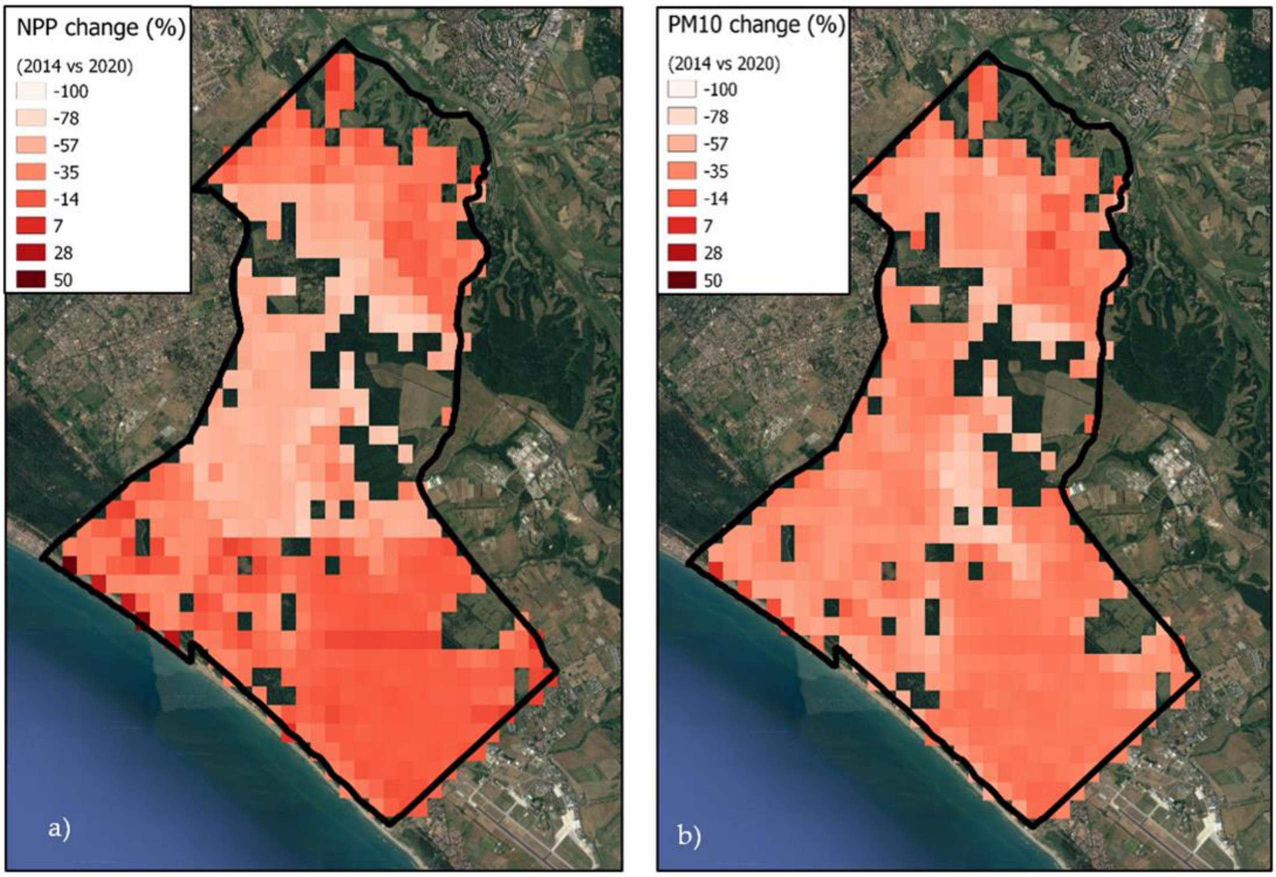


Figure 21: Percent changes (%) in (**a**) NPP (kg CO_2 y^{-1}) and (**b**) PM_{10} deposition (kg PM y^{-1}) estimated by the AIRTREE model between 2014 and 2020. The vividness of the pixels (from dark red to white) indicates the increased or decreased sequestration capacity of the Estate, while empty (transparent) pixels represent non-natural LCT not considered in this study. *(From Conte et al., 2022).*

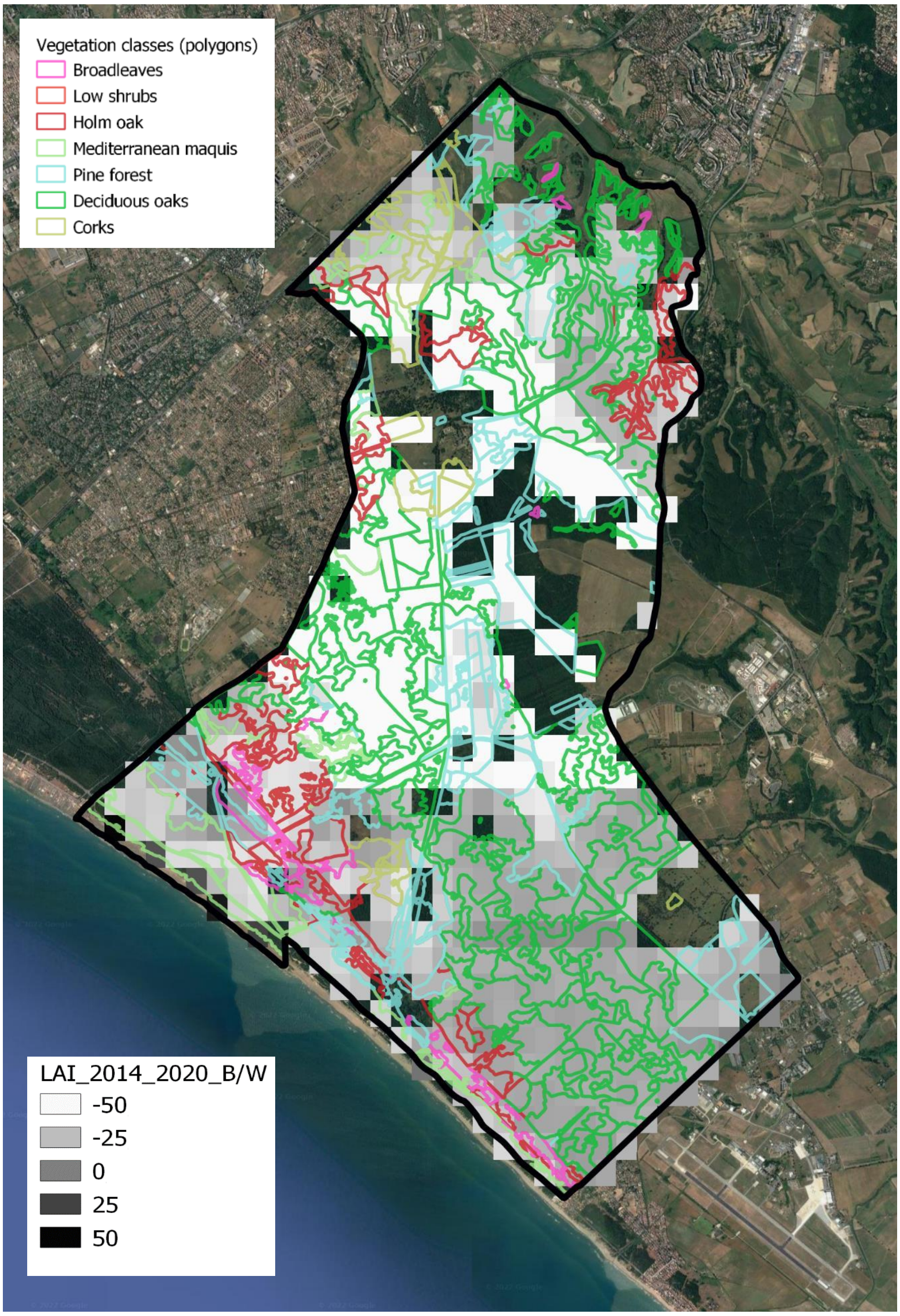


Figure 22: Superimposed maps of vegetation (colored shapes) to NPP (black and white pixels). Colored shapes represent homogeneous groups of vegetation (LCT). Pixels represents percent change of satellite values of NPP between 2014 and 2020. The vividness of color represent loss or increase of NPP. *(From Conte et al., 2022).*

Considering that pollutants such as ozone have been found to reduce carbon assimilation in the Holm oak forest up to 10% (Fares et al., 2019), is not exclude a possible beneficial effect of reducing atmospheric pollutants in 2020 due to lockdown status in the first part of the year. Other LCTs instead, showed a decrease in carbon uptake up to 17%, and such decrease can be highly appreciated by comparing 2014 and 2020.

Such behavior can be explained considering the different temperature recorded during the warm seasons in these two dry years. Otherwise, in 2019, the average temperature during summer was higher than in 2020 (Figure 8), and accordingly, the high vapor pressure deficit may have caused stomatal closure and reduced photosynthesis, as previously discussed in Conte et al. (2019). Looking at single LCT, the data show (Table 9) that broadleaf species and Holm oaks are the ecosystems with the higher NPP, with values up to 2.46 kg CO_2 m^2 y^{-1} in the wet year (2014), with abrupt decreases in productivity in the dry years. These model results, especially for Holm oak, are in line with data measured with Eddy Covariance at the ICOS Holm oak It-Cp2 site at the Estate (data not shown). Like Italy, warmer and drier conditions linked to increases in Atlantic multidecadal oscillations are associated with the increase in post-1990 defoliation in the forests of Spain (Sánchez-Salguero et al, 2016). These results are in agreement with previous studies that have clearly indicated that the decreases such as dieback, defoliation and lower growth in Mediterranean oaks (*Quercus spp*.) and pines (*Pinus spp.*) in southern Europe are mainly due to more frequent drought, often interacting with higher temperatures (higher water demand) and pathogenic attack (Braisier et al., 1996; Corcobado et al., 2014; Gea-Izquierdo et al., 2014; Peña-Gallardo et al., 2018; Camarero et al., 2018). Such a cumulative effect of stressors has been associated with pathogen infestations in Mediterranean forests (Erkan et al., 2011; Avila et al., 2016; Camarero et al., 2018; Mecheri et al., 2018).

LCT	**Year**	**NPP (Kg CO^2 m^2 y^{-1})**	**O_3 Flux ($g\ m^2\ y^{-1}$)**	**PM_{10} Flux ($g\ m^2\ y^{-1}$)**	**$PM_{2.5}$ Flux ($g\ m^2\ y^{-1}$)**	**NO_2 Flux ($g\ m^2\ y^{-1}$)**	**SO_2 Flux ($g\ m^2\ y^{-1}$)**	**CO Flux ($g\ m^2\ y^{-1}$)**
Low shrubs	2014	1.53 ± 0.44	2.52 ± 0.30	5.78 ± 0.27	0.36 ± 0.02	1.60 ± 0.24	0.08 ± 0.013	0.005 ± 0.0003
	2019	0.74 ± 0.26	2.22 ± 0.07	3.26 ± 0.40	0.23 ± 0.06	1.22 ± 0.02	0.03 ± 0.003	0.005 ± 0.0003
	2020	0.94 ± 0.09	1.65 ± 0.12	3.97 ± 0.64	0.25 ± 0.04	0.84 ± 0.05	0.02 ± 0.003	0.004 ± 0.0002
Other broadleaves	2014	2.46 ± 0.31	3.23 ± 0.19	6.69 ± 0.58	0.42 ± 0.04	1.79 ± 0.11	0.10 ± 0.006	0.006 ± 0.0002
	2019	2.10 ± 0.38	3.14 ± 0.13	4.72 ± 0.59	0.34 ± 0.04	1.60 ± 0.05	0.04 ± 0.003	0.006 ± 0.0002
	2020	1.81 ± 0.20	2.32 ± 0.12	4.63 ± 0.61	0.29 ± 0.04	1.05 ± 0.04	0.03 ± 0.002	0.004 ± 0.0001
Holm oak forest	2014	2.45 ± 0.43	3.17 ± 0.32	6.35 ± 0.96	0.40 ± 0.06	1.69 ± 0.18	0.09 ± 0.012	0.005 ± 0.0002
	2019	1.91 ± 0.54	2.91 ± 0.32	4.27 ± 0.80	0.31 ± 0.07	1.41 ± 0.11	0.04 ± 0.005	0.005 ± 0.0002
	2020	1.75 ± 0.38	2.11 ± 0.30	4.28 ± 0.83	0.27 ± 0.05	0.95 ± 0.11	0.03 ± 0.005	0.004 ± 0.0002
Mediterranean maquis	2014	1.86 ± 0.65	2.74 ± 0.34	5.61 ± 1.72	0.35 ± 0.11	1.66 ± 0.21	0.08 ± 0.015	0.005 ± 0.0003
	2019	1.32 ± 0.57	2.56 ± 0.26	3.57 ± 1.31	0.26 ± 0.09	1.38 ± 0.10	0.03 ± 0.005	0.005 ± 0.0003
	2020	1.43 ± 0.36	1.95 ± 0.17	3.78 ± 1.20	0.23 ± 0.07	0.96 ± 0.07	0.03 ± 0.004	0.004 ± 0.0002
Pine forest	2014	2.09 ± 0.38	5.62 ± 0.35	15.78 ± 2.72	1.56 ± 0.27	3.42 ± 0.20	0.14 ± 0.016	0.008 ± 0.0001
	2019	1.69 ± 0.49	5.43 ± 0.43	10.89 ± 2.45	1.20 ± 0.27	2.96 ± 0.15	0.06 ± 0.008	0.008 ± 0.0001
	2020	1.54 ± 0.39	4.02 ± 0.41	9.52 ± 2.35	0.96 ± 0.24	2.03 ± 0.16	0.04 ± 0.008	0.006 ± 0.0001
Deciduous oaks	2014	2.36 ± 0.61	3.20 ± 0.42	5.95 ± 0.90	0.38 ± 0.06	1.83 ± 0.22	0.09 ± 0.014	0.006 ± 0.0002
	2019	1.43 ± 0.68	2.92 ± 0.26	4.21 ± 0.84	0.30 ± 0.06	1.55 ± 0.08	0.03 ± 0.005	0.006 ± 0.0002
	2020	1.55 ± 0.36	2.21 ± 0.23	4.05 ± 0.79	0.26 ± 0.05	1.03 ± 0.07	0.03 ± 0.005	0.004 ± 0.0002
Corks	2014	2.03 ± 0.50	2.66 ± 0.30	6.60 ± 1.41	0.45 ± 0.10	1.47 ± 0.17	0.08 ± 0.012	0.005 ± 0.0004
	2019	1.52 ± 0.32	2.42 ± 0.25	4.33 ± 0.82	0.31 ± 0.07	1.21 ± 0.10	0.03 ± 0.004	0.005 ± 0.0004
	2020	1.41 ± 0.37	1.70 ± 0.28	4.48 ± 0.81	0.31 ± 0.06	0.81 ± 0.11	0.02 ± 0.005	0.003 ± 0.0003

Table 9: Mean annual estimate and standard deviation of Carbon uptake (NPP), Ozone (O_3 flux), Particulate matter fluxes (PM_{10} and $PM_{2.5}$), fluxes of Nitrogen dioxide (NO_2), Sulphur dioxide (SO_2) and Carbon monoxide (CO) for the three years of study (2014, 2019 and 2020), for each LCT are reported. *(From Conte et al., 2022).*

4.2.3 Sequestration of pollutants

Total yearly estimates of ozone deposition (O_3) indicate that the Estate sequestrated 159, 145 and 103 t O_3 y^{-1} for the years 2014, 2019 and 2020, respectively (Figure 12). Deciduous Oaks, Pine forests and Holm oak forests were the LCTs that contributed most. Deciduous Oaks contributed up to 46% to total ozone deposition (70 t y^{-1}) and Pine Forest up to 33% (51 t y^{-1}). The contribution to total ozone deposition by the other ecosystems (Holm oak forest up to 9%, Mediterranean maquis up to 7%, Other broadleaves up to 3%, corks up to 4% and low shrubs <1%) is below 25% (Figure 23).

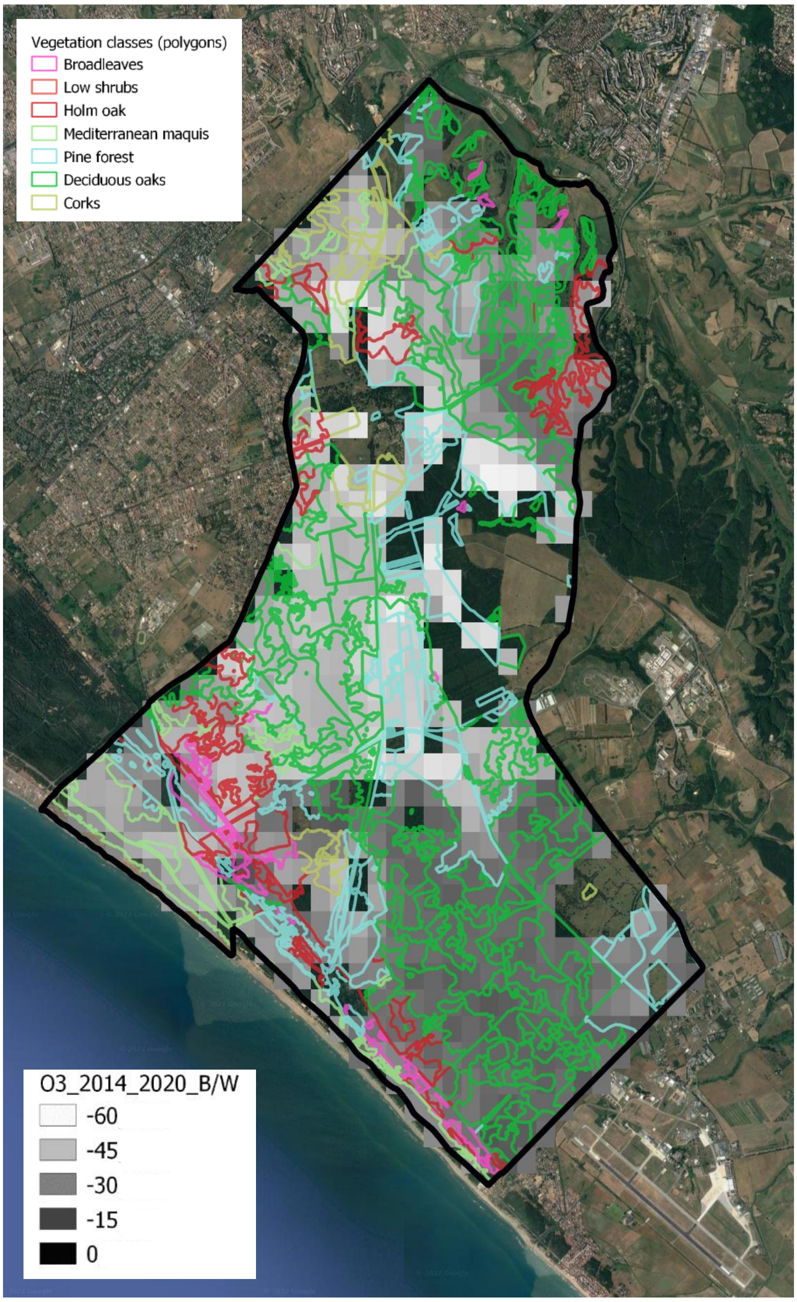


Figure 23: Superimposed maps of vegetation (colored shapes) to O_3 fluxes (black and white pixels). Colored shapes represent homogeneous groups of vegetation (LCTs). Pixels represents percent change of satellite values of O_3 fluxes between 2014 and 2020. The vividness of color represent loss or increase of O_3 fluxes. *(From Conte et al., 2022).*

The modelled ozone deposition on Holm oak agrees with our previous measured with the Eddy Covariance carried out in 2013 and 2014 (Fusaro et al., 2015).

Compared to 2014, the Estate showed a decrease in ozone deposition by 8.8% and 35.2% in 2019 and 2020, respectively. In particular, Pine Forest, Deciduous oaks, other broadleaves and Corks showed a decrease in ozone uptake by 32.7%, 25.2%, 29.4% and 35.3%, respectively. Such a decrease in deposition (Figure 18), in part due to the loss of canopy cover and in part due to physiological limitations as demonstrated by NPP decreases (Table 8), suggests that land use changes and environmental stressors heavily compromise the capacity of the Estate and, in general, Mediterranean peri-urban forests. It must point out, however, that most of the decreases in 2020 compared with 2014 are due to a strong reduction in atmospheric ozone concentration due to the lockdown status in the first part of the year. Nevertheless, ozone concentrations were similar between 2014 and 2019, highlighting evidence of stomatal limitation to ozone deposition in the dry year as stressed in our previous work (Savi et al., 2014; Conte et al., 2021).

Total yearly estimates of particulate matter (PM) indicate that the Estate sequestrated 357, 241 and 215 t PM_{10} y^{-1} and 27, 21, and16 t $PM_{2.5}$ y^{-1} for the years 2014, 2019 and 2020, respectively (Figures 20 and 23a, b). Deciduous oaks and Pine forests were the LCTs that contributed equally to PM_{10} deposition (up to 41% Pine Forest in 2014 and Deciduous oaks in 2020, Figure 24a, b), while a higher contribution (up to 52%) of Pine forest was found for $PM_{2.5}$ (Figure 24a, b). The contribution to total PM deposition by the other ecosystems is below 25% (Other broadleaves up to 3%, Corks up to 4%, Mediterranean maquis up to 6%, Holm oak forest 8% and low shrubs <1%) (Figure 24a, b).

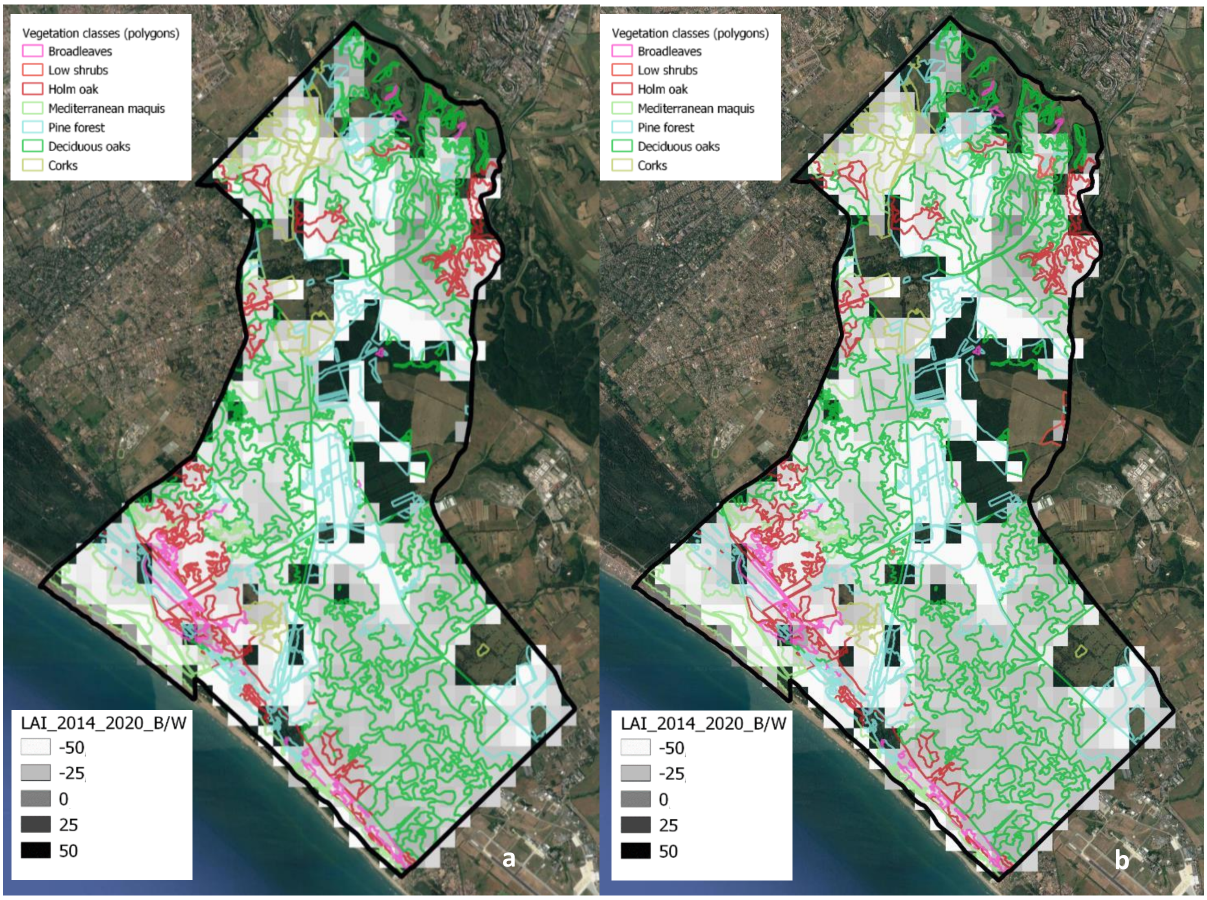


Figure 24 (a, b): Superimposed maps of vegetation (colored shapes) to PM_{10} (left) and $PM_{2.5}$ (right) deposition on canopies (black and white pixels). Colored shapes represent homogeneous groups of vegetation (LCT). Pixels represents percent change of satellite values of LAI between 2014 and 2020. The vividness of color represent loss or increase of LAI. *(From Conte et al., 2020).*

Compared to 2014, the Estate showed a decrease in PM_{10} deposition by 32.5% in 2019 and 39.6% in 2020 (Figure 21b), while there was a decrease in $PM_{2.5}$ deposition of 25% in 2019 and 40% in 2020. Between 2019 and 2020, a decrease of 10.5% and 19.9% was observed for PM_{10} and $PM_{2.5}$, respectively. In particular, the Pine Forest showed a reduction of 19.8% and 26% for PM_{10} and $PM_{2.5}$ in 2020, respectively (Figure 12). The same conclusions drawn for ozone work here, with a significant reduction in PM emissions during the lockdown in 2020 (Figure 18). To note, several hectares (0.7 km^2 in 2017 and 2022) of forests have been cut in 2017 and 2020 (and another 1 km^2 is going to be cut in the near future) probably due to a strong infection of *Tomicus destruens* and *Toumayella parvicornis* on *Pinus pinea* stands, as visible in the white pixels in Figure 24. Moreover, as shown in Table 9 (and also found elsewhere in the Case Study 1, by Fares et al. (2020)), individual performances in PM removal are higher in Pine forests; therefore, a loss of these forest stands represents a major loss in RES provision. This loss is particularly relevant for cities such as Rome, where *Pinus pinea* is a key tree of the urban landscape.

Concerning Nitrogen Dioxide (NO_2), we found values of 92, 76 and 49 t NO_2 y^{-1} sequestration for the years 2014, 2019 and 2020, respectively (Figure 20). Deciduous Oaks removed up to 45%, and Pine forests removed up to 34%. Mediterranean maquis and holm oak forest contribution to total fluxes were constant (up to 8%) in the three years of study. The contribution to total NO_2 deposition by the other ecosystems (Other broadleaves up to 3%, Corks up to 4% and Low shrubs <1%) was below 10%. Compared to 2014, the Estate showed a decrease in NO_2 deposition of 17% and 46.8% in 2019 and 2020, respectively. In particular, the highest decreases in NO_2 deposition were observed for Holm oak forest (up to 36.2%) and Corks (up to 38%), Pine Forest (37.9%) and other Broadleaves (36.9%). Similar to ozone, the same conclusions can be drawn for NO_2, since 2020 recorded much lower emissions due to lockdown status (Figures 8 and 25).

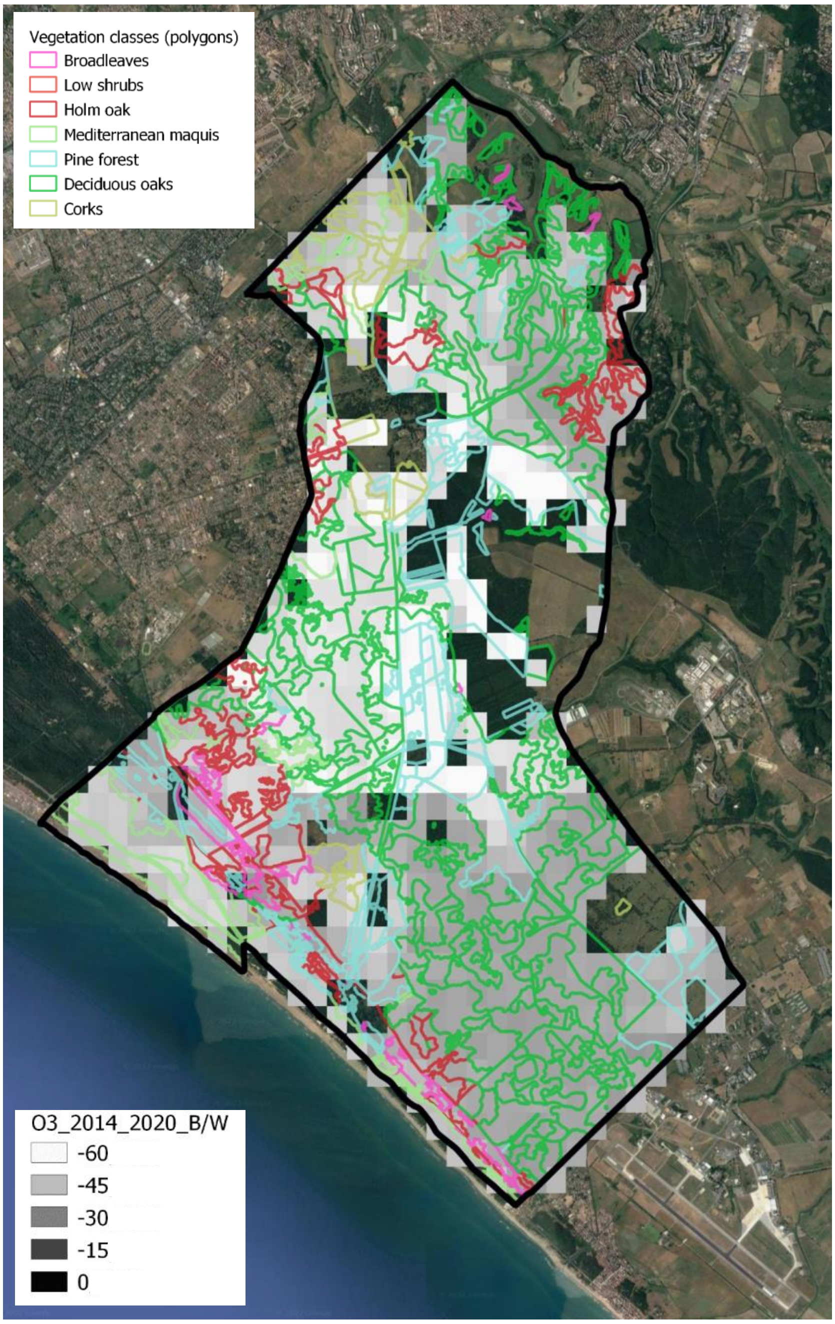


Figure 25: Superimposed maps of vegetation (colored shapes) to NO_2 deposition on canopies (black and white pixels). Colored shapes represent homogeneous groups of vegetation (LCT). Pixels represents percent change of satellite values of LAI between 2014 and 2020. The vividness of color represent loss or increase of LAI. *(From Conte et al., 2020).*

As a minor entity, the Estate sequestrated other pollutants such as sulfur dioxide (SO_2) and carbon monoxide (CO). In agreement with the removal dynamics observed for the other pollutants, a strong decrease was observed, with values decreasing from 4.4 to 1.6 and 1.26 t SO_2 y^{-1} and from 0.26 to 0.25 and 0.18 t CO y^{-1} for the years 2014, 2019 and 2020, respectively (Figure 18).

4.3 Case Study 3: Urban Street Trees

4.3.1 Statistical analysis

Taking into consideration the list of species of Bologna and Milano, the different frequency of the various species was analyzed using the Pareto criterion (criterion 80/20) (Hochman, 1969). The first 16 and 21 species are the peculiar ones for Milan and Bologna respectively. These species are representative of 80% of the variety of street trees and public parks.

Tables 12 and 13 show the absolute frequencies of trees.

It can be noted that *Celtis australis* and *Platanus x acerifolia* are the most frequent species in the two italian cities. *Celtis australis* is generally used successfully in street trees and city parks, for its resistance to urban pollution and for the dense shading canopy, despite the risks for road paving, due to the fact that its root system can also develop on the surface. *Platanus x acerifolia* is actually a hybrid obtained from two species: *Platanus occidentalis* and *Platanus orientalis*. This tree tolerates well cold and heat, and its high resistance to air pollution, has contributed to its spread as a road tree. *Platanus x acerifolia* can be pruned and this makes it suitable for any type of road, even if crossed by high and bulky public transport. It resists weathering and pollution and is very long-lived. Even after the most traumatic prunings it can resprout and emit new shoots. *Ulmus Pumila* tolerates rapid changes in atmospheric conditions (it also survives after winter frosts and prolonged periods of drought).

An in-depth botanical analysis was performed with the intent to define the state of the art of urban trees and shrubs planted in the municipalities. The families of each species present were also mentioned, to obtain the most frequent ones.

The most represented family is the *Rosaceae.* This distribution could be linked to the different uses of the species themselves; the *Rosaceae* family includes a large number of most of the species of great importance for the human economy (Christenhusz, 2016), such as the most common fruit trees, rich in colored flowers and fruits, which in the context of street trees reflect an aesthetic taste which favors them compared to other species.

Family	Species Frequency
Rosaceae	35
Pinaceae	27
Cupressaceae	20
Sapindaceae	19
Salicaceae	15
Betulaceae	15
Leguminosae	14
Fagaceae	13
Malvaceae	10
Oleaceae	9
Ulmaceae	8
Moraceae	7
Arecaceae	6
Magnoliaceae	5
Platanaceae	4
Bignoniaceae	3
Cannabaceae	3
Juglandaceae	3
Ebenaceae	2
Rutaceae	2
Cornaceae	2
Altingiaceae	2
Lythraceae	2
Taxaceae	2
Paulowniaceae	2
Aquifoliaceae	1
Tamaricaceae	1
Anacardiaceae	1
Simaroubaceae	1
Hamamelidaceae	1
Ginkgoaceae	1
Cercidiphyllaceae	1
Lauraceae	1
Adoxaceae	1
Juglandaceae	1
Lamiaceae	1
Meliaceae	1
Total	**242**

Table 10: Families with relative frequency in the municipality of Milan.

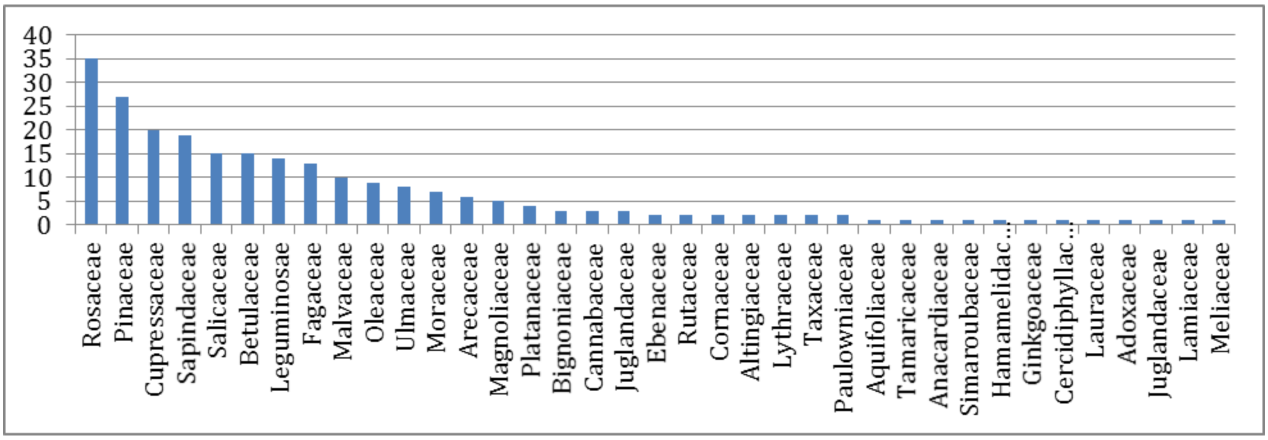


Figure 26: Graph of the distribution of families in the municipality of Milan.

Family	Species Frequency
Rosaceae	23
Pinaceae	14
Cupressaceae	12
Sapindaceae	10
Fagaceae	9
Oleaceae	7
Salicaceae	6
Malvaceae	6
Leguminosae	5
Betulaceae	4
Cornaceae	3
Magnoliaceae	3
Ulmaceae	2
Juglandaceae	2
Anacardiaceae	2
Platanaceae	2
Adoxaceae	2
Lythraceae	2
Moraceae	2
Apocynaceae	1
Tamaricaceae	1
Bignoniaceae	1
Meliaceae	1
Lamiaceae	1
Lauraceae	1
Altingiaceae	1
Aquifoliaceae	1
Cannabaceae	1
Paulowniaceae	1
Taxaceae	1
Asparagaceae	1
Pittosporaceae	1
Ginkgoaceae	1
Total	130

Table 11: Families with relative frequency in the municipality of Bologna.

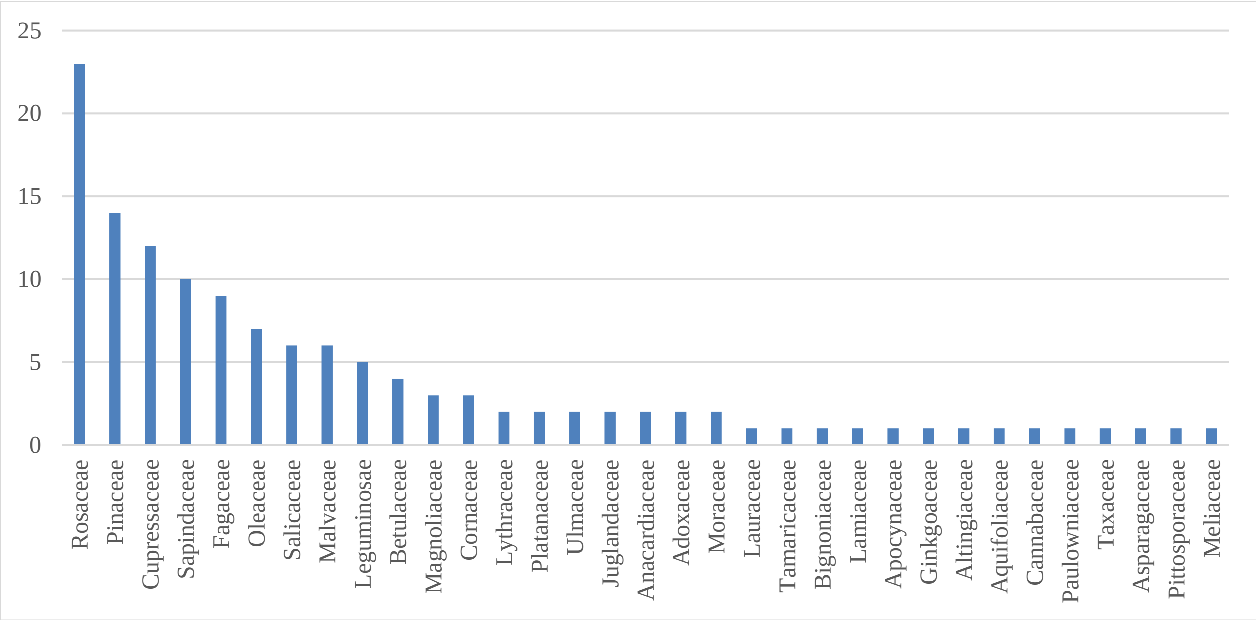


Figure 27: Graph of the distribution of families in the municipality of Bologna.

Between the various individuals of each species an average of heights was made, so as to obtain a single middle class of belonging. These height classes are inspired to the SIA classification (Italian Arboriculture Society), and are 4: (I), trees that can exceed 30 m in height; second greatness (II), m trees between 20 and 30 meters; (III), trees between 10 and 20 m height; (IV), small trees that reach a maximum height of 10 m.

From the analyzes concerning the size classes it turned out that the most frequent class is the one between 0 and 10 m (IV class). Probably this choice is justified by the fact that the trees of this size could be the most suitable for the dimensions of the sidewalks and available spaces, as well as pruning practices.

Classes	Frequency Classes of Size
I	0,09%
II	6,38%
III	44,83%
IV	48,70%
Total	**100,00%**

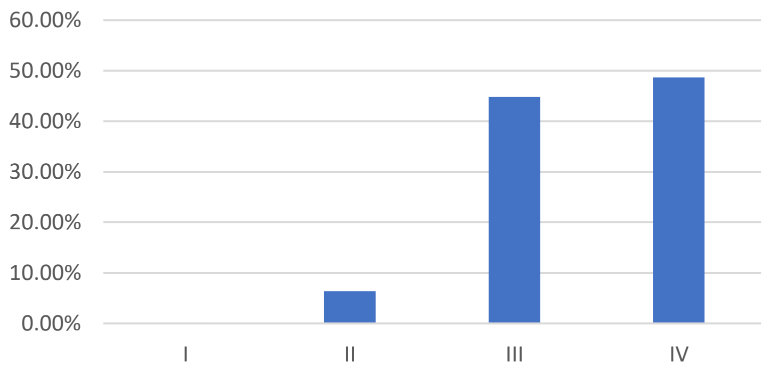


Figure 28: Height classes in the municipality of Milan.

Classes	Frequency Classes of Size
I	11,76%
II	22,79%
III	28,68%
IV	36,76%
Total	**100,00%**

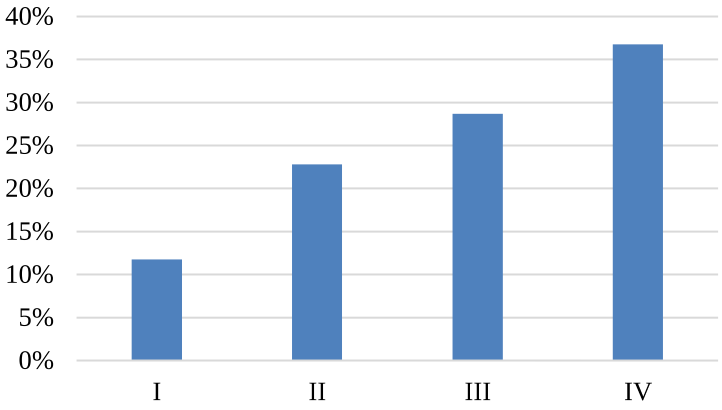


Figure 29: Height classes in the municipality of Bologna.

The corotypes were obtained from Encyclopedia of Life (http://www.eol.org). They have been merged into 14 major chorological types (Africa, Asiat, Australia, E-Asiat, Eurimedit, Euro-Asiat, Europ, Europ-Caucas, N-Amer, Pontica, S-Amer, Steno-Medit, Boreali, Paleotemp). For artificially created species (such as commercial hybrids), since it is not possible to assign a corresponding corotype, it was decided to define them as "Horticultural Origin" or "Silvicultural Origin".

The calculation of the normal chorological spectrum is interesting as it is generally used in vegetation studies and not in contexts of street trees. Furthermore, in order to evaluate the percentage of exotic species used in this context, the 14 corotypes were grouped into three groups: European (Eurimedit, Euro-Asiat, Europ, Europ-Caucas, Pontica, Steno-Medit, Paleotemp); Exotic (Africa, Asiat, Australia, E-Asiat, N-Amer, S-Amer, Boreali); Silvicultural Origin (Horticultural Origin).

For the data of chorotypes, two types of parallel analyzes have been made, one taking into account the species only in qualitative terms, another keeping the quantitative information (how many individuals for each species).

Qualitative analysis means that only the species itself is taken into account, and not the number of trees present in that city for that species. On the other hand, in the quantitative analysis, the number of individuals for each species was also considered.

For Milan see Figures 30, 31, 32, and 33. For Bologna see Figures 34, 35, 36 and 37.

As far as the chorological spectrum is concerned, the exotic and the native are even if the datum is considered qualitative, but the native species quantitatively prevail. The presence of exotic species, in addition to the local growth conditions, and origin of the species, could be linked to historical and cultural reasons as well as to personal preferences for some species (Thomsen et al., 2016).

Chorotypes	% Chorological Spectrum
(Avv)	1
Asiat	19
Boreali	10
E-Asiat	44
Eurimedit	17
Euro-Asiat	11
Europ	21
Europ-Caucas	16
N-Amer	53
Paleotemp	10
Pontica	8
Steno-Medit	12
Horticultural Origin	16
Australia	2
S-Amer	2
Total	**242**

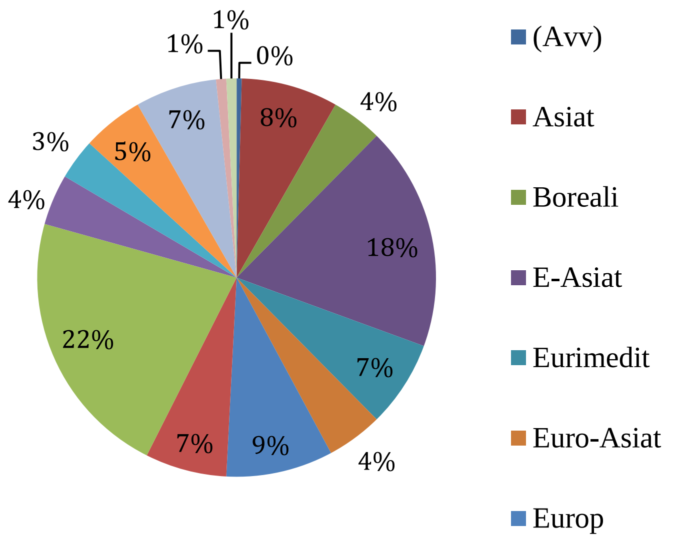


Figure 30: Table and pie chart of the chorological spectrum at a qualitative level in the municipality of Milan.

Chorotype Group	Species Count
EXOTIC	127
EUROPEAN	98
SILV-CULT	17
Total	**242**

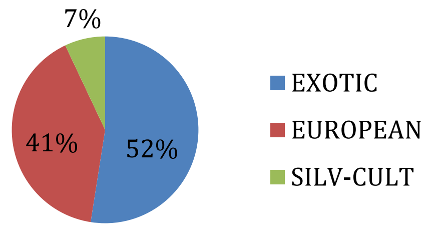


Figure 31: Table and pie chart with grouping of chorotypes, quality level in the municipality of Milan.

Chorotypes	Plant Distribution
N-Amer	15595
Europ-Caucas	10800
Eurimedit	9298
Horticultural	7961
E-Asiat	6564
Europ	4300
Paleotemp	3359
Pontica	3018
Asiat	1409
Euro-Asiat	1351
Boreali	1293
Steno-Medit	424
(Avv)	31
Australia	10
S-Amer	3
Total	**65416**

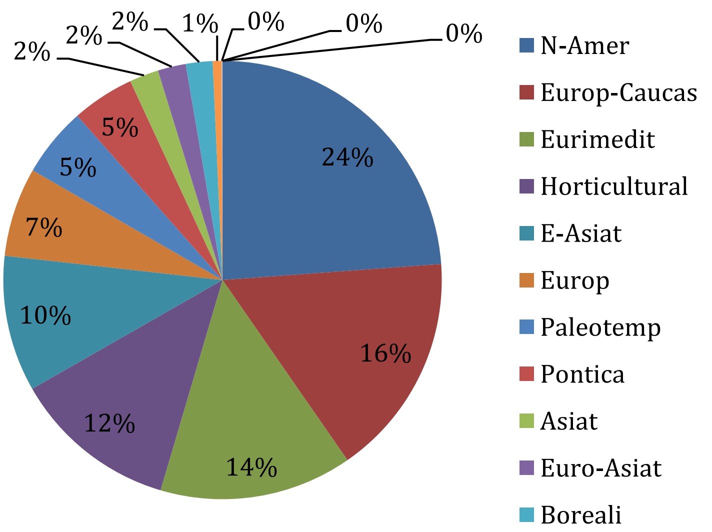


Figure 32: Table and pie chart of the chorological spectrum at a quantitative level in the municipality of Milan.

Chorotype Group	Species Count
EXOTIC	24846
EUROPEAN	32578
SILV-CULT	7992
Total	**65416**

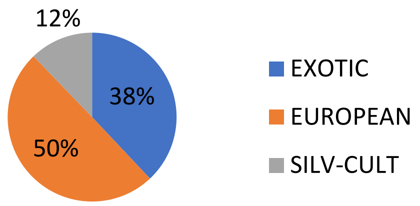


Figure 33: Table and pie chart with grouping of chorotypes, quantitative level in the municipality of Milan.

Chorotypes	% Chorological Spectrum
(Avv)	1
Horticultural Origin	7
Asiat	15
Boreali	5
E-Asiat	21
Eurimedit	14
Euro-Asiat	9
Europ	12
Europ-Caucas	8
N-Amer	23
Paleotemp	6
Pontica	5
Steno-Medit	10
Total	**136**

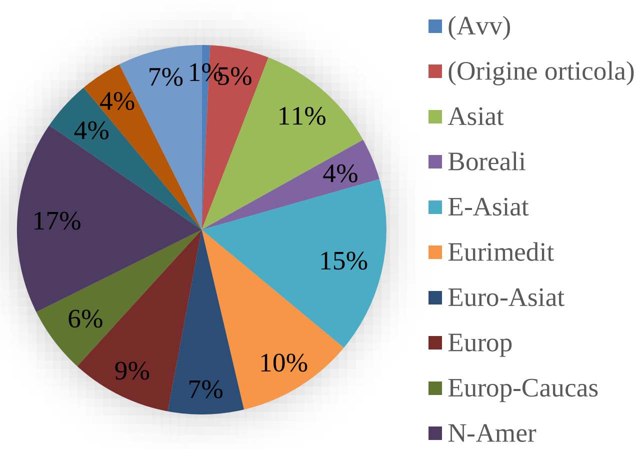


Figure 34: Table and pie chart of the chorological spectrum at a qualitative level in the municipality of Bologna.

Chorotype Group	Species Count
EXOTIC	64
EUROPEAN	64
SILV-CULT	8
Total	**136**

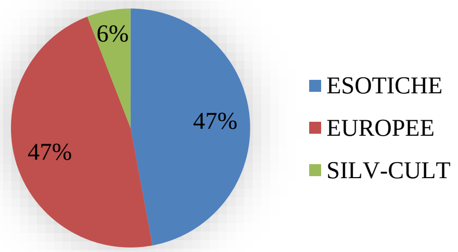


Figure 35: Table and pie chart with grouping of chorotypes, quality level in the municipality of Bologna.

Chorotypes	Plant Distribution
(Avv)	17
Horticultural Origin	11411
Asiat	2641
Boreali	571
E-Asiat	3309
Eurimedit	13456
Euro-Asiat	2135
Europ	1560
Europ-Caucas	14658
N-Amer	4181
Paleotemp	3195
Pontica	3010
Steno-Medit	1071
Total	**61215**

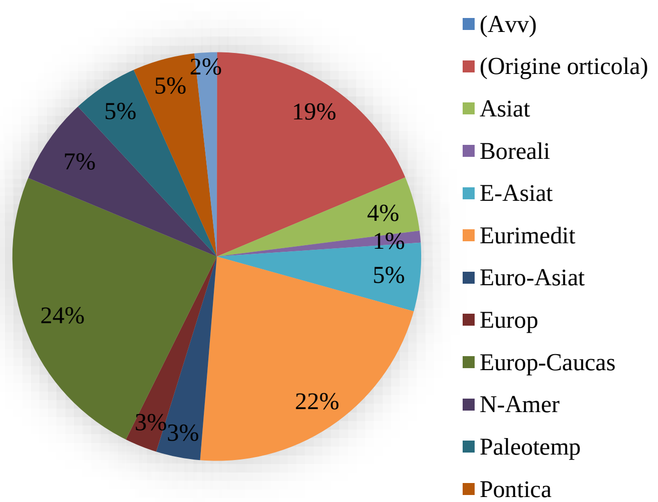


Figure 36: Table and pie chart of the chorological spectrum at a quantitative level in the municipality of Bologna.

Chorotype Group	Species Count
EXOTIC	10702
EUROPEAN	39085
SILV-CULT	11428
Total	**61215**

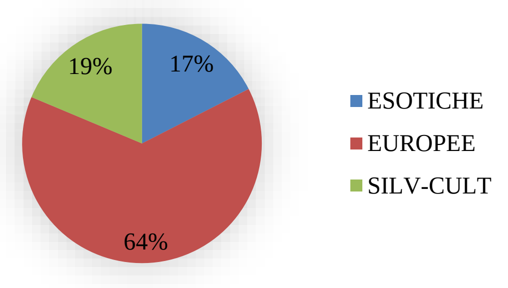


Figure 37: Table and pie chart with grouping of chorotypes, quantitative level in the municipality of Bologna.

4.3.2 Specie-specific performances in Carbon and pollutant removal

Tables 11 and 12 show the 30 most abundant species in Milano and Bologna, that are representing the variety of street trees and public parks in these two cities.

In total, the Municipality of Milan reports 264558 individuals, while that of Bologna 66216.

Tables 11 and 12 present the average values (considering the whole year) of the sequestration of CO_2 and pollutants (O_3, PM_{10}, $PM_{2.5}$, NO_2) for each species for both cities.

The tree family that seems to sequestrate the higher amount of CO_2 in Milan, considering the most abundant species (Table 12) is the *Platanus* family, with values of 1025.470 g CO_2 m^2 y^{-1}. High sequestration values are also reached by *Ulmus pumila* and *Quercus rubra*, with values of 1024.465 g CO_2 m^2 y^{-1} and 1023.284 g CO_2 m^2 y^{-1}, respectively. Similar values were found in the urban parks above mentioned in the Case Study 1 (Fares et al., 2020) when simulating ES for the Valentino urban park in the city center of Turin. Interestingly, while in Milan these species showed the highest carbon uptake capacity (Table 12), in Turin they were not even in the top three of performance classes. This might be explained as different planting choices. While in urban parks ornamental trees, that can reach high canopies and LAI, can represent the most occurring choice, this might be not true for tree-lined row (Caneva et al., 2020). *Platanus x acerifolia* (also called “London Plane”), indeed, well tolerates cold and heat, and its high resistance to air pollution, has contributed to its spread as a road tree (Li et al, 2011). This species can tolerate severe pruning compared to other street trees though in the long term this can result in wood decay and stability issue (Smithers et al, 2018). Interestingly, although scarcely used (only 772 individual censed in the city of Milan, representing only 0.3% of the total), *Cupressus arizonica* is one of the tree that show highest carbon sequestration (3272.828 g CO_2 m^2 y^{-1}), followed immediately by *Cedrus libani* and *atlantica* (representing the 0.9% of the total with 2831.016 g CO_2 m^2 y^{-1} of average sequestration) and *Taxodium* (representing the 0.1% of the total trees with 2565.978 g CO_2 m^2 y^{-1} sequestration). These results are consistent with those seen in the Valentino urban park in the city of Turin in Case Study 1 (Fares et al., 2020) and the values for

Cupressus, are in the same order of magnitude of González-Cásares et al. (2019). In Bologna, on the other hand, the highest carbon uptake considering the most abundant species (Table 13) was provided by *Cedrus atlantica* and *deodara*, with values of 2581.468 g CO_2 m^2 y^{-1} and 2594.612 g CO_2 m^2 y^{-1} respectively. Similar values were found in Case Study 1 (by Fares et al., 2020) for *Cedrus deodara*, thus suggesting that this species is particularly suitable for CO_2 sequestration, because it can reach high dimension and LAI, sequestrating carbon almost all year long.

Surprisingly, *Celtis australis*, between the top two most frequent trees in both cities (together with *Platanus x acerifolia*), showed poor capacity to sequestrate CO_2 when compared to *Platanus x acerifolia* (about half), probably due to the lower dimension of the canopy and LAI (one of the most important parameters for the AIRTREE model) (Fares et al., 2019). Although *Celtis* is generally used successfully in street trees and city parks for its resistance to urban pollution and for the dense shading canopy (Caneva et al., 2020), this PhD work suggest that other species could be more indicated for the urban environment (Li et al, 2011; Smithers et al., 2018). Indeed, although among the least used tree in Bologna, the tree that provides the higher carbon uptake (3938.288 g CO_2 m^2 y^{-1}) is also a cypress, in particular *Cupressus lutisanica*, which is indeed large (so called "monumental tree") but is present in only three individuals in the whole city.

In this work, the attention is focused on PM since the latter is the pollutant regulated by European legislation (UNI EN12341/2014) with the greatest health concerns. Particularly worrying is the fact that in 2015, the limit for annual average concentration of PM_{10} (equal to 40 µg/m^3) was exceeded in Milan (an average of 42 µg/m^3). Moreover, from 25 November to 30 December, average daily PM_{10} concentrations remained constantly above 50 µg/m^3, for a total of 36 consecutive days of the year 2015, on which the legal limit for daily average concentration was exceeded. The exceedances of the limits in 2015 is related to the strong and prolonged atmospheric stability and low rainfall in the winter season, particularly in the last months of the year, but certainly also to intensive traffic. (https://www.arpalombardia.it/). Bologna, on the other hand, exceeded this limit for an average of measuring stations of about 20 days in 2015. (https://www.arpae.it/).

Focusing on sequestration of particles (PM_{10} in particular), species-specific values remain homogeneous among the most present species in the city of Milan (Table 12) with an average of 2.164 g PM_{10} m^2 y^{-1}, with peaks of sequestration by species with large canopies such as *Platanus* (2.921 g PM_{10} m^2 y^{-1}). Regarding the most present species in the city of Bologna (Table 13), however, the PM_{10} and $PM_{2,5}$ sequestration peaks relate to species such as *Cedrus deodara*, *Cedrus atlantica* and *Cupressus sempervirens*. These trees are examples of large, evergreen trees with dense needle foliage, well maintained and often occurring in historic villas, which explains the high sequestration of PM and other pollutants (Table 13).

The highest performance in PM_{10} sequestration was observed for Evergreen trees in both Milan and Bologna. In Milan, the highest performance was showed by *Cedrus libani* (31.060 g PM_{10} m^2 y^{-1}), *Pinus jeffreyi* (27.318 g PM_{10} m^2 y^{-1}) and *Taxodium distichum* (27.305 g PM_{10} m^2 y^{-1}). In Bologna the highest performance is from *Taxodium distichum* (14.536 g PM_{10} m^2 y^{-1}) and *Cupressus lusitanica* (12.865 g PM_{10} m^2 y^{-1}). This because these trees have large and dense canopies (high values of LAI) with needle leaf previously described as suitable vegetated surfaces for particulate sequestration (Dzierżanowski et al., 2011; Speak et al., 2012; Manes et al., 2016; Beckett et al., 2000; Muhammad et al., 2022). These results are also in line with a study performed in the Italian city of Florence with a climate (Cfb) near the one of Bologna (Cfa), by Bottalico et al. (2017).

After PM, O_3 is the air pollutant that, due to its toxicity and the concentration levels that can be reached, has the greatest impact on human health. Italian legislative Decree 155/2010 defines an information threshold (180 µg/m^3) and an alert threshold (240 µg/m^3) for O_3 for the protection of human health. Milan and Bologna both exceed the alert threshold for the year 2015 for more than 25 days. (https://www.arpae.it/; https://www.arpalombardia.it/). As seen above, 2015 was a particularly warm year. The strong sunlight favors the photochemical reactions that generate O_3. For this reason, this pollutant is particularly critical during the summer period, especially during the hottest hours of the day, or occurs massively throughout the year if temperatures remain higher in winter. The criticality of O_3 is therefore closely linked to the meteorology that characterizes each year (Baraldi et al., 2019).

O_3 is also very evenly sequestered by trees. *Platanus x acerifolia* and *Populus nigra* take the lead in O_3 sequestration in both cities, when considering the most frequent species (Tables 12 and 13) (Baraldi et al., 2019).

Looking instead at the less frequent species, it is very interesting to see how in Bologna, Deciduous broadleaf sequester the most O_3 (1.459 g m^2 y^{-1}), followed then by Evergreen Needleleaf and finally Evergreen Broadleaf. *Cupressus lusitanica* (1.692 g m^2 y^{-1}) and *Taxodium distichum* (1.461 g m^2 y^{-1}), however, show a high O_3 sequestration capacity, but this is always attributable to the size of these trees (i.e., large and dense canopies with high LAI). In general, the data on Deciduous Broadleaf is well explained by the AIRTREE model formalism: dry deposition is driven by stomata and therefore the same species that have a high capacity to sequester CO_2 (with high stomatal conductance) also tend to have a higher stomatal deposition. It is also interesting to note that some trees species are sensitive to O_3 exposure. In particular, accounting for O_3 stress in AIRTREE formalisms led to a mismatch between O_3 and carbon sequestration for ozone-sensitive species. A typical example is the genus *Populus* which has a high sequestration of O_3 (*Populus deltoides* is 1.010 g m^2 y^{-1} in Milan and *Populus nigra* is 1.211 g m^2 y^{-1} in Bologna), but a low sequestration of CO_2

(808.820 g m^2 y^{-1} in Milan and 862.381 g m^2 y^{-1} in Bologna, respectively). Other trees showing the same behavior are among conifers: *Pinus* and *Abies*; while other broadleaf trees are *Acer, Cercis, Fraxinus, Platanus, Quercus* and *Fagus*. All these species are in fact sensitive to O_3 and the damage they receive from it affects their ability to sequester CO_2 (Coulston et al., 2003). For example, *Pinus* has an average of 0.803 g m^2 y^{-1} of sequestrated O_3 and an average of 989.325 g m^2 y^{-1} of sequestrated CO_2 in Milan; and an average of 0.986 g m^2 y^{-1} for O_3 and 1081.858 g m^2 y^{-1} for CO_2 in Bologna. About broadleaf, the same evidence can be found in *Cercis*, which has an average of 0.627 g m^2 y^{-1} of sequestered O_3 and 272.766 g m^2 y^{-1} of sequestered CO_2 in Milan; and then 0.787 g m^2 y^{-1} for O_3 and 223.370 g m^2 y^{-1} for CO_2 in Bologna.

As regard NO_2, Legislative Decree 155/2010 sets an hourly limit value (200 µg/m^3 not to be exceeded more than 18 times in a year) and an annual limit value (40 µg/m^3) for the protection of human health. In Bologna, the annual limit value is exceeded, but not the hourly limit value, while in Milan both were exceeded. (https://www.arpae.it/; https://www.arpalombardia.it/).

NO_2 sequestration by urban vegetation is homogeneous in both cities. This is probably because most of the trees are Deciduous Broadleaf. As such, these trees have broadleaf, hence a greater stomatal density and cuticle thickness, and consequently a greater capacity to absorb pollutants in a gaseous state, when compared with needleleaf (Baraldi et al., 2019).

On average, we found that in Milan the evergreen seems to be the most indicated species in sequestration of PM (mean of 15.9 g m^2 y^{-1}) and even for gases whose deposition is affected by plants stomata (O_3 mean of 1.06 g m^2 y^{-1}; CO_2 mean of 1.5 g m^2 y^{-1}; NO_2 mean of 0.46 g m^2 y^{-1}). Conversely, in Bologna the pollutant sequestration of O_3 and NO_2 between conifers and broadleaf trees is roughly equal. This implies that, in light of species selection for new forest plantation, the choice of tree may go towards needleleaf trees, but attention should be paid to the emission of Biogenic Volatile Organic Compounds (BVOCs), which are emitted in particular by needleleaf trees and may cause local episode of pollution (de la Paz et al., 2022).

Species	Number of Individuals	d.b.h.	Height	Mean NPP	Mean O_3	Mean PM_{10}	Mean $PM_{2.5}$	Mean NO_2
		cm	m	g CO_2 m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}
Platanus x acerifolia	16510	51	17	1008.417	1.018	2.688	0.356	0.395
Celtis australis	12632	53	15	582.344	0.924	1.927	0.257	0.336
Acer platanoides	12357	32	10	849.697	0.898	2.113	0.283	0.350
Carpinus betulus	11812	25	8	530.409	0.785	1.935	0.263	0.322
Robinia pseudoacacia	11,133	33	10	539.993	0.823	2.088	0.286	0.327
Populus nigra	10249	60	16	810.649	0.971	2.510	0.342	0.375
Liquidambar styraciflua	8492	29	10	674.763	0.855	2.126	0.284	0.342
Prunus cerasifera	7332	24	5	249.012	0.613	1.445	0.193	0.282
Ulmus spp	6843	41	14	792.059	0.917	2.515	0.338	0.367
Fraxinus excelsior	6837	25	7	503.281	0.769	1.720	0.233	0.315
Quercus rubra	6798	39	14	1023.284	0.978	2.448	0.329	0.376
Acer pseudoplatanus	6750	30	9	598.583	0.796	2.073	0.275	0.331
Platanus spp	6599	48	18	1025.470	0.997	2.921	0.376	0.405
Acer negundo	5832	36	10	672.343	0.838	2.146	0.287	0.340
Tilia spp	5780	39	14	745.138	0.906	2.469	0.329	0.360
Quercus robur	5760	32	12	874.515	0.914	2.321	0.312	0.359
Acer saccharinum	5689	35	10	667.615	0.836	2.142	0.287	0.338
Prunus serrulata	5567	16	4.5	142.167	0.571	1.360	0.180	0.272
Tilia cordata	5352	38.20	12	694.857	0.878	2.283	0.306	0.347
Aesculus hippocastanum	4905	46	14	715.952	0.905	2.524	0.334	0.364
Liriodendron tulipifera	4751	25	8	487.146	0.769	1.906	0.254	0.319
Cercis siliquastrum	4225	30	6	445.550	0.695	1.653	0.219	0.304
Ulmus pumila	4183	46	15	1024.465	1.006	2.708	0.356	0.399
Ailanthus altissima	3345	44	12	654.776	0.864	2.346	0.314	0.350
Prunus avium	2938	24	6	290.794	0.655	1.655	0.220	0.291
Platanus x hybrida	2799	49	17	1011.862	0.990	2.853	0.368	0.405
Prunus spp	2594	20.5	5	209.672	0.607	1.431	0.191	0.276
Pyrus calleryana	2585	13	5	148.155	0.584	1.509	0.199	0.277
Ulmus carpinifolia	2566	47	15	808.942	0.919	2.667	0.356	0.381
Tilia americana	2557	43	13	721.477	0.894	2.449	0.324	0.361

Table 12: Most abundant species presence in the city of Milan with number of individuals, diameter at breast height (dbh), height and average pollutant deposition. The dbh, heights, NPP and pollutants are species-specific average values, and have no correlation with the abundance or number of individuals of that species in the city. *(From Zappitelli et al., 2023).*

Species	Number of Individuals	d.b.h.	Height	Mean NPP	Mean O_3	Mean PM_{10}	Mean $PM_{2.5}$	Mean NO_2
		cm	m	g CO_2 m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}	g m^2 y^{-1}
Celtis australis	8135	23.5	9	337.942	0.908	0.496	0.102	0.323
Platanus x acerifolia	6485	40	19.5	1033.446	1.252	0.862	0.178	0.451
Tilia x intermedia	4857	23.5	9	492.788	0.978	0.638	0.131	0.353
Fraxinus excelsior	4031	7.5	5	244.090	0.845	0.448	0.093	0.318
Acer campestre	3819	7.5	5	216.256	0.810	0.449	0.093	0.313
Aesculus hippocastanum	3270	23.5	9	468.190	0.940	0.651	0.133	0.361
Populus nigra	3177	80.43	23.94	862.381	1.211	0.965	0.200	0.477
Quercus robur	2361	7.5	5	302.873	0.851	0.451	0.093	0.324
Populus alba	2222	14	5	221.588	0.797	0.451	0.093	0.326
Fraxinus angustifolia	1817	7.5	5	224.888	0.775	0.462	0.095	0.326
Cercis siliquastrum	1643	7.5	5	223.370	0.787	0.458	0.094	0.316
Styphonolobium japonicum	1641	23.5	9	335.382	0.876	0.782	0.162	0.358
Tilia platyphyllos	1567	7.5	5	182.188	0.772	0.456	0.094	0.315
Cedrus deodara	1461	59	19.5	2594.612	1.251	11.476	2.526	0.463
Tilia cordata	1401	7.5	5	184.095	0.772	0.452	0.093	0.308
Robinia pseudoacacia	1345	17	9	386.954	0.966	0.670	0.139	0.358
Acer negundo	1189	23.5	9	529.712	0.934	0.670	0.138	0.372
Fraxinus ornus	1075	7.5	5	222.708	0.785	0.462	0.095	0.316
Carpinus betulus	1058	12.5	9	447.757	0.942	0.716	0.148	0.385
Acer pseudoplatanus	899	23.5	9	541.095	0.975	0.657	0.135	0.357
Cupressus sempervirens	880	23.5	9	2244.193	1.432	6.170	1.359	0.453
Quercus ilex	873	7.5	5	155.846	0.695	2.588	0.558	0.268
Ulmus carpinifolia	860	23.5	9	521.669	0.962	0.681	0.141	0.365
Acer platanoides	770	7.5	5	324.786	0.816	0.452	0.093	0.325
Morus nigra	707	7.5	5	196.303	0.681	0.455	0.094	0.275
Cedrus atlantica	706	59	19.5	2581.468	1.336	11.630	2.548	0.479
Pinus pinea	647	40	9	1071.029	1.083	5.614	1.229	0.384
Prunus avium	626	14	5	141.520	0.751	0.474	0.098	0.318
Pinus nigra	586	23.5	9	979.277	0.908	5.062	1.109	0.351
Acer saccharinum	471	12.5	9	441.598	0.927	0.679	0.140	0.389

Table 13: Most abundant species presence in the city of Bologna with number of individuals, diameter at breast height (dbh), height and average pollutant deposition. The dbh, heights, NPP and pollutants are species-specific average values, and have no correlation with the abundance or number of individuals of that species in the city. *(From Zappitelli et al., 2023).*

4.3.3 Differences in pollutant sequestration depending on macrotype

The Radar graphs (Figure 38) show the normalized overall sequestration of pollutants by each macrotype in the two cities. Regarding the relative abundance of the various macrotypes, the reference is the list provided by the municipalities, excluding the species that AIRTREE does not have in its inventory. The most present macrotype is Deciduous Broadleaf (240965 and 59489 individuals in Milan and Bologna, respectively), followed by Evergreen Needleleaf (15059 and 5343 in Milan and Bologna, respectively) and finally Evergreen Broadleaf (4263 and 1189 in Milan and Bologna, respectively).

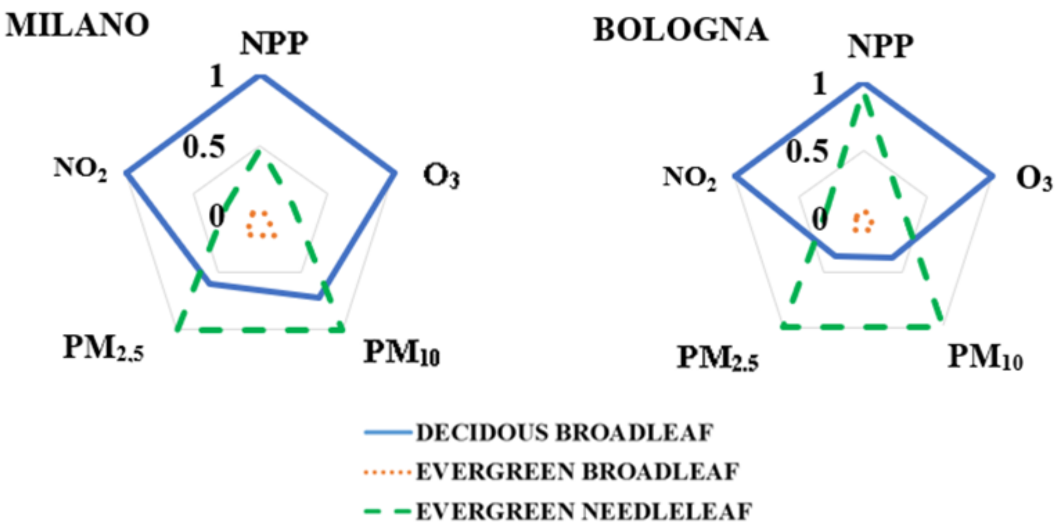


Figure 38: Radar Plot showing the sum performance of the different macrotype in sequestrating pollutants in the city of Milano and Bologna. The values (expressed in g m^2 y^{-1} for O_3, NO_2 and particulate matter; and g CO_2 m^2 y^{-1} for NPP) were normalized for the maximum value. The maximum values (which equals 1 in the normalization) are for NPP: 3953 kg m^2 y^{-1}; O_3 5677 g m^2 y^{-1}; PM_{10}: 20,491 g m^2 y^{-1}; $PM_{2.5}$: 3261 g m^2 y^{-1}; NO_2: 2358 g m^2 y^{-1}. *(From Zappitelli et al, 2023).*

The ability of plants to reduce pollutants depends on the deposition rate and the pollutant capture efficiency, which are species-specific parameters (Beckett et al., 2000). For example, a high sequestration of gaseous pollutants is provided by trees with large leaves, as the greater the number of stomata, the greater the potential to absorb gaseous pollutants (Beckett et al., 2000). For this reason, in both cities, the evergreen needleleaf trees display a reduced capacity for CO_2 sequestration (0.483 and 0.946 in Milan and Bologna, respectively), and this numerical difference could be due to the relative abundance of conifers, the latter previously described as a poor CO_2 sequestration capacity (Ma et al., 2022). On the other side, our results confirm previous findings showing that conifers are more efficient at intercepting particulate matter than broadleaf trees due to their greater leaf area and structural complexity (Chen et al., 2017). It was found that conifers have a very high sequestration capacity for dust (PM_{10} and $PM_{2.5}$), and a low impact on sequestration capacity for NO_2 (0.250 in Milan and 0.274 in Bologna) and O_3 (0.241 in Milan and 0.299 in Bologna).

Vice versa, broadleaf trees recorded the maximum sequestration for NO_2, O_3 and CO_2, although in Milan they are also efficient to sequestrate PM_{10} (0.717).

Evergreen broadleaf trees behave very similarly in both municipalities and, because of their limited abundance in both cities, their contribution on overall sequestration resulted to be negligible.

4.3.4 Sequestration of Carbon Diaxide and Particulate Matter at Municipality scale

The generation of georeferenced maps allowed to make important considerations about city pollution and the distribution of trees in the city. Figure 39 A, B shows the percent fraction of the city occupied by trees (1 km resolution) and their distribution over the city.

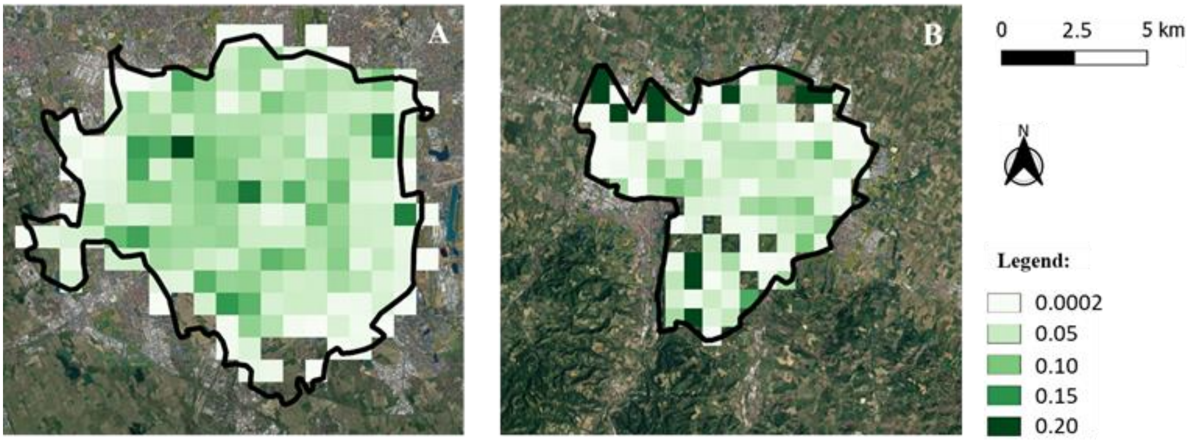


Figure 39 (A, B): Fraction of tree occupancy per square kilometer in the city of Milan (A) and in the city of Bologna (B) for the year 2015. Each pixel is the size of a square kilometer. In the legend, the fraction of green goes from the lightest color (less presence of green) to the darkest color (more presence of green). *(From Zappitelli et al., 2023).*

Concerning Milan, it can be noted how the distribution of green spaces is homogeneous towards the city center (this is also due to the large parks of the city, where there is up to 20% green coverage in a single pixel), while in the suburbs (with a green coverage of only 0.02%), less treed areas are due to agricultural fields (Milan boasts a significant agricultural territory) or industrial zones. In the case of Bologna, on the other hand, vegetation is homogeneous in the city center (remaining around a percentage of green presence between 10% and 15%), but abundant in the suburban areas surrounding the city.

As for Milan, for the estimation and updating of the regional inventory of atmospheric emissions, the INventario EMissioni Aria (IN.EM.AR) system has been used for years. The IN.EM.AR developed under the Regional Air Quality Plan (PRQA) and managed, since 2003, by the U.O. "Atmospheric Modeling and Inventories of ARPA Lombardia." (https://www.arpalombardia.it/) and is useful to chart the pollution levels of all pollutants in the city's air year by year.

Due to the higher-than-normal summer temperatures in 2015, there was a large accumulation of PM in Milan during that period. The strong solar radiation and the greenhouse effect due to the status of an urban metropolis also facilitated the formation of O_3 (Paoletti, 2009). The winter, as usual, favored the use of heaters and cars, and thus an increase in CO_2 and NO_2 emissions (https://www.arpalombardia.it/).

As far as Bologna is concerned, according to ARPA data (https://www.arpae.it/), the year 2015 was characterized by an overall worsening of air quality compared to the previous year. This worsening is said to be attributable to the marked differences between the meteorological conditions that occurred in the two years, with more frequent situations favorable to the accumulation of particulate material in autumn/winter 2015 and more days characterized by good weather, ideal for the formation of O_3 in summer 2015 (Nali et al., 2001).

ARPA Emilia Romagna also adopts the INEMAR inventory, the two emission inventories were therefore constructed applying the same methodology.

In Bologna, moreover, the winter-fall seasons of 2015 had 68% of days particularly favorable to the accumulation of atmospheric particulate matter and was therefore a year with particularly critical weather conditions (high percentage of days without rain and poor ventilation, high use of heating and cars), while in the period April-September, the percentage of days with maximum temperatures above 29 °C was over 35%, conditions promoting O_3 formation.

The average background concentration of PM and O_3 in the city of Bologna depends, in part, on the large-scale pollution typical of the Po Valley (Raffaelli et al., 2020). On a more local scale, road traffic and non-industrial combustion (heating) are the main sources of emissions related to direct dust pollution (PM_{10} and $PM_{2.5}$), followed by transport of other mobile sources (e.g., aircraft) and industry. Nitrogen oxides (NO_x), which is also an important precursor to the formation of secondary particulate matter and O_3, are contributed by road transport and other mobile sources, as well as combustion in industry and energy production (11% and 9% respectively). In contrast, 32% of CO_2 is produced by road transport, both light and heavy (https://www.arpae.it/).

Figure 40 A–D shows the CO_2 and PM_{10} sequestrations in the two cities. In particular, Figure 40 A and B show the NPP (t CO_2 km^2) of Milan and Bologna, respectively, during the year 2015. Figure 40 C and D show the PM_{10} (kg km^2) during the same year.

In line with tree coverage, PM deposition and CO_2 sequestration is highest in the center of Milan (reaching values as high as 800 Kg PM_{10} Km^2 y^{-1} and 500 t CO_2 Km^2 y^{-1}), which may be justified by the presence of urban parks in those areas, but also by the higher emissions in winter for combustion processes (Quah and Roth, 2012).

Assuming an average vehicle circulation of 10,000 Km y^{-1} and an emission of CO_2 of 108.6 g y^{-1}, we estimate that trees in Milan are able to sequester the CO_2 emission of 460 vehicles per year (0.06% of the car fleet circulating in Milan in one year). For PM, on the other hand, assuming an emission of 0.005 g y^{-1}, we estimate that trees in Milan abate the emissions of 16,000 vehicles per year (2.28% of the car fleet circulating in Milan in one year).

In Bologna, it was observed an almost opposite trend, with PM deposition and NPP higher in the treed suburban areas (700 Kg PM_{10} Km^2 y^{-1} and 550 t CO_2 Km^2 y^{-1}) than in the city center (around 10 Kg PM_{10} Km^2 y^{-1} and 150 t CO_2 Km^2 y^{-1}). This effect is due to the highly concentrated peri-urban greenery on the borders of this municipality.

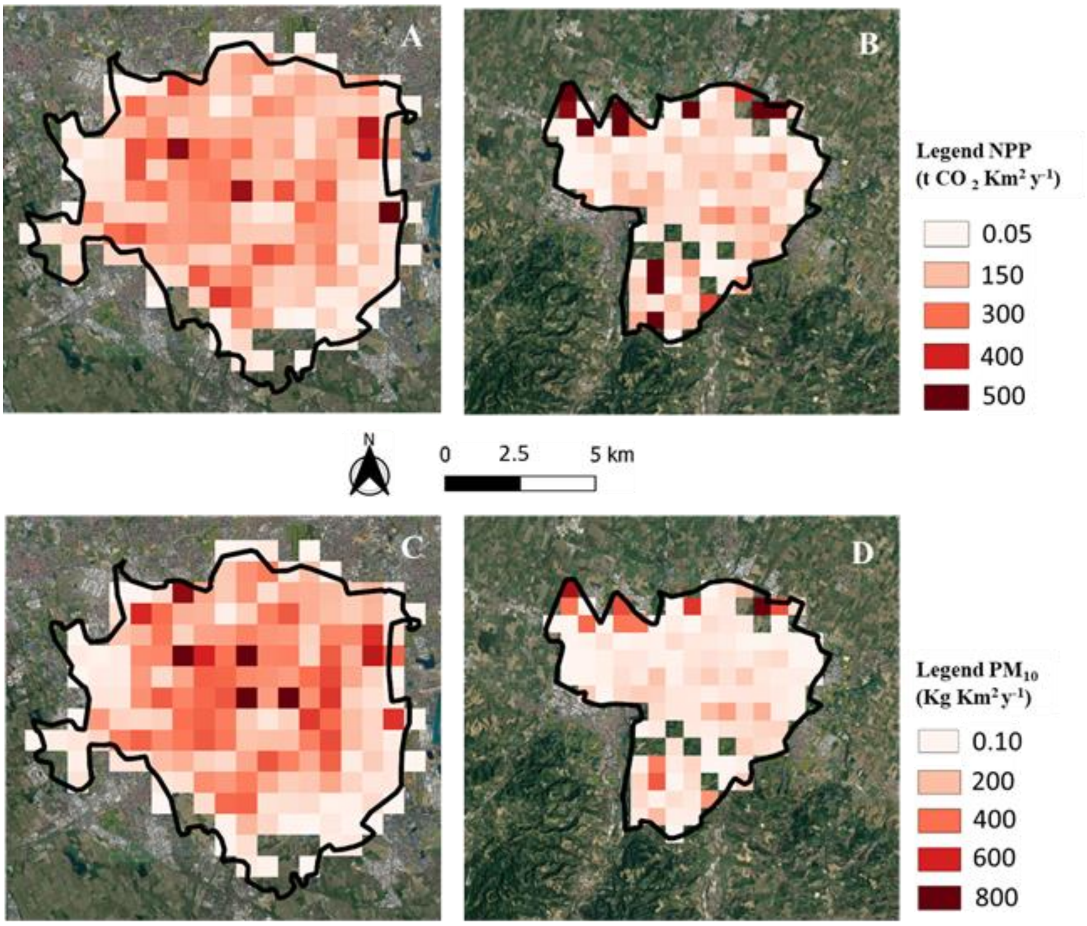


Figure 40 (**A**–**D**): CO_2 and PM_{10} sequestrations in the two Municipalities of Milano and Bologna. (**A**) shows Milan's NPP expressed in tonnes of CO_2 captured per pixel (per square kilometer) during the year 2015. (**B**) shows the same but for Bologna. The legend of the numerical values can be consulted alongside these two figures. (**C**) shows instead the PM_{10} particulate sequestration per pixel (per square kilometer) during the year 2015. (**D**) shows the same for Bologna. The legend of the numerical values can be consulted alongside these two figures. *(From Zappitelli et al., 2023).*

5. Conclusions and Future Perspectives

Throughout Europe, the Mediterranean region is facing many challenges, some of which can be addressed by nature-based solutions such as urban forests and green spaces. However, there is still poor literature concerning detailed studies on tree composition and ES delivery in Mediterranean countries (Krajter Ostoić et al., 2018), with research more related to peri-urban forests (Barbati et al, 2014; Bentsen et al., 2010), or parks in the city, and few works related to the beneficial action of street trees and the effect of general greenery in the city (Gill et al., 2007). People's health and well-being are closely linked to the state of the environment (EEA, 2019). A good quality natural environment meets basic needs for clean air, especially in cities where sources of pollution are abundant (Gualtieri et al., 2020).

Most of the urban green areas, in the past, were designed and created for social and economic purposes, but it is very important to also take into consideration the environmental benefits that these green lungs can bring to city contexts. Knowledge of ES is essential, especially in the planning stages, to justify a choice and a consequent investment, which generally falls on public accounts. For example, carbon sequestration is very important, because urban green area managers are considering whether tree planting projects in urban areas can be financed through the carbon market, especially since it is now an internationally accredited market. To this end, such an environmental challenge does not only concern individual cities, but is reflected on a larger scale, as stressed in the European Green Deal of the EU committed to plant 3 billion trees by 2030. This will increase forest area and tree cover in the EU, improve the resilience of forests and their role in reversing biodiversity loss, and mitigate and help adapt to climate change (Nespor, 2016; di Cadore, 2016) (https://unfccc.int/).

These trees could also be planted in urban or peri-urban forests, and even if urban and peri-urban forests represent a very minor quota of available forests to mitigate climate changes, at the same time they are close to the sources of city pollution and can therefore mitigate the effect of polluting emissions (Romano, 2020). It was also seen in this study that the trees currently chosen for street trees are not the most sequestering. This is justified by the choices also made by tree size. In any case, this does not detract from the fact that the species in question could be chosen for the urban or peri-urban park closest to the source of urban pollution.

Above all, species and species associations must be chosen consistently and carefully to maximize Ecosystem Services and minimize Disservices. It is important to give space to species associations and species diversity, especially with regard to trees that are typically found in the urban fabric. The obtained results can have important implications for future forest management, and to raise awareness on the high ecological and economic value of RES. With this study, light was shed on important ecosystem services provided by urban trees. In this study, some of the ESs produced by species in urban parks and roads were quantified, highlighting differences in their performance based on their biometric characteristics and ecophysiological properties. However, careful attention should be paid while selecting new species to plant in urban areas, since some of these species emit BVOCs, previously described as Ecosystem Disservice when they are blended in the atmosphere with precursors such as NO_x to form O_3 and aerosols (de la Paz et al., 2022).

A limitation of this study is in fact that the AIRTREE model does not contain a parameterization for the calculation of the BVOCs emitted by tree species, but it is aimed at expanding the model in the future also towards this addition.

In addition to BVOCs, it is planned to integrate other important Ecosystemic Disservices, such as allergen emissions, which are very important in the city due to allergic factors in the population, and thus affect whether or not the plant is accepted by citizens.

Modifying future tree planting to favor lower-emitting species can further support municipality in abating atmospheric pollutants. On this end, research is being addressed to evaluating the potential negative impact of Biogenic emissions in our target cities. One tool to help in this is the VEG-GAP portal (https://www.lifeveggap.eu/), an information platform that evaluates the multiple services provided by plant ecosystems in an integrated manner, through multi-scale and multi-pollutant approaches that resemble the real world as closely as possible.

The future perspective is the application of the method learnt in the period abroad in ARGANS of high-resolution maps, on at least one of the target cities between Milan and Bologna, to test the differences between the method applied in this PhD and the one applied in AIRFRESH. It is planned to produce a paper comparing the method I used in this study and the one learned in AIRFRESH, on the same cities of Milan and Bologna, of which I was able to acquire the very high-resolution maps.

Awaiting the publication by Dr Coulibalì of the method she developed for the construction of these very high-resolution maps, there is also already the prospect of a publication on the data obtained by running AIRTREE on Florence and Aix-an-Provence (that is, part of the work I carried out during my stage in France).

After the defense of my PhD, I also intend to apply the method used in this doctorate to the city of Madrid (also part of the VEG-GAP project), something that could not happen before, as the AIRTREE database needs to be expanded even more than it already has been. This is because Madrid is home to a large number of exotic and allochthonous species.

Furthermore, in future research, it is planned to consider the weight that each Ecosystem Service or Ecosystem Disservice has in relation to another, also on an ecological level.

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