UNIVERSITÀ DI PARMA

UNIVERSITA’ DEGLI STUDI DI PARMA

DOTTORATO DI RICERCA IN

“BIOLOGIA EVOLUZIONISTICA ED ECOLOGIA”

CICLO XXXIV

Urban green and ecosystem services: not just a tree

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Anni Accademici 2018/2019 – 2021/2022

Il dottorato in Biologia Evoluzionistica ed Ecologia – XXXIV ciclo è in convenzione tra le Università di Ferrara, Firenze e Parma (sede amministrativa)

To my parents, Marinela and Liviu Muresan, who have always supported me even if they did not share my decisions.

Ai miei genitori, Marinela e Liviu Muresan, che mi hanno sempre appoggiato anche quando non condividevano le mie decisioni.

Abstract

Natural and anthropogenic ecosystems are highly touched by ongoing climate change. In particular, urban ecosystems are more vulnerable due to the demographic density and natural structure lack. Environmental and human well-being in urban ecosystem depend on mitigating the harmful effect given by climate change and urbanization. Nature based solutions (NBS), particularly urban green infrastructures (UGI), are well known to provide ecosystem services that improve the ecological quality of urban areas. UGI regulate the environmental conditions leading to significantly mitigated consequences of urbanization. This work aimed to analyse the ecosystem services at three different scales (macro-, meso- and micro-scale), using different models, in order to understand which strategies are more efficient to improve ecosystem services provision.

The results showed a complex interaction between tree traits and background environmental condition for the provision of ecosystem services in urban ecosystems. Moreover, foresting actions increase the ecosystem services provision. Lastly, functional traits play a key role of ecosystem services provision, especially for $PM_{2.5}$ removal. Furthermore, considering the functional traits, models like i-Tree can reduce its limitations.

From an urban management point of view, the results emphasize the importance to consider the environmental context when urban green areas management is performed. Moreover, considering the specificity of tree as well as the ecosystem services demand is crucial. Finally, diverse tree composition would provide vast spectrum of ecosystem services.

Keywords: Biodiversity, Ecosystem Services, Urban Green Infrastructure, Ecosystem Services Modelling, Tree Traits

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Introduction

Biodiversity loss, climate change intensification and nature requalification for human well-being are some of the problems that decision makers have to face. Considering the link between urbanization and the environmental problems is mandatory. For example, urban ecosystem is the major source of greenhouse gases and air pollution. Moreover, cities are one of the main responsibles of land use – land cover (LULC) variation due to the worldwide population increase and migration to urban areas. The mitigation of urbanization negative impacts are very important for the environmental quality and human health and welfare. This thesis aims to analyse the ecosystem services (ES) provided by urban green infrastructure (UGI) at different scales in order to understand how the use of these nature-based solutions can be applied to increase the ecosystem services. Moreover, this study intents to give some inputs in terms of urban management and urban restoration.

Biodiversity and Ecosystem Services

Edward O. Wilson, during the United Nation Convention on Biological Diversity (1992), defined biodiversity as "*the variability among living organisms from all sources […] and the ecological complexes of which they are part*". This definition covers the genetic diversity (variation at genetic and population levels), species diversity (variability between species) and the ecosystem diversity (variability at landscape and regional level). It includes also the complex relation between species and ecosystem, which means, that it recognizes the ecological interaction as both being a cause and consequence of biodiversity.

Biodiversity is known to be the root of ecosystem dynamics and functions. Particularly, biodiversity influences the processes of an ecosystem such as primary production, nutrient cycling, decomposition, etc. Additionally, the functional diversity (FD) is another biodiversity component that plays an important role in ecosystem functions. FD quantifies the value and range of organismal traits that affect species performance and so the ecosystem functioning (Díaz and Cabido 2001).

The actions on ecosystem functions manipulate the ecosystem services (ES) (Balvanera et al. 2006; Cardinale et al. 2012; Costanza et al. 2007). ES are the benefits provided by ecosystems to humans (Millennium Ecosystem Assessment 2005a) and they are responsible to maintain and promote human health and well-being. ES are classified in provisioning services, such as food, timber, fibre, but also supporting services like nutrient cycling or soil formation and regulating services that involve water and air quality, climate, etc. Lastly, there are also cultural services such as recreational and aesthetic benefit (Figure 1)

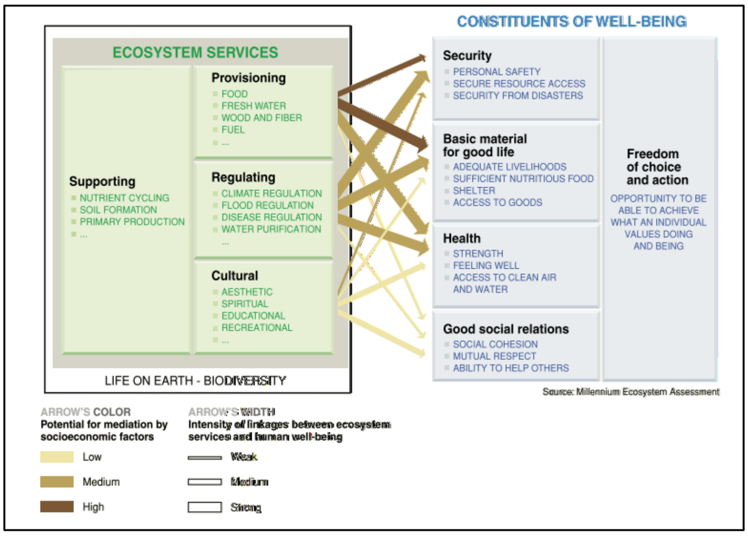

Figure 1 Linkages among ES and human well-being (MAE, 2005)

There are different points of view when it comes to analyse the biodiversity – ecosystem services interaction. On one hand, biodiversity and ES might be equal. This means that managing one will affect the other. So, if biodiversity increases, ES increase as well and, at the same time, if there is a biodiversity loss, it will lead to an ES provision decline (Perrings et al. 2010). On the other hand, biodiversity is an ES, whereby organisms have an unquantifiable value. However, the ecosystem complexity consists of biotic and abiotic interactions. These interactions, as well as biodiversity, influence the quality and quantity of ES (Mace, Norris, and Fitter 2012). The importance of biodiversity within ES can be read by agreeing that there are different relations at different levels. Biodiversity can be a regulator if we consider for example the nutrient cycle dynamics in which the composition of the biological community in the soil plays a key role. Moreover, the resilience to habitat changes, for instance, depends on the biological community, too (Mori, Furukawa, and Sasaki 2013). Biodiversity is also a good. Indeed, there are policies for its conservation since it has cultural and recreational value (i.e. birdwatching).

Biodiversity loss is one of the main problems nowadays and it is directly related to habitat change (ie. land use and land change, watercourse modification, etc.), overexploitation, pollution, climate change (Hooper et al. 2012; Mantyka-Pringle et al. 2015). Moreover, the worldwide population increase is another important driver to biodiversity loss. For example, demographic increment intensifies food production demand, which means an increase of fertilizer use and pest control, as well as water withdrawal for irrigation. In addition, due to demographic increase and people migration into cities, urban ecosystems are continuously expanding. Cities development accentuates the urbanization effect such as changes in land usage by soil sealing and fragmentation, microclimatic modifications, growth of water and air pollution, etc. All these urbanization negative impacts affect ES provision and, therefore, affect citizen health and well-being.

Climate Change

Climate change is occurring globally and it is another important challenge for decision makers and scientists. This concept is often interchanged with the term “global warming” although global warming represents only a part of the whole problem. Indeed, “climate change”, refers to long-term changes at local, regional or global climates. Climate is dynamic so the climate change is a natural process but since 19th century, it has speeded-up and it does not occurs anymore in geological period time but quicker. The main event that started this climate change acceleration is the intensification of human activity after the Industrial Revolution. This has produced a large amount of carbon dioxide and other greenhouse gases that caused changes on Earth climate. Greenhouse gases are compounds present in the atmosphere and they are responsible of the greenhouse effect. The temperature on Earth depends on the equilibrium of the entering and leaving energy on the planet. Greenhouse gases create a sort of “blanket” that interferes with the exit of heat towards space and provokes a warm growth in the tropospheric part. The high concentration of anthropogenic greenhouse gases in the atmosphere intensifies the greenhouse effect, leading to Earth climate modifications. Climate change affects the environment, economy and human health. The main impacts of climate change are temperature and rainfall patterns variation with extreme temperature and precipitation, in terms of duration, intensification and frequency. Warmer temperature promotes heat waves that affect health risk especially for young children and elderly people, while rainfall intensification affects water quality, the water supplies and stream flow. Moreover, the increase of sea level, ocean temperature and ocean acidification, represent other impacts of climate change. Ocean temperature increases, for example, the metabolic rate of organisms affecting the reproductive period (Lamont and Fujisaki 2014; O’Connor et al. 2007), while the acidification influences the carbonate equilibrium (Guinotte and Fabry 2008; Koch et al. 2013; Mollica et al. 2018). Lastly, climate change is responsible also for ecosystem modifications by modifying the habitat, animal or plant behaviour (i.e. bird migration or leaf growth). All these impacts on ecosystem have consequences on the provided ecosystem services and biodiversity (Díaz et al. 2019). According to MAE (2005), due to the global changes caused by human activity, over 60% of services provided by biotic system decreased (Millennium Ecosystem Assessment

2005b). ES can also be used as climate change mitigation by increasing the ES importance and management in order to respond to climate change consequences (Munang et al. 2013).

European Environmental Agency 2017 (EEA 2016) mapped the observed and projected climate changes and their effects in Europe based on the biogeographic regions (Figure 2). As the EEA emphasizes, the entire European continent is highly touched, even if the Mediterranean region seems to be the most touched and the most to suffer from climate change.

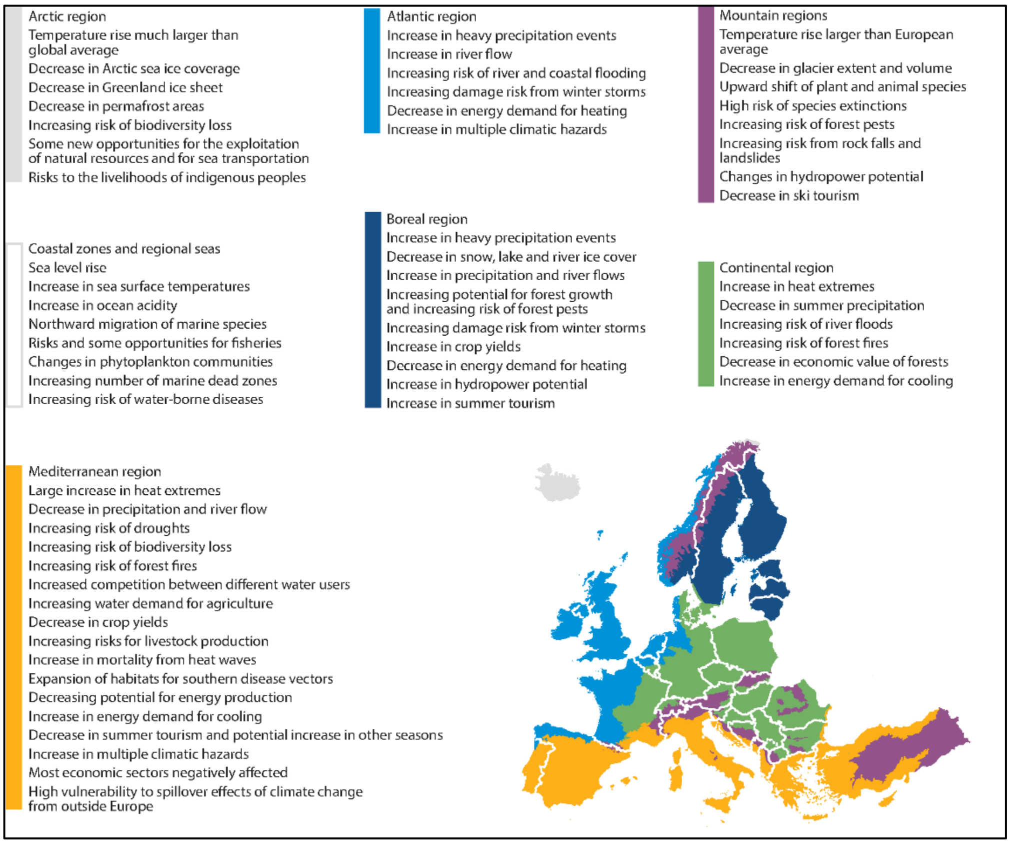

Figure 2 Climate change observed and projected map (EEA, 2016)

Anyways, all natural and anthropogenic ecosystems suffer the climate change effect in all regions.

Natural Ecosystems and climate change

Climate change increases the vulnerability of ecosystem to extreme events and at the same time, their natural capacity to modulate these events is reduced. The influences of climate change on natural ecosystems and the response of the ecosystems depend on the ecosystem we consider. For example, global warming may drive species to migrate to higher elevation (and latitude) where temperatures are more suitable to their survival. This effect is commonly seen in mountain ecosystems. Indeed, tree line shift upwards, as well as pasture area expansion are some of the main effects on mountain ecosystems (Parmesan 2006). In terrestrial ecosystem, earlier greening and leaf emergence, but also modification of birds' migration and reproduction are consequences of climate change. In marine ecosystems, phytoplankton and fish migrate towards the poles. Furthermore, coral reef decline and blanching is observed in tropical waters while temperate seas are exposed to the so-called “tropicalization effect”. The climate change effects on wetland instead, regard variations in hydrology field, such as decrease of recharge of some floodplain aquifers. Moreover, sea level increase cause saltwater intrusion phenomenon along the alluvial aquifer (Siaka et al. 2018). Another aspect of climate change is the onset of biogeochemical changes; due to increase of decomposition and primary production, a shift occurs from carbon sink to carbon source (Kayranli et al. 2010). Here are presented two studies showing the impact of climate change on two natural wetlands and how these water bodies respond to such pressures. The first example, Muresan et al. 2019, regards small coastal wetlands (pools) and how macroinvertebrates respond to the phenomenon of salty water intrusion (full paper in Appendix 1). Small wetlands sustain a high variety of animals and plants. Therefore, they are considered biodiversity hotspots. These ecosystems, despite their high diversity level, are not very studied. Macroinvertebrates have a key role in all the ecosystem processes and they are excellent functional and taxonomical bioindicators. The results emphasize that salinity increment causes functional and taxonomical diversity loss.

The second example, Gaglio et al. 2022, concerns an inland freshwater wetland and the vegetation mitigation capacity to climate change (full paper Appendix 2). This wetland has the water storage function so during the rainfall period this wetland is used against flooding, while during the dry period it is used as water source for the agriculture. This hydrological variation, amounted to the effect of the invasive species (i.e. common carp), generates vegetation loss. The results show a negative impact on water vegetation and a consequence on the carbon storage and sequestration capacity.

Urban Ecosystems

Urban ecosystems are socio-ecological systems where most people live (Maes J et al. 2016). They are essentially artificial and built by humans to satisfy their needs but, however, they contain also different ecosystems such as forests, agricultural areas, aquatic ecosystem, etc. Cities are eutrophic ecosystems characterized by a high-energy input demand necessary to maintain themselves (Odum 1977). The natural ecosystems are characterized by a circular metabolism. Cities, instead, are generally characterized by linear metabolism (Figure 3a), which means that the energy coming from the outside (i.e. energy and materials) is brought into the city, transformed, thrown out as waste, and not recycled. Indeed, urban areas are recognized as consumption and waste generation hotspot (Adel and Magdy 2016).

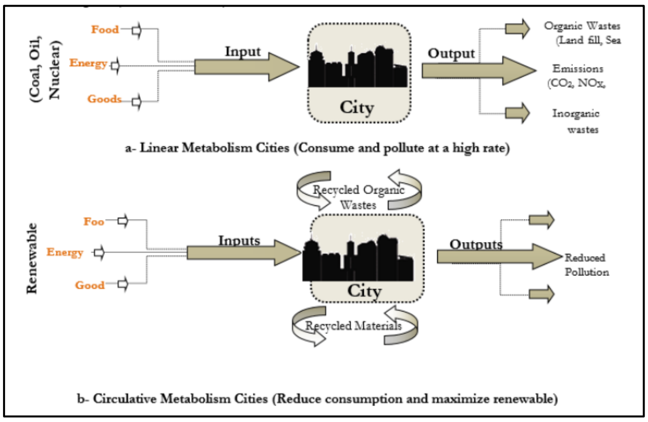

Figure 3 Urban area metabolism (Adel & Magdy, 2016)

The urban waste is nothing more than pollutants such as heat, greenhouse gas, wastewater, etc. that have negative impacts on human well-being and environmental quality.

Urbanization is increasing due to the demographic growth and urban density expansion. According to United Nations 2018, 55% of the population lives in cities with the highest urbanization levels in Northern and Latin America (>80% each) followed by Europe with 74% of people living in urban areas. Cities consume about 78% of the world energy and it is responsible of 60% of the greenhouse gas emissions. Considering that urban ecosystems represent 3% of the Earth surface, the footprint of this ecosystems is remarkable. Urbanization growth leads to the rise of pollutants output and air pollution is the main issue related to urban ecosystems. According to Word Health Organization (WHO) air pollution is the major environmental risk to health, in particular $PM_{2.5}$. High concentrations of particulate matter (PM_{10} and $PM_{2.5}$), ozone (O_3) and nitrogen dioxide (NO_2) cause cardiovascular and respiratory problems. In particular, particulate matter is related to mortality and morbidity increase and it is responsible of cardiorespiratory diseases and tumours, while ozone leads to asthma and reduces lung function. Nitrogen dioxide health effect is associated to long term exposure. Heat increase is other problem related with human settlements that affect health and it is responsible of premature death during summer time, especially for elder people. Moreover, there is a negative correlation between urbanization and water quality (Nagy et al. 2012).

Cities are the major author in global climate change, but urbanization amounted to climate change accentuate the negative impacts. In other words, climate change widely affects urban life and, at the same time, urban areas are responsible of greenhouse gas production and emission. Urban areas are vulnerable because of buildings density and sealed surfaces, that lead to increased local temperature and rainfall flooding.

Global decision makers encourage the "climate change resilience" concept as a mixture between mitigation, adaptation and sustainable development that bring co-benefits. This context is gaining importance within current political agreements (i.e. Paris Agreement, 2015). Indeed, increasing cities resilience will increment environmental quality responsible to have the highest level of ecosystem function provided by urban ecosystem and therefore maximize ecosystem services provision as McPhearson et al. (2015) shows (Figure 4).

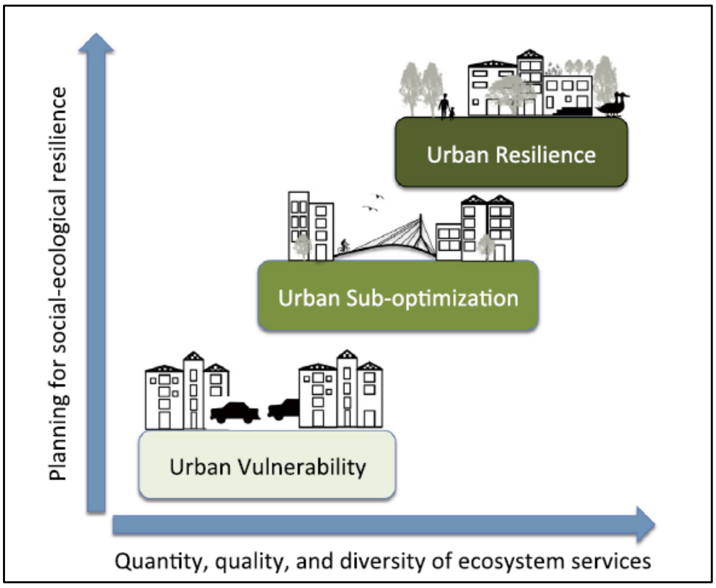

Figure 4 Relationship between social-ecological resilience and ecosystem services (McPhearson et al., 2015)

Nature based solutions, in particular urban green infrastructure, as we will see below, play a key role.

Nature Based Solutions and Urban Green Infrastructure

Anthropogenic pressures are the main cause to all the environmental problems humanity has to deal with nowadays. The answer to mitigate these pressures is hidden in nature. Nature Based Solutions (NBS) are defined by the IUCN (2016) as "*actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits*". European Commission, instead, defines them as "*solutions that are inspired and supported by nature, which are cost-effective, simultaneously provide environmental, social and economic benefits and help build resilience. Such solutions bring more, and more diverse, nature and natural features and processes into cities, landscapes and seascapes, through locally adapted, resource-efficient and systemic interventions*". Even if the two definitions seem to be different, both suggest the use of nature processes to manage and restore the ecosystems designed to maximize environmental and human well-being benefits. For example, Griscom et al. (2017) showed how nature solutions can provide 37% of cost-effective CO_2 mitigation to keep warming below 2 °C by 2030. NBS adopt nature conservation norms and they are site-specific determined by the natural and cultural context. Moreover, NBS preserve biological and cultural diversity, ecosystem capacity to evolve and supply social benefits (Canedoli, Crocco, et al. 2018; Xie and Bulkeley 2020).

Eggermont et al. (2015) proposed three main types of NBS:

- Type 1 – solutions that improve the existing nature or preserve the ecosystems. This type is aimed to improve or maintain the ES provision range (i.e. maintaining costal wetland to reduce the risks related to extreme weather conditions)
- Type 2 – solutions that are set up on sustainable management or ecosystem restoration. This type of NBS generates sustainable management, which improves the provision of a selected ES.
- Type 3 – solutions that require creating new ecosystems. This type is an invasive NBS and it is usually associated to green and blue infrastructures.

Wide varieties of NBS are employed to deal with different environmental pressures. For urban ecosystems, which represent the main anthropogenic pressure source, green infrastructures are the most used. Green infrastructure (GI) is widely recognized as a tool for the ecological conditions and improvement of human well-being. They are defined as *"a strategically planned network of high quality natural and semi-natural areas with other environmental features, which is designed and managed to deliver a wide range of ecosystem services and protect biodiversity in both rural and urban settings"* (European Commission 2013). Therefore, GI is a spatial network created to improve nature's ability to provide multiple important ecosystem goods and services. Urban forests, parks, botanical garden, tree alley, green roofs and walls and horticulture are some examples of urban green infrastructures (UGI).

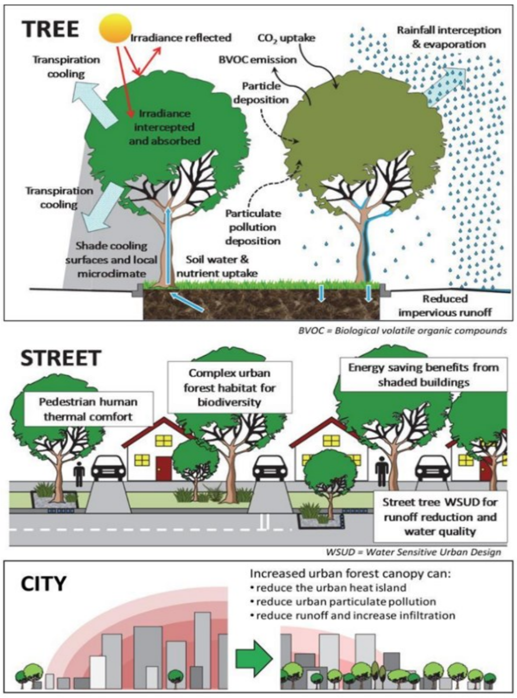

Figure 5 Urban green infrastructure ecosystem services (Livesley et al., 2016)

For definition, UGI provide numerous ES that support citizens' health and well-being. Indeed, it plays a key role by regulating the environmental conditions and it can mitigate the urbanization negative impact so that inhabitants' welfare is maintained (Duan et al. 2018; Elmqvist et al. 2015). As previously mentioned, soil sealing and fragmentation, microclimatic modification, water and air pollution growth are urbanization effects, that can be mitigated by using UGI.

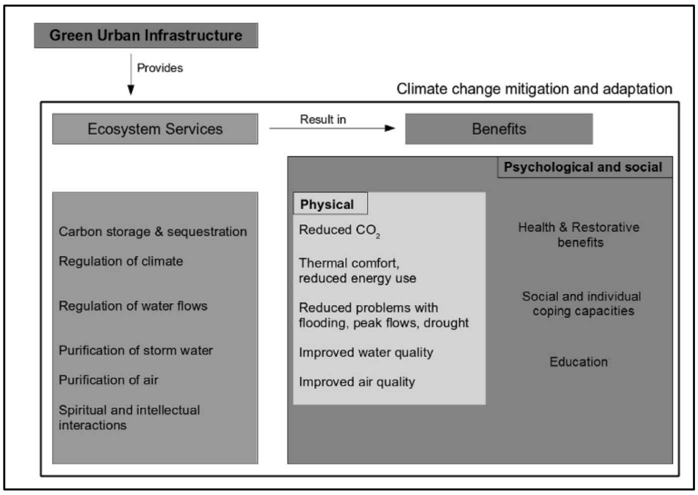

Figure 6 Urban green infrastructure and benefits (Demuzere et al., 2014)

The ecosystem services provided by UGI can be categorized as:

- Regulating Services (i.e. carbon storage, air quality regulation, temperature regulation and runoff regulation)
- Support Services (i.e. biodiversity support)
- Cultural Services (i.e. recreational)
- Provisioning Services (i.e. timber)

As mentioned above, air quality degradation caused by anthropogenic activities linked to urbanization generates high pressure on the environmental quality and human health. Indeed, according to WHO (2021), 7 million people die prematurely annualy and 80% of death related to $PM_{2.5}$ could be avoided if the air pollution levels were reduce to 5 µg/m^3 annual mean and 15 µg/m^3 24-hour mean (actual limit 10 µg/m^3 annual and 25 µg/m^3 24-hour mean). UGI provides significant air quality recovery. Depending on the air pollutant, GI remove differently the pollutants. For PM removal, mechanical force is used. Urban trees capture PM on leaf surface (i.e. Escobedo and Nowak 2009; Nowak et al. 2006; Selmi et al. 2016). Gaseous air pollutants instead are absorbed through the stomata of leaves (Anderson and Gough 2020; Manes et al. 2016).

Due to the climate change, extreme temperature and precipitation become an issue. This problem is more evident in urban areas. For instance, heat island effect is a consequence of higher temperature increase in urban areas rather than in outlying areas. The grey infrastructure (buildings, roads, etc.) absorbs the sun's heat during the day and radiates it back during the night. In the last decades, tropical nights and heat waves have become more frequent in particular in southern Europe (EEA 2016). Marando et al. (2019), showed how different GIs have different capacity to reduce the heat in urban area. Moreover, authors underlined a buffer zone where the cooling effect is still present.

A further aspect is the ability of UGI to mitigate climate change by removing CO_2 from the atmosphere. UGI uptakes CO_2 during the day via photosynthesis and discharge CO_2 during night respiration. In addition, it uptakes carbon also from below-ground biomass and soils (Demuzere et al. 2014; Velasco and Roth 2010). According to Nowak et al. (2013), carbon storage densities average 7,69 kg Cm^{-2} of tree cover while carbon sequestration average 0,28 kg Cm^{-2} of tree cover per year among 28 cities.

Water runoff is the water overflow due to soil sealing. Rainfall water cannot penetrate into the soil so there is a water surplus flowing. In a perspective in which extreme precipitations are frequent due to the climate change, runoff can cause damage to public (i.e. flooding) and it impacts also human safety. UGI contribute to the flood reduction by absorbing water through the roots (Hammond et al. 2015; Maragno et al. 2018; McPhearson, Hamstead, and Kremer 2014).

Further, UGI provide cultural services such as recreational (Chen, Qi, and Qiu 2018; Terkenli et al. 2017), but also support services by creating habitat for species (Canedoli et al. 2018; Capotorti et al. 2020).

Analysis at different scales

Macroscale: Landscape Scale

Landscape ecology is the proper scale for efficient planning. It integrates people and nature can be used for all ecosystem types. Landscape ecology is more and used to plan management and restoration of sustainable environments. Land mosaic regards land use spatial arrangement that integrates and sustain nature and people for both long-term benefit. Small spatial study and planning is common used but is not possible to ignore a wider scale like region. The cumulative effect of small spaces can be measurable and might be representing at regional scale. For urban ecosystem, this interconnection of land mosaic is represented by build space and green space. Urban areas can be considered as many different small spaces and the cumulative effect of each space might increment the environmental quality in an entire region and increase resilience.

Mesoscale: Ecosystem Scale

The ecosystem approach is “a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way”. This definition born in 2000 during the Convention on Biological Diversity. A similar concept is the Ecosystem based management that aims to maintain or restore ecosystem structure, composition function and services provision in order to archive sustainability (Millennium Ecosystem Assessment 2005b). So the ecosystem approach is based on the relationship between the biodiversity and ecosystem services as we already described before.

Microscale: Organism Scale

If we consider the organism scale of analysis, we have to take into account the biological characteristic and functional traits of the organism itself. Indeed, the functional diversity is a biodiversity quantifying aspect. The range and value of organismal trait have an effect on their efficiency and consequently on the ecosystem functioning (Díaz and Cabido 2001). Regarding UGI, for example, tree characteristics influence their performance in terms of pollutants removal (see Fusaro et al. 2017; Pace and Grote 2020; Zhang et al. 2017).

The pandemic has re-established the need to review the urban environmental quality and the importance of urban green restoration. In agreement to the financing provided by the Recovery Plan, it is necessary to comprehend the best way to restore the environmental quality in cities. In particular, it is crucial to understand the provided ecosystem services ad different scales, as well as how to assess ES at different scales. Moreover, it is fundamental to find out which tree species might help to increase ES provision.

The following three chapter, show which strategies can work better to increase the urban environmental quality using green areas. A three-scale approach was chosen in order to advise public decision makers about the possible actions.

Chapter 1 Macroscale Analysis

Parks and public gardens in general, represent a GI component and it is important to understand how to take action on them. Moreover, it is necessary to give guidelines for the creation of new parks and green areas. This work considers three parks from three cities located in Po River Plain (Northern Italy) and it assesses the ES provided by this parks. This analyse aims to understand how different tree species influence the ecosystem services supply. Moreover, the urban background is considered too. The results emphasize how difficult is to separate the specie-specific effect from the environmental background.

Regulating services provided by Urban Parks: how tree composition influence human health in Po River Plain (Italy)

Abstract

Urbanization phenomenon, because of human densification increase, lead to natural ecosystems modification that have negative impacts on the society.

Urban green infrastructures can significantly mitigate such effects by regulating environmental conditions in cities and their development is strongly supported by recent European and National policies.

The study aims to assess the regulating services provided by urban park of three cities located in the Po river Plain (Northern Italy), one of the most important industrial and agricultural areas, yet environmentally degraded region of Europe. Four ecosystem services (climate mitigation, flooding prevention, temperature regulation and air quality regulation) were modelled using iTree-eco tool,

then their relations with tree community composition were investigated by means of a functional analysis based on leaf and crown traits.

The analysis emphasized that the provision of regulating services is a result of a complex interaction between tree characteristics (i.e. tree shape, canopy width, foliage shape, etc.) and climatic context. Nonetheless, the results suggested some general relevant guidelines for environmental strategies. Flooding prevention and temperature regulation were provided by species presenting high leaf areas during rainiest and summer months, respectively. Evergreen species were found to be the most effective to mitigate air pollution, while monumental trees were important carbon pools for climate change mitigation.

Overall, the environmental management of urban areas urgently needs to the modelling of tree-environment interactions to inform decision processes concerning tree species selection, in order to ameliorate the quality of degraded urban areas. Lastly, the study brings to light the necessity to improve models in terms of PM2.5 removal performed that consider species-specific effects.

Introduction

Worldwide population considerably increase and people living in urban areas increment as well. Europe is the third region most urbanized with 74% of its population living in urban areas after Northern America with 82% of population and Latin America and the Caribbean with 81% (United Nations, 2018). This urbanization development lead to land use and land cover (LULC) variation that induce modifications in terms of microclimate and hydrological conditions. Moreover, environmental pollution is another important impact of urban areas especially in terms of air pollution (de Souza et al., 2016; Intergovernmental Panel on Climate Change, 2014; Jiang et al., 2015; Nowak et al., 2006). The negative effects of urbanization and the mitigation of this negative effects influence the environmental health and citizen well-being. Urban green infrastructure (UGI) plays a key role by regulating the environmental conditions and it can mitigate the negative impact. Indeed, it is known that UGI provides numerous ecosystem services (ES) that supports citizens' health and well-being (Duan et al., 2018; Elmqvist et al., 2015; Su et al., 2010). The predominant ES provided by UGI are regulating and cultural services. UGIs are responsible of air purification, temperature and storm water regulation and noise mitigation as well. In addition to these regulating services, UGIs provide cultural services such as social development and connections, educational and recreation. Moreover, UGI improves for example mental health (WHO, 2016).

Since the main impact of urbanization and cities' progress is LULC variation, it has an important impact on biodiversity. Habitat alteration and fragmentation induced by urbanization leads to biodiversity loss. Biodiversity support ecosystem functioning (i.e. primary production) and this relationship influence and support the ecosystem services provided (Balvanera et al., 2006; Cardinale et al., 2012; Costanza et al., 2007). Functional diversity is a biodiversity component that plays an important role to support ecosystem functioning. Indeed, it promotes ecosystem stability, dynamics, productivity and other ecosystem functioning aspects.

Ecosystem services can be read as multitude functions of ecosystem essential for the human well-being (Kremen, 2005). Moreover, Millennium Ecosystem Assessment highlighted the positive relation between ecosystem services and biodiversity that means that if one increases the other increase as well (Millennium Ecosystem Assessment, 2005; Perrings et al., 2010). Urbanization, in addition to the negative impacts above, is the main causes linked to biodiversity loss (Díaz et al., 2019; McKinney, 2006).

Considering the new policies in terms of urban environmental quality and the importance of urban green restoration as well as the financing provided by the Recovery Plan for Europe (NextGenerationEU), it is important to understand how to increase the urban quality. This study aims to analyse from a functional point of view: a. which are the tree life traits most efficient in terms of ES provision; b. if the same tree traits group provide the same ES in different location.

Material and Methods

Study Area

The study focused on three parks located in three cities of the river Po Plain (Northern Italy) (Figure 1). The cities (Pavia, Ferrara and Rimini) were selected along a longitudinal axis to capture the local environmental variability within the Po Plain. The parks were selected according to their representativeness of the respective municipal green areas. The Po Plain is a 47820 km² alluvial plain and is one of the areas with the highest concentrations of air pollutants in Europe (Bigi & Ghermandi, 2014; Bigi et al., 2012; Diémoz et al., 2019). This region is characterized by intense anthropic impacts, such as intensive agriculture and animal farming, extensive industrial districts and wide urbanization. Moreover, due to its peculiar geo-morphological structure, air pollution stagnation is heavily present especially during winter period, when low ventilation increases primary pollutants accumulation (Carbone et al., 2010). Because of the poor environmental quality of the area, the national government is promoting greening actions to mitigate the ongoing climate change and to tackle environmental degradation in urban areas. For these reasons, understanding the contribution of different urban green areas to human well-being in the Po Plain is fundamental to inform decision-makers concerning reforestation actions and creation of new urban parks. According to WHO (2016), Po Plain is one of the most polluted areas in Europe which makes the area an excellent case of study that can provide suitable information for others areas facing with this issue.

Along the longitudinal axis, Pavia is located in the upper part of the Po Plain, being the western city considered for this study. Pavia municipality (45°11’07”N - 9°09’18”E; 77m a.s.l.) is located in Lombardy Region and covers an area of 63.25 km^2 and has 71,882 inhabitants (ISTAT, 2020). The annual precipitation average is 935mm and the average annual temperature is 13.8 °C. The considered park is Giardini del Castello Visconteo, an historical garden built at the end of 14th century. It is located in the city centre and it covers around 6 ha. The second city is Ferrara, located in the centre of the Po Plain. Ferrara municipality (44°50’07.07”N - 11°37’11.51”E; 9-13m a.s.l) covers an area of 405.16 km^2, of which about 4.74 km2 is included within the ancient city walls. Ferrara has 132,899 inhabitants (ISTAT, 2020), an annual precipitation of 814mm and average annual temperature of 14,3 °C. Parco Massari is a historical garden build in 1780s of about 5 ha is representative of local green areas. It is located in the city centre and it is the biggest public park within the ancient city walls of Ferrara. Lastly, Rimini is in the eastern part of the Po Plain (44°03’34”N - 12°34’06”E; 5m a.s.l) and located on the Adriatic coast. It covers 135.707 km^2 and has 149,335 inhabitants. The annual precipitation is 838 mm, while the mean annual temperature is 15.1 °C. Parco Fellini built at the end of 19th century, was a hotel garden open to the public since 1947. It covers 3 ha and it is located nearby on the beach.

For better understanding and identification, parks will be referred to by the name of the city and not by their real name.

Figure 1: Map showing the selected areas for this study. a – Pavia Park; b – Ferrara Park; c – Rimini Park. Brown circle represents tree position in the park.

The different history of each park leads to different tree community composition. Therefore, the respective public tree census were used for the analysis and all trees are mature. Table 1 shows the community composition of the 3 parks in terms of tree species richness, total trees number, tree density and leaf area. Moreover, diversity level for each park was calculated using the Shannon-Wienner index.

Table 1: Tree composition for each park

	Tree species richness	**Trees Number**	**Tree Density (n° trees/ha)**	**Total Leaf Area (m^2)**	**LAI**	**Total Canopy Covare (m^2)**	**Shannon-Wienner Index**
Pavia Park	37	311	52	67926,8	5,1	13228,4	3,12
Massari Park	45	417	83	126022,9	5,8	22665,9	3,18
Fellini Park	24	372	124	71176,9	4,8	15107,7	2,40

Data analysed

The popular iTree-Eco tool (https://www.itreetools.org/tools/i-tree-eco) was used to model the ecosystem services provided by the considered parks, using the complete tree census as input data per park. The ecosystem services assessed were:

- Climate regulation
- Flooding prevention
- Urban heat island mitigation
- Air quality regulation

Carbon storage, avoided runoff, evapotranspiration, as well as $PM_{2.5}$, ozone and nitrogen dioxide removal was assessed.

The climate regulation service is provided by urban green by sequestering and storing carbon by the atmosphere. Carbon storage is calculated by multiplying tree biomass by 0.5 with an upper limit at 7,500 kg of carbon to prevent overestimation for very large trees. Tree biomass is computed based on the five parameters: tree species, diameter breast height, total tree height, crown dieback and light exposure.

Flooding prevention was assessed as the avoided runoff due to the presence of trees on the green area. The tool processes precipitation data and leaf and bark area to model the interaction between rain events and vegetation. Specifically, the outcome estimates the hourly rain interception by urban trees, assuming a mean value of impervious coverage of 25.5%. Estimates are generated based on current tree conditions and then without trees in order to estimate the impact of trees on surface runoff.

Based on the same inputs, the model also calculate evaporation from leaf surfaces and transpiration processes. The sum of these two components was assumed to contribute to the regulation of urban temperatures during heat waves.

Lastly, air quality regulation was quantified based on air pollutants removal by trees. The i-Tree Eco tool evaluates the hourly dry deposition of pollutants of leaves. The removal pollutant flux is calculated as:

[1] $$F = V_d C$$

where V_d is the deposition velocity (m/s) and C is the pollutant concertation (Nowak 2021). The $PM_{2,5}$ deposition velocity derives from bibliographic average values, while deposition velocity of ozone and nitrogen dioxide the deposition velocity were calculated by the model according the canopy resistance that, in turn, is the result of tree components: stomatal resistance, mesophyll resistance and cuticular resistance.

In order to assess the functional features of vegetation and test their relations with ES provision, four traits related to tree canopy and leaves were considered: tree shape, foliage type, canopy width and foliage shape (Table 2). The traits were assigned according the database provided by "Urban Forest Ecosystem Institute" (https://ufei.calpoly.edu/). All the analyses were carried out considering the 90% of the community abundance in terms of tree species, to avoid undesired effects due to the presence of rare species.

Table 2 Tree traits related to canopy and leaves

Tree_Shape	**Foliage_Type**	**Canopy_width**	**Foliage_Shape**	
Columnar	Deciduous	Small (≤ 8m)	Bipinnately	Ovate
Conical	Evergreen	Medium (> 8 m and ≤ 11,50 m)	Cordate	Palmate
Palm		Large (> 11,50 m)	Deltoid	Palmately
Rounded			Lanceolate	Pinnately
			Lobed	Round
			Needle	Scalelike
			Obovate	

The combination of these four traits was used to classify trees in different functional groups (Supplementary 1). The ESs provided each group were as assessed as the average ES values of all trees belonging to the same group. Additionally, four functional indices were calculated using the "FD" package for R (Laliberté & Legendre, 2010), which takes into account multidimensional (i.e. multiple traits) functional diversity: functional richness (FRic), functional evenness (FEve), functional dispersion (FDis) and RaoQ (Rao's quadratic entropy). Furthermore, an analysis of means (ANOM) plot at 95% confidence was used to evaluate the direction of the significant differences based on the deviation from the grand mean of the ANOM plots (Gaglio et al., 2017). Lastly, a Kruskall-Wallis

Test was applied to test possible differences among weather and air pollutants concentration in the three cities (Supplementary 2).

Results

To obviate the effect of rare species (species with low number of trees), the community representing 90% of the total trees has considered.

Structural and functional characteristics of the three parks are presented in Table 3. Pavia Park has the lowest number of trees with 17% of the total leaf area represented by evergreen species. Ferrara Park has the largest number of trees and 21% of leaf area represented by evergreen taxa. Rimini Park denotes an appreciable number of trees, even if belonging to only 13 species, and the larger percentage of evergreen trees on the total leaf area (65%). Species composition of the three parks is presented in Table 4.

Table 3: To avoid the effect of rare species (species represented by a single individual), 90% of the trees community composition has took into account for each park.

	Pavia Park	**Ferrara Park**	**Rimini Park**
Tree species	24	23	13
Tree Number	283	376	339
Evergreen Leaf Area	10203.7	24500.2	43547
Deciduous	51585.3	90024.6	22980
Shannon-Wienner Index	2.86	2.85	2.08
FRic	0.57	0.60	0.37
FEve	0.58	0.57	0.62
FDis	0.34	0.39	0.38
RaoQ	0.14	0.17	0.18

Table 4: Species representing 90% of the community and tree diameter at breast height (DBH ± standard errore)

Pavia Park	**DBH (cm)**	**Ferrara Park**	**DBH (cm)**	**Rimini Park**	**DBH (cm)**
Acer negundo	28.7±1.4	*Acer campestre*	22.8±1.7	*Celtis australis*	20.2±1.3
Acer pseudoplatanus	25.0±4.7	*Acer pseudoplatanus*	26.7±3.8	*Pinus halepensis*	52.9±2.6
Aesculus hippocastanum	12.2±5.2	*Aesculus hippocastanum*	43.4±2.7	*Pinus nigra*	26.0±5.6
Ailanthus altissima	29.8±1.1	*Betula pendula*	26.5±2.2	*Pinus pinea*	49.3±1.6
Betula alba	23.0±0.4	*Carpinus betulus*	20.3±0.1	*Populus alba*	39.3±4.4
Broussonetia papyrifera	23±1.1	*Cedrus atlantica*	48.8±4.8	*Populus canescens*	39.6±2.6
Cedrus atlantica	46.2±0	*Cedrus libani*	55.8±7.1	*Populus nigra*	29.0±2.2
Cedrus deodara	43.3±2.5	*Celtis australis*	44.8±2.5	*Populus x hybrida*	43.2±4.0
Celtis australis	56.8±2.4	*Fraxinus excelsior*	24.6±3.1	*Quercus ilex*	28.4±1.0
Cercis siliquastrum	13.3±1.1	*Gleditsia triacanthos*	47.4±3.2	*Robinia pseudoacacia*	31.1±2.6
Fraxinus angustifolia	21.1±1.4	*Koelreuteria paniculata*	33.9±2.7	*Tamarix sp.*	10.9±2.4
Fraxinus excelsior	24.0±2.0	*Liriodendron tulipifera*	32.0±2.7	*Tilia x europaea*	45.3±4.7
Fraxinus ornus	16.1±0.1	*Magnolia grandiflora*	43.9±5.0	*Trachycarpus fortunei*	25.7±1.0

Gleditsia triacanthos	66.0±4	*Picea abies*	49.1±5.0		
Ilex aquifolium	14.7±0	*Platanus x acerifolia*	52.0±7.3		
Liquidambar styraciflua	26.6±2.0	*Populus canescens*	62.5±0		
Magnolia grandiflora	26.4±1.5	*Quercus robur*	50.7±6.9		
Magnolia x soulangeana	12.2±2.6	*Robinia pseudoacacia*	40.0±3.0		
Pinus nigra	26.1±0.6	*Sambucus nigra*	25.3±2.5		
Platanus x acerifolia	53.5±2.7	*Styphnolobium japonicum*	23.6±2.3		
Prunus cerasifera	18.1±0.8	*Taxus baccata*	58.5±3.8		
Quercus robur	30.8±4.1	*Thuja orientalis*	31.3±0		
Taxus baccata	19.8±8.4	*Tilia x europaea*	36.5±5.7		
Ulmus minor	40.1±2.3				

The average ES provided by the analysed parks are shown in Figure 2. The Figure shows different trends for each of the ES assessed.

The Kruskal-Wallis test highlighted that the three cities significantly differ for mean annual precipitation (p-value <0.05). No differences were observed for the others weather parameters and air pollutant concentrations.

Table 5: Annual weather and air pollution concentrations (± standard deviation) and Kruskal - Wallis Test (95% significance level)

	Pavia	**Ferrara**	**Rimini**	**Kruskal - Wallis Analysis**
Precipitation (cm/hr)	0.0077 ± 0.005	0.0011 ± 0.001	0.0034 ± 0.003	<0.05
Temperature (°C)	14.74 ± 8.213	15.23 ± 9.181	15.38 ± 7.993	n.s.
Solar radiation	309.87 ± 175.133	300.75 ± 164.894	308.86 ± 173.329	n.s.
NO_2 (ppm)	0.02 ± 0.005	0.02 ± 0.005	0.02 ± 0.003	n.s.
O_3 (ppm)	0.02 ± 0.017	0.02 ± 0.015	0.03 ± 0.013	n.s.
$PM_{2.5}$ (ppm)	23.40 ± 13.059	19.38 ± 10.599	22.56 ± 11.583	n.s.

A total of 30 different groups resulted after combining the functional traits (Supplementary 1). 17 groups were found in Pavia, 18 in Ferrara and 11 in Rimini parks.

The ES provided by the three parks computed for each functional group are represented in Figure 3 and Figure 4. ES values were normalized per canopy cover (m^2) of each group.

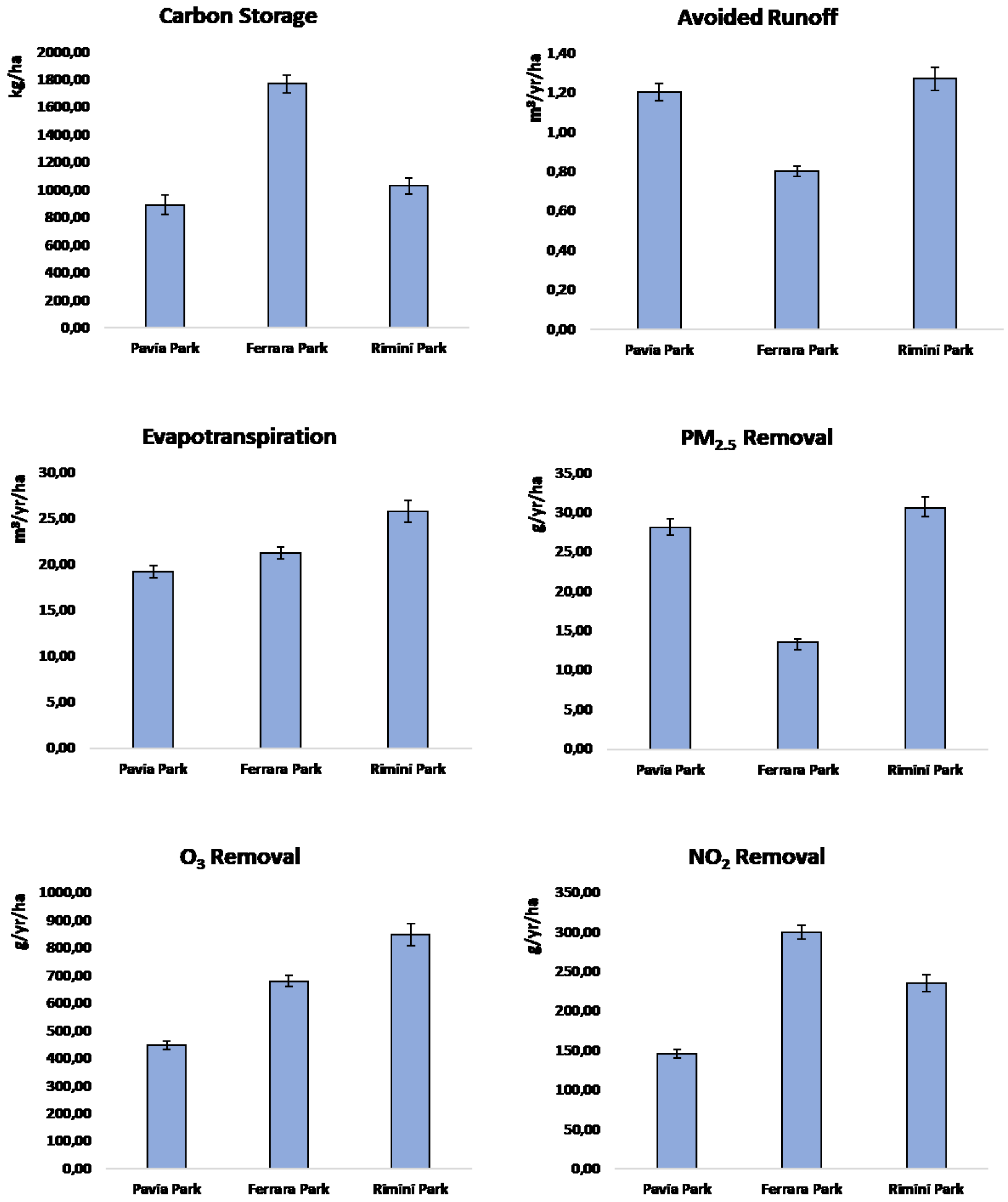

Figure 2 *Average ES provision for each park*

Variations in terms of ES provision can be observed according to arboreal composition. At Pavia Park (Figure 3a and 4a), the trees that mainly store carbon are the those belonging to the G18, i.e. species characterized by rounded tree shape, medium canopy width, bipinnately foliage shape and deciduous foliage. For the other ES, trees from G6 seems to be the most efficient. G6 is characterized by conical tree shape, evergreen, large canopy width and needled shape. Deciduos trees with small canopy width, rounded tree shape (G24, G21) , store the highest amount of carbon in Ferrara Park, but there are other groups such as G28 that are large evergreen trees with ovate foliage shape or G6 that are conifers (i.e. *Cedrus libani*) with high carbone storage. For the other ES analysed large conical trees with lobed and deciduous foliage are the most efficient. Rimini Park instead, have trees with rouded deciduous large trees with lobed foliage shape with highest carbone storage. The other services are mainly provided by evergreen type, small canopy width, palmate foliage shape and palm tree shape.

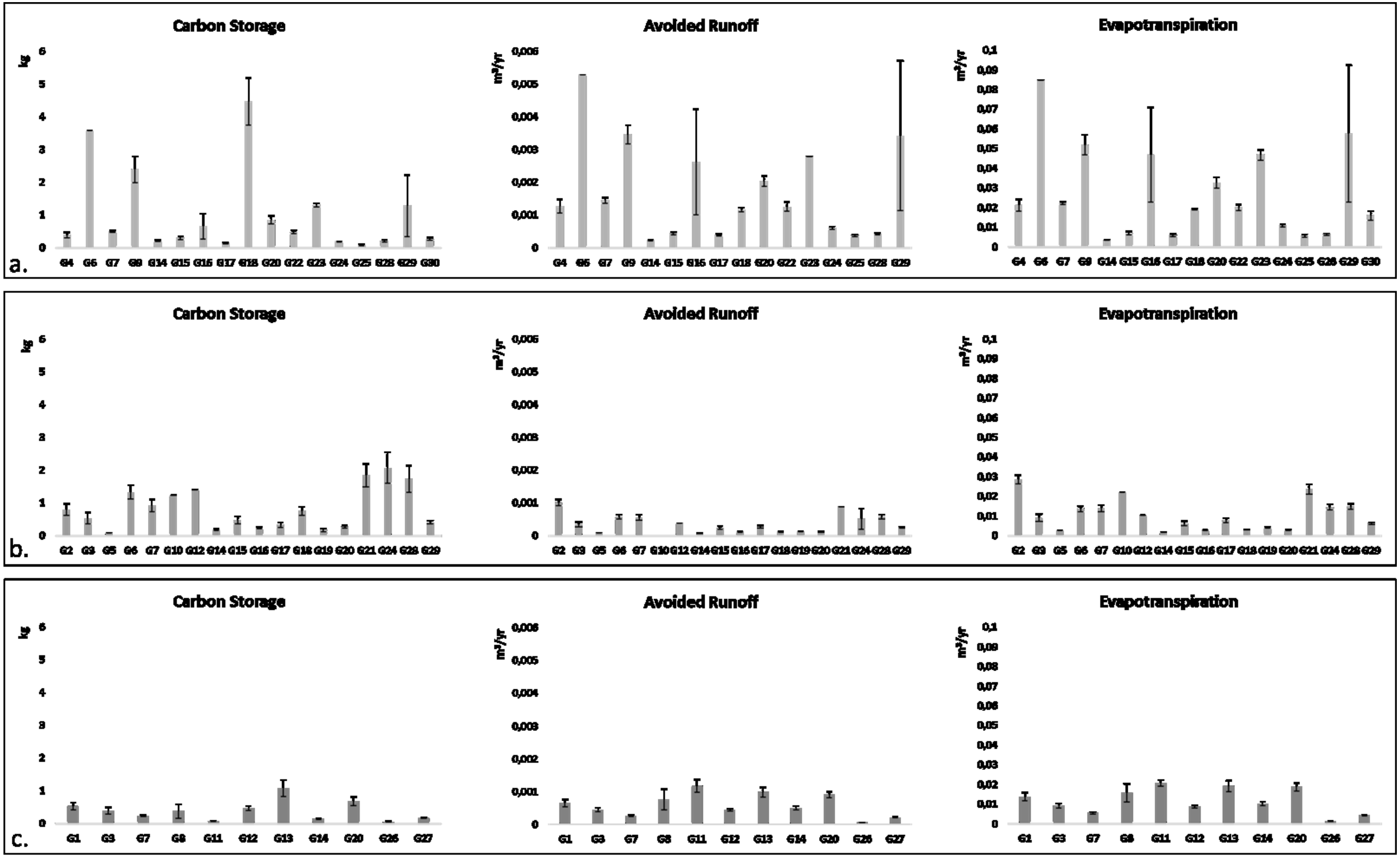

Figure 3 Carbon storage, avoided runoff and evapotranspiration removed per hectare, a – Pavia Park, b – Ferrara Park and c – Rimini Park. Data normalized on squared meter of canopy coverage

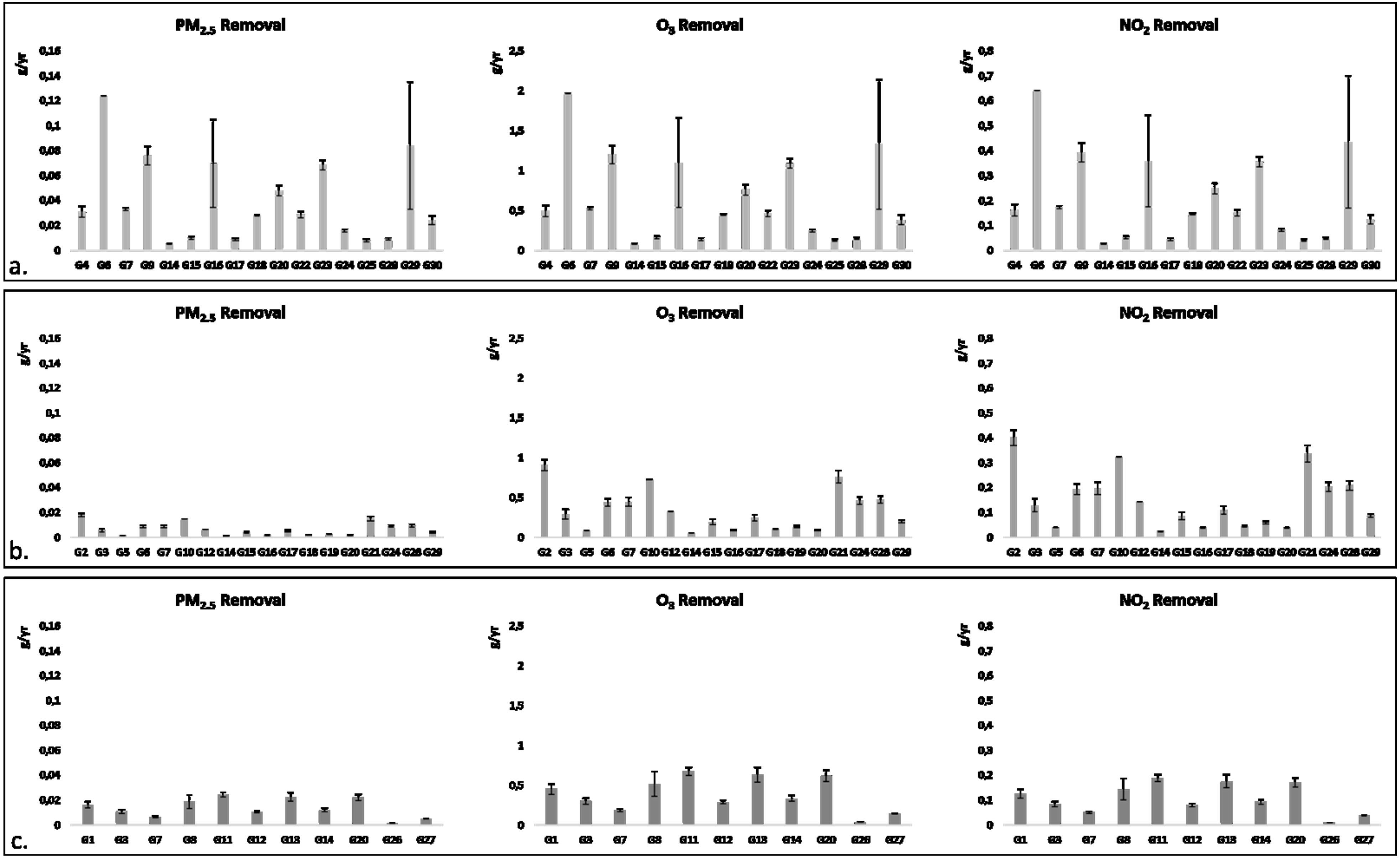

Figure 4 PM$_{2.5}$, ozone and nitrogen dioxide removal per hectare, a – Pavia Park, b – Ferrara Park and c – Rimini Park. Data normalized for squared meter of canopy coverage.

The ANOM analysis was carried out for testing significantly differences among the three cities in ES provision, functional indexes and leaf area (Table 6). Significant variations were found for 5 out of the 6 ESs and for leaf area. No differences were observed for functional indices.

Table 6 Analysis of means (ANOM) for the different ES modelled in the three parks. The three codes a, b and c denote the respective location of the proportion values (above, inside, below the upper and lower 95% confidence limits) 0

	χ^2 (df=2)	P-Value	Pavia Park	Ferrara Park	Rimini Park
Carbon Storage (kg)	*409895.55*	*<0.001*	*c*	*a*	*c*
Avoided Runoff (m^3/yr)	-	*n.s.*	*b*	*b*	*b*
Evapotranspiration (m^3/yr)	*21.94*	*<0.001*	*a*	*b*	*c*
$PM_{2,5}$ Removal (g/yr)	*192.63*	*<0.001*	*a*	*c*	*a*
O_3 Removal (g/yr)	*84438.57*	*<0.001*	*c*	*a*	*a*
NO_2 Removal (g/yr)	*12555.27*	*<0.001*	*c*	*a*	*a*
Leaf area (m^2)	*110016.5*	*<0.001*	*c*	*a*	*b*
FRic	-	*n.s.*	*b*	*b*	*b*
FEve	-	*n.s.*	*b*	*b*	*b*
FDis	-	*n.s.*	*b*	*b*	*b*
RaoQ	-	*n.s.*	*b*	*b*	*b*

Ferrara Park showed significantly higher values for carbon storage and leaf area, and lower values for $PM_{2.5}$ removal when compared with other two green areas. Pavia Park ranked lower for O_3 and NO_2 removal, as well as for leaf area, and higher for evapotranspiration. Avoided runoff is the only ES that does not present significant differences among parks.

Only 3 groups are present in all tree park. Pavia Park and Ferrara Park share 8 groups (not present in Rimini Park), while Ferrara Park and Rimini Park have in common only 2 groups (not present in Pavia Park). No group is shared only by Rimini Park and Pavia Parks. The Kruskal-Wallis test shows significant differences for all groups except Avoided Runoff in G7 and Carbon Storage and NO_2 Removal in G20, both groups present in all three parks. For the groups shated by Pavia and Ferrara Park, the non significant test result in Carbon strage for G15, Avoided Runoff in G16, Evapotraspiration for G6 and G28, PM2.5 Removal in G16 and G29, O3 removal G6 and G28 are not significantly diverce while NO2 removal is not significant in G15, G17 and G18. The ES provided by the two groups shared by Ferrara and Rimini Park (G3 and G12) are all statiticaly different (Supplementary 2).

Discussion

The growing demand for environmental quality in cities recently addressed researcher efforts to the study of ESs provided by urban green infrastructures (i.e. Escobedo et al., 2011; Fusaro et al., 2015).

This introduced the pressing need for the understanding of how the different park configurations support human well-being. More specifically, species composition of urban parks is a fundamental aspect for ecological functioning. Parks tree composition is often a result of different urban history and socio-environmental context that cannot be ignored.

The results of this study provide relevant knowledge concerning the ecological functioning of urban parks in one of the most environmentally impacted yet populated area of Europe. The provision of ESs in the three parks differs substantially even though the analysis of functional diversity of tree community alone was not sufficient to explain such variability. Rather, the presented evidences suggest that the variation in terms of ESs is a result of complex interrelations between functional dimension of vegetation and local environmental context, which cannot be reduced in simple paired relations. In facts, even though no significant differences were observed for weather conditions and air pollutant concentrations, except for precipitation depths, ES provision varies among parks. This finding can be attributed to: i) different functional composition of trees species and ii) complex interactions among pollutants concentrations, local climate and tree community composition. Paradoxically, avoided runoff is the only ES that does not significantly change among parks, despite the fact that precipitation is the only climatic parameter that significantly changes among considered cities. This further confirms the complex nature of trees-environment interactions for the provision of ESs in urban areas. In particular, the higher precipitations occurred in Pavia during summer season were absorbed by the broadleaved vegetation, which provided higher leaf and bark areas during this this period. Such characteristic also results in significantly higher evapotranspiration process (Table 6), which underpin temperature regulation service.

The higher climate regulation capacity of Ferrara Park might be related to the presence of some monumental trees, which represent important carbon pools. Indeed, ancient trees provide numerous ecosystems functions. Monumental trees are extraordinary to maintain carbon balance (Gunn et al., 2014; Janisch & Harmon, 2002; Ma et al., 2021). For this reason, old trees plays a key role in climate change regulation. Ferrara Park is the only park that hosts monumental trees. In fact, one of the groups most efficient in term of carbon storage is represented by G6 defined by *Cedrus libani*, one of the species having monumental trees.

Rainfall intensification is another important issue related to climate change. Increasing the precipitation interception is crucial to prevent flooding. Pavia has the high rain events during the summer period, but still the presence of the groups G6 and G9 buffers the rainfall effect. These two groups are both conical needled evergreen species. The different between the groups is represented by the canopy width that is large in G6 (i.e. *Cedrus libani*) and small in G9 (i.e. Cedrus deodara). Leaf surface is an important parameter that increase water retention and shed canopy. Higher leaf area have been identified with higher interception rate (Frischbier et al., 2019; Livesley et al., 2014; Nooraei Beidokhti & Moore, 2021).

Maximizing leaf area can be a strategy for mitigating the effect of urban heat waves too (Tamaskani Esfehankalateh et al., 2021). In Pavia Park and Ferrara Park groups characterized by board-leaves have a higher evapotranspiration rate. For Rimini instead, evapotranspiration is lower due to the climatic condition.

Considering Figure 2 and Table 6, $PM_{2.5}$ is removed more at Pavia and Rimini Parks while for the other pollutants analysed, Ferrara Park and Rimini Park are the most efficient to remove O_3 and NO_2. Air depuration is highly related to the air pollution concentration. Anyways, trees have different removal capacity in accord to their traits. Evergreen trees are more efficient in term of air depuration in particular to remove $PM_{2.5}$ (Pace & Grote, 2020). Rimini Park is characterized by high evergreen leaf area while Pavia has the Parks with the lower evergreen leaf area. Air depuration ranked lower despite high levels of pollution. Indeed, Rimini has the highest removal rate and the dominant group is G7 represented by needled evergreen species. Ferrara Park have low $PM_{2.5}$ removal and it is also the city with the lowest concertation of $PM_{2.5}$. O_3 and NO_2 removal is linked to the stomatal conductance (Manes et al., 2008). Moreover, the relationship between O_3 and NO_2 as well as the temperature influence the removal capacity (Fares et al., 2010; Han et al., 2011).

Environmental condition seems to be the main parameter that influence ES provision as well as leaf area. Leaf area is limitat for the model. i-Tree Eco, in addition to the parameters shown in Methods, consider leaf area as the main tree characteristic that influence the ES provision. Moreover, $PM_{2.5}$ removal consider an average Vd (deposition velocity) for all tree species while several authors discovered higher Vd for conifers instead than boarder leaf (i.e. Freer-Smith et al., 2005; Popek et al., 2013; Pace & Grote, 2020). The model limits underrate the real tree potential in terms of services provision. However, the model is a good method to assess ecosystem services even if it should be "polished" in order to adapt ES estimation to the real capacity of trees to provide this ES.

More generally, the findings demonstrate that no specific guidelines for improving ES in urban areas can be provided to decision-makers without consider the complexity of tree-environment interaction. This requires the modelling of ecological functions, whose reliability is based on model accuracy and input data quality concerning both environmental context and tree community. Nonetheless, the analysis allows detecting some general trends useful for addressing urban green management.

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SUPPLEMENTARY 1 *List of species with the related traits and the assigned group*

Group	Specie	Tree Shape	Foliage Type	Canopy width	Foliage Shape
G1	*Populus nigra*	Columnar	Deciduous	Small	Cordate
G2	*Liriodendron tulipifera*	Conical	Deciduous	Large	Lobed
G3	*Tilia x europaea*	Conical	Deciduous	Large	Ovate
G4	*Liquidambar styraciflua*	Conical	Deciduous	Large	Palmate
G5	*Carpinus betulus*	Conical	Deciduous	Small	Ovate
G6	*Cedrus atlantica*	Conical	Evergreen	Large	Needle
G6	*Cedrus libani*	Conical	Evergreen	Large	Needle
G7	*Picea abies*	Conical	Evergreen	Medium	Needle
G7	*Pinus halepensis*	Conical	Evergreen	Medium	Needle
G7	*Pinus nigra*	Conical	Evergreen	Medium	Needle
G8	*Tamarix sp*	Conical	Evergreen	Medium	Scalelike
G9	*Cedrus deodara*	Conical	Evergreen	Small	Needle
G10	*Thuja orientalis*	Conical	Evergreen	Small	Scalelike
G11	*Trachycarpus fortunei*	Palm	Evergreen	Small	Palmate
G12	*Populus canescens*	Rounded	Deciduous	Large	Cordate
G13	*Populus alba*	Rounded	Deciduous	Large	Lobed
G14	*Broussonetia papyrifera*	Rounded	Deciduous	Large	Ovate
G14	*Celtis australis*	Rounded	Deciduous	Large	Ovate
G14	*Populus x hybrida*	Rounded	Deciduous	Large	Ovate
G14	*Quercus robur*	Rounded	Deciduous	Large	Ovate
G14	*Ulmus minor*	Rounded	Deciduous	Large	Ovate
G15	*Acer pseudoplatanus*	Rounded	Deciduous	Large	Palmate
G15	*Platanus x acerifolia*	Rounded	Deciduous	Large	Palmate
G16	*Aesculus hippocastanum*	Rounded	Deciduous	Large	Palmately
G17	*Ailanthus altissima*	Rounded	Deciduous	Large	Pinnately
G17	*Fraxinus excelsior*	Rounded	Deciduous	Large	Pinnately
G17	*Styphnolobium japonicum*	Rounded	Deciduous	Large	Pinnately
G18	*Gleditsia triacanthos*	Rounded	Deciduous	Medium	Bipinnately
G19	*Acer campestre*	Rounded	Deciduous	Medium	Palmate
G20	*Acer negundo*	Rounded	Deciduous	Medium	Pinnately
G20	*Koelreuteria paniculata*	Rounded	Deciduous	Medium	Pinnately
G20	*Robinia pseudoacacia*	Rounded	Deciduous	Medium	Pinnately
G21	*Betula pendula*	Rounded	Deciduous	Small	Deltoid
G22	*Prunus cerasifera*	Rounded	Deciduous	Small	Obovate
G23	*Betula alba*	Rounded	Deciduous	Small	Ovate
G24	*Fraxinus angustifolia*	Rounded	Deciduous	Small	Pinnately
G24	*Fraxinus ornus*	Rounded	Deciduous	Small	Pinnately
G24	*Sambucus nigra*	Rounded	Deciduous	Small	Pinnately
G25	*Cercis siliquastrum*	Rounded	Deciduous	Small	Round
G26	*Quercus ilex*	Rounded	Evergreen	Large	Lanceolate
G27	*Pinus pinea*	Rounded	Evergreen	Large	Needle
G28	*Magnolia grandiflora*	Rounded	Evergreen	Large	Ovate
G29	*Taxus baccata*	Rounded	Evergreen	Small	Needle
G30	*Ilex aquifolium*	Rounded	Evergreen	Small	Ovate
G30	*Magnolia x soulangeana*	Rounded	Evergreen	Small	Ovate

SUPPLEMENTARY 2 Kolmogorov-Smirnov Test for the groups shared (n.s. – not significant; * - <0.05; ** - <0.001)

Groups shared by all three Parks				
	G7			
	P-Value	**Pavia Park**	**Ferrara Park**	**Rimini Park**
Carbon Storage (kg)	**	*c*	*a*	*b*
Avoided Runoff (m^3/yr)	*n.s.*	*b*	*b*	*b*
Evapotranspiration (m^3/yr)	*	*c*	*b*	*b*
$PM_{2.5}$ Removal (g/yr)	*	*b*	*b*	*a*
O_3 Removal (g/yr)	*	*c*	*b*	*b*
NO_2 Removal (g/yr)	**	*c*	*a*	*b*

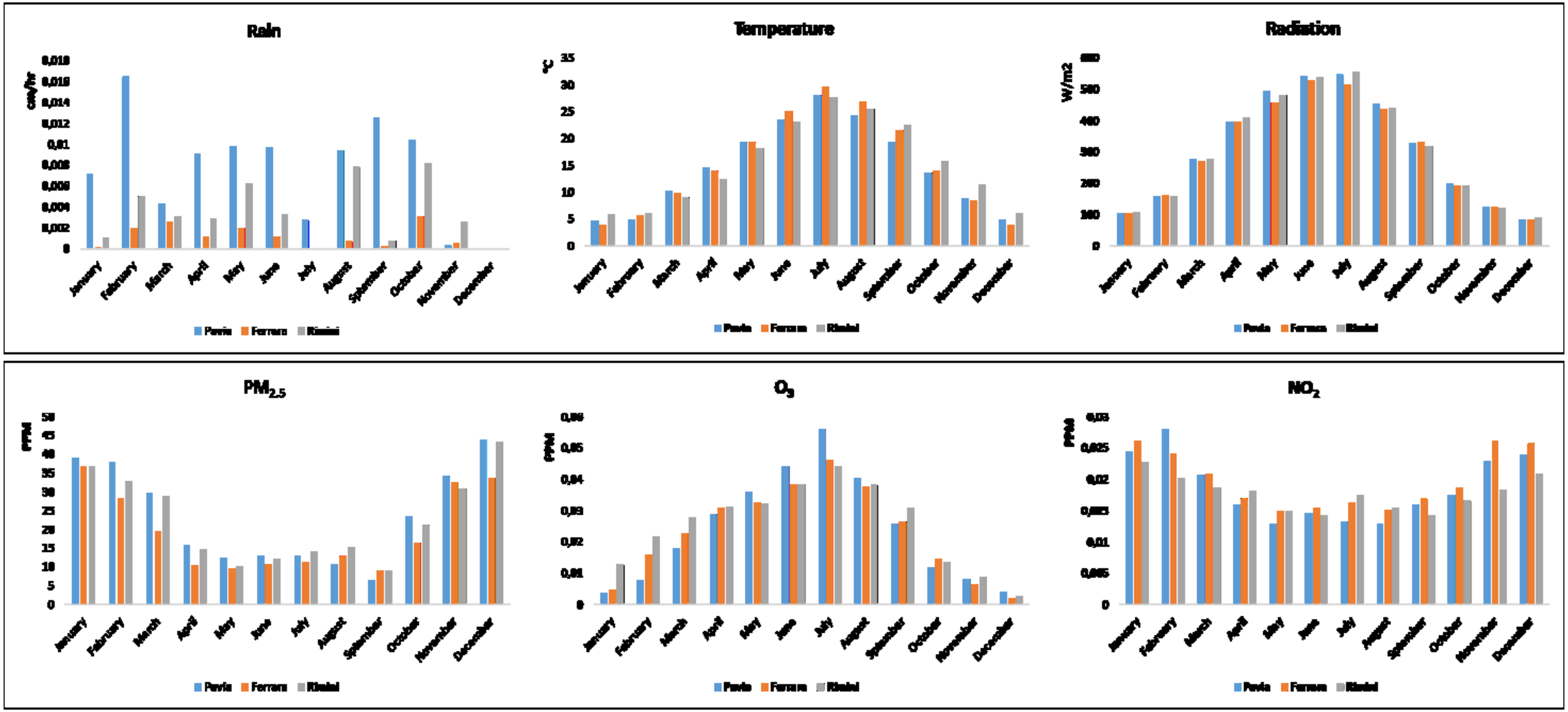
	G14			
Carbon Storage (kg)	**	*b*	*b*	*c*
Avoided Runoff (m^3/yr)	**	*a*	*c*	*c*
Evapotranspiration (m^3/yr)	**	*a*	*c*	*c*
$PM_{2.5}$ Removal (g/yr)	**	*a*	*c*	*c*
O_3 Removal (g/yr)	**	*a*	*b*	*c*
NO_2 Removal (g/yr)	**	*b*	*a*	*c*

	G20			
Carbon Storage (kg)	*n.s.*	*b*	*b*	*b*
Avoided Runoff (m^3/yr)	**	*a*	*c*	*a*
Evapotranspiration (m^3/yr)	**	*a*	*c*	*a*
$PM_{2.5}$ Removal (g/yr)	**	*a*	*c*	*a*
O_3 Removal (g/yr)	*	*b*	*c*	*a*
NO_2 Removal (g/yr)	*n.s.*	*b*	*b*	*b*

Groups shared in Pavia Park and Ferrara Park						
	Carbon Storage (kg)	**Avoided Runoff (m^3/yr)**	**Evapotranspiration (m^3/yr)**	**PM2.5 Removal (g/yr)**	**O3 Removal (g/yr)**	**NO2 Removal (g/yr)**
G6	*	**	*n.s.*	**	*n.s.*	*
G15	*n.s.*	**	*	**	*	*n.s.*
G16	*	*n.s.*	*	*n.s.*	*	*
G17	*	**	**	**	*	*n.s.*
G18	*	*	*	*	*	*n.s.*
G24	**	**	**	**	**	*
G28	*	*	*n.s.*	**	*n.s.*	*
G29	*	*	*	*n.s.*	*	*

Groups shared in Ferrara Park and Rimini Park						
G3	*	**	**	**	**	**
G12	**	**	**	**	**	*

SUPPLEMENTARY 3 Annual trend of weather and air pollutants concentration in the three cities



Chapter 2 Mesoscale Analysis

Integrating geospatial data based on field measurements allowed us to analyse the provided ecosystem services at municipality scale. This scale approach allows a broader overall evaluation of the offered ES. In particular, this study assess the PM_{10} and O_3 removal (air quality regulation) at the current stage. Furthermore, in order to inform the decision makers about the urban foresting potential, it was planned a possible future scenario of tree planting in public green areas. The study prove the key role of green infrastructure for the air quality regulation. Moreover, low cost foresting actions can significantly increase the ES provision capacity.

(Published as: Alexandra Nicoleta Muresan, Alessandro Sebastiani, Mattias Gaglio, Elisa Anna Fano, Fausto Manes. *Assessment of air pollutants removal by green infrastructure and urban and peri-urban forests management for a greening plan in the Municipality of Ferrara (Po river plain, Italy)* Ecological Indicators, 2022, 135, 108554. https://doi.org/10.1016/j.ecolind.2022.108554)

Assessment of air pollutants removal by green infrastructure and urban and peri-urban forests management for a greening plan in the Municipality of Ferrara (Po river plain, Italy)

Abstract

Air pollution is a serious concern for human health and is even more worrying in areas that are known to be "pollution hotspots", such as the Po Plain in northern Italy. The Urban Green Infrastructure (UGI), which includes urban and peri-urban forests, enhances human health and wellbeing delivering a wide range of ecosystem services, including air quality improvement. In this research, we analyzed, in biophysical and monetary terms, the role of the UGI in removing PM_{10} and O_3 from the atmosphere in the Municipality of Ferrara using established removal models. We used a multiscale approach that includes geospatial data, field sampling and laboratory analysis. Then, using a local green areas database, we located public areas that could potentially undergo forestation actions without requiring any land conversion and evaluated the benefit in terms of ESs provision that these actions may exert. We found that, in 2019, the UGI in the Municipality of Ferrara removed about 19.8 Mg of PM_{10} and 8.6 Mg of O_3, for a monetary benefit of € 2.12 million € and $147*10^3$ respectively. We then identified about 121 ha within the urban core of the Municipality that could potentially be forested. Such an action would increase the PM_{10} and O_3 removal by about 49% and 18%, respectively. Our findings comply with the EU Biodiversity strategy for 2030, which calls for the development of an ambitious greening plan for cities with more than 20,000 inhabitants.

Keywords

Urban Green Infrastructure; Regulating Ecosystem Services; Biodiversity and urban green; air pollutants removal; urban green management; multi-scale experimental approach; PM_{10}; O_3

1. Introduction

Air pollution is a global challenge, and is currently considered to be the most important risk to human health (EEA, 2020). According to the European Environment Agency (EEA, 2020), PM pollution caused about 52,300 premature deaths in 2018 in Italy, by enhancing the incidence of lung diseases, cancer, and stroke; in the same period, the O_3 caused about 3,000 premature deaths. In addition, long-term exposure to air pollutants affects the respiratory and cardiovascular systems, leading to an increase in the COVID-19 death rate (Zhu et al., 2020; Accarino et al., 2021). In Europe, sectors such as transport, commercial, household, energy production, industry, and agriculture highly contribute to the emission of pollutants into the atmosphere (EEA, 2019); the formation and accumulation of air pollutants also depend on local factors such as climatic conditions (Ferrero et al., 2011).

In this framework, alongside emissions reduction policies (Lafortezza et al., 2018), the EU has encouraged the use of *Nature-based Solutions* (NBS) to effectively contrast air pollution. NBS enhance the environmental quality, as well as the growth and conservation of the Natural Capital and may help to shift towards a greener economy (Capotorti et al., 2020). In particular, the Urban Green Infrastructure (UGI), which encompasses urban and peri-urban forests and other elements such as street trees, provides a wide range of Ecosystem Services (ESs) (EC, 2013). For example, the UGI plays a pivotal role in improving the air quality of urban ecosystems by removing different air pollutants (Manes et al., 2012a and 2016; Baumgardner et al., 2012) and by mitigating the Urban Heat Island effect (Marando et al., 2019).

Increasing the provision of ESs is a major challenge in compact cities (Hansen et al., 2019; Grêt-Regamey et al. 2020). Dense urban environments demand ES of urban trees, yet existing green spaces to plant new trees are scarce and the conversion of non-green areas is expensive and complex. Nonetheless, while urban green spaces are limited, increasing their quality is a pressing need (Haaland and Van der Bosch 2015). Raising the presence and coverage of trees are widely recognized as valuable measures to increase the multifunctionality of UGI (Reid et al., 2017; Jim et al., 2018; Belmeziti et al., 2018; Manes et al., 2012a). This is also evident in the EU biodiversity strategy for 2030 (EC, 2020), which remarks the need for protecting and restoring well-functioning ecosystems and requires cities with at least 20,000 citizens to develop an ambitious urban greening plan by the end of 2021.

The biophysical assessment of ESs has increasingly been used to support decision-makers in defining management policies (Burkhard & Maes, 2017); the economic evaluation of ESs may also represent a valuable tool for highlighting the relevance of natural ecosystems to society (Capriolo et al., 2020). Notwithstanding, mapping Regulating ES is complex (Wei et al., 2017), as it requires a deep understanding of those structures and processes that underpin the considered ES (Crossman et al., 2013). As a consequence, the NBSs are not systematically included in sustainable planning strategies (Woodruff and BenDor, 2016; Sebastiani et al., 2021a).

This research aimed firstly to assess the role of the urban and peri-urban forests of the Municipality of Ferrara (northern Italy) in removing the PM_{10} and O_3 from the atmosphere. The Municipality of Ferrara was selected as an ideal case study, since it is representative of the river Po plain, an area that suffers from high air pollution levels due both to direct emissions and its geomorphological features. Secondly, by accessing a local green areas database (data provided by the Municipality of Ferrara), we identified areas that could potentially undergo forestation actions without requiring land conversion, and evaluate a scenario in which the UGI of the Municipality is enhanced. A monetary evaluation of air quality improvement service is also performed, to demonstrate the relevant socio-economic benefits of planting trees in urban areas and provide arguments to support future investments in enhancing the UGI. We adopted a multi-scale approach that relies on field sampling, laboratory analysis, geospatial data such as remote sensing data and the Corine Land Cover (CLC) inventory, and modeling techniques. Our findings are also oriented to support stakeholders toward the development of an urban greening plan for the Municipality, according to the EU biodiversity strategy for 2030.

2. Materials and methods

2.1 Study area

The Municipality of Ferrara (44°50’07.07”N; 11°37’11.51”E, Figure 1) is located in the middle of the Po plain, Northern Italy, at approximately 9-13m a.s.l. The Po plain is one of the major air pollution hotspots in Europe (Bigi et al., 2014; Dièmoz et al., 2019); indeed, several factors concur to raise the air pollution in that area. First, the agricultural sector, which has a strong influence on the local air quality, is highly developed (Lonati et al., 2021). Then, winters are characterized by a very low wind speed, and temperature inversion is frequent (Caserini et al., 2017); therefore, due to the reduced ventilation, the accumulation of primary pollutants and the formation of secondary pollutants are favored (Carbone et al., 2010; Ferrero et al., 2011).

Ferrara covers an area of 405.16 km^2, of which about 4.74 km^2 is included within the ancient city walls, and has 132,899 inhabitants who reside mostly in the compact urban core (ISTAT, 2020), with a population density of about 328 inhabitants km^{-2}. The annual precipitation is about 814 mm yr^{-1}; the area can be classified as “Cfa” according to the Köppen-Geiger climate classification (Peel et al. 2007). Following the recently updated ecoregional classification of Italy (Blasi et al., 2018), the Municipality falls in the Po plain province (lagoon subsection, code 1B1a), which embeds the northern part of the Italian Adriatic coast.

The potential vegetation in the inner part of the Po plain is essentially made by Querco-carpinetum forests with *Quercus robur*, *Carpinus betulus* and *Acer campestre* (Blasi et al., 2010). The natural vegetation has almost entirely been removed to maximize the space for agricultural activities. A major role is played by the riparian vegetation along the Po riverside, which includes *Populus alba*, *Fraxinus angustifolia,* and *Quercus robur* (Blasi et al., 2010).

According to the National Census of urban green areas (ISTAT 2019), the public green areas of Ferrara account for 8.2 km^2. The ancient city walls extend for 9 km and are surrounded by an urban green fringe with appreciable levels of biodiversity (Pelizzari et al. 2015). In addition, other small public and private parks and gardens are present on the municipal territory (Figure 1).

The urban flora of Ferrara encompasses 589 vascular species, more than 100 of which are aliens (Pelizzari et al. 2015). Following the public trees’ census (data provided by the Municipality of Ferrara, 2018), *Celtis austalis, Tilia x europea, Platanus x acerifolia, Populus nigra, Acer negundo,* and *Robinia pseudoacacia* are the most abundant species. Evergreen broadleaves and conifers are poorly represented in the Municipality, being respectively *Magnolia grandiflora, Quercus ilex,* and *Picea abies*, *Cedrus atlantica, Thuja sp*.

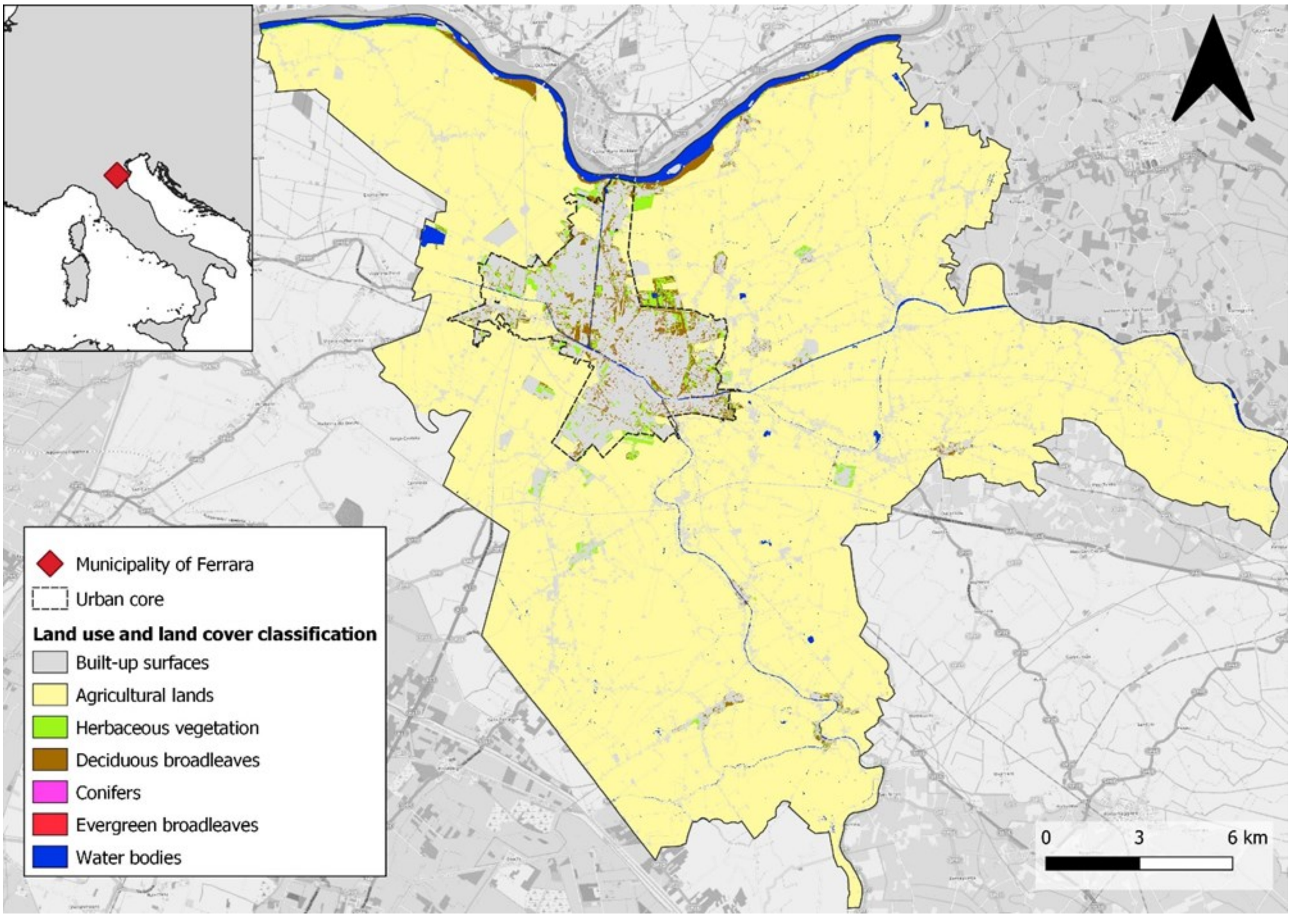


Figure 1: Land use/land cover classification of the Municipality of Ferrara

2.2 Land use/land cover classification

We identified 7 land use/land cover classes for the study area: built-up surfaces, agricultural areas, herbaceous vegetation, water bodies, and 3 different functional groups of vegetation, that is, deciduous broadleaves, evergreen broadleaves, and conifers (Manes et al., 2012a). Ecosystem mapping was conducted using different data sources.

The built-up surfaces were detected based on the National Land Consumption map produced by ISPRA (Munafò et al., 2019), which is made through photointerpretation and semiautomatic classification of satellite images and is delivered as a 10 m resolution raster. The agricultural areas were extracted from the Corine Land Cover 2018 dataset, which has a minimum mapping unit of 25 hectares; specifically, we extracted class 2, which refers to the agricultural areas, including pastures, arable lands, and permanent crops. The herbaceous and woody vegetation was mapped performing a supervised classification of two Sentinel-2 images, acquired during summer 2019. We adopted the maximum-likelihood algorithm to assign each pixel to its land use and land cover class; the classification was performed utilizing the SCP plug-in implemented in QGIS (Congedo, 2020). We used the Sentinel-2 Level-2A products, which provide Bottom Of Atmosphere (BOA) reflectance and are systematically generated over Europe and freely delivered by the European Space Agency. At last, conflicts between different data sources were solved using ground truth data and field observations carried out by the experts of the University of Ferrara. The accuracy of the land use/land cover classification was evaluated using field observations and ground-truth data.

2.3 Seasonal Leaf Area Index retrieval

Direct measurements of the Leaf Area Index (LAI) are time-consuming and labor-intensive, and can only be conducted within small areas like isolated trees, single fields, or parks (Xie et al., 2019). In this research, the seasonal LAI was computed processing the Level-2A Sentinel-2 data (two cloud-free images for each season) within the biophysical processor module of the Sentinel Application Platform (SNAP) software. The biophysical processor uses 8 different spectral bands and other information such as solar zenith, viewing zenith, and relative azimuth angles (Kganyago et al., 2020); it is based on a pre-trained artificial neural net. The output consists of a 10 m resolution raster covering the study area, which we used to estimate the LAI of the different functional groups of vegetation. Due to the unprecedented spatial resolution, the LAI product derived from the described methodology will be increasingly used for a wide range of applications. Preliminary studies have shown that Sentinel-2 derived LAI has a good accuracy when compared to field data (Brown et al., 2021); however, this product still needs comprehensive validation over different regions and for different periods of the year (Hu et al., 2020).

2.4 Pollution data

The Monthly PM_{10} and O_3 concentrations were retrieved from the Emilia-Romagna ARPA (Regional Environmental Protection Agency) monitoring system. Pollution data was taken from the monthly report of Air quality, freely available on the ARPAE website (https://www.arpae.it/it). We used monthly data from the monitoring stations named Barco Nuova (urban-industrial), Villa Fulvia (urban background) and Corso Isonzo (urban traffic) to estimate the seasonal average concentrations. To avoid any bias caused by the 2020 pandemic, we decided to use 2019 pollution data. Then, we assumed that the seasonal PM_{10} and O_3 concentrations were homogeneous for the study area.

2.5 Air pollutants removal by vegetation

To estimate the PM_{10} removal, we adopted the methodology proposed by Manes et al. (2016), Fusaro et al. (2017), and Nowak et al. (1994). The following equation was used:

$$Q = F * L * T \qquad (1)$$

Where *Q* is the quantity (kg) of the pollutant removed by the vegetation; *F* ($\frac{\mu g}{m^2 * s}$) is the flux of the pollutant, which is a function of the PM_{10} concentration and the deposition velocity (V*d*) of particles on the leaves; *L* is the LAI; *T* is the vegetative period of each functional group of vegetation, which was set to 365 days for evergreen species and 214 days (from early spring to mid-autumn) for deciduous species.

The deposition velocities of the three functional groups (deciduous broadleaves, evergreen broadleaves, and conifers) were experimentally measured (Eq.2). The 11 most representative tree species for the Municipality of Ferrara were selected. Twelve samples of branch leaves were taken for each tree species (three samples x four trees), close to the monitoring stations for PM_{10}, to obtain reliable measures of PM air concentrations before and during sampling events. The PM_{10} deposition velocity for the sampled leaves was estimated using the vacuum filtration method, following the methodology proposed by Sgrigna et al. (2015). The total leaf area of broadleaf samples was measured through ImageJ (Schneider et al. 2012) upon direct leaf scan and considered as double-sided. The leaf area of conifers was estimated by assuming that needles have a cylindrical shape. The surface of a single needle was estimated by multiplying needle length by needle girth. The latter was measured making the cross-sectional slice in the middle part of the needle. The individual surfaces of 20 needles were measured to obtain a mean value and then multiplied by the number of needles present in each branch sampling. Samplings were carried out after at least 10 dry days (i.e. days occurred after the last precipitation event). The deposition velocity was computed considering the mean PM_{10} adsorbed by leaf samplings of the three functional types and the PM_{10} air concentration in the dry days before samplings, according to the following equations:

$$V_d = \frac{F_h}{PM_{air} * 0.000036} \qquad (2)$$

Where Vd is the deposition velocity (cm s^{-1}), F_h is the mean hour flux (g m^{-2} h^{-1}) and PM_{air} is the mean PM_{10} air concentration during the *d* dry days before sampling (µg m^{-3}).

$$F_h = \frac{PM_{leaf}}{100 * 24 * d} \qquad (3)$$

Where PM_{leaf} is the amount of PM_{10} measured on the leaf sampling (µg cm^{-2}) after *d* dry days before sampling.

The O_3 removal was computed following the methodology proposed by Manes et al., (2012a). First, we calculated the stomatal O_3 flux referred to 1 m^2 of forested soil surface:

$$FO_{3(i,p)} = gs_{(i)} * [O_3]_{(i,p)} * 0.613 \qquad (4)$$

Where FO_3 is the stomatal ozone flux, *gs* is the mean annual stomatal conductance $\left(\frac{mol\ H_2O}{m^2 * s}\right)$, retrieved from field and laboratory data, $[O_3]$ is the mean seasonal Ozone concentration, *0.613* is the ratio of diffusivity between Ozone and water vapor, *i* and *p* refer to the day of the year and location respectively. $FO_{3(i,p)}$ was then integrated over time for the entire study area, to estimate the cumulative amount of O_3 which is seasonally removed by the vegetation:

$$FO_{3cum(p)} = \left(\sum_{i=1}^{n} \quad FO_{3(i,p)} * Ph * \frac{1}{R}\right) \qquad (5)$$

Where *n* is the number of cumulative days (which is equal to 183), *Ph* is the photoperiod expressed in *s*, and *R* is the $\frac{stomatal}{total}$ flux ratio. A number of 183 cumulative days was chosen since Ozone is mostly concerning during the hot periods of the years, which in our case occur from April to October (Fusaro et al., 2017). All the modelling elaborations were carried out using the GRASSGIS software.

2.6 Monetary evaluation

The monetary evaluation was conducted following the methodology proposed by Fusaro et al. (2017). We used the externality values, that is, an estimated social cost of air pollution that is not included in the market price, including the detrimental effect on human health, environment, and man-made materials (EEA, 2020). In our case, the externality is expressed in euros per Mg of pollutants removed by the vegetation. The externality values are equal to € 107,384 and € 17,110 per PM_{10} and O_3 respectively. Those are calculated based on the conservative value of a life year (VOLY), integrating different scientific and economic disciplines (EEA, 2020).

2.7 Identification of spaces for the urban forestation and enhancement of ESs

The area for the urban forestation was extracted from the public green space's database of the Municipality, which is spatially explicit and reports the current intended use of the area (e.g., parks, sports facilities, street trees, etc.). Potential conversion from agricultural land or brownfields to forested areas was not considered in the analysis. In selecting suitable spaces, we excluded all the areas smaller than 1 ha, which are in most cases connected to residential buildings and schools as well as street greening; we also excluded private areas and sports facilities such as football fields and racecourses. Spaces devoted to religious purposes were ignored, as those areas provide a wide range of cultural and recreational ESs (Langermeyer et al., 2015). Lastly, we did not consider areas close to high historical-values sites such as the ancient city walls, which may need specific conservation and management measures and are subject to landscape and historical constraints. After this first screening, we mapped the portion of the suitable area which is not currently covered by woody vegetation, assuming that it can be subject to forestation actions.

We then hypothesized to reforest the above-mentioned areas with equally distributed evergreen broadleaves, deciduous broadleaves, and conifers; precisely, young plants of autochthonous species should be preferentially chosen (Loures et al., 2007). Soil and climate conditions in the study area can support species from the three functional groups of vegetation. In fact, the soil of Ferrara municipality is quite homogeneous (Emilia-Romagna Region, 2010) and there are no relevant restrictions for urban forests. Thus, no significant spatial differences in vegetation growth and performances are expected.

We focused on the urban core of the Municipality of Ferrara since such area is densely populated and highly demanding for ESs; therefore the enhancement of the NBSs would provide benefits to the majority of the people living in the city. Eventually, we estimated the benefit in the PM_{10} and O_3 removal, in biophysical and monetary terms, that this forestation action would provide to the Municipality.

3. Results

3.1 Ecosystem mapping

Figure 1 shows the land cover classification of the study area. Agricultural areas cover the majority of the Municipality (almost 82%). Built-up surfaces cover slightly more than 13% of the Municipality and are mostly located in the urban core. The combination of the functional groups of vegetation, covers about 697 ha, which equates to 1.7% of the total surface of the Municipality. The three functional groups of vegetation are not equally present: 97% (677 ha) of the total vegetation belongs to the deciduous broadleaves; about 3% (21 ha) belongs to the conifers group; the presence of evergreen broadleaves is very scarce (about 1 ha). About 43% (about 297 ha) of the woody vegetation can be found within the urban core. Water bodies cover almost 3% of the Municipality. The accuracy of the classification was estimated to be slightly higher than 80%.

3.2 Seasonal Leaf Area Index

Figure 2 shows the seasonal LAI maps for the urban core of the Municipality; due to the scarcity of evergreen species, winter data is not shown. Quantitative values are reported in Table 1. Deciduous broadleaves show moderate values in spring and autumn (2.1 and 1.7 respectively) and peak during summer (3.0). The seasonal LAI of conifers spans from a minimum of 1.4 in winter to a maximum of 2.4 during summer, whereas evergreen broadleaves show small LAI variations amongst the different seasons, with the lowest value (1.5) in winter, and the highest (2.3) in summer.

Figure 2: Seasonal LAI for spring (A), summer (B) and autumn (C). Due to the scarcity of evergreen species, the winter map is not shown

Table 1: Mean seasonal Leaf Area Index derived from Sentinel-2 data

Vegetation group	Season			
	Winter	**Spring**	**Summer**	**Autumn**
Deciduous broadleaves	0	2.1	3.0	1.7
Evergreen broadleaves	1.5	1.8	2.3	1.7
Conifers	1.4	2.1	2.4	2.0

3.3 Pollutants levels and PM_{10} deposition velocity

PM_{10} levels (year 2019) are highest in winter (50.4 µg m^{-3}), followed by spring (36.9 µg m^{-3}), summer (30.3 µg m^{-3}), and autumn (29.1 µg m^{-3}). O_3 levels (year 2019) are higher during summer (53.8 µg m^{-3}), followed by autumn (48.7 µg m^{-3}), spring (41.3 µg m^{-3}) and winter (19.3 µg m^{-3}). Table 2 summarizes the V*d* and the related standard deviation for each functional group. The V*d* of deciduous broadleaves is equal to 0.545 cm s^{-1} (st.err. ±0.0024); V*d* for conifers is 1.458 cm s^{-1} (±0.109); V*d* for evergreen broadleaves is 0.79 cm s^{-1} (±0.152).

Table 2: Mean PM_{10} deposition velocity experimentally measured and the corresponding mean sampled leaf area (± st.err.)

Vegetation group	**PM_{10} Deposition Velocity (cm s^{-1})**	**Sampled Leaf Area (cm^2)**
Deciduous broadleaves	0.54 ± 0.02	5806.3 ± 766.3
Evergreen broadleaves	0.79 ± 0.15	1536.3 ± 149.6
Conifers	1.46 ± 0.11	1816.0 ± 248.0

3.4 Air pollutants removal by vegetation and monetary evaluation

Figures 3 and 4 show the seasonal PM_{10} and O_3 removal by the urban and peri-urban forest in the Municipality; quantitative results are reported in Tables 3 and 4. Due to the scarcity of evergreen species, winter maps are not shown.

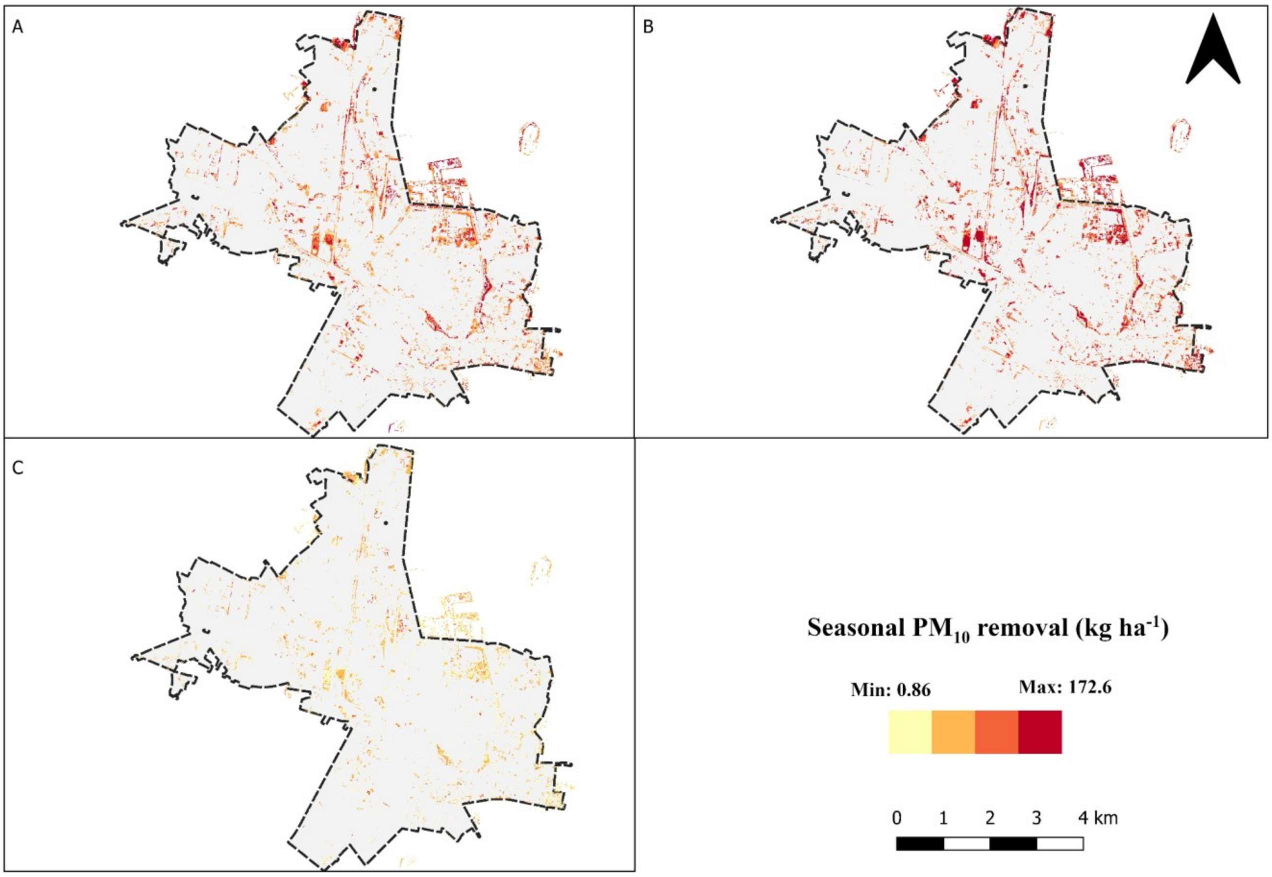


Figure 3: PM_{10} removal for spring (A), summer (B) and autumn (C). Due to the scarcity of evergreen species, the winter map is not shown

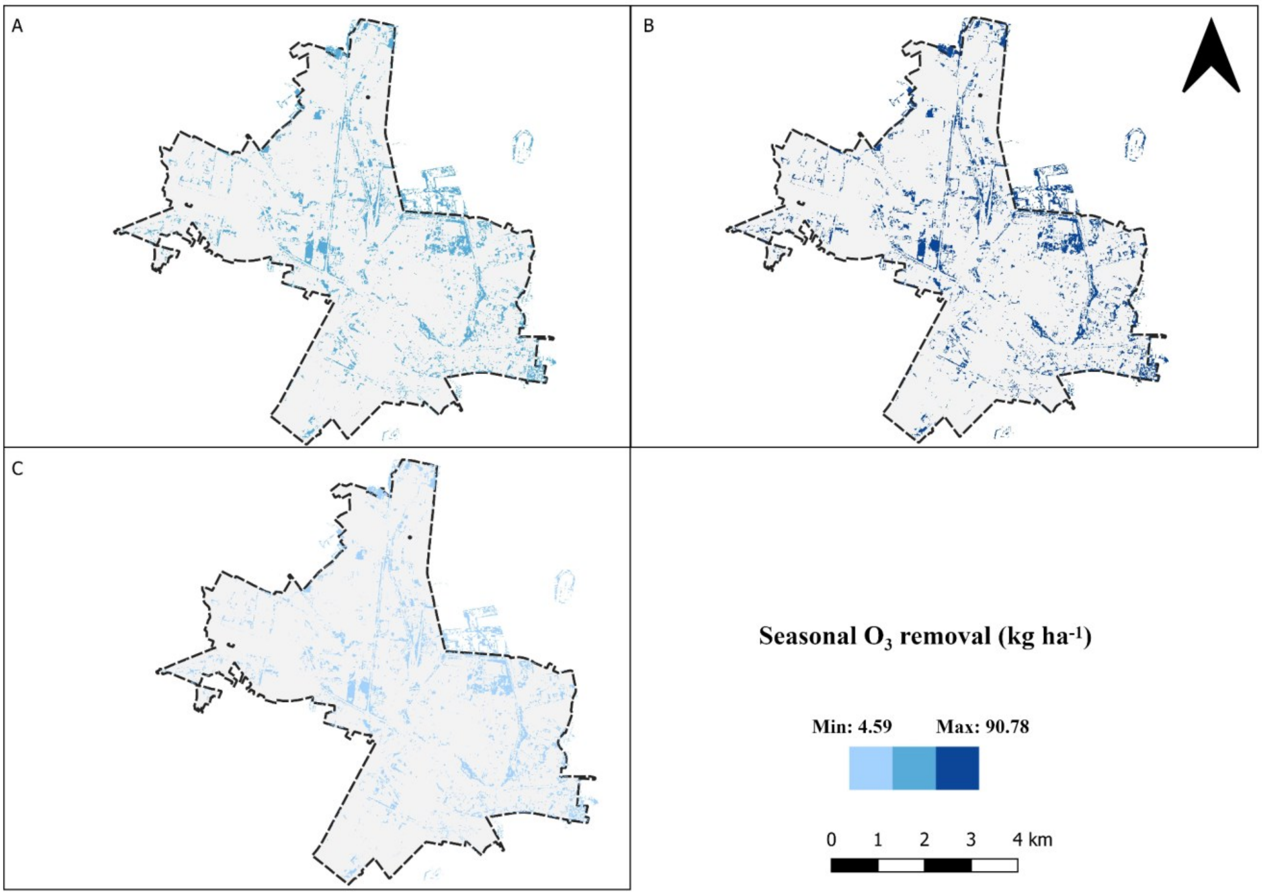


Figure 4: O_3 removal for spring (A), summer (B) and autumn (C). Due to the scarcity of evergreen species, the winter map is not shown

As for the PM_{10}, a total of 19.78 Mg have been removed by the vegetation in the Municipality for the considered year. Conifers have by far the highest annual removal rate (145.4 kg ha^{-1} y^{-1}), and their total contribution to the PM_{10} removal is about 7% of the total. The major contribution in absolute terms is provided by the deciduous broadleaves, which remove about 18.35 Mg of PM_{10}, that is, 93 % of the total. Deciduous broadleaves show the highest removal rate in summer, which is equal to 21.0 kg ha^{-1} y^{-1}. The role of the evergreen broadleaves is negligible, as those are very scarce for the study area. The monetary benefit derived from the PM_{10} removal amounts to about € 2.12 million for the considered year (2019).

Concerning the O_3 removal, a total of 8.6 Mg have been removed by the vegetation in the Municipality for the considered year (Table 3). The highest removal efficiency has been observed for deciduous broadleaves (17.52 kg ha^{-1} y^{-1}), followed by evergreen broadleaves (13.03 kg ha^{-1} y^{-1}) and conifers (9.20 kg ha^{-1} y^{-1}). The monetary benefit derived from the O_3 removal amounts to about € 147.1 *10^3 euros for the considered year (2019).

Table 3: Seasonal and annual PM_{10} removal and monetary evaluation

	Winter			***Spring***			***Summer***			***Autumn***			***Annual***		
	Kg	kg/ha	€ * 10^3	kg	kg/ha	€ * 10^3	kg	kg/ha	€ * 10^3	kg	kg/ha	€ * 10^3	kg	kg/ha	€ * 10^3
Deciduous broadleaves	0.00	0.00	0.00	6700.90	13.81	210.11	10202.30	21.02	319.90	1448.04	2.98	45.40	18351.24	37.81	575.42
Evergreen broadleaves	7.79	12.17	0.38	9.25	14.45	0.29	12.15	18.98	0.38	6.67	10.42	0.21	35.86	56.03	1.12
Conifers	283.22	29.50	0.93	518.14	53.97	16.25	315.18	32.83	9.88	279.72	29.14	8.77	1396.26	145.44	43.78
Total	291.01		1.31	7228.29		226.65	10529.63		330.17	1734.43		54.38	19783.36		620.33

Table 4: Seasonal and annual O_3 removal and monetary evaluation

	Spring			**Summer**			**Autumn**			**Annual**		
	kg	kg/ha	€*10^3	kg	kg/ha	€*10^3	kg	kg/ha	€*10^3	kg	kg/ha	€*10^3
Deciduous broadleaves	3283.43	6.76	25.61	4798.88	9.89	37.423	423.57	0.87	3.30	8505.88	17.52	66.33
Evergreen broadleaves	3.09	4.83	0.024	4.52	7.06	0.035	0.73	1.14	0.005	8.34	13.03	0.06
Conifers	34.28	3.55	0.267	50.11	5.19	0.39	4.44	0.46	0.034	88.83	9.20	0.69
Total	3320.80		25.89	4853.51		37.84	428.74		3.34	8603.05		67.09

3.5 Identification of spaces for the urban forestation and enhancement of ESs

The area respecting the established criteria for being subject to forestation actions covers about 151.5 ha within the Municipality; about 80.2% (121.5 ha) of this area is not currently covered by woody vegetation (Figure 5). In the scenario in which the 121.5 ha are forested using equally distributed evergreen broadleaves, deciduous broadleaves, and conifers, the woody vegetation of the urban core would increase by about 40.9%, passing from about 297 ha to about 418.5 ha. The yearly PM_{10} removal would increase by 9.69 Mg, that is, about 49% of the current removal. Interestingly, the winter removal would increase by 579%, which is by far the highest increase among all the seasons. As for the O_3 removal, it would increase by 1.61 Mg, that is, 18.70% of the current removal, with no relevant differences for the different seasons. Concerning the monetary benefit, its yearly increment would be equal to € 1.04 million and € $27.54*10^3$ for PM_{10} and O_3 removal respectively.

Figure 5: public green areas selected for forestation actions.

4. Discussion

Unlike other bigger cities like Rome (Manes et al., 1997; Fusaro et al., 2015), Milan (Sanesi et al., 2017), and Naples (Appolloni et al., 2018; Sebastiani et al., 2021b), the Municipality of Ferrara cannot rely on large urban parks or historical villas. This may be attributable to the highly agricultural vocation of the territory, and to the tendency in maximizing the available space for agriculture. Within the urban core, the compactness of the urban settlement reduces the available space for forestation. The current UGI in the Municipality lacks functional diversity since more than 97% of the total woody vegetation belongs to the deciduous broadleaves. According to different research (Cadotte et al., 2011; Manes et al., 2012a, 2016), the functional diversity of vegetation stabilizes the provision of regulating ESs due to differences in phenology, inherent functional traits such as stomatal conductance, and leaf morphology. Therefore, we argue that any forestation action within the Municipality should keep this in mind.

Overall, for all functional groups, most of the PM_{10} removal came during summer. This is owing to the fact that in summer the vegetation has completed the vegetative growth; thus if conditions such as water or heat stress don't take place, a high LAI, which is related to the PM_{10} removal by vegetation (Manes et al., 2016a) can be observed. The superior PM_{10} removal efficiency of conifers is due to the higher deposition velocity. The deposition velocity is influenced by the leaf's surface structure as well as by the chemical characteristics of cuticle wax (Chen et al., 2015), and several authors found out that V*d* is higher for coniferous species rather than broadleaved species (Freer-Smith et al., 2005; Popek et al., 2013; Petroff et al., 2008).

As for the O_3, the majority of the removal for all the functional groups of vegetation occurs during summer; this result was expected, as the O_3 concentrations are notoriously favored during the hot season. The highest removal efficiency is observed for deciduous broadleaves, mostly due to their higher stomatal conductance compared to the other functional groups (Manes et al., 2012b).

It is important to remark that an ESs assessment alone is not sufficient to properly select tree species for forestation actions; indeed, trees may also provide the so-called ecosystem disservices, thus generating health problems for the people living nearby (Velasco-Jim´enez et al., 2020). For example, trees may contribute to the formation of tropospheric O_3 by emitting biogenic volatile organic compounds (BVOCs), which can act as an O_3 precursor (Calfapietra et al., 2013; Fitzky et al., 2019). Furthermore, it is necessary to assess both ESs supply and demand to choose areas where the ESs should be primarily enhanced (Sebastiani et al., 2021a).

New forestation plans should pursue multiple objectives besides the environmental quality of the area, including social, economic, and cultural development (Guarini et al., 2020). In this regard, the Municipality of Ferrara is promoting participatory approaches for the reforestation of specific areas, with the involvement of citizens, local stakeholders, technicians, and other experts to identify the demand for different uses of new forested areas. Such an approach aims to design urban green areas that intercept the needs of the local population while increasing its awareness and knowledge of the ecosystem services provided by UGI.

Suitable areas for forestation (Fig. 5) could be extremely valuable, also to enhance the functional diversity of vegetation. The scenario would mostly benefit from the winter PM_{10} removal, which is the lowest in the current condition, therefore stabilizing the provision of this ES and limiting the extreme fluctuations that have been observed across different seasons. It's worth remarking that, due to the large coverage of agricultural land in the peri-urban territory and to the presence of marginal areas in the urban core, much more potential for reforestation measures relies on the possible conversion of croplands and brownfields to green areas. A feasible solution could be represented by Payments for ecosystem services (PES) schemes, which are increasingly used to preserve urban and peri-urban forests and the related ecosystem services in rural settings (Richards et al., 2019). In this sense, the monetary evaluation provided in this study may represent a justification to sustain the economic costs of agricultural land acquisition, the de-paving actions, and the PES implementation. The monetary evaluation, which is in line with the ones reported in similar studies (Manes et al., 2012a; 2016; Fusaro et al., 2017), represents a powerful instrument for connecting the ecological and economic perspectives of ESs, and is widely used for defining urban planning and management strategies (Nikodinoska et al., 2018). Our results may indicate how worth it is to invest in the enhancement of the UGI, and could easily be used by stakeholders and policy-makers as a decision support tool. Nonetheless, it should be borne in mind that other ESs such as carbon storage, flood prevention, and urban heat island mitigation (Velasco et al., 2016; Li et al., 2020; Marando et al., 2019; Maragno et al. 2018) should also be assessed for a comprehensive evaluation. Interestingly, according to the municipal budget of 2020, the Municipality invested about € 4.3 million in the maintenance of public green spaces and enhancement of the natural environment (https://openbilanci.it/), which equates to about 2.7% of the total budget. The expenses for health protection account for about € 805,000, which is lower than the monetary benefits assessed in this study. Moreover, the municipal budget does not include the benefits derived from the UGI, nor monetarily nor under any other perspective.

Our approach presents some general limitations that should be addressed. First, the removal models simplify the processes of PM_{10} and O_3 removal by the vegetation. Factors such as rainfall, wind speed, temperature, and humidity were not addressed. Then, due to the lack of more suitable data, we used average seasonal PM_{10} and O_3 concentration for running the model; this kind of input does not consider the spatial variation of pollutants within the city. As for the O_3 removal, we used the annual average g_s value of the different functional groups of vegetation; this may lead to an overestimation of the removal since conditions such as drought stress significantly reduce g_s (Fusaro et al., 2015). Finally, the air pollutants removal modelled in the reforestation scenario is based on average annual parameters applied in this study (e.g LAI, pollutants concentration, Vd, gs) for the three functional groups.

5. Conclusions

This analysis quantified the benefits in terms of air quality improvement provided by the UGI in the Municipality of Ferrara and proposes a forestation scenario aimed at enhancing the provision of ESs. The scenario involves about 121 ha of existing green areas that are not subject to landscape and historical constraints, and can be used as a starting point for a more refined land planning strategy. The forestation would generate relevant benefits by increasing the PM10 and O3 removal by about 49% and 18%, respectively.

The plantation of autochthonous tree species represents a feasible and cost-effective solution to tackle air pollution in cities with a dense urban fabric, as well as to enhance its resilience (Jim et al., 2013; Tian et al., 2014; Grote et al., 2016). The results demonstrated that the social costs of air pollution can be significantly reduced by foresting the existing scattered green areas in a city where those are fragmented and scarce, such as the case of the Municipality of Ferrara. Our findings may also help in complying with the EU Biodiversity strategy for 2030, which calls cities to develop an ambitious greening plan by the end of 2021.

The monetary benefit, which is remarkable, represents an argument for enhancing the UGI in forthcoming land planning strategies and could hopefully be included in the Municipality's budget.

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Chapter 3 Microscale Analysis

$PM_{2.5}$ is by far the most harmful pollutant for human health and for this reason, it is crucial to evaluate, as precisely as possible, the contribution of tree species to remove this pollutant. The current models used to quantify air pollution removal do not consider any difference in term of performances between different species. The study presented, modifies the currently most used model for the urban ecosystem services estimation, iTree-ECO (https://www.itreetools.org/tools/i-tree-eco), in order to consider the performance of different tree species. In particular, an index based on the leaf traits was generate. This index was used to modify the four parameters that the model consider for the removal assessment (deposition velocity, resuspension rate, washing and potential leaf water storage). Furthermore, field measurements was used to calibrate the index. The results show that tree specificity plays a fundamental role in the ability to remove pollutants, in particular $PM_{2.5}$. Moreover, this aspect cannot be ignored when ES are analysed and green areas projected. In particular, there is a greater efficacy of conifers, which, however, is highly underrated by the conventional model.

*(**Submitted as**: Gaglio M., Pace R. Muresan A.N., Grote R., Castaldelli G., Calfapietra C., Fano E.A. Species-specific efficiency in PM2.5 removal by urban trees: from leaf measurement to improved modeling estimates. Science of The Total Environment)*

Species-specific efficiency in $PM_{2.5}$ removal by urban trees: from leaf measurements to improved modeling estimates

Abstract

The growing population in cities is causing a deterioration of air quality due to the emission of pollutants, causing serious health impacts. Trees and urban forests can contribute through the interception and removal of air pollutants such as particulate matter (PM). The dry deposition of PM by vegetation depends on air pollutant concentration, meteorological conditions, and specific leaf traits. Several studies explored the ability of different plant species to accumulate PM on leaf structures leading to the development of models to quantify the PM removal. The i-Tree Eco is the most used model to evaluate ecosystem services provided by urban trees. However, fine particulate matter ($PM_{2.5}$) removal is still calculated with a poorly evaluated function of deposition velocity (which depends on wind speed and leaf area) without differentiating between tree species. Therefore, we present an improvement of the standard model calculation introducing a leaf trait index to distinguish the species effect on PM deposition. We also compared model results with measurements of leaf PM deposition by vacuum filtration. The index includes the effect of morphological and functional leaf characteristics of tree species using four parameters: leaf water storage, deposition velocity, resuspension rate and leaf washing capacity. Leaves of 11 common urban tree species were sampled in representative areas of the city of Ferrara (Italy) and at different times of the year from 2018 to 2021. This includes four deciduous broadleaf trees (*Tilia cordata*, *Platanus acerifolia*, *Acer platanoides*, *Celtis australis*), three evergreen broadleaf trees (*Quercus ilex*, *Magnolia grandiflora*, *Nerium oleander*), and four conifers (*Thuja orientalis*, *Cedrus libani*, *Pinus pinaster*, *Picea abies*). The results provide significant advancement in assessing PM removal using decision support tools such as models to properly select tree species for future urban tree planting programs aimed at improving air quality.

Introduction

As urban population is dramatically increasing worldwide, life quality and health in urban areas play a role of paramount importance for human well-being. Unfortunately, the latter is seriously affected by air pollution originating from a variety of anthropogenic sources such as auto-motor vehicles, indoor heating, industries, etc. In order to improve air quality levels, EU legislation imposed legal limits for different air pollutants (EU Directive 2008/50), which targets were often not achieved or criticized as insufficient to protect human health (Wise, 2018; Yamineva, 2017).

According to the EEA Report No 10/2019, fine particulate matter (PM2.5) caused 412,000 premature deaths in Europe in 2018, being the air pollutant with the highest impact in terms of mortality in EU. In fact, exposure to high PM2.5 levels is associated to elevate risk of cancer, cardiovascular and respiratory mortality (Pun et al., 2017) and cause severe economic costs for the society (Trejo-González et al., 2019). Moreover, the ongoing climate change is expected to further exacerbate the mortality risk in the future (Fang et al., 2013; Silva et al., 2017). The exposure to high levels of PM2.5 air concentrations also seems to increase the mortality due to COVID-19 during pandemic (Brunekreef et al., 2021). Therefore, effective measures are urgently needed to improve urban environment and protect its inhabitants. In this regard, Urban Green Infrastructures (UGI) are gaining popularity as a cost-effective solution to improve the environmental quality of cities (Keesstra et al., 2018), supplying a comprehensive set of ecosystem services to support citizens' well-being, including air purification, storm water retention, temperature regulation, recreation and habitat provision for species (Capotorti et al., 2020) with the urban tree component representing the major (Livesley et al., 2016). Among these, the capacity of urban vegetation to intercept and remove airborne particulate matter has been identified as an important process in polluted cities, where PM emissions cannot be easily reduced (Baldacchini et al., 2017; Manes et al., 2016). Airborne

PM removal by vegetation is a complex result of multiple processes encompassing background pollution levels (i.e. PM air concentrations), meteorological and climatic conditions (e.g. wind speed, temperature and precipitation intensity) (Beckett et al., 2000; Nowak et al., 1998) and specific plant morphological characteristics (Corada et al., 2021; Sgrigna et al., 2020). In particular, two main processes can be identified: i) the gravimetric sedimentation of particles on leaves or complex inertial impacts and ii) their subsequent retention on leaf surface to avoid resuspension until a precipitation event.

In the last decade, a growing number of studies demonstrated that PM removal depends on specific morphological traits of tree leaves and crown (Corada et al., 2021; Grote et al., 2016; Sgrigna et al., 2020) and that, consequently, plant species perform differently in removing airborne PM. However, the available literature provides a fragmented knowledge of the interaction between morpho-functional traits and PM removal, as well as their upscaling towards a quantification of PM removed by vegetated areas. Some leaf attributes, as trichomes density and cuticle wax content, were associated with a higher fine PM removal capacity. Foliar trichomes have the first contact with atmospheric PM thus favoring their adsorption and reducing resuspension (L. Zhang et al., 2019). In addition, it seems that the presence of trichomes also affect leaf wettability (Bickford, 2016) which, in turn, influences PM deposition (Wang et al., 2013). Leaves with higher content of cuticle waxes can adsorb and retain (i.e. avoid or mitigate resuspension phenomena) larger amounts of fine PM (Dzierzanowski et al., 2011; Sgrigna et al., 2015), while no analogous effects were clearly demonstrated for coarser PM (Popek et al., 2013). Some authors also stated that composition and structure of the wax may influence PM deposition (Leonard et al., 2016; Wang et al., 2015a), further supporting the evidences for species-specific properties. Similar effects can be also attributed to leaf secretions. Even though more investigations are needed to clarify their role, some studies observed that sticky leaf surfaces capture and retain more PM than smooth leaves (Corada et al., 2021; Sgrigna et al., 2020). PM removal is also influenced by macro-level morphological and geometrical traits. Higher dry deposition levels are associated with leaves with roughness surfaces, due to the presence of veins, wrinkles and ridges (Zhao et al., 2019), as well as complex leaf shape (Son et al., 2019). Also, petiole length was found to be negatively correlated with PM deposition velocity (v) (Zhang et al., 2021): a shorter petiole may reduce leaf movement thus enhancing the velocity of particle deposition (Prusty et al., 2005), while a longer petiole enables leaves to flutter and causes resuspension (Leonard et al., 2016).

A further parameter that influences PM removal capacity is the washing performed by rainfall. In fact, intense precipitation events cause PM to be washed off from leaves, recovering their capacity to filter airborne PM. Even though some studies demonstrated that rainfall intensity and duration are most relevant parameters affecting PM deposition (Weerakkody et al., 2018; R. Zhang et al., 2019), it should also be noted that plant can retain part of the deposited PM according to their leaf traits after a rain event (Xu et al., 2017) - particularly in the case of fine PM (Xu et al., 2019). Leaf traits such as presence of sticky secretions, epidermal waxes, trichomes density and patterns, stomatal density, shape of epidermal cells, and leaf water status, may help avoiding a complete leaf washing during moderate rain events possibly due to their influence on leaf wettability (Wang et al., 2015b).

The application of UGI as nature-based solutions for improving air quality and its implementation in policy requires reliable information of PM retention at a large scale (e.g. city or neighborhood levels). The biophysical upscaling of airborne PM2.5 removal assessment is often based on the use of dedicated models and tools as a basis for decision-making processes. Nonetheless, the reliability of such assessments depends on model accuracy, which is seldom validated with field measurements and is seriously affected by simplistic model assumptions. Among such models, i-Tree Eco (www.itreetools.org) is the most popular software suite for the estimation of a number of ecosystem services provided by urban trees and has been widely used to estimate PM2.5 removal (Pace et al., 2018; Selmi et al., 2016; Song et al., 2020). This tool models dry deposition of particles as a function of air pollution levels, meteorological data, and Leaf Area Index (LAI) of trees. However, the model does not consider any variability in species-dependent deposition, but rather uses a standard average value of deposition velocity obtained from a range of literature values (Nowak et al., 2013). Pace and Grote (2020) found that even just implementing differences of deposition velocities between tree functional types (i.e. conifers, broadleaves and evergreen broadleaves) in i-Tree Eco leads to consistent variations in model outputs. Model estimates were also compared with experimental measurements at the canopy and leaf level of a Mediterranean holm oak forest (Pace et al., 2021), but there are no studies in the literature that systematically evaluate and compare the impact of different tree species properties on the calculation of PM deposition.

With the aim to rank and predict the PM removal performance of different urban tree species, some authors combined a set of leaf morphological traits by means of statistical models (Chen et al., 2017; Li et al., 2019; Muhammad et al., 2019), ranking criteria (Yang et al. 2015) or to compute indexes that quantify PM deposition (Sgrigna et al., 2020). However, such efforts were not implemented in existing models to improve the assessment of air purification capacity of UGI to support decision processes. In fact, despite the rapid increasingly body of literature demonstrating their relevance, to date there is no clear and effective application of morpho-functional leaf traits in PM removal modelling (Corada et al., 2021).

The aim of this work is to modify/improve the i-Tree Eco model by the implementation of different species-specific properties that influence the removal of PM2.5 from the air. By doing this, leaf and crown traits of each species were combined to compute a score-based index, which was validated/calibrated using primary measures obtained from field samplings of different urban trees species. The index is based on four model parameters that influence PM2.5 deposition on leaves: i) potential leaf water storage, ii) dry deposition velocity, iii) resuspension rate and iv) rain washing. We assume that this work represents a significant advancement on PM removal modelling that is urgently needed for improving our estimates on PM deposition and to guide planners and management to the choice of the best tree species for this purpose.The bio-physical upscaling of airborne $PM_{2.5}$ removal assessment is often based on the use of dedicated models and tools as a basis for decision making processes. Nonetheless, the reliability of such assessments relies on model accuracy, which is seldom validated with field measures and is seriously affected by simplistic assumptions on which the models are based. Among these, iTree Eco (www.itreetools.org) is the most popular software suite for the estimation of a number of ecosystem services provided by urban trees and has been widely used to estimate $PM_{2.5}$ removal (Pace et al. 2018; Selmi et al. 2016; Song et al. 2020). This tool models dry deposition of particles as a function of air pollution levels, meteorological data and Leaf Area Index (LAI) of trees. However, the model does not consider any variability in species performance, but it rather uses a standard average value of deposition velocity obtained from a range of literature values. Pace and Grote (2020) found that even just implementing differences of deposition velocities between tree functional types (i.e. conifers, broadleaves and evergreen broadleaves) in iTree Eco lead to consistent variations in model outputs.

With the aim to rank and predict the PM depuration performance of different urban tree species, some authors combined a set of leaf morphological traits by means of statistical models (Chen et al. 2017; Li, Wang, and Chen 2019; Muhammad, Wuyts, and Samson 2019), ranking criteria (Yang et al. 2015) or to compute indexes that quantify PM accumulation (Sgrigna et al. 2020). However, such efforts were not implemented in existing models to improve the assessment of air depuration capacity of UGI and support decision processes. In fact, despite the rapid increasingly body of literature demonstrating their relevance, to date there is no clear and effective application of morpho-functional leaf traits in PM removal modelling and hence decisions for species selection can be hardly based on them (Corada et al. 2021).

The aim of the work is to modify/improve the iTree Eco model by the implementation of the performances of different species in removing $PM_{2.5}$ from the air. By doing this, leaf and crown traits of each species were combined to compute a score-based index, which was validated/calibrated using primary measures obtained from field samplings of different urban trees species. The index is based on four model parameters that influence $PM_{2.5}$ accumulation on leaves: i) potential leaf water storage, ii) dry deposition velocity, iii) resuspension rate and iv) washing. The work provides a significant advancement on PM removal modelling by the inclusion of leaf traits and species-related effects that is urgently needed for the quantification of air depuration services ant to inform the environmental management of urban areas.

2. Material and methods

2.1 Study area and laboratory measures

The city of Ferrara is located on the Po plain (Fig.1), an extended flat area in the Northern Italy which is subjected to high concentration levels of air pollutants, due both to the regional emissions and its particular geomorphological conformation. The municipality of Ferrara covers an area of 405.16 km^2, at approximately 9-13m a.s.l. The annual precipitation is about 814 mm yr^{-1}, while the mean annual temperature is 14.6 °C. The area can be classified as "Cfa" according to the Köppen-Geiger climate classification (Peel et al. 2007). Prevalent wind directions are W in winter and NE in summer, with an annual mean speed of 3.4 m s^{-1}. According to the municipality census of public trees (Ferrara Municipality, 2018), *Celtis austalis*, *Tilia europea*, *Platanus acerifolia* are the most abundant species.

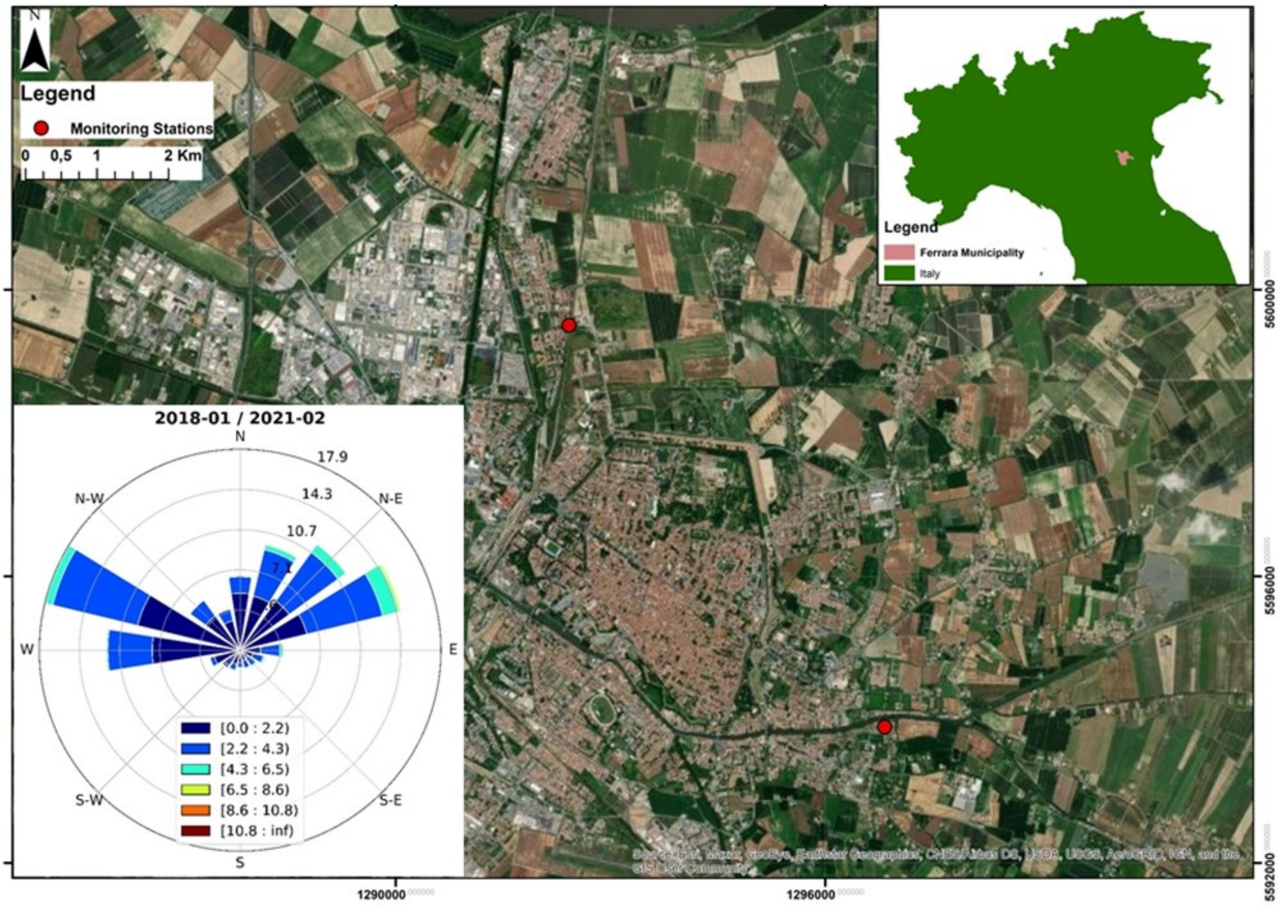

Figure 1: Ferrara municipality and the two monitoring stations for $PM_{2.5}$ air concentration and climate data. The panel in the back-left reports wind speed and direction for the sampling period (summer 2018 – winter 2021).

Experimental measures of $PM_{2.5}$ adsorbed by 11 urban trees species were used to validate the modelled results. These include four deciduous broadleaves (*Tilia cordata, Platanus acerifolia, Acer pseudoplatanus, Celtis australis*), three evergreen broadleaves (*Quercus ilex, Magnolia grandiflora, Nerium oleander*) and four conifer species (*Thuja orientalis, Cedrus libanii, Pinus pinaster, Picea abies*). An additional measure for *Q. ilex* and *T. orientalis* was performed to obtain a second value for evergreen and conifer species. The trees were located nearby the two city monitoring stations for $PM_{2.5}$ (Fig.1) in order to obtain reliable measures of PM air concentrations before and during sampling events. For each tree, diameter at breast height and crown diameter were measured *in situ*. Samplings were carried out between June 2018 and February 2021. Twelve samples of branch leaves were taken for each tree species (three samples x four trees) in the city of Ferrara, for a mean leaf area of 3151 cm^2 per sample (min 691 cm^2, max 14182 cm^2), stored in plastic bags, placed in portable fridges, and quickly taken to the laboratory. The amount of $PM_{2.5}$ deposited on leaf surface was measured by vacuum filtration method. Leaves were singularly washed in 500 ml of micro-distillated water. The solution with suspended particles was filtered in a plastic sieve (100 µm pore size) to remove coarser matter and subsequently forced through filters using a vacuum filtration system. PM>10 µm was removed from the solution using filters with 10-13 µm porosity (Anoia S.A., Barcelona, code 1250). Coarse PM (between 2.5 and 10 µm) was collected using filters with 2.5 µm size (Whatman No. 42 Cat No. 1442-055, Maidstone, UK), while fine PM fraction (<2.5 µm) was collected on nitrocellulose filter with 0.2 µm pore size (Whatman NC20). All filters were dried at 50°C for 72h before and after the vacuum filtration and then stabilized in glass boxes before being weighted through electronic precision balance (precision 0.0001 g, Crystal 100 CAL, Gibertini). The difference between the first and second weight expresses the PM deposition on the analyzed amount of leaves of the same sample. The $\mathrm{PM}_{0.2–2.5}$ load was assumed to be representative of the total $\mathrm{PM}_{2.5}$, while PM_{10} loads were obtained by summing the $\mathrm{PM}_{0.2–2.5}$ and $\mathrm{PM}_{2.5–10}$ size fraction loads. The results were then normalized by dividing the PM mass by the sample leaf area and expressed in µg cm^{-2}. Because of the different morphology of their leaves, the total leaf area of each sample was measured differently for broadleaves and conifers. Total leaf area of broadleaf samples was measured through ImageJ (Schneider et al. 2012) upon direct leaf scan. Leaf area has been considered as double sided. Leaf area of conifers was estimated according to the method described in the patent n. CN104990530A (2015), except for *T. orientalis* samples that was measured by leaf scanning due to their specific leaf conformation. Specifically, leaf area of conifers was estimated by assuming that needles have a cylindrical shape. The surface of a single needle was estimated by multiplying needle length by needle girth. The latter was measured making the cross-sectional slice in the middle part of needle. The individual surfaces of 20 needles were measured to obtain a mean value and then multiplied by the number of needles present in each branch sampling.

2.2. Removal index and model implementation

The PM2.5 deposition flux was calculated according to the method implemented in the i-Tree Eco model (Nowak et al., 2013). Morphological and functional leaf traits were combined to obtain an index that, in turn, was implemented in the model to include the species-related effects in PM2.5 removal. The index encompasses four parameters (j, k, i, w) j, that influence deposition velocity (j), resuspension (k), leaf washing (w), and canopy water storage (i):

$$f_t = Vd_t \cdot j \cdot C \ \cdot LAI$$

$$R_t = (A_{t-1} + f_t) \cdot \frac{rr_t}{100 \ \cdot \ k}$$

$$A_t = \begin{cases} A_{t-1} + f_t - R_t & \text{if } rain \leq plws \ \cdot LAI \cdot i \\ (A_{t-1} + f_t - R_t) \cdot \left(1 - \frac{1}{w}\right) & \text{otherwise} \end{cases}$$

$$F_t = \begin{cases} f_t - R_t & \text{if } rain \leq plws \ \cdot LAI \cdot i \\ 0 & \text{otherwise} \end{cases}$$

where ft is the $\mathrm{PM}_{2.5}$ flux at time t (g m−2 s−1), Vdt is the deposition velocity at time t (m s−1), C is the $\mathrm{PM}_{2.5}$ air concentration (g m−3), LAI is the leaf-area index, Rt is the PM2.5 flux resuspended in the atmosphere at time t (g m−2 s−1), At is $\mathrm{PM}_{2.5}$ mass deposited on leaves at time t (g m−2) depending on previous hour deposition as well as precipitation (At-1), rrt denotes a 'resuspension

class', which is the relative amount of deposited $PM_{2.5}$ that is resuspended at a specific wind speed at time t (%), and Ft is the net $PM_{2.5}$ removal at time t after considering resuspension.
The leaf area index of tree species has been calculated based on the Beer-Lambert Law (Nowak et al., 2008):

$$LAI = \frac{\left[\frac{\ln(1-x_s)}{-k}\right] \cdot \pi r^2}{CC}$$

Where xs is the species-specific shading factors (Nowak, 1996a), k is the light extinction coefficient (0.52 for conifers and 0.65 for hardwoods) (Nowak, 1996b), r is the crown radius (m), and CC is the crown cover (m2).

The deposited PM2.5 on leaves (At) refers to square meters of crown cover and therefore has been rescaled by the LAI to compare it with leaf measurements.

Deposition velocities (vdt) and resuspension classes (rrt) both depend on wind speed and are defined based on the i-Tree Eco model standards (Nowak et al., 2013). When precipitation events are higher than the maximum water storage of the canopy (Ps in mm), which is calculated according to the potential leaf water storage plws (0.2 mm) and LAI (Ps = plws * LAI), PM2.5 deposition on leaves (At) is assumed to be washed off according to the washing rate (w), and Ft are set to 0.

Leaf on/off dates were set based on the dates of the first (DOY 63) and last frost events (DOY 320), assuming a 30-day transition period in spring when trees produce leaves reaching maximum LAI and in fall when they lose leaves leading LAI to 0.

The assessment of leaf traits was performed with direct measures and information derived by literature (Kattge et al., 2020; Keating, 2009), as specified in Table 1. Leaf size, needle length, shape index and petiole length were measured by data previously obtained by scanned images and needle measurement. Trichome density (number of trichomes per area unit) and surface roughness (% of leaf surface covered by groves, ridges and wrinkles) were quantified by means of scanning electron microscopy (SEM) images. After drying and mounting, the leaf specimens were cutted into squares (1 cm2), coated with gold in an Edwards S150 sputter coater and examined with a Zeiss EVO 40 SEM. Three images of each species were captured as replications. Presence of veins and cuticle waxes were assessed using literature information (Manual of leaf architecture, 2009; Kattge et al., 2020). Stickiness was evaluated by direct observations (presence of sticky organic material on leaves).

Table 1 Set of leaf traits used in model improvement. Method of assessment, scores rationale and assumption on which they were attributed are reported

Leaf trait	Assessment	Description	Assumptions
Leaf size	Direct measure	Measured with leaf scanning	Larger leaves need larger water amounts for a complete washing
Needle length	Direct measure	Directly measured	Larger needles need larger water amounts for a complete washing
Veins	MLA, 2009	Veins order (from 1 to 4)	Veiny leaves have higher deposition velocity due to their rougher surfaces
Trichomes	Direct measure	Measured by SEM images	Trichomes increase particles capture by increasing their deposition velocity
Cuticle waxes	Kattge et al. (2020)	Abundant/Mod. present/absence	Waxes increase particles capture by increasing their deposition velocity
Petiole length	Direct measure	Measured with leaf scanning	Longer petiole favor particle resuspension by increasing leaf movement
Shape index	Direct measure	Measured with leaf scanning (length/width ratio)	Narrow leaves are less subjected to resuspension phenomena
Roughness	Direct measure	Measured by SEM images	Presence of groves, ridges and wrinkles on leaf surface prevents resuspension of trapped particles
Stickiness	Direct observation	Presence/absence	Sticky secretions can retain PM even after rain events

A score was attributed to each leaf trait according to Table 2 and then computed to obtain a final value for each of the four parameter (i, j, k, w) as following:

$$\text{i} = \begin{cases} leaf\ area & if broadleves \\ needle\ lenght & if conifer \end{cases}$$

$$\text{j} = \begin{cases} {(veins + trichomes + waxes)}/{3} & if broadleves \\ 3 & if conifer \end{cases}$$

$$\text{k} = \begin{cases} {(petiole\ lenght + shape\ index + roughness)}/{3} & if broadleves \\ 3 & if conifer \end{cases}$$

$$\text{w} = \begin{cases} 2\ \times\ Stickyness & if broadleves \\ 3 + Stickyness & if conifer \end{cases}$$

Table 2: Model parameters and the respective leaf traits used to attribute the respective scores

Model parameter	Leaf trait	Classes	Score
Canopy water storage (i)	Leaf area (mm^2)	<225	1
		225-4500	2
		>4500	3
	Needle length (cm)	<2	1
		2-4	2
		>4	3
Deposition velocity (j)	Veins	1° order	1
		2° order	2
		3°- 4° order	3
	Trichomes (number/mm^2)	<33	1
		33-66	2
		>66	3
	Waxes	Absent	1
		Moderately present	2
		Abundant	3
Resuspension (k)	Petiole length (cm)	>2	1
		1-2	2
		<1	3
	Shape index (adimentional)	>3	1
		1.5-3	2
		<1.5	3
	Roughness (% coverage)	<33	1
		33-66	2
		>66	3
Washing (w)	Stickyness	Presence	1
		Absence	0

2.3 Comparative analysis

In order to test the performance of the leaf trait-based approach, the experimental measures were matched with the $PM_{2.5}$ accumulated flux computed by the modified model. Additionally, the latter was also compared with the results of the conventional iTree Eco model to detect the differences induced by the modified approach. Both models were run for the period ranging from 1st January 2018 to 22th February 2021. This period covers more than three complete years until the date of the last field sampling, thus to evaluate the seasonal performances of models for the different species. Moreover, the results of the modified model allowed the calculation of the mean annual $PM_{2.5}$ removal per space unit (g m^{-2} yr^{-1}) for each species. Such outcome can inform environmental managers about the removal performance of different species and address urban planning towards the planting of most effective species to improve air quality in cities.

3. Results

3.1. Meteorological conditions and experimental analysis

The meteorological conditions recorded during the sampling period are shown in Figure 2. Mean monthly $PM_{2.5}$ concentrations followed seasonal patterns, with higher values observed in winter because of heating and traffic emissions. $PM_{2.5}$ concentrations in Barco and Villa Fulvia stations were comparable, even though the first recorded slightly higher concentrations in the whole period. For the years 2018, 2019 and 2020, the mean annual $PM_{2.5}$ concentrations were respectively 21.8, 21.1 and 23.3 $\mu g\ m^{-3}$ for Barco station and 16.4, 17.0 and 16.9 $\mu g\ m^{-3}$ for Villa Fulvia station. These values fall below the 25 $\mu g\ m^{-3}$ legal limits foreseen by EU Directive 2008/50/CE, but exceed the recommended WHO threshold (10 $\mu g\ m^{-3}$). Mean monthly wind speed for the sampling period was 2.21 $m\ s^{-1}$, ranging from 1.71, to 2.77 $m\ s^{-1}$. The highest monthly precipitations were 162.6 mm in May 2019 and 160 mm in November 2019.

The results of $PM_{2.5}$ mass deposited on leaves (At) are reported in Table 3. The higher depositions were found in *T. orientalis* (44.6 ±10.1 $\mu g\ cm^{-2}$), while the lowest were measured in *P. x acerifolia* (2.1 ±0.8 $\mu g\ cm^{-2}$). It worth to be noted that these values do not represent the air depuration capacity of species but rather the $PM_{2.5}$ deposited on leaves at the sampling date, as a function of previous meteorological conditions and air PM concentrations at the sites. Therefore, the observed values were used to test and calibrate the improved version of the model.

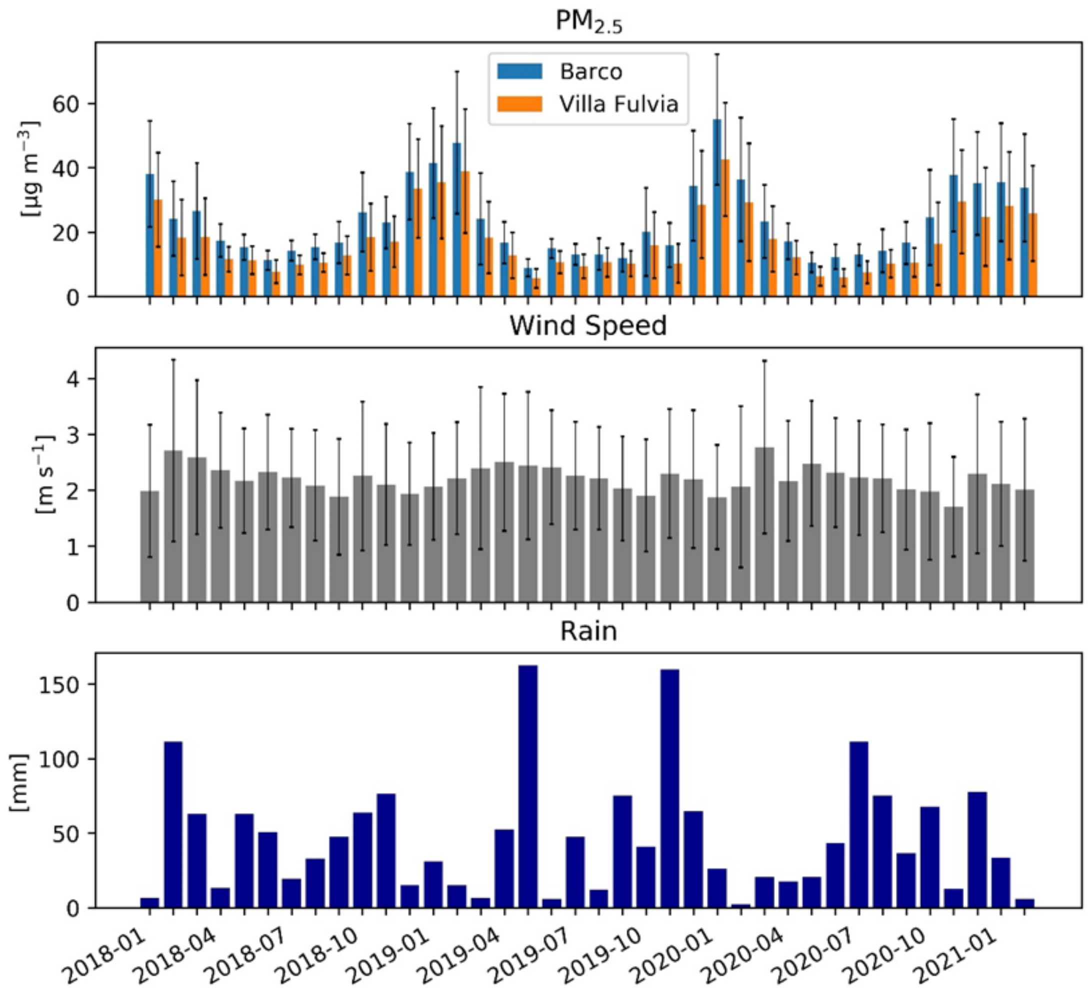

Figure 2 Mean monthly PM2.5 concentrations in Barco and Villa Fulvia stations (upper panel), mean monthly wind speed (central panel) and monthly precipitations (lower panel). Values represent mean daily values (±st.dev) for PM concentration and wind speed, while cumulated values for rain.

Table 3 Mean PM2.5 mass deposition per leaf unit (At) for each sampled species. Sampling stations and dates are reported

Species	Type	Station	Sampling date	Mean At (±St.err.) (µg cm^{-2})
Tilia cordata	deciduous	Barco	6/21/2018	3.5 (±0.6)
Quercus ilex	evergreen	Villa Fulvia	1/17/2019	2.5 (± 0.5)
Acer platanoides	deciduous	Villa Fulvia	7/26/2019	2.6 (±0.6)
Celtis australis	deciduous	Villa Fulvia	7/26/2019	3.2 (±0.7)
Platanus x acerifolia	deciduous	Villa Fulvia	7/26/2019	2.1 (±0.8)
Thuja orientalis	conifer	Barco	7/26/2019	14.2 (±2.7)
Cedrus libanii	conifer	Villa Fulvia	1/13/2020	25.0 (±3.0)
Pinus pinaster	conifer	Villa Fulvia	1/13/2020	26.9(±2.7)
Magnolia grandiflora	evergreen	Barco	1/13/2020	14.6 (±1.0)
Picea abies	conifer	Villa Fulvia	2/22/2021	26.3 (±2.4)
Nerium oleander	evergreen	Villa Fulvia	2/22/2021	26.2(±4.3)
Quercus ilex	evergreen	Villa Fulvia	2/22/2021	3.6 (±0.5)
Thuja orientalis	conifer	Barco	2/22/2021	44.6 (±10.1)

Table 4 Scores attributions for the four parameters (i, j, k and w) based on leaf traits values (Shape index =leaf length/leaf width).

Species	***plws***			***vds***				***rr***				***washing***	
	Leaf size (mm^2)	Needle length (mm)	**i-Index**	Veins	Trichomes (n./mm^2)	Cuticle waxes	**j-Index**	Petiole length (cm)	Shape index	Roughness (%)	**k-Index**	Stickyness	**w-Index**
Tilia cordata	2174.4	-	**2.0**	4°	>100	Absent	**2.5**	3.02	1.24	0	**2.0**	Present	**2.0**
Quercus ilex	664.1	-	**1.0**	2°	>100	Moderate	**2.0**	1.22	2.74	0	**2.0**	Absent	**0.0**
Acer platanoides	5981.5	-	**3.0**	3°	16	Absent	**2.0**	10.00	0.93	60	**2.0**	Absent	**0.0**
Celtis australis	237.2	-	**1.0**	2°	83	Absent	**2.0**	1.53	2.83	5	**2.0**	Absent	**0.0**
Platanus x acerifolia	6936.4	-	**3.0**	1°	0	Absent	**1.0**	1.79	1.08	65	**2.5**	Absent	**0.0**
Thuja orientalis	-	118.6	**3.0**	-	-	-	**3.0**	-	-		**3.0**	Present	**4.0**
Cedrus libanii	-	24.4	**1.0**	-	-	-	**3.0**	-	-		**3.0**	Absent	**3.0**
Pinus pinaster	-	144.2	**3.0**	-	-	-	**3.0**	-	-		**3.0**	Absent	**3.0**
Magnolia grandiflora	8047.9	-	**3.0**	1°	>100	Abundant	**2.5**	2.35	1.97	0	**1.5**	Absent	**0.0**
Picea abies	-	16.8	**1.0**	-	-	-	**3.0**	-	-		**3.0**	Absent	**3.0**
Nerium oleander	1908.0	-	**2.0**	4°	>100	Moderate	**2.5**	0.77	6.99	85	**2.5**	Absent	**0.0**

3.2. Leaf traits and model results

The four model parameters (i, j, k and w) were computed on the basis of the specific leaf traits classified for each of the 11 considered species, as are reported in detail in Table 4. Conifers have larger values for i, j and w indices that respectively influence deposition velocity (vd), resuspension rate (rr) and washing. Some traits were peculiar of few of the considered species. For example, an abundant amount of sticky substances was evident in leaves of *T. cordata* and *T. orientalis*. Higher coverage of rough structures on leaf surfaces was observed in *N. oleander* and, in lesser extent, in *P. acerifolia* and *A. platanoides*. Interestingly, these two species were also those with lower trichome density. Afterwards, the modified model was run to be tested with the experimental At reported in Table 3. Its performance is shown in Fig. 3, which contrasts experimentally observed vs. model predicted values of PM deposition. Figures 4, 5 and 6 respectively show $PM_{2.5}$ deposition results for deciduous broadleaves, evergreen broadleaves, and conifers. The improved model successfully predicted PM depositions for deciduous and conifer species, while was less accurate for evergreen broadleaves. The improved model underestimated both experimental measures of *Q. ilex*. In contrast, PM deposition of *M. grandiflora* and *N. oleander*, as well as the second measures of *T. orientalis,* resulted slightly overestimated. However, the model reached such values during winter season in all the previous years (Fig. 5 and 6).

3.3. Models comparison and species performances

The modified model was also compared with the conventional i-Tree-Eco model to describe differences in $PM_{2.5}$ deposition for different species. The standard i-Tree-Eco model markedly underestimates PM deposition for all the considered species, except fot *Q. ilex* (Fig. 4). In particular, conifers accumulated higher amount on PM mass by one factor (Fig.5). In general, PM deposition is significantly higher in conifers than deciduous broadleaves.

The results also highlighted difference in PM deposition among species and tree types. Conifers reached higher PM values both in all seasons when compared with evergreen and deciduous broadleaves. Evergreen broadleaves revealed contrasting results. Deposited flux of *Q. ilex* was similar to those of deciduous species, while the performance of *N. oleander* was comparable with those of conifer species. Interestingly, higher PM deposition fluxes do not necessary correspond to higher PM removal, as demonstrated by computing the total $PM_{2.5}$ removal per year for each species (Fig. 7). For instance, although *T. orientalis* showed higher PM deposition (Fig.5), higher annual PM removal values were observed for *P. abies*, *C. libanii* and *N. oleander*. It has to be mentioned, however, that the total PM removal reported in Fig. 7 is calculated for the whole tree and is therefore influenced by the related leaf area index, while PM accumulation fluxes are normalized per leaf area unit.

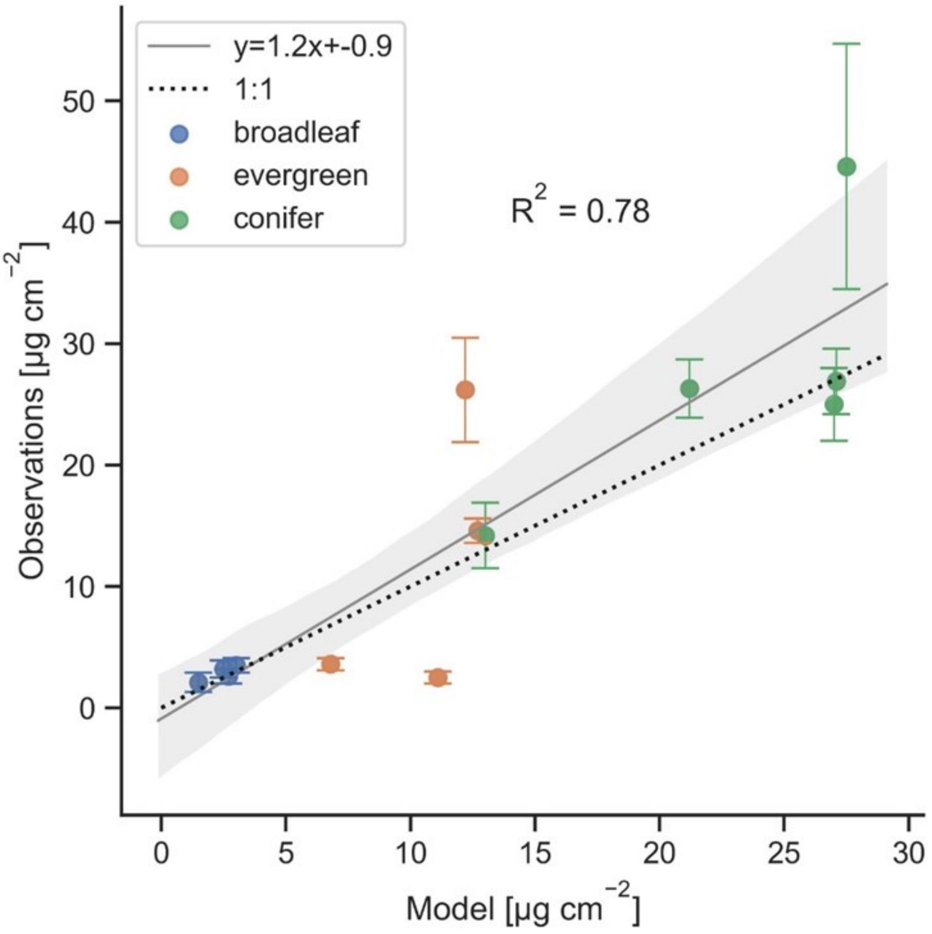

Figure 3: Mean leaf PM deposition values observed experimentally (bars represent st.err.) vs. values predicted by the improved model. Dashed line represents the ideal condition of perfect match between observed vs. predicted values (1:1 ratio). Solid line represents the fittest regression curve (grey area corresponds to 95% confidence interval) (R2=0.78, p<0.05).

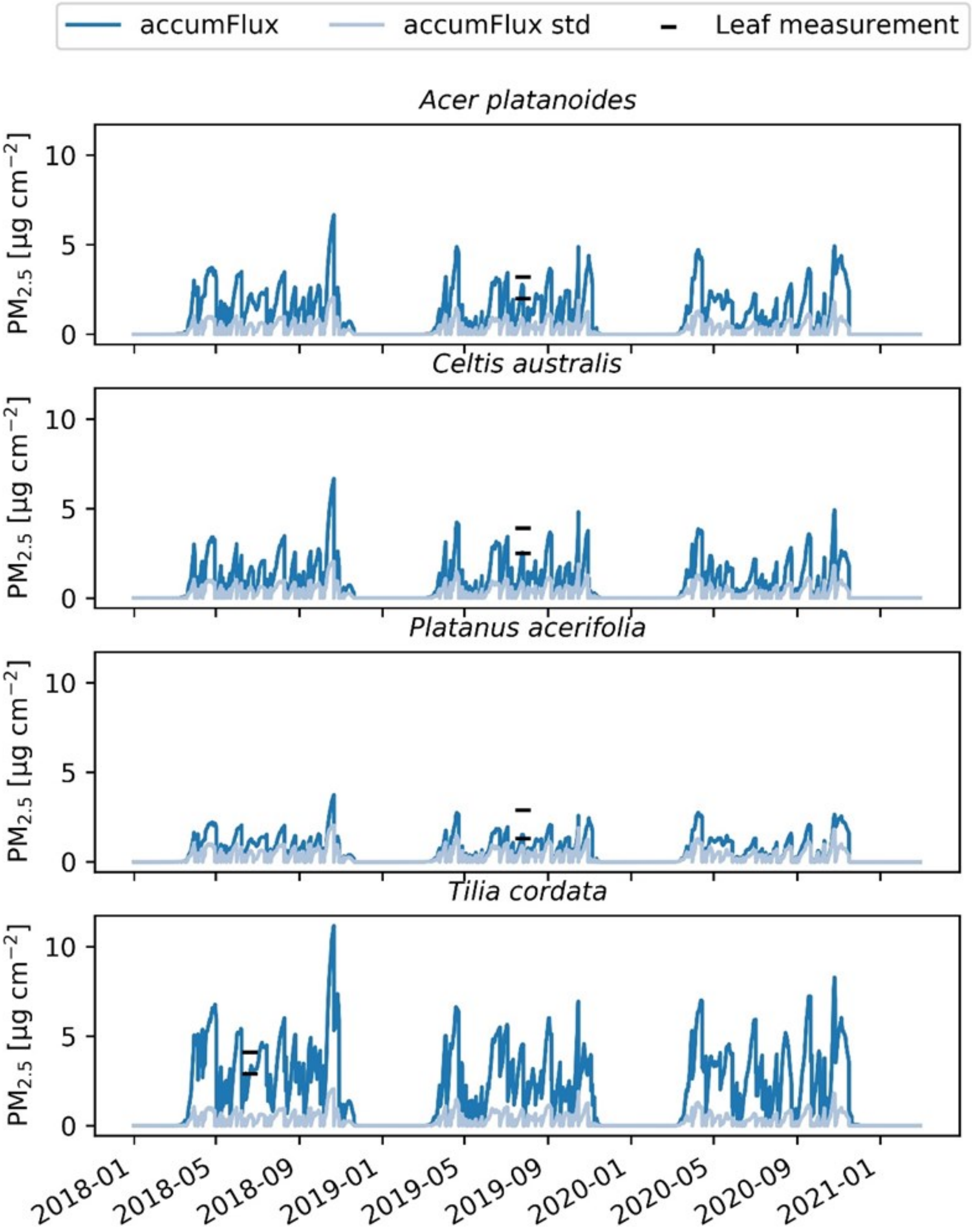

Figure 4 $PM_{2.5}$ deposition fluxes for deciduous broadleaf species during the considered period (01-2018; 02-2021) as modelled by standard (light blue line) and modified (bright blue line) models. The experimental measures were shown in black line (mean ± st.err.).

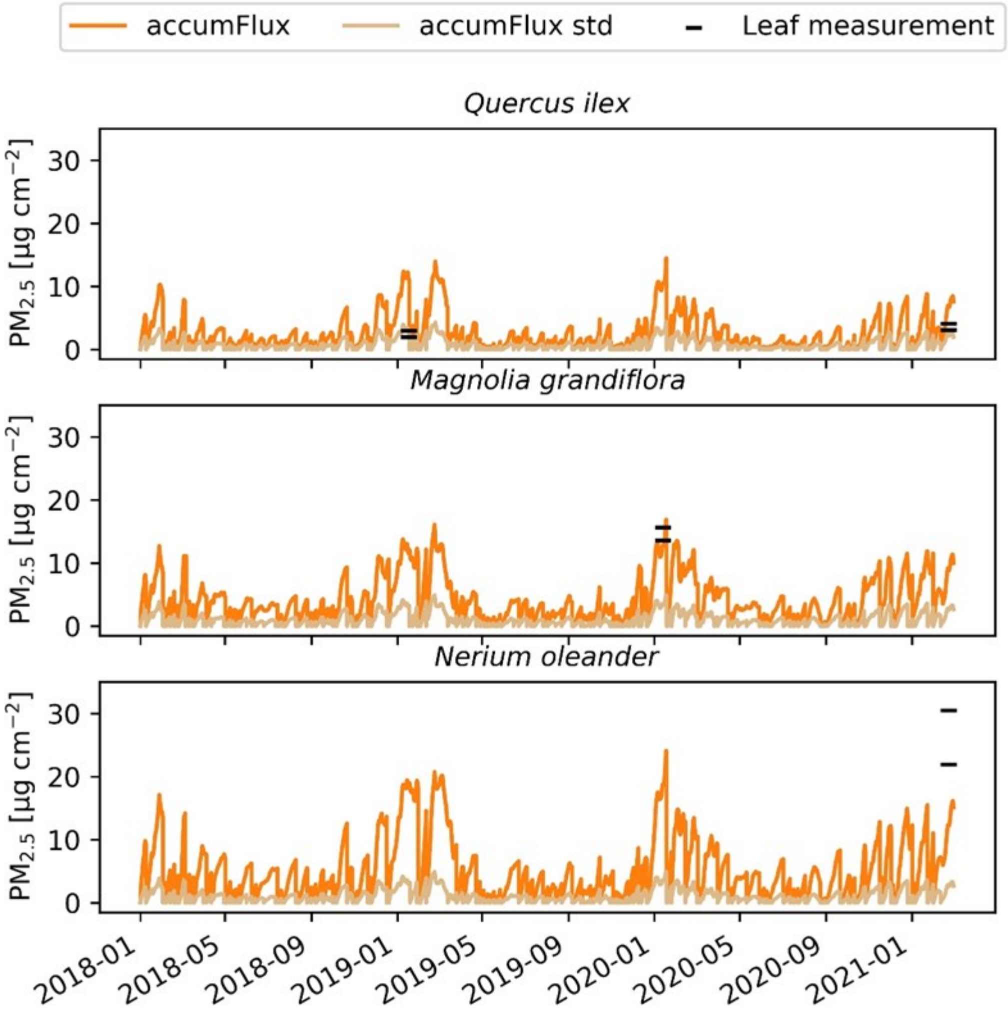

Figure 5 $PM_{2.5}$ deposition fluxes for evergreen broadleaf species during the considered period (01-2018; 02-2021) as modelled by standard (light orange line) and modified (bright orange line) models. The experimental measures were shown in black line (mean ± st.err.).

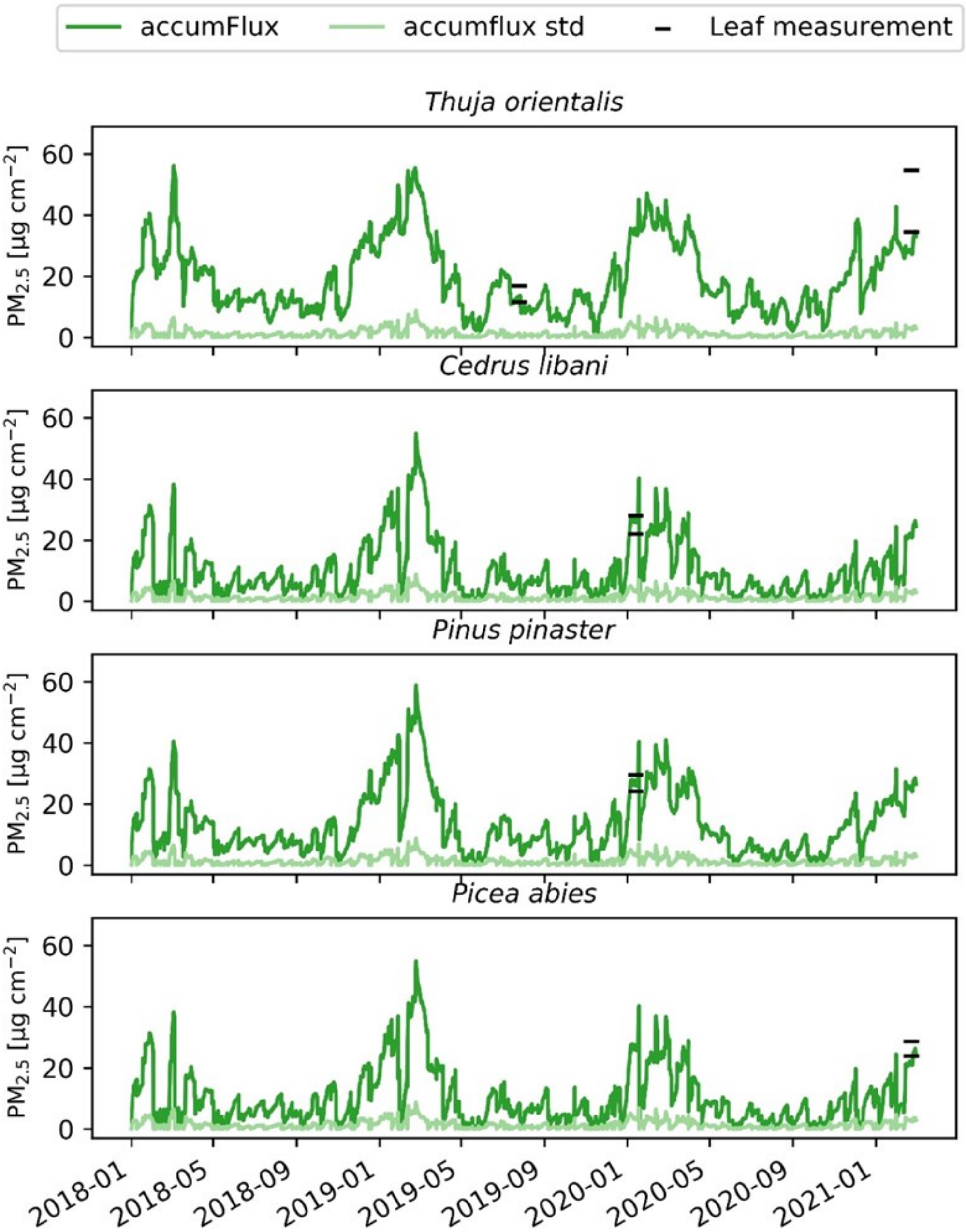

Figure 6 $PM_{2.5}$ deposition fluxes for conifer species during the considered period (01-2018; 02-2021) as modelled by standard (light green line) and modified (bright green line) models. The experimental measures were shown in black line (mean ± st.err.).

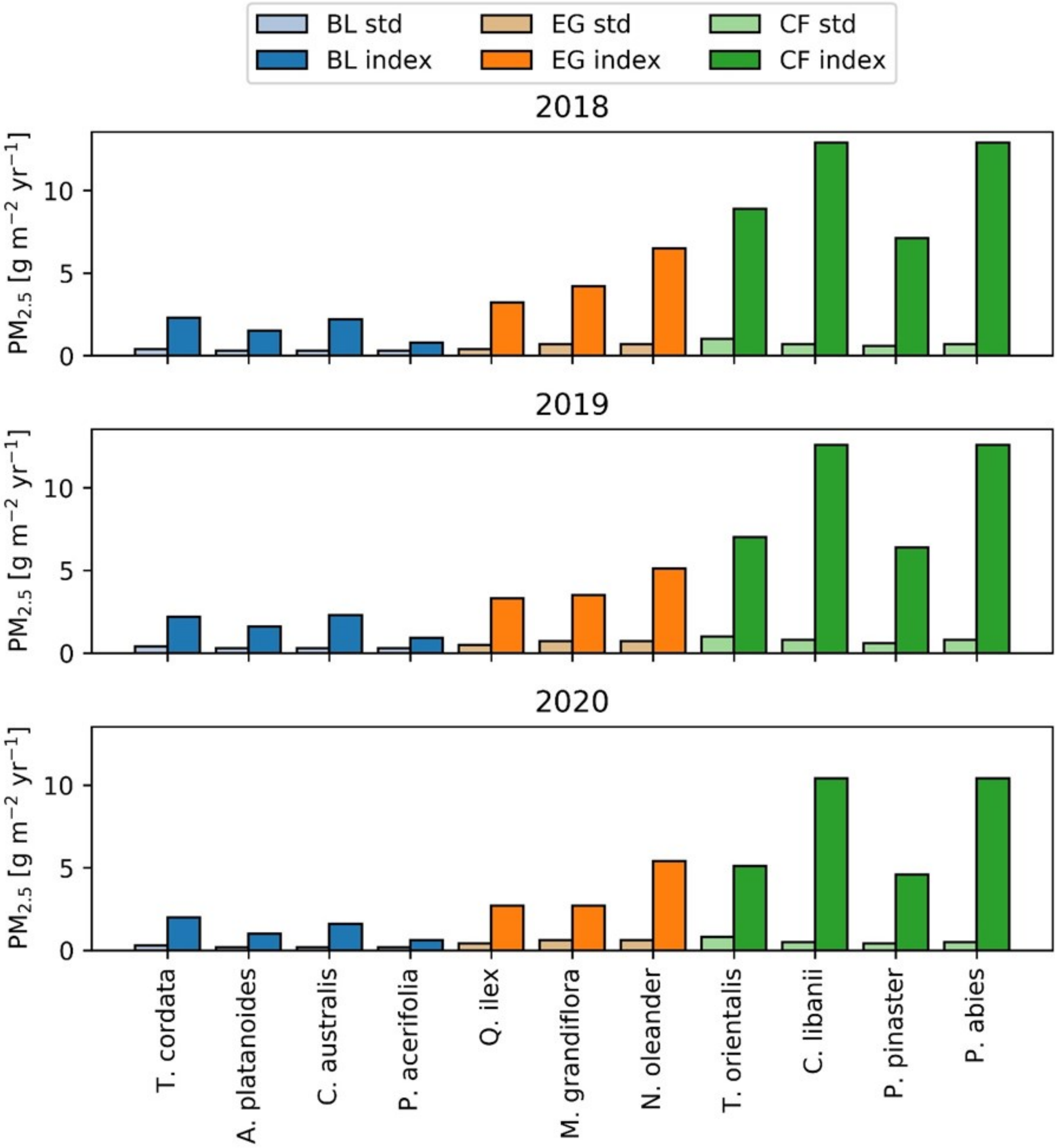

Figure 7 Total annual $PM_{2.5}$ removal of the different species as calculated by standard (std) and modified (index) models (BL = deciduous broadleaves; EG = evergreen broadleaves; CF = Conifers).

4. Discussion

4.1 Model performance and improvements

The results demonstrate the importance of including the properties of different tree species when assessing the contribution of UGI to air depuration. While the role of specific leaf traits in determining PM deposition by trees is often demonstrated in literature (Chávez-García and González-Méndez, 2021; Xu et al., 2021), the introduction of a comprehensive set of traits into PM removal modelling is done here for the first time. Despite difficulties, this model approach has been evaluated under conditions of a wide range of environmental dynamics, such as wind speed and rainfall, considering washing and resuspension as separated processes that affect fine particulates to a larger extent than coarse material (Yan et al., 2019; Zhang et al., 2018). This work therefore provides important advances towards a more reliable quantification of $PM_{2.5}$ removal by trees, significantly improving the popular i-Tree-Eco tool. The effects of specific leaf traits on $PM_{2.5}$ accumulation and removal were implemented by means of multi-criteria scores that, in turn, adjust the four environmental processes computed by the model. This approach allows accounting the composite interactions between leaf traits and environmental conditions, a challenging aspect for the quantification of the PM removal service provided by urban trees.

The modified model well matched the experimental $PM_{2.5}$ accumulation measurements of deciduous broadleaves and conifers, thus representing an effective update of current tools for these functional categories. In fact, the comparison with the conventional i-Tree-Eco model highlighted that the latter strongly underestimates $PM_{2.5}$ deposition on leaves and fails to distinguish species effects on it. In particular, deposition values were found to be a factor of 4.6 to 10.3 higher in deciduous broadleaves, 7.6 to 11.9 in evergreen broadleaves, and 23.9 to 32.2 in conifers, than the conventional model estimates (Fig. 4, 5 and 6). The higher PM deposition on conifer species is in line with previous studies (Chen et al., 2017; Muresan et al., 2022; Zhang et al., 2019), showing that longer and narrows needles may be more easily hit by airborne particles than large and flat leaves (Sæbø et al., 2012).

These outcomes are also reflected in the annual PM2.5 removal values of sampled trees (Fig.6), which further highlights the very good performance of *P. abies* and *C. libanii*. Although *T. orientalis* showed the highest accumulation values in winter season, the total annual removal was lower than for *P. abies* and *C. libanii*. Similarly, Yang et al. (2015) observations suggest higher PM deposition on cupressaceous species. The presence of sticky substances on leaf surface favored fine PM deposition by retaining fine particles also during intense precipitation events, resulting in a higher amount of PM deposited, which does not correspond to equally higher removal performances. This finding demonstrates that PM deposition and removal should be considered separately, depending on the scope of analysis. For instance, in an ecosystem services perspective, the contribution of trees to air depuration (a regulation service) is represented by PM removal value, while PM deposition can be a suitable indicator of air pollution (Chung et al., 2021). The lower PM removal of *P. pinaster* can be justified by its lower leaf area measured on sampled plants, compared to the other considered conifers.

Among the other species, N. oleander is the only species reaching comparable levels of PM2.5 deposition (Table 3). This unexpected result could be due to the combined performance in terms of deposition velocity and avoided resuspension, related to the presence of fourth order veins, roughness surface and short petiole length. Interestingly, conventional model better predicts deposition values of *Q. ilex* compared to the modified model, despite the low scores obtained for potential leaf water storage and deposition velocity parameters (Table 4). In general, the shorter vegetation period and the overall low trait scores are the reasons for the lower values of deciduous broadleaves. Among them, the highest deposition values reached by *T. cordata*, as also observed by Steinparzer et al., 2022, are explained by higher deposition velocity and presence of sticky substances on leaf surface. The effect of the latter trait was partially compensated when assessing total annual PM removal because of lower leaf washing capacity, as observed also in *T. orientalis*.

The controversial outcomes of improved model for evergreen broadleaves may found on their leaf history. Older leaves tend to accumulate larger amounts of fine PM because their exposure time to air pollutants is longer (Memoli et al., 2020) and their leaf microstructure is more differentiated (Niu et al., 2020). At the same time, some traits, such as the presence of epicuticular waxes, can prevent PM wash off after rain events over time, thus leading to larger PM deposition in older leaves. Accordingly, younger leaves, are supposed to have lower PM deposition values. Nonetheless, some recent studies demonstrated that leaf age significantly affects PM deposition only in combination with other factors, e.g. local emission sources, chemical composition of particulates and air pollution intensity (Esposito et al., 2020; Memoli et al., 2020; Niu et al., 2020). Therefore, although samplings were supposed to be representative, the combined effects of leaf age, traits and local environment may have caused the deviations of observed and modelled deposition for evergreen broadleaves. Additionally, it should be considered that leaf washing before vacuum filtration may not completely wash off the fine PM attached on epicuticular wax layer, which maybe a likely explanation for the overestimation of experimentally defined deposition values in case of *Q. ilex*, although this issue was not observed with seemingly relatively similar leaves of *M. grandiflora*, for which the model underestimated the measured deposition.

4.2 Management implications

Nature-based solutions are now mainstream considerations in environmental management to improve ecosystem services provision and protect human health (Keesstra et al., 2018). Their application requires a reliable evaluation of the ecological performances which, however, is not available for many currently applied decision support tools (Kangas et al., 2018; Schröter et al., 2015). Our results supported the notion that models may often and significantly underestimate the contribution of urban trees to air quality, thus limiting their reliability for application in urban management. We therefore recommend using the improved model for assessing PM2.5 removal for decision making in urban green management in order to select the most effective taxa for reducing air pollution levels. Regarding to the presented model setup, conifers are generally the most effective plants in terms of PM2.5 removal because of both their higher accumulation capacity per leaf unit and higher leaf area. However, species selection needs to consider local climate and air pollution conditions. Advantages of broadleaf trees include their ability to regenerate leaves each year regaining their full deposition capacity and potentially provide renewed leaf structures to interact with particulate matter. This may particularly provide advantages compared to evergreens in environments where air pollution removal is more important in the summer.

A further highlight of this study is that it emphasizes the close dependence of particulate removal on leaf washing capacity. Species with sticky substances such as waxes or resin are able to accumulate more particulate matter than others, but, at the same time, they do not allow for complete leaf washing and thus reducing the amount of particulate matter removal. This feature is particularly important when considering climate conditions for selecting tree species to promote PM deposition. Species that accumulate and retain particulates are recommended for dry environments, where precipitation events are sparse and concentrated, while leaves with better wetting characteristics are recommended in rainy environments, where a lower accumulation is balanced by continuous PM removal. Such information can be suitable for planning reforestation measures as for instance those targeted by the European Strategy for 2030 (European Commission, 2019), which should contribute to mitigate air pollution effects on human health. Nonetheless, it is worth mention that although greening measures offer opportunities for reducing air pollution levels, a solution of the air quality problem requires the integration with other actions targeted to reduce pollution emissions (Jennings et al., 2021).

Furthermore, urban trees provide a set of other valuable services that should be considered as well in urban green management. For example, although less effective in PM removal when compared to conifer species, deciduous broadleaves contribute largely to tropospheric ozone absorption (Fusaro et al., 2017; Muresan et al., 2022) and produce higher benefits in terms of urban heat island mitigation (Yilmaz et al., 2018). For these reasons, a functional selection of tree species requires analyzing the ecosystem service demand, since mismatches between urban services supply and demand generate ineffectiveness and inequalities in urban planning strategy (Maragno et al., 2018; Sebastiani et al., 2021). In this sense, conifer plantations should be primarily destined to zones with higher PM emissions and presence of most vulnerable people, while deciduous species may be better suited for areas with higher temperature during summer season. Another aspect for species selection may be to focus on autochthonous species to support local biodiversity. In the case of Ferrara municipality, 2 out of 4 conifers sampled in this analysis are non-native species.

In general, species should be always selected according to the local environmental context. In fact, local climatic conditions not only interact with vegetation in regulating ecosystem services provision (Cimburova and Pont, 2021) but also affect plant health, thus determining the success of plantation efforts and maintaining total leaf area over time. These aspects can perhaps best be captured by considering the potential vegetation of the area (Blasi et al., 2010). Finally, species selection should be coherent with urban cultural history and landscape configuration to support cultural services, as sense of place, aesthetic values, recreation, and education.

4.3 Future researches

Improving current models for the quantification of air depuration is challenging. The presented work provides important model improvements by introducing species-specific sensitivity based on leaf traits that can be obtained by both direct measurements as well as literature sources. The relatively certain availability of leaf trait knowledge increases model accuracy, although some traits may be better known than others. We thus recommend future research to also consider the interaction among different traits, as well as interactions with local environmental conditions. Moreover, the integration of trait-based and fluid dynamics models could be an additional field of investigation (Vos et al., 2013).The results of this study also highlighted the importance of experimental measures to calibrate commonly used models. Finally, future studies are called for that enable the evaluation of model processes regarding PM deposition and removal processes.

5. Conclusions

The combination of experimental measurements and modeling enabled the development of an index, based on morphological and functional characteristics of leaf traits for a more accurate estimate of the contribution of different tree species to fine particulate removal. The outcome provides a significant advancement for improving decision support tools such as the i-Tree-Eco model and represents a basis for a broader classification of species useful for improving urban air quality. These results are considered relevant to planning and management of urban green spaces, and important for more comprehensive analyses of plant-PM interactions. We recommend that future studies should focus on the leaf history of deposited PM, the characteristics of the particulate matter, leaf structures, as well as the saturation levels of leaves and their ability to regenerate after precipitation events.

6. References

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Conclusions

The study uses different scales to analyse ecosystem services in urban areas. Specifically, the study aims to understand how the use of green infrastructures contributes to the provision of important regulation services, focusing on the tree community composition. Additionally, the findings can inform decision-makers by the identification of some guidelines for urban green management.

As showed by Appendices 1 and 2, the ongoing climate change generates important consequences on biodiversity and ecosystem services provision in natural ecosystems. The increased frequency of extreme climatic events affects also urban ecosystems, where their effects are exacerbated by demographic density and the scarcity of natural components.

Cities are vulnerable due to the high population living in these areas so increasing cities resilience, intended as cities tolerance level to alterations before re-adapt to new set of structure and processes (Alberti et al. 2008), is urgently necessary to protect human well-being. Strategies founded on Nature-based solutions (NBS), in particular green infrastructures, guarantee the supply of a number of ecosystem services useful to the society.

Techniques for ecosystem services analysis are vary and different approaches and models should be used according the scale of analyses and the scope. In fact, the scale approach, as well as the used method, are important in terms of accuracy and obtained information (Grêt-Regamey et al. 2014). Functional classification based on the combination of leaves and crown traits was a successfully method to characterize tree community and link it with ES provision.

Macroscale analysis (Chapter 1) allowed evaluating how different parks located in the same region (Po Plain) contribute differently to the provision of ES according to tree community composition and its complex interaction with local environmental conditions. Therefore, such interactions cannot be omitted when analysing ES provided by UGI. However, the following general guidelines arise from the analyses and represent suitable indications for decision-makers:

- Monumental trees are important carbon pools for climate mitigation service
- Increasing leaf area during the rainiest periods is an effective strategy to prevent floodings
- Maximize leaf area during summer can be strategical for mitigating the effects of urban heat waves
- Air depuration service (i.e. air pollutants removal) is enhanced by evergreen species

Using remote sensing, even combined with field measurements, is an excellent mesoscale approach (Chapter 2). This method allows mapping air quality regulation services on the entire municipality (Sebastiani et al. 2021; Tavares et al. 2019). The modelling considers mean deposition velocity values for pollutant removal, also deriving from experimental analysis. Such simplifications may affect result reliability at finer scale but can be acceptable when applied at municipal scale. At the same time, experimental measuring for determining mean PM_{10} deposition velocity for each vegetation functional class (conifers, deciduous broadleaves, evergreen broadleaves) allow to improve analysis performance by increasing model sensitivity to vegetation types. Results highlight that potential foresting areas could be identified and possible future scenario of foresting was projected. These low-cost foresting actions would increase significantly the air depuration capacity without requiring expansive land use conversions. Therefore, urban planners could consider existing green areas for future plantations in order to meet Biodiversity 2030 strategy target.

Finally, microscale approach (Chapter 3) allows understanding how functional leaf traits influence $PM_{2.5}$ removal and accumulation. As for the finding presented in Chapter 1, the combination of traits, rather than single leaf traits, influences ES, specifically $PM_{2.5}$ removal. Current models fail to capture species related performances, resulting in lower accuracy of PM removal. In particular, the results of improved iTree-Eco model revealed that the current model version underestimates this ecosystem service, as already observed by Pace and Grote (2020). This has important implication for ecosystem management. Improved model version can better valorise the role of urban trees in air depuration. Specifically, conifers are highly efficient in terms of particulate removal and their presence in urban areas with high levels of air pollution should be promoted in cities subjected to extended drought periods.

Decision makers must face biodiversity loss, climate change and urban intensification, which require important efforts for nature requalification for human well-being. The study offers suggestions and principles about the possible actions in urban areas. First, it is important to consider the background environmental conditions before carrying out restore actions or when new urban green areas are projected. Rounsevell et al. (2010), show how ES can either mitigate or adapt to environmental conditions. Another important aspect is to consider the ecosystem services demand and adapt the tree species to maximize the ES provision. This study emphasizes the specificity of tree species to provide ES.

Lastly, a diverse tree composition covers a wider range of ecosystem services. Studies emphasize how different species number provide and support multiple ES (Gamfeldt et al. 2008; Zavaleta et al. 2010). Moreover, in a scenario of global change, keeping a different community composition promote ecosystem functioning under different change scenarios (Craine et al. 2003; Isbell et al. 2011). The relationship between ecosystem functioning and ecosystem services is highly studies (i.e. Balvanera et al. 2006; Costanza et al. 2007) and considered in this work. The functional diversity is responsible for ecosystem functioning and services (Cardinale et al. 2012; Díaz et al. 2006) and therefore, high level of functional variability should be promoted in urban green areas to deliver a wide set of ES.

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APPENDIX 1

(Published as: Muresan, A. N., M. Gaglio, V. Aschonitis, G. Nobili, G. Castaldelli, and E. A. Fano. 2020. "Structural and Functional Responses of Macroinvertebrate Communities in Small Wetlands of the Po Delta with Different and Variable Salinity Levels." Estuarine, Coastal and Shelf Science 238(August 2019):106726. doi: 10.1016/j.ecss.2020.106726)

Structural and functional responses of macroinvertebrate communities in small wetlands of the Po delta with different and variable salinity levels

Abstract

Coastal areas often host small water bodies described by high levels of biodiversity, which are threatened by environmental changes such as saltwater intrusion. This work evaluates the salinization effects on macroinvertebrate communities of 16 permanent small wetlands (ponds) located in a coastal Mediterranean forest in Northern Italy, characterized by different salinity levels. From a preliminary multivariate analysis (CCA), salinity was detected as the only important parameter affecting taxa distribution. Thus, diversity in terms of taxa, biological and functional traits of macroinvertebrate communities were analyzed considering three salinity classes (freshwater, oligo-mesohaline and polyhaline). The threshold indicator taxa analysis (TITAN) was used for detecting changes in taxa abundance and trait distributions within the salinity range and for assessing synchrony among their change points as evidence of community thresholds. Taxonomic and functional diversity indices and single functional/biological traits among the three salinity classes were also compared. The findings demonstrated that ponds' macroinvertebrate communities are resilient to moderate increases of salinity, but salinization increase to polyhaline levels causes loss of biodiversity and a drastic community simplification in terms of functional eveness. Since climate change and anthropogenic activities are expected to exacerbate salinization, management measures are required for the conservation of aquatic biodiversity in coastal small wetlands.

Introduction

Climate change is expected to generate significant ecosystems modifications worldwide, causing changes in environmental conditions and ecosystem processes (Scholze et al., 2006). Transitional environments as deltaic areas are markedly vulnerable to climatic and environmental changes because of their sensitive hydrological balances and the increasing presence of human settlements and activities (Gu et al., 2011; House et al., 2016; Gaglio et al., 2019). Deltas host different typologies of aquatic environments, as river branches, wetlands, salt marshes and mudflats, that host a high environmental diversity and provide several ecological functions, such as water retention and depuration, habitat provision for species and recreation (Gaglio et al., 2017a). Since the changes of environmental and climatic conditions are predicted to affect aquatic biotic and abiotic components underpinning deltas ecological functions, the investigation of management consequences on river deltas are of paramount importance to guarantee biodiversity conservation and human well-being.

The deltaic areas of the Mediterranean region are particularly vulnerable to climate change, since an increase of temperature and a decrease of precipitation has been already identified (Cramer et al. 2018). The combined effects of rising water demand and water scarcity will significantly affect future water availability in Mediterranean basins (Saadi et al. 2015). Raising temperatures will increase evapotranspiration rates, while the reduced amount of rainfall will enhance plant water stress, requiring higher amounts of water withdrawals for crop irrigation. This trend will significantly influence the wetlands biota by favoring species more tolerant to drought (Johansen et al. 2018; Oliver et al. 2015). At the community level, the responses to such phenomena can be observed by investigating macroinvertebrate assemblages. In Mediterranean coastal systems, climate change was observed to cause losses of taxonomic and functional diversity in macroinvertebrate communities (Cardoso et al. 2008; Pitacco, Mistri, and Munari 2018).

The resilience of aquatic ecosystems to environmental changes relies on the capacity of aquatic biota to re-establish living communities after perturbations (Downing and Leibold 2010; Schaffner 2010). Nonetheless, the newly established communities may present different levels of taxonomic and functional diversity, as a consequence of adaptation to the new environmental conditions (Macleod, Moltschaniwskyj, and Crawford 2008). This may lead to a general loss of both biodiversity and capacity to respond to additional perturbations, which further threaten these ecosystems (Oliver et al. 2015). Moreover, aquatic ecosystems are widely subjected to increasing human pressures harming their ecological status. The intensification of human activities, such as agriculture, aquaculture, water withdrawals, in combination with ongoing climatic changes can lead to detrimental effects in their ecological conditions and capacity to support human well-being (Blann et al. 2009; Day et al. 2008; Gaglio et al. 2019; Xenopoulos et al. 2005).

Salinization is one of the main stressors affecting deltaic areas that occur as an amplified result of climatic changes and anthropogenic effects (Colombani et al. 2016). These transitional environments are highly sensitive to changes of both terrestrial and marine components (Harley et al. 2006), as well as climatic factors (Nielsen and Brock 2009; Scavia et al. 2002). Different natural and human-related factors concur to the increase of salinity levels of water bodies. Primary salinization is referred to natural salt accumulation from rainwater and leached from terrestrial sources unaffected by human activities. On the contrary, secondary salinization is caused by human-induced mechanisms, such as vegetation clearance, intensive irrigation, river regulation and land reclamation (Gaglio, Vassilis G. Aschonitis, et al. 2017; Herbert et al. 2015). Unlike primary salinization, secondary salinization occurs on a time frame of decades or less, under the consequence of hydrological cycle alteration due to human effects (Herbert et al. 2015).

Salinity affects the presence of species both directly, for example through osmoregulation physiology, and indirectly, by influencing abiotic factors and biotic interactions (Liancourt, Callaway, and Michalet 2005; Pinder et al. 2005; Withers 1992). Freshwater invertebrates can withstand small salinity increases maintaining constant iso-osmotic conditions between haemolymph and external solutions. With the increase of external solute concentrations, many freshwater invertebrates suffer from dehydration, while salt-tolerant aquatic invertebrates respond to the increased salinity by adopting osmoregulation strategies for maintaining constant the osmotic concentration of body fluids (Evans 2008).

Salinity also influences biotic interactions. For instance, the role of competition may vary with the level of abiotic stress, such as salinity, as a result of different tolerance and competitive response of species (La Peyre et al. 2001; Liancourt et al. 2005). Variation of salinity levels mediates trophic cascade by influencing predators' abundance, thus altering their top-down control on preys (Cañedo-Argüelles et al. 2016; Herbst 2006; Herbst and Blinn 1998). Moreover, salinity may interact with other environmental factors to influence species composition (Larson and Belovsky 2013).

The effects of increasing salinity levels on macroinvertebrate communities were described in literature for coastal marine habitats (Zettler et al. 2014), estuarine (Little, Wood, and Elliott 2017; Ritter, Montagna, and Applebaum 2005) and lagoons (Como, van der Velde, and Magni 2018), but very few studies exist for pond systems (Boix et al. 2008).

Ponds are small and isolated ecosystems ranging from 1 m2 to few hectares, which temporarily or permanently store water (De Meester et al. 2005). They are biodiversity hotspots both in terms of species composition and biological traits, and provide ecosystem services to support human well-being (Céréghino et al. 2012, 2014). Pond ecosystems host a large number of species and rare species, exceeding those of other aquatic ecosystems such as streams and lakes

(Williams et al. 2004). Consequently, despite their limited dimensions, ponds are fundamental features for biodiversity conservation at the landscape scale (Céréghino et al. 2014; Coccia et al. 2016). In fact, in addition to the diversity of their own communities, ponds also play a role as stepping stones for aquatic mobile species, thus mitigating the negative effects of habitat fragmentation and increasing aquatic habitats' connectivity (Pereira, Segurado, and Neves 2011; Rothermel 2004). Moreover, ponds are ideal sentinels and early warning systems of environmental changes due to their sensitivity to environmental changes, which is associated to their small size and the high interaction with the groundwater (De Meester et al. 2005). Particularly, pond macroinvertebrate communities can be sensitive indicators of how living communities respond to environmental variations in coastal systems. However, the set of environmental variables governing pond community composition and species traits is specific for each climatic/biogeographic region (Céréghino et al. 2012; De Marco et al. 2014; Ruhí et al. 2013). Therefore, site-specific investigations are needed to assess the response of macroinvertebrate communities of pond systems.

The aim of this study is to investigate macroinvertebrate communities' changes in a coastal permanent system of ponds located in the Po river delta (Northern Italy) with different salinity levels, by means of taxonomic and functional (i.e. biological traits) analyses. Overall, the assessment of macroinvertebrate community responses to salinity can shed light on how biodiversity and ecological functions of aquatic ecosystems of deltaic areas are expected to change due to salinity alterations.

Material and Methods

Study area

This study was carried out at "Bosco della Mesola" (44°50′ 28′′ N, 12°15′ 12′′ E), a National Natural Reserve of 1058 ha located in the province of Ferrara (Northern Italy), with an altitude ranging from -1 to +3 m a.s.l (Fig. 1).

The Nature Reserve "Bosco della Mesola" hosts a ponds' system formed by small water bodies mainly located among the ancient dunes, where water stagnation is fed either by rainfall or by the coastal aquifer that is hydrologically connected to the sea or, in a lesser extent, by incoming water from the canal system. Their aquatic biota includes macroinvertebrate, amphibian and reptile communities. No fishes were observed in ponds. The ponds' system is characterized by different salinity levels. Water bodies near the shoreline are influenced by seawater rising from groundwater table and by occasional floods, while the ponds at north receive freshwater from drainage canals.

The surrounding area was strongly altered by human interventions during the last century, e.g. wetland reclamations, that affected the local hydrological balance (Gaglio et al., 2017a). Additionally, the area is subjected to subsidence that causes the progressive intrusion of saltwater towards the inner part of the Reserve, due to the progressive difficulties in recharging water table with freshwater, affecting the communities of living organisms of local water bodies (Gerdol et al. 2018). Moreover, climatic changes could speed up the salinization level of ponds' system, thus exacerbating such impacts. Macroinvertebrate communities were studied as suitable indicators for detecting salinization effects on aquatic biota of small wetlands of deltaic areas, such as the case of the "Bosco della Mesola" ponds in the river Po delta.

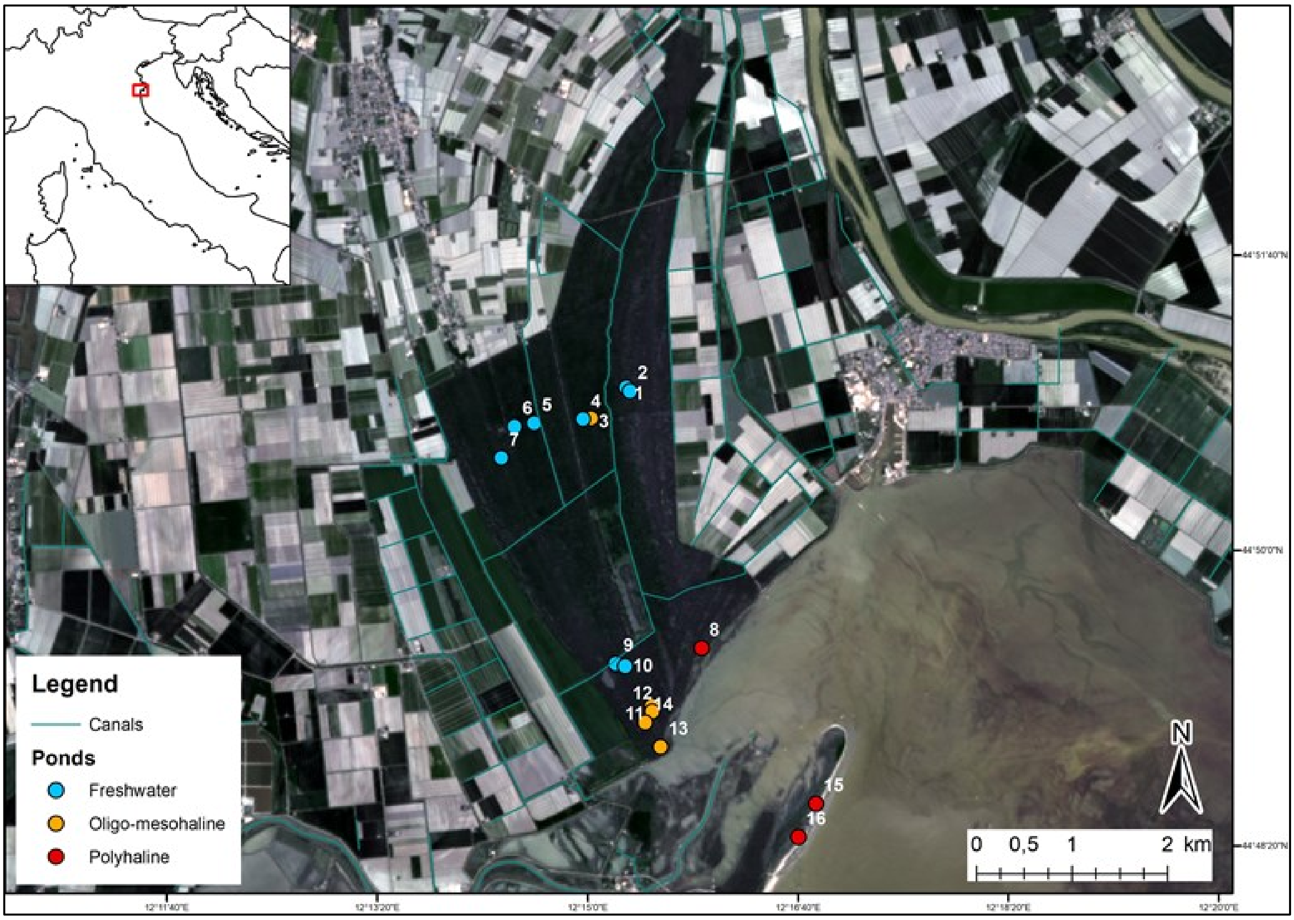

Fig.1 The study area of the 16 ponds where samplings were performed.

Sampling procedures

The macroinvertebrate communities of 16 permanent coastal ponds were sampled during May 2017. Among these ponds, 14 are located inside the Nature Reserve of "Bosco della Mesola" in north-eastern Italy. Moreover, two additional ponds were sampled outside the Reserve, located south in a coastal outlet (Fig. 1). Temporary ponds were not considered in the analysis. The salinity levels ranged between 0.2 and 29.3 psu. The ponds were classified into three salinity classes according to the classification of Por (1972): freshwater (<0.5 psu) (8 ponds), oligo-mesohaline (0.5-18 psu) (5 ponds) and polyhaline (>18 psu) (3 ponds). Benthic macrofauna was collected sweeping a distance of 2.5 m with a 40 cm-wide D-frame hand net (mesh size = 500 µm) (sampling area of 1 m^2). Samplings were carried out in 3 different sites within each pond, one in central and two in the outer parts, to capture intra-pond variability. Animals were preserved in a 4% formalin solution and later identified in the laboratory up to the genus level, and when it was not feasible, the family level was reached. Hence, the respective biological/functional traits (feeding, mobility, adult life habitat, body size, life span, reproductive frequency, habitat choice) were attributed to each taxon by means of bibliographic information using the databases of Horton et al. (2017) and (MarLIN 2006). When observed taxa were not covered by these sources, Thorp and Covich (2010) was used as alternative literature. Table 1 provides the 39 taxa observed in the sampling ponds while Table S1 in the Supplemetary material provides the trait modalities attributed to each taxon and Table S2 their mean abundance for each pond.

Eight environmental factors were measured/assessed during samplings (Table 2). Water temperature, oxygen saturation and salinity were measured using a multi-parameter probe (YSI Model 85). Ponds' surface and maximum water depth were also measured in situ. The shape of the small ponds was approximated as circle/ellipse. The area of circle is $A=\pi R^2$ while of ellipse is $A=\pi ab$ (where a and b are the smaller and the larger radius). In order to avoid assumptions of shape, a laser meter was used to take measurements of diameter from many positions of the pond shore. These measurements were used to derive the mean diameter and consequently the mean radius (R_{mean}) of each pond. The area A was then estimated as $A=\pi(R_{\text{mean}})^2$. For the larger ponds which were visible in Google Earth Pro, the area was measured with the Ruler polygon tool of the respective software.

Euclidean nearest-neighbor distance (ENN) was computed using Fragstat 4.2 (McGarigal 2014) as a measure of pond isolation. ENN can be defined as the shortest straight-line distance between the focal patch and its nearest neighbor of the same class. ENN was calculated for each pond, also considering the presence of other water bodies, such as channels and coastal lagoons. Shapefiles of ponds and channels were used to develop a reclassified raster file where water bodies were coded as "1" and the remaining part of the landscape as "0". FRAGSTAT software used this raster file to calculate ENN, considering the code "1" as foreground and "0" as background.

Both canopy coverage (i.e. the % of the pond surface shaded by surrounding trees) and aquatic vegetation coverage (i.e. the % of the pond surface where aquatic plants were present) were measured by analyzing photos taken on site (photos of water surface and photos of the sky at the shoreline were elaborated in Arc-GIS).

Table 1. Taxa observed in the 16 sampling ponds of Bosco della Mesola.

Phylum	**Order**	**Family**	**Family abbr.**	**Genus**
Anellida	Haplotaxidae	Naitidae	Nai	-
Anellida	Hirudinida	Hirudinidae	Hir	*Hirudo*
Anellida	Polychaeta	Nereidae	Ner	*Nereis*
Crustacea	Amphipoda	Corophiidae	Cor	*Corophium*
Crustacea	Amphipoda	Gammaridae	Gam	*Echinogammarus*
				Gammarus
Crustacea	Decapoda	Cambaridae	Cam	*Procambarus*
Crustacea	Decapoda	Portunidae	Por	*Carcinus*
Crustacea	Isopoda	Asellidae	Ase	*Asellus*
Crustacea	Isopoda	Sphaeromatidae	Sph	*Sphaeroma*
Insecta	Coleoptera	Dytiscidae	Dyt	-
Insecta	Coleoptera	Haliplidae	Hal	-
Insecta	Coleoptera	Hydrophilidae	Hyd	*Helochares*
Insecta	Diptera	Ceratopogonidae	Cer	-
Insecta	Diptera	Chaoboridae	Chi	*Chaoborus*
				Chironomus
				Cladopelma
				Cryptochironomus
				Parachironomus
				Polypedilum
Insecta	Diptera	Orthocladiinae	Ort	*Orthocladius*
Insecta	Diptera	Tanyponidae	Tan	*Procladius*
				Psectrotanypus
				Tanypus
Insecta	Ephemeroptera	Baetidae	Bae	*Baetis*
				Cloeon
Insecta	Heteroptera	Corixidae	Crx	*Cymatia*
				Micronecta
Insecta	Heteroptera	Nepidae	Nep	*Nepa*
Insecta	Lepidoptera	Crambidae	Cra	*Cataclysta*
				Paraponyx
Insecta	Odonata	Lestidae	Les	*Chalcolestes*
Insecta	Odonata	Libellulidae	Lib	*Libellula*

Mollusca	Bivalvia	Corbulidae	Crb	*Corbula*
Mollusca	Gasteropoda	Bithyniidae	Bit	*Bithynia*
Mollusca	Gasteropoda	Lymnaeidae	Lym	*Lymnae*
Mollusca	Gasteropoda	Physidae	Phy	*Physa*
Mollusca	Gasteropoda	Planorbidae	Pla	*Planorbis*
Nematoda	-	-	Nem	-

Table 2. Environmental parameters used in this study for analyzing taxa responses.

Parameter	**Unit**	**Transformation**	**Abbrev.**	**Max**	**Min**	**Mean**	**SD**
Temperature	°C	Log (x+1)	Temp	29.6	12.4	17.2	5.7
Oxygen saturation	%	arcsin (x/100)$^{0.5}$	O2	100.0	8.5	56.3	30.5
Surface area	m^2	Log (x+1)	Area	6191.5	24.0	641.9	1514.0
Depth (maximum)	cm	Log (x+1)	Dep	63.0	10.0	34.3	16.6
Salinity	psu	Log (x+1)	Salt	29.3	0.2	6.3	9.8
Vegetation	%	arcsin (x/100)$^{0.5}$	Veg	40.0	0.0	4.1	10.8
Canopy	%	arcsin (x/100)$^{0.5}$	Can	50.0	0.0	14.4	18.6
ENN	m	Log (x+1)	ENN	356.5	28.3	103.9	88.0

Methods of analysis

Detection and evaluation of salinity gradient effects on taxa abundance

All environmental parameters and taxa abundance were transformed to reduce normality departures. Environmental parameters, which are ratios/percentages, were transformed using arcsin(x/100)0.5 while the rest environmental parameters and taxa abundance (ind. m-2) using log(x+1) (Aschonitis et al. 2016). Spearman correlations were performed among environmental variables of Table 2.

Multiple gradient analysis was performed for assessing the effect of multiple descriptors (environmental parameters) on multiple target variables (taxa) using Canonical Correspondence Analysis (CCA) (ter Braak and Smilauer 2002; Lepš and Šmilauer 2003). CCA was performed using CANOCO 4.5, based on target variables correlations and their standardized scores (ter Braak and Smilauer 2002). The method was applied following the same steps as those described in a similar case study (Aschonitis et al. 2016) and significant descriptors were identified using CANOCO's forward selection procedure and Monte Carlo permutation test (499 permutations) (a default option in the CANOCO software). Collinear variables with variance inflation factor VIF>8 (Zuur, Ieno, and Smith 2007) or variables with statistical significance p>0.5 were excluded from multiple gradient analysis. The multivariate analysis was performed for macroinvertebrate communities at the family level, which is considered a sufficient level for invertebrate community analysis (Gayraud et al. 2003). Moreover, when applied to multivariate analyses, the results obtained considering higher taxa may more closely reflect effects related to gradients or stresses than those based on finer level (e.g. genera or species), that are more affected by natural "noise" (Warwick 1988).This analysis detected that the only statistically significant descriptor variable at $p<0.05$ level is salinity. For this reason, a method of single gradient analysis is more appropriate.

The threshold indicator taxa analysis TITAN (Baker and King 2010) is a single gradient analysis method that is used in ecological studies for detecting changes in taxa abundance distributions along a unique environmental gradient (i.e. salinity) and for assessing synchrony among taxa abundance change points as evidence of community thresholds (Baker and King 2010). TITAN uses bootstrapping for estimating purity and reliability criteria as well as uncertainty of change points related to individual taxa abundances along the salinity gradient. Usually, a cut off value of 95% is used in both purity and reliability criteria for identifying statistical robust responses of taxa abundance versus an environmental gradient (i.e. salinity). The purity cut off value defines what is considered a pure response direction. A purity value of 0.95 indicates that 95% of the results from bootstrap replicates agree with the observed response direction. The reliability cut off value defines what is considered a reliable response magnitude. A reliability value of 0.95 indicates that 95% of the results from bootstrap replicates have an IndVal p-value less than or equal to 0.05, indicating a response magnitude at a given change point location that is significantly different from what would expect from random permutation (Baker and King 2010) {for more explanations about IndVal p-value, see Dufrêne and Legendre (1997) and Baker and King (2010)}. In this study, the purity and reliability values were estimated and the response plots versus salinity gradient were developed for all taxa without considering cut off values. The TITAN analysis was performed with TITAN2 version 2.1 (Baker et al. 2015) in R language using 500 random permutations of taxa abundances and creating 1000 new bootstrap datasets created by resampling the observed data with replacement.

Effects of salinity on biological traits and diversity measures

The analysis considered both taxonomic and functional indices. The functional indices were calculated using the "FD" package for R (Laliberté and Legendre 2010), which takes into account multidimensional (i.e. multiple traits) functional diversity. The differences in population indices among the three salinity classes of ponds (i.e. Freshwater, Oligo-mesohaline, Polyhaline) were evaluated with the non-parametric Kruskal-Wallis Test. When statistical significanltly differences were found, a post-hoc test was carried out (Bonferroni test). The analysis considered:

- Four taxonomic indices: species richness (S), Shannon's diversity index (H′), Pielou evenness index (J′) and Simpson index (D), and
- Four functional indices: Functional richness (FRic), Functional evenness (FEve), Functional divergence (FDiv) and Rao quadratic entropy index (RaoQ).

Moreover, significant differences in single biological traits among the three respective salinity classes of ponds were evaluated through the comparison of proportions with χ2 test for P-value ≤ 0.01. The comparison of proportions takes into account the number of individuals characterized by each specific trait modality. For each trait, the comparison was performed between the proportion of each trait modality of the three salinity classes versus the overall proportion of the remaining modalities in the respective salinity class (e.g. for the “adult life habitat” trait, the significance of difference of “aquatic” type in the three salinity classes was evaluated comparing “aquatic” vs. “aeric”). The null hypothesis was that the proportion of the two trait modalities (or of the proportion of the one trait type versus the remaining ones) did not differ over the three salinity classes. Afterward, an analysis of means (ANOM) plot with 99% confidence was applied to provide indications about the direction of the significant differences based on the deviation from the grand mean of the ANOM plots (Fedrigotti, Aschonitis, and Fano 2016; Gaglio, Vassilis George Aschonitis, et al. 2017). The ANOM analysis was also followed by TITAN analysis of biological traits using salinity as a single gradient for assessing change points of trait groups abundance. TITAN was perfomed following the same methodological steps as described in the case of taxa.

Results

Effect of salinity gradient on taxa abundance

As it was mentioned in the Methods section, CCA detected that the most important parameter from Table 2 but also the only parameter with statistical significance at $p<0.05$ level describing the taxa variance was salinity (Table S3 in the Supplementary material). According to Table S3, the CCA analysis with all the variables of Table 2 explained the 58.7% of taxa variance while only salinity explained 14.9% of taxa variance.

The Spearman correlations among the environmental variables (Table 3) showed that salinity is significantly positively correlated with temperature and pond area while it is significantly negatively correlated with canopy coverage. Temperature is positively related to salinity due to higher evapo-concentration (accumulation of salts due to higher water temperature that enhances evaporation). Pond area is positively related to salinity, because larger ponds were closer to the coastline or to lower elevation indicating higher salt water intrusion. Canopy coverage is negatively related to salinity because larger canopy coverage reduces solar radiation interception by the water that leads to lower evapoconcentration (i.e. lower solar radiation leads to lower evaporation). Thus, salinity can also be used as a general surrogate descriptor of the aforementioned environmental parameters.

Table 3. Spearman correlations among environmental variables of Table 2.

	Temp	O2	Area	Dep	Salt	Veg	Can	ENN
Temp	1							
O2	0.421	1						
Area	0.444	0.313	1					
Dep	-0.394	-0.345	-0.372	1				
Salt	0.519*	0.450	0.599*	-0.105	1			
Veg	0.004	0.303	-0.216	-0.111	-0.063	1		
Can	-0.462	-0.589*	-0.247	0.409	-0.556*	-0.282	1	
ENN	-0.092	0.142	0.389	-0.055	0.175	0.007	0.124	1

* Statistical significance for $0.01<p<0.05$ level.

The results of TITAN analysis that concern the effects of salinity gradient on taxa abundance are provided in Table 4 and Fig. 2. Table 4 provides the indicator change point (CP) along the salinity gradient expressed in psu (median of 1000 bootstrap replicates), the purity and reliability % of CP, and the response (positive + or negative -) of each taxon versus the increase of salinity gradient. Fig. 2 shows the declining taxa on the left axis and the increasing taxa on the right axis. The observed change point is indicated by the circular symbol (its size corresponds to the median z-value of bootstrap replicates), while the horizontal line behind each circular symbol describes the 5-95% quantiles from the bootstrapped change-point distribution. Taking into account Table 4 and Fig. 2, the following observations were made:

- 18 out of 28 taxa showed purity >95%, while 9 out of 28 taxa showed both purity and reliability >95% (other taxa showing only reliability >95% were not observed). From the nine highly pure (>95%) and highly reliable (>95%) taxa, four showed a positive and three a negative response versus the salinity gradient.
- From the nine highly pure (>95%) and highly reliable (>95%) taxa, some of them (e.g. *Naitidae* - Nai) showed very low z-score (response magnitude) in the CP and large bootstrap interval. This

indicates a very clear positive relationship (response) between this taxon and salinity but with high uncertainty in the CP value.

- The most sensitive taxa to salinity were found to be Cambaridae (Cam), Sphaeromatidae (Sph), Dytiscidae (Dyt), Haliplidae (Hal), Orthocladiinae (Ort), Bithyniidae (Bit) and Planorbidae (Pla), showing negative responses for indicator change points (CP) of salinity <0.5 psu.
- The taxa of highest tolerance to salinity were Naitidae (Nai), Nereidae (Ner), Chaoboridae (Chi), Corbulidae (Crb) and Nematoda (Nem) showing positive responses for indicator change points (CP) of salinity >10 psu.

Table 4. Indicator change point (CP) along the salinity gradient expressed in psu (median of bootstrap replicates), values of purity and reliability criteria, and response (positive + or negative -) of each taxon (see Table 1 for abbreviations) versus the increase of salinity gradient according to TITAN analysis.

Taxon	CP (psu)	Purity	Reliability	Response group	Taxon	CP (psu)	Purity	Reliability	Response group
Nai**	14.25	0.978	0.958	+	Ort	0.3	0.886	0.654	-
Hir**	8.15	0.998	0.985	+	Tan**	3.05	0.994	0.985	-
Ner**	10.2	1	1	+	Bae	4.8	0.82	0.871	-
Cor*	8.15	0.962	0.751	+	Crx*	8.15	0.96	0.939	-
Gam**	0.5	1	1	+	Nep	1.85	0.87	0.306	+
Cam*	0.3	0.967	0.76	-	Cra	1.85	0.665	0.668	-
Por*	8.15	0.962	0.719	+	Les	0.45	0.733	0.922	+
Ase**	2.6	1	1	-	Lib	1.85	0.87	0.317	+
Sph	0.25	0.862	0.807	-	Crb*	14.25	0.982	0.938	+
Dyt	0.25	0.921	0.857	-	Bit*	0.3	0.967	0.734	-
Hal*	0.3	0.967	0.734	-	Lym**	4.8	0.997	0.97	+
Hyd	3.05	0.771	0.841	-	Phy*	4.8	0.964	0.616	+
Cer*	0.8	0.982	0.936	+	Pla**	0.4	1	0.997	-
Chi**	10.2	1	1	+	Nem	22.5	0.754	0.694	+

*Statistically significant purity (>95%) or reliability (>95%)

**Statistically significant purity (>95%) and reliability (>95%)

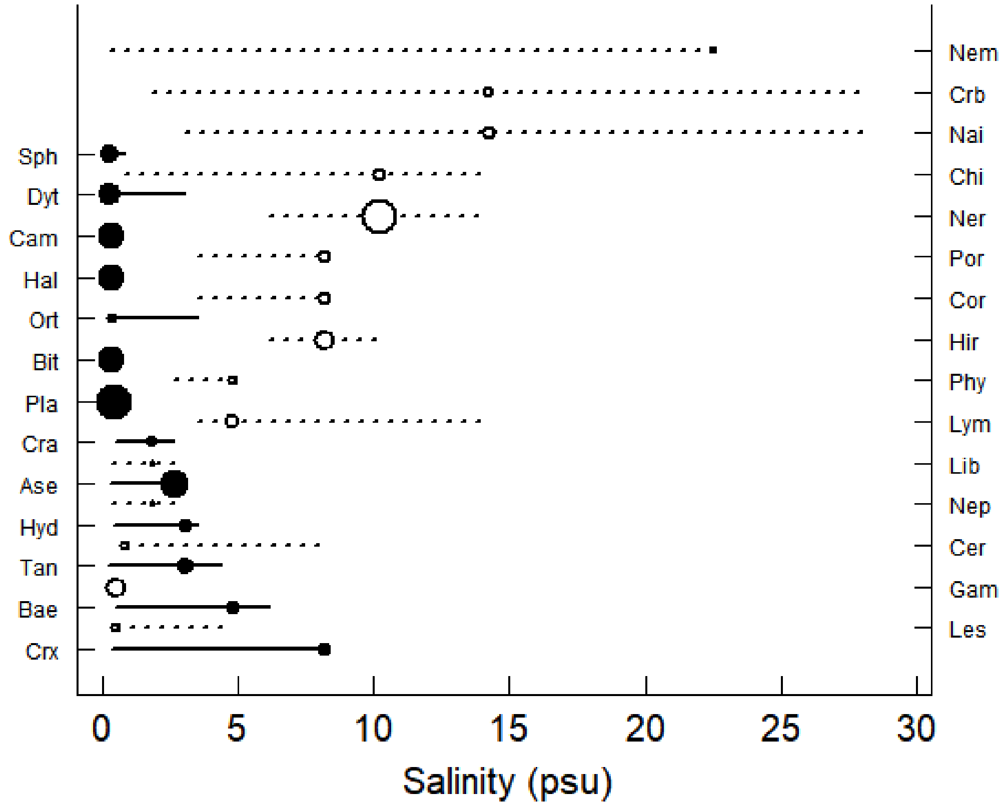

Fig.2 CP and response plots of each taxon versus the salinity gradient according to TITAN analysis. Black circles are aligned and correspond to the negative-reponse taxa of the left vertical axis, whereas white circles are aligned and correspond to the positive-reponse taxa of the right vertical axis. The size of the circle is proportional to median z-score of bootstrap replicates. Horizontal lines overlapping each circular symbol represent the interval of 5th and 95th percentiles among bootstrap replicates.

Effect of salinity on diversity indices and biological traits

The values of taxonomic and functional population indices of the three salinity classes of ponds (i.e. Freshwater, Oligo-mesohaline, Polyhaline) are given in Table 5. The only statistically significant differences among salinity classes were observed for the case of functional evenness (FEve) (p-value<0.05), when shifting from freshwater to oligo-mesohaline conditions. Even though not highlighted by statistical tests, a decreasing trend in the majority of the other indices and especially in species and functional richness values along the salinity gradient was observed.

Considering the ANOM analysis of differences in biological traits among the three respective salinity classes, all the considered traits showed significant variations (Table 6), highlighting taxa sensitivity to salinity in terms of functional and biological characteristics. From the 22 traits, 21 showed significant variations where 7 showed a positive response and 14 a negative response to salinity increase. The changes mainly occurred in the transition from oligo-mesohaline to polyhaline class (Table 6). The TITAN analysis of traits versus salinity is given in Table 7 and Fig.3, from which it was observed that 12 out of 22 traits showed CPs with both purity and reliability >95%. TITAN analysis complements the ANOM analysis since it provides the possible range where a CP may appear.

Table 5. Average values (±Standard error) of taxonomic (S, H', J' and D) and functional indices (FRic, FEve, FDiv and RaoQ) observed in each salinity class.

Pond salinity classes	**Freshwater**	**Oligo-mesohaline**	**Polyhaline**	**p-value Kruskal-Wallis**
Species Richness (S)	7.625±0.73a	5.8±2.853a	3.333±0.882a	0.074
Shannon's Diversity (H')	0.954±0.154a	0.82±0.201a	0.546±0.082a	0.235
Pielou Evenness (J')	0.481±0.07a	0.622±0.049a	0.561±0.205a	0.254
Simpson index (D)	0.466±0.07a	0.482±0.078a	0.376±0.129a	0.678
Functional Richness (FRich)	7.625±0.73a	5.6±2.65a	3.333±0.882a	0.074
Functional Evenness (FEve)	0.609±0.064a	0.345±0.092b	0.361±0.118b	0.029
Functional Divergence (FDiv)	0.219±0.034a	0.237±0.042a	0.116±0.045a	0.178
Rao index (RaoQ)	0.081±0.017a	0.088±0.026a	0.022±0.011a	0.112

Table 6. Analysis of means (ANOM) for the different functional attributes in the three salinity classes. Statistical significances are highlighted in bold (p<0.05). The three codes a, b and c denote the respective location of the proportion values (above, inside and below the upper and lower 95% confidence limits) of the three salinity classes for each trait type. The arrows show the positive (↑), negative (↓) or no trend (n.t.) of trait response to salinity increase.

Trait modality	***Trend with***	***χ^2 (df=2)***	***P value***	***Freshwater***	***Oligo-mesohaline***	***Polyhaline***

	salinity increase					
FEEDING						
Predator	↓	1035.52	**<0.001**	a	a	c
Grazer	↓	63.86	**<0.001**	a	b	b
Shredder	↓	6158.39	**<0.001**	a	a	c
Scraper	↓	814.6	**<0.001**	a	a	c
Deposit feeder	↑	6038.5	**<0.001**	c	c	a
Filter feeder	↓	8.3	**0.0158**	a	b	b
MOBILITY						
sessile	↓	8.3	**0.0158**	a	b	b
swimmer	↓	8245.1	**<0.001**	a	a	c
burrower	↑	7195.53	**<0.001**	c	c	a
walker	↓	3242.25	**<0.001**	a	a	c
ADULT LIFE HABITAT						
aquatic	↓	830.57	**<0.001**	a	a	c
aeric	↑	830.57	**<0.001**	c	c	a
BODY SIZE (g AFDW)						
small (<0.01)	*n.t.*	2.71	0.2575	b	b	b
medium (0.01-0.05)	↑	1723.75	**<0.001**	c	c	a
large (>0.05)	↓	134.61	**<0.001**	a	a	c
LIFE SPAN (years)						
short (< 1)	↓	7709.88	**<0.001**	a	a	c
medium (1-5)	↑	8024.5	**<0.001**	c	c	a
long (>5)	↓	1518.22	**<0.001**	a	a	c
REPRODUCTIVE FREQUENCY						
Iteroparous	↑	1905.59	**<0.001**	c	c	a
Semelparous	↓	1905.59	**<0.001**	a	a	c
HABITAT CHOICE						
generalist	↑	1679.05	**<0.001**	c	c	a
specialist	↓	1679.05	**<0.001**	a	a	c

Table 7. Indicator change point (CP) along the salinity gradient expressed in psu (median of bootstrap replicates), values of purity and reliability criteria, and response (positive + or negative -) of each trait class versus the increase of salinity gradient according to TITAN analysis (traits' abbreviations correspond to the traits of Table 6 in the same order).

Trait	CP (psu)	Purity	Reliability	Response group	Trait	CP (psu)	Purity	Reliability	Response group
F_pred	22.5	0.709	0.917	-	AL_aer*	8.15	1	0.912	+
F_graz*	1.85	0.985	0.912	-	BS_small**	0.8	1	1	+
F_shred*	0.4	0.875	0.998	+	BS_medium**	1.85	0.993	0.981	+
F_scr	22.5	0.529	0.919	-	BS_large*	0.35	0.961	0.758	-
F_depfeed**	8.15	1	0.994	+	LS_short**	0.8	1	1	+
F_filtfeed*	14.25	0.979	0.937	+	LS_medium**	1.85	1	0.999	+
M_sess*	14.25	0.979	0.937	+	LS_long**	4.8	1	1	-
M_swimm**	0.8	1	1	+	RF_iterop**	0.8	1	1	+
M_bur**	8.15	1	0.994	+	RF_semelp**	4.8	1	1	-
M_walk	22.5	0.876	0.804	-	HC_gener**	0.8	1	0.999	+
AL_aq**	0.8	1	1	+	HC_special	0.8	0.851	0.567	+

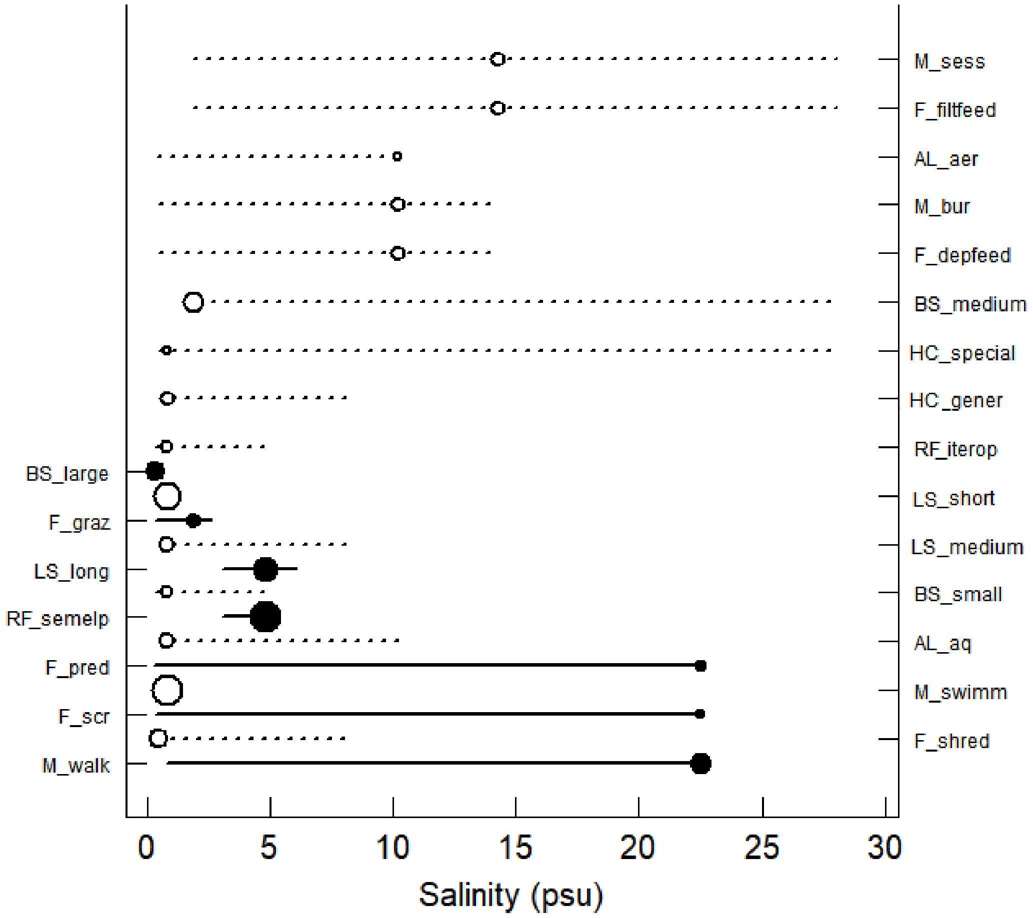

Fig.3 CP and response plots of each trait versus the salinity gradient according to TITAN analysis. Black circles are aligned and correspond to the negative-reponse traits of the left vertical axis,

whereas white circles are aligned and correspond to the positive-reponse traits of the right vertical axis. The size of the circle is proportional to median z-score of bootstrap replicates. Horizontal lines overlapping each circular symbol represent the interval of 5th and 95th percentiles among bootstrap replicates.

Discussion

Small wetlands are often under-investigated biodiversity hotspots that should be instead regarded with great attention under a conservation perspective (Viaroli, Bartoli, and Vymazal 2016). Particularly, the permanent ponds system of Bosco della Mesola hosts a high number of taxa (39) if compared with ponds surrounded by other land use types. Hill et al. (2016) investigated macroinvertebrate diversity in urban, arable and floodplain ponds, recording 22, 30 and 32 taxa, respectively. The positive effect of surrounding forest is further corroborated by the comparison of our data with those of Bazzanti (2015), who found a similarly high number of taxa in temporary and permanent ponds located in a Mediterranean Tyrrenian coastal forest ecosystem. However, biodiversity of coastal ponds is seriously threatened by the pressing environmental changes occurring in these areas. Our findings highlighted the key role of salinity in shaping macroinvertebrate community compositions in coastal permanent ponds and that these communities could be significantly affected by salt water intrusion, which causes a relevant simplification of taxonomic and functional diversity.

This finding is consistent with other studies describing community variations in different aquatic environments (Castillo et al. 2018; Little et al. 2017; Piscart, Moreteau, and Beisel 2005; Zettler et al. 2014), including temporary wetlands (Waterkeyn et al. 2008), and with Kefford et al. (2016) who found a limited tolerance of aquatic insects to salinity increases in freshwater habitats.

Despite the fact that CCA detected salinity as the only statistically significant parameter of Table 2 for describing taxa variation, the Spearman correlations among the environmental variables (Table 3) showed that salinity is significantly positively correlated with temperature and pond area while it is significantly negatively correlated with canopy coverage. Thus, the observed salinity effect on macroinvertebrate communities partly includes some effects of these parameters. Water depth and pond isolation (described through ENN) were among the least important parameters affecting the communities (according to preliminary CCA analysis) and the least associated parameters to salinity. The insignificant effect of water depth was probably the result of the generally shallow profile of all analyzed ponds. In the case of pond isolation, its insignificant effect can be attributed to the fact that the recruitment of flying insects may be guaranteed by the surrounding forested landscape rather than by other water bodies. Moreover, the potential role of isolation could be not evident at finer scale, as the one applied in this study.

Taxonomic diversity indices were negatively influenced by salinity depicting an overall loss of taxa diversity due to saltwater intrusion. Although freshwater ponds show higher values of taxonomic diversity indices, oligo-mesohaline ones hosted higher taxa richness according to Table 5. This partially corroborates to the ‘intermediate disturbance hypothesis’ (Connell 1978) of salinity gradient proposed by Piscart, Lecerf, et al. (2005), according to which an intermediate level of salinity promotes a higher level of biodiversity because of the co-occurrence of both halotolerant and freshwater species. Higher values of taxa richness detected in oligo-mesohaline ponds are due to the occurrence of more insect and crustacean taxa. However, this does not equal to higher values of taxonomic diversity indices because of the decrease of eveness of taxa distribution along the salinity gradient. The increasing proportions of crustacean taxa and crustaceans/insects ratio along salinity levels observed by Boix et al. (2008) were confirmed by our findings only up to oligo-mesohaline levels, while opposite trends were observed at polyhaline level.

As it was shown from Table 6, salinization of permanent ponds above the polyhaline level leads to a drastic loss of functional diversity. The results highlighted that functional traits’ analysis is sensitive to depict community responses mainly to high levels of salinity transitions. This also provides evidences that functional variables can be used as indicators of drastic environmental perturbations and should always be studied when assessing disturbance impacts on biota (Sandin and Solimini 2009).

The salinization of the ponds system toward polyhaline conditions caused the shift of macroinvertebrate communities towards assemblages dominated by r-strategist taxa (sensu Pianka 1970), which are generalist taxa with smaller body size and higher reproductive frequency. Polyhaline ponds are dominated by burrowing deposit feeder taxa, mainly Chironomidae, Tubificinae and Nereidae. All other taxa with different feeding and mobility attributes were rarely found in polyhaline conditions. Piscart et al. (2006) found an increase in filter-feeding at intermediate salinity levels and an increase in deposit-feeding thereafter, in accordance with an energy transfer from water column (i.e. suspended organic material) to sediment (deposited organic material) along a salinity gradient. However, in our case the low filter-feeding abundances observed in oligo-mesohaline ponds were insufficient to justify such food web modification. On the other hand, other changes in the proportion of the feeding traits were observed such as higher proportion of deposit feeder and lower proportions of predator/shredder/scraper in polihaline ponds.

The increase of ionic concentrations requires specific adaptations of macroinvertebrate communities. The faster metabolic rates induced by the elevated maintenance costs for osmoregulation are reflected by smaller body size (Woodward et al. 2005), supporting its use as an effective indicator for assessing community variations in transitional environments (Basset et al. 2012). Some flying insect taxa, as the case of Diptera, have aeric adult life stage in order to reduce the permanence in saltwater environments.

Contrarily to the results presented by Venâncio et al. (2019), who found changes in community structures and trophic relations even with small increments of salinity in laboratory experiment, the macroinvertebrate communities of the permanent ponds of Bosco della Mesola were found to be resilient to moderate salinization. This could be due to the functional redundancy phenomena (i.e. maintaining functional traits over time when facing taxa extinctions) and, in a lesser extent, to the tolerance of some taxa to moderate salinity levels. In fact, when analyzing single traits, the communities of oligo-mesohaline ponds maintain all the functional traits occurring in freshwater ones, except for a lower grazer abundance, which has found to be sensitive even at low salinity variations. Moreover, the observed results for filter feeders and sessile mobility attributes are due to abundance variations of a single species (Corbula sp.) that is the only observed organism with these functional traits, rather than to a general pattern. Since the relation between salinity and abundance of grazer organisms is mediated by the occurrence of microphytobenthos (De Jonge and Van Beuselom 1992; Juneau et al. 2015), the relative decrease of such functional attribute may highlight the sensitivity of microphytobenthic assemblages to salinity variations (Waska and Kim 2010). Therefore, changes in grazer relative abundances could be regarded as a sentinel of salinization effects on microphytobenthic assemblages in the permanent ponds' systems.

When considering functional multi-traits indices, only functional evenness was found statistically sensitive to moderate salinization. This confirms that functional diversity tends to be a better predictor of impacts of environmental changes on ecosystem functioning, and therefore ultimately on ecosystem services, than taxonomic diversity (Sandin and Solimini 2009). The results also suggest that a single trait functional approach, as those performed with the ANOM and TITAN, can be more informative than a multi-traits approach when linking ecological functions with environmental gradients, as also stated by Butterfield and Suding (2013). In fact, while multi-traits functional evenness expresses the overall evenness of traits abundance distribution in filled niche space providing a comprehensive measure of functional modifications, ANOM and TITAN, based on single traits, provide insights on which specific trait is sensitive to environmental change and also allow to better identify the effects due to a single taxa extinction.

Since the salinity levels observed in the study area significantly increased during last two decades (Gerdol et al. 2018), and this trend is expected to keep on in the next years, this investigation can provide evidence to be exported on other deltaic contexts on how biodiversity of the pond systems is expected to also respond to climate change. From an environmental conservation perspective, the ongoing salinization of coastal water bodies observed in our and other study cases (Ketabchi et al. 2016) is a serious threat to aquatic biodiversity. The described effects on macroinvertebrate communities in terms of taxonomic and functional diversity are expected also to have consequences on higher trophic levels and ecosystem stability, functioning and services (Landuyt et al. 2014; Pinto, De Jonge, and Marques 2014; Schratzberger and Ingels 2018). Future management measures should be designed for mitigating the impact of salinization phenomena, which could be also the result of climate change, through the control of human activities in coastal areas and through targeted environmental restoration works.

For instance, in the case of Bosco della Mesola ponds' systems new management measures have been recently undertaken to decrease salinity levels in the near coastal lagoon (Gaglio et al. 2019). Such interventions are expected to have beneficial outcomes in the coastal water bodies which are in contact with groundwaters. However, this does not represent a definitive solution to the conservation of biodiversity in a long period vision.

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APPENDIX 2

(Published as Gaglio, M.; Bresciani, M.; Ghirardi, N.; Muresan, A.N.; Lanzoni, M.; Vincenzi, F.; Castaldelli, G.; Fano, E.A. Aquatic Vegetation Loss and Its Implication on Climate Regulation in a Protected Freshwater Wetland of Po River Delta Park (Italy). Water 2022, 14, 117. https://doi.org/10.3390/w14010117).

Aquatic Vegetation Loss and Its Implication on Climate Regulation in a Protected Freshwater Wetland of Po River Delta Park (Italy).

Abstract: Aquatic vegetation loss caused substantial decrease of ecosystem processes and services during the last decades, particularly for the capacity of these ecosystems to sequester and store carbon from the atmosphere. This study investigated the extent of aquatic emergent vegetation loss for the period 1985–2018 and the consequent effects on carbon sequestration and storage capacity of Valle Santa wetland, a protected freshwater wetland dominated by *Phragmites australis* located in the Po river delta Park (Northern Italy), as a function of primary productivity and biomass decomposition, assessed by means of satellite images and experimental measures. The results showed an extended loss of aquatic vegetated habitats during the considered period, with 1989 being the year with higher productivity. The mean breakdown rates of *P. australis* were 0.00532 d^{-1} and 0.00228 d^{-1} for leaf and stem carbon content, respectively, leading to a predicted annual decomposition of 64.6% of the total biomass carbon. For 2018 the carbon sequestration capacity was estimated equal to 0.249 kg C m^{-2} yr^{-1}, while the carbon storage of the whole wetland was 1.75×10^3 t C (0.70 kg C m^{-2}). Nonetheless, despite the protection efforts over time, the vegetation loss occurred during the last decades significantly decreased carbon sequestration and storage by 51.6%, when comparing 2018 and 1989. No statistically significant effects were found for water descriptors. This study demonstrated that *P. australis*-dominated wetlands support important ecosystem processes and should be regarded as an important carbon sink under an ecosystem services perspective, with the aim to maximize their capacity to mitigate climate change.

Keywords: *Phragmites australis*; carbon storage; carbon sequestration; remote sensing; vegetation indexes; Po river delta; wetland management; climate change

1. Introduction

Wetlands are important environmental components for human well-being and sustainable development since provide a multiplicity of ecosystem services and support aquatic biodiversity

(Mitsch, Bernal, and Hernandez 2015; Xu et al. 2020). Nonetheless, they are affected by several pressures and impacts deriving from human activities and climate change, which cause their loss and the dramatic degradation of their environmental quality worldwide (Kingsford, Basset, and Jackson 2016; Xi et al. 2021). According to the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005b), about the 50% of inner wetlands were lost during the XX century in Europe, Northern America and Australia, while habitats and species they host are among those that suffered the most negative impacts. In order to halt this trend, different policy measures were adopted at international and local levels. The Ramsar Convention (1971) was the first international initiative specifically focused to wetlands protection and had an important pulse since the early 1980s with the adoption of the 'wise use' principle, according to which ecosystem properties should be maintained and restored (Hettiarachchi, Morrison, and McAlpine 2015). EU legislation provided further opportunities to strengthen wetland conservation with the establishment of the Natura2000 network, under the framework of EU Birds and Habitat Directives. Additionally, wetlands have undergone local protection through the widespread establishment of National and Regional parks, as well as other local protection initiatives. Despite the number of protection initiatives undertaken during the last decades, the decline of environmental quality of wetlands is still ongoing (Gaglio, Vassilis G. Aschonitis, et al. 2017; T. Xu et al. 2019). Particularly, aquatic vegetation loss represents one the most significant forms of ecological deterioration in fresh and brackish wetlands, which can be observed in terms of both reduction of vegetated area and net primary production (i.e., decreased biomass production per spatial unit) (Fogli, Marchesini, and Gerdol 2002; Lorenzo Lastrucci et al. 2017; Y. Zhang et al. 2017). Aquatic vegetated habitats, including emergent, floating and submerged macrophytes, are key functional features, being essential energetic inputs for the whole ecosystem. The degradation of their organic matter is an important part of the aquatic carbon cycle, supporting food and debris chains and ensuring the storage of carbon that was previously sequestered from the atmosphere during the vegetative period. Many studies described leaf breakdown dynamics under different conditions and influencing factors, including variations of salinity, surrounding land use, water depth and species (Arroita et al. 2015; Martínez et al. 2013; de Mello Cionek et al. 2021; Paul, Meyer, and Couch 2006; Sauer et al. 2016), but the consequences of aquatic macrophytes loss in terms of reduced organic inputs to support wetland processes and functions are often overlooked. Additionally, aquatic vegetation plays an important role in supporting biodiversity and water quality. In fact, it provides habitat for fishes and invertebrates, as well as nesting sites for several aquatic birds, often of high conservation or economic interest

(Cunha et al. 2019; Gaglio et al. 2019). Aquatic plants release oxygen along water column and remove nutrients by promoting the presence of biofilms on their water-root surfaces (Caraco et al. 2006; Soana et al. 2018).

Due to their importance for wetland functionality and conservation, aquatic macrophytes are considered by the EU Water Framework Directive (WFD, 2000/60/EC) as indicators of the ecological status of water bodies. Therefore, the monitoring of aquatic vegetation is fundamental to inform environmental managers on the ecological conditions of wetlands and link them to the ecosystem functionality. Remote sensing technics rapidly enhanced their potential to assist wetland monitoring during last years to estimate aquatic vegetation extension, biomass, biophysical and biochemical parameters, biodiversity and exotic species presence (Guo et al. 2017; Klemas 2013). At present, a variety of optical and radar remotely sensed images are available for mapping wetland vegetation at different levels by a range of Unmanned Aerial Vehicle (UAV), airborne and space-borne sensors from multi-spectral to hyperspectral sensors, with different temporal and spatial resolution from daily to weeks or months and from meter to hundreds of meters, respectively (e.g., [23,24]). Multi-temporal and spatial remote sensing images have also been positively applied to characterize aquatic vegetation cover and temporal processes (Liira et al. 2010; Liu et al. 2015; Villa et al. 2015).

The common image analysis techniques used in mapping wetland vegetation include digital image classification (e.g., [28]), various change analysis methods and spectral vegetation index (e.g., [29]).The archived moderate resolution Landsat time-series data provide an exclusive opportunity to detect and identify wetland changes as a result of the extensive historic imagery library—free of cost, in fact. Landsat, since 1972, is the longest running uninterrupted Earth observation program (Wulder et al. 2016) and the Landsat archive was the first to offer global imagery at 30 m resolution without restriction in a free and open manner (Woodcock et al. 2008). Landsat images are used in different work to maps the wetlands and their changes over time (e.g., [32,33]). The newly-launched (23 June 2015) and free-available Sentinel-2 (S2) sensor offers a new opportunity to integrate and increase the Landsat dataset, with the advantage of having a higher spatial resolution (10 m of S2 compared to 30 m of Landsat). For example, Pinardi et al. (Pinardi et al. 2018) and Bhatnagar et al. (Bhatnagar et al. 2020) mapped spatial and temporal dynamics of vegetation communities inside wetlands using S2 imagery.

While different studies described aquatic vegetation losses worldwide and many others measured breakdown rates in water bodies, there is an existing gap in linking the consequences of ecological degradation of wetlands on carbon cycle over time. Given the role of aquatic vegetation biomass as energetic input for aquatic biota and wetlands functioning, this connection can assist in estimating the degradation of ecological functioning of wetlands during last decades and in highlighting the role of vegetated habitats in environmental conservation. This work aims to: (i) describe the loss of emergent vegetation in a protected freshwater wetlands of the Po river delta Park (Northern Italy) during a long-term period (1985–2018), chosen as case study of a general pattern occurred in the whole delta in the same period, accounted as net aboveground production measured by means of field calibrated satellite images and (ii) estimate the consequences on carbon cycle by considering both estimated aboveground production, observed breakdown rates and abiotic conditions.

2. Materials and Methods

2.1. Study Area

Valle Santa is a freshwater wetland of 250 ha which is part of the Valli di Argenta, a 3 wetlands-system located in the province of Ferrara (Northern Italy) at 6–9 m a.s.l. (Figure 1). The wetland system was declared as site of national interest in 1976 according to the Ramsar Convention, recognized as Special Protected Areas (SPA) in 2006 and recently designated as Special Areas of Conservation (SAC) in 2019. Although part of river Reno basin, the wetlands are included within the River Po delta park, as a remnant environment of the ancient landscape of the river Po delta. In fact, the river Po delta area was subjected to extensive reclamations since the end of 19th century to the 1969 that drastically reduced the original marshes and inner wetlands in the region (Gaglio, Vassilis G. Aschonitis, et al. 2017).

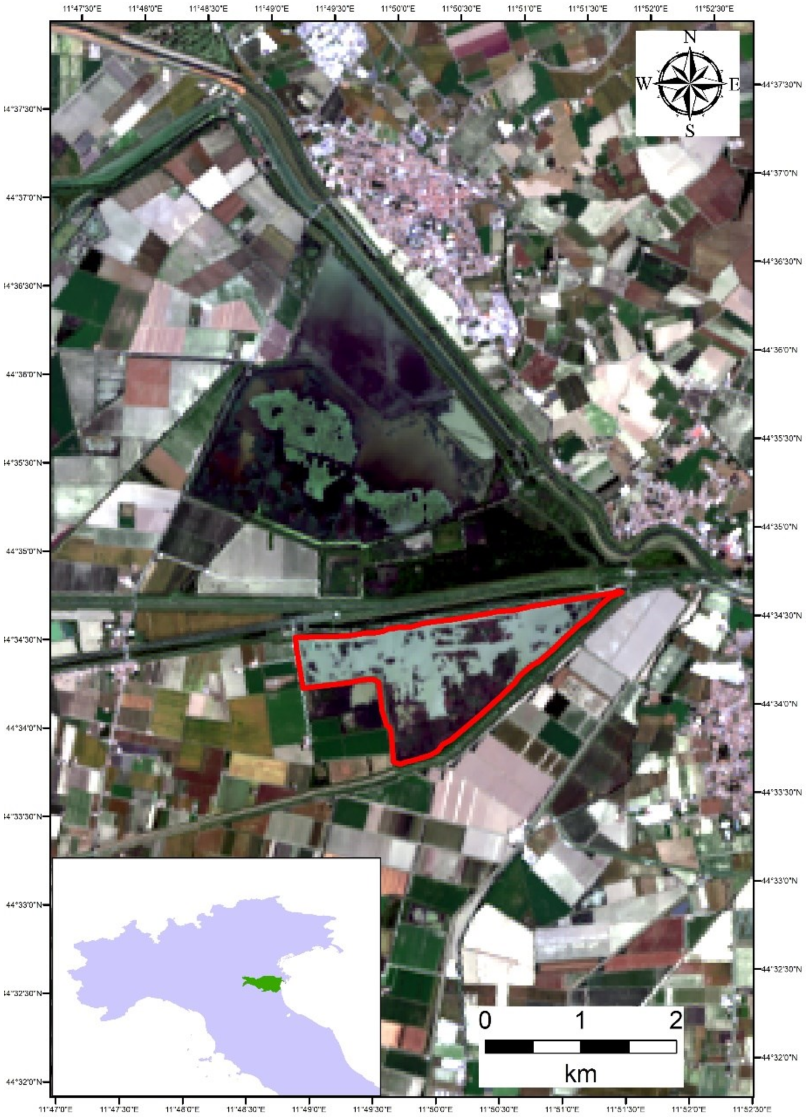

Figure 1. Valle Santa wetland (highlighted in red) and Ferrara province in Italy (in the left back panel).

From a hydraulic point of view, the Valle Santa wetland is devoted to a water storage function. The water input deriving from the Idice stream is stored in the wetland during peak events to prevent floods in the nearby agricultural lands, while it is released when needed for irrigation. The wetland is dominated by monospecific *Phragmites australis* habitats with a sparse presence of floating plants (*Nuphar lutea*) and sparse trees on inner dry zones (Piccoli 1979). This vegetation provides nesting habitat, food and shelter to a large number of birds, amphibians and invertebrates with high conservation value and is a key source of organic matter for the whole ecosystem. Unfortunately, aquatic vegetation suffered important losses during the last decades due to different contributory factors, such as the variation of hydraulic regimes and the grazing activity related to the increasing abundance of common carps. As a consequence, the wetland faced a reduction of ecological functionality which could seriously affects ecosystem services and biological conservation.

2.2. Loss of Aquatic Vegetation and Aboveground Biomass

Due to their dominance on the total biomass of the wetland*, P. australis* was selected as model plant species for the analysis. In fact, the contribution of other macrophytes (e.g., *N. lutea*) and trees in terms of total biomass is assumed to be negligible. Aboveground biomass loss was estimated by the processing and comparing satellite images at different dates after calibration and validation routines based on field measures. The dates considered were: 1985, 1989, 1997, 2010, 2016, 2017 and 2018. The 1985 represents the first date covered by satellite images with suitable quality. The most recent dates (2016, 2017, 2018) were selected for capturing a more reliable description of current situation. Other dates were selected for covering coherently the period of analysis, according the availability of cloud-free images. Aboveground Ash Free Dry Biomass (AFDB) per spatial unit (g AFDB m^2) was measured in 7 georeferenced sampling sites in September 2017 for the calibration of biomass predictions (Figure S1). *P. australis* plants were cut on a 1 m^2 surface and brought to the laboratory for the measure of AFDB, and carbon content. Biomass samples were dried at 50°C for 72 h and subsequently put in a muffle furnace at 375 °C for 3 h to obtain dry weight and ash content, respectively. Carbon content (%) was assessed analyzing 15 g samples of biomass with TOC-V SHIMADZU (solid module SMM-5000A), previously shredded with a 0.2 mm mesh mill (FRITSCH, pulverisette 14).

Abiotic conditions were measured by means of water descriptors at each sampling date to test possible effects due to vegetation presence: NH_{4+}, NO_2^-, NO_3^-, PO_4^{3-}, suspended sediments (total, inorganic and organic fractions), O_2, pH and temperature. Water temperature and dissolved oxygen concentration were measured using a multiparameter probe (YSI Model 85), pH was measured with pH meter (Hanna Instruments HI 9026). NH_4^+, NO_2^- and PO_4^{3-} were measured using the Bower and Holm-Hansen protocol (Bower and Holm-Hansen 1980). NO_{3-} was measured with automatic colorimeter method using AutoAnalyser II (APHA, AWWA, and WEF 1992; Armstrong, Stearns, and Strickland 1967). Total suspended solids (TSS) were quantified filtering water (GFF Whatman filters 0.7 µm porosity) using a vacuum filtration system. The inorganic (ISS) and organic (OSS) fractions of TSS were also measured by drying samples at 105 °C for 72 h and putting them in a furnace at 375 °C for 3 h.

Based on research literature of different vegetation index applied to multispectral optical satellite data, we selected six vegetation indexes used for mapping wetland vegetation and calculated from seven satellite scenes previously corrected to remove atmospheric disturbances. The images were selected from Landsat database (https://earthexplorer.usgs.gov/) and Sentinel-2 Open Access Hub (https://scihub.copernicus.eu) for the months of September-November according the availability of cloud free scenes (Table 1). The satellite images were radiometrically calibrated and converted to surface reflectance after atmospheric correction performed with the 6SV code (Second Simulation of the Satellite Signal in the Solar Spectrum—Vector, (Vermote et al. 1997)). We selected the Continental aerosol model available for 6SV code and the values of Aerosol Optical Thickness (AOT) were retrieved (where available) from daily MODIS products and Ozone concentration from OMI-Aura (Ozone Monitoring Instrument), via NASA Giovanni interface (Acker and Leptoukh 2007). The different vegetation index was mentioned in the Table 2. Their performances on predicting aboveground biomass were tested using the fittest regression model. The Chlorophyll Index Green (CIGreen) was found to be the best predictor ($R^2 = 0.827$, $p < 0.01$) and was therefore chosen for calibration and validation routine with field measures (Table 2). Notably, all the vegetation indices significantly fitted field data and could be suitable for calibration.

Table 1. Dates and satellite source of the processed satellite images. The Landsat-5 images have a spatial resolution of 30 m, the Sentinel-2 images have a spatial resolution of 10 m.

Date	Satellite
9 October 1985	Landsat 5-TM
4 October 1989	Landsat 5-TM
24 September 1997	Landsat 5-TM
12 September 2010	Landsat 5-TM
15 November 2016	Sentinel 2-MSI
21 September 2017	Sentinel 2-MSI
31 October 2018	Sentinel 2-MSI

Table 2. Spectral vegetation indexes tested and fittest models results. CIGreen showed the highest R^2 value.

Vegetation Index	**Reference**	**Fittest Model**			
		R^2	***p*-Value**	**Model Type**	**Equation**
Chlorophyll Index Green (CIGreen)	(Gitelson et al. 2003)	0.837	<0.01	Exponential	$y = e^{(6.29595+0.660924x)}$
MERIS Terrestrial Chlorophyll Index (MTCI)	(Dash and Curran 2004)	0.803	<0.01	Squared	$y = 595.21 + 4747.09x^2$
Modified Chlorophyll Absorption in Reflectance Index (MCARI)	(Haboudane et al. 2004)	0.792	<0.01	Exponential	$y = e^{(6.32046+9.87445x)}$
Normalized Difference Vegetation Index (NDVI)	(Rouse et al. 1973)	0.731	0.014	Exponential	$y = e^{(5.82568+2.94848x)}$
Normalized Difference Aquatic Vegetation Index (NDAVI)	(Villa et al. 2013)	0.709	<0.01	Exponential	$y = e^{(5.8476+2.85724x)}$
Enhanced Vegetation Index (EVI)	(Huete et al. 2002)	0.706	0.018	Exponential	$y = e^{(5.89928+5.66156x)}$

2.3. Breakdown Rate and Climate Regulation

The ecological functions of carbon sequestration and storage that underpin the climate regulation capacity of the wetland were assessed as a function of primary productivity and breakdown rate. The breakdown rates of stems and leaves of *P. australis* were assessed using the litterbag method (Petersen and Cummins 1974). A total of 72 litterbags were filled with 5 g of stems and leaves. Three replicates for both leaves and stems were collected after 7, 30, 60, 119, 182 days and brought to the laboratory for the measure of remaining ash-free dry biomass (AFDB) and carbon content, following the abovementioned procedure. Experiments were replicated in two stations (covered by vegetated habitats and bare sediment) (Figure S1) and with different mesh sizes (10 × 10 mm and 1 × 1 mm) in order to capture local variability.

The decay rates of leaves and stems were calculated according Olson (1963):

$$\boldsymbol{M_t = M_0\, e^{-kt}} \tag{1}$$

where Mt is the AFDB at time t, M_0 is the initial AFDB at the day 0, k is the decay rate (days^{-1}) and t are the number of days spent by the litterbags in water.

The capacity of the ecosystem to sequester and store carbon was estimated according the following formulas adapted from Duke et al. (2015) (Duke, Francoeur, and Judd 2015) and Gaglio et al. (2019) (Gaglio et al. 2019):

$$\boldsymbol{C_{seq} = Cf \times \Big(AFDB - (\Delta_{stem} \times AFDB \times 0.75) + \big(\Delta_{leaf} \times AFDB \times 0.25\big)\Big)} \tag{2}$$

$$\boldsymbol{C_{stored} = Cf \times AFDB} \tag{3}$$

where C_{seq} is the amount of carbon that remain in the system after 1 year (C yr^{-1}), C_{stored} is the amount of carbon stored by the system, AFDB is the aboveground biomass expressed as ash free dry biomass, Cf is the carbon fraction of aboveground biomass, Δ_{stem} and Δ_{leaf} are the annual estimated fraction of carbon loss of stems and leaves, respectively. A 3:1 stem/leaf ratio on the total biomass was considered. C_{seq} and C_{stored} were estimated for each considered date on the basis of AFDB values derived from satellite images. Their monetary value was also assessed using the global social cost of carbon. As calculated by Rickle et al. (2018), 1 t of CO_2 generates a global social cost of 418 US$ (according to a SSP2/RCP60 with discount growth adjusted scenario), corresponding to 96.9€ t^{-1} C (US$-€ exchange rate of 0.86).

3. Results

3.1. Aboveground Biomass over Time

Satellite images were calibrated using aboveground AFDB values, measured in 7 sample sites in November 2017, ranging from 667.7 to 2429.7 g m^{-2} (Table 3). It is worth mentioning that the values reported in Table 3 are not related to net primary production (i.e., biomass production per time), rather they represent the epigean biomass currently present in the sampled site. The 1989 was the year with the highest mean AFDB value (2.49 kg m^{-2}), while 2016 was the lowest (0.97 kg m^{-2}). The progressive vegetation loss over time is spatially showed in Figure 3. The peak of >15 kg AFDB m^2 is due to the sparse presence of terrestrial vegetation (i.e., trees) in the dryer zones.

Table 3. Aboveground biomass, expressed as fresh matter and Ash Free Dry Biomass (AFDB), in the 7 sampling sites used for calibration of satellite images.

Sampling Site	**Aboveground Biomass**		**Coordinates**	
	Fresh Matter ($g\ m^{-2}$)	**AFDB ($g\ m^{-2}$)**	**N**	**E**
1	3219.6	2289.1	44.56938	11.83117
2	2649.6	1739.5	44.57368	11.81999
3	1639.8	1056.1	44.57347	11.82222
4	1178.1	715.1	44.57371	11.82299
5	2166.4	1337.3	44.57289	11.81776
6	1112.4	667.7	44.57372	11.82539
7	3744.5	2429.7	44.57388	11.82720

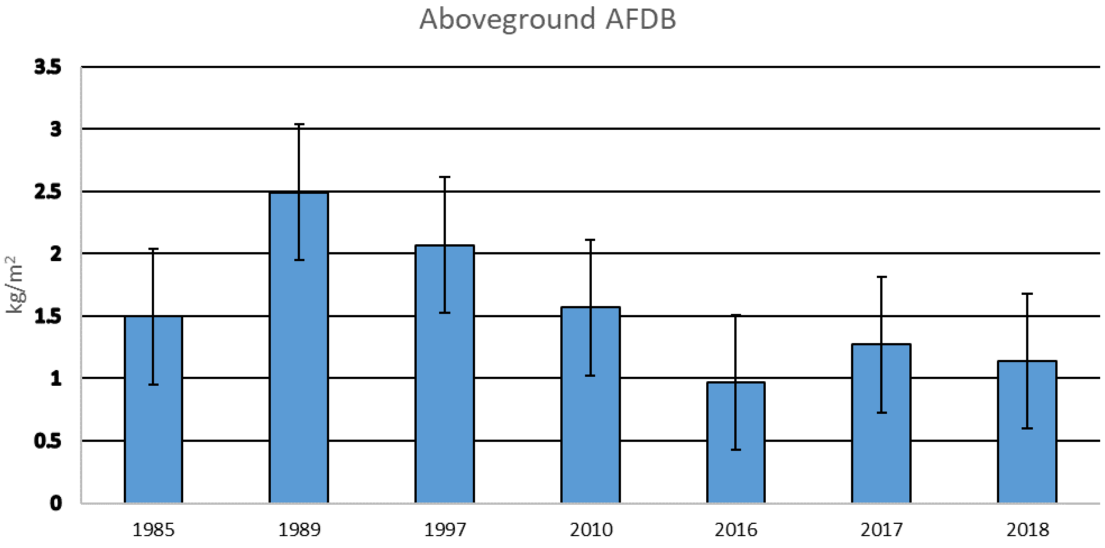

Figure 2. Estimates of aboveground ash free dry biomass (AFDB) per m^2 (±st.err.) over time (1985–2018) for each considered year.

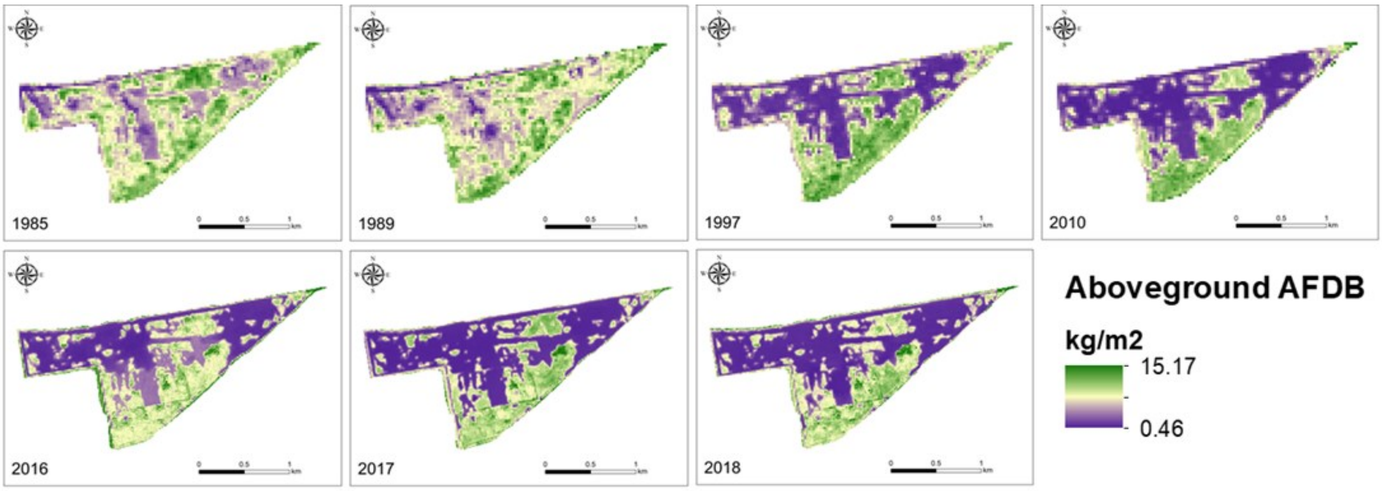

Figure 3. Maps of aboveground ash free dry biomass (AFDB) for each considered year.

3.2. Biomass Decomposition

P. australis bags of two different mesh sizes were collected in two sites. Since no statistical significant effect was observed for mesh sizes nor for sampling sites (Wilcoxon signed-rank test, $p > 0.05$), the data were pooled together to investigate biomass and carbon decomposition dynamics over time. Leaves and stem decomposition was ruled by different decay rates. The observed leaf decay rates were 0.00573 d^{-1} and 0.00532 d^{-1} for AFDB and carbon content, respectively, while stem decay rates were 0.00268 d^{-1} and 0.00228 d^{-1}. After 365 days the estimated remaining fraction of AFDB and carbon was 11.21% and 13.41% of initial leaves biomass and 42.64% and 44.32% of initial stem biomass, respectively.

3.3. Carbon Storage and Sequestration

The mean carbon fraction of *P. australis* plants resulted equal to 42.8% (±0.51 st.err) of dry biomass and 61.9% (±0.01) of AFDB. The latter value was used to calculate carbon sequestration and carbon storage of the wetland in biophysical and monetary terms (eq. 2 and 3) (Table 4). All the values showed a clear pattern of increase between 1985–1989 and a decrease in the remaining period, with the lowest values for 2016. This trend caused a loss of 54.3% of carbon sequestration and storage. Specifically, 2085 t C stored and 736.7 of C sequestered per year were lost during the period 1989–2018. In monetary terms, the loss of ecosystem services corresponds to 202,140€ for carbon storage and 71,414€ yr^{-1} for carbon sequestration.

Table 4. Carbon sequestration and storage provided by Valle Santa wetland and related monetary values over time.

	C Sequestration					**C Storage**		
Year	**g AFDB m^{-2}**	**g C m^{-2}**	**g C m^{-2} yr^{-1}**	**t C yr^{-1}**	**€ 10^3 yr^{-1}**	**t AFDB**	**t C**	**€ 10^3**
1985	1495.9	925.7	327.0	814.3	78.9	3724.9	2305.0	223.4
1989	2492.5	1542.3	544.9	1356.8	131.5	6206.3	3840.4	372.3
1997	2069.0	1280.3	452.3	1126.3	109.2	5151.7	3187.9	309.0
2010	1570.5	971.8	343.3	854.9	82.9	3910.5	2419.8	234.6
2016	967.7	598.8	211.5	526.8	51.0	2409.5	1491.0	144.5
2017	1270.4	786.1	277.7	691.5	67.0	3163.3	1957.4	189.7
2018	1139.1	704.9	249.0	620.1	60.1	2836.4	1755.2	170.1

	Unit	**E**	**N-V**	**V**
NH_{4+}	mg/l	0.14 (±0.03)	0.07 (±0.01)	0.13 (±0.01)
NO_2^-	mg/l	0.12 (±0.03)	0.09 (±0.03)	0.09 (±0.03)
NO_3^-	mg/l	4.27 (±1.11)	2.45 (±1.00)	2.41 (±0.93)
PO_4^{3-}	mg/l	0.01 (±0.001)	0.01 (±0.001)	0.01 (±0.001)
TSS	mg/l	69.15 (±8.75)	71.44 (±9.66)	147.27 (±17.06)
ISS	mg/l	54.18 (±7.49)	52.70 (±7.03)	117.18 (±14.35)
OSS	mg/l	14.97 (±1.52)	18.75 (±2.90)	30.08 (±2.93)
O_2	mg/l	8.63 (±0.70)	9.70 (±0.75)	8.54 (±0.47)

pH	-	7.30 (±0.09)	7.60 (±0.05)	7.60 (±0.04)
Temp	°C	11.30 (±1.53)	11.60 (±1.54)	11.70 (±1.38)

Because of their high variability no statistically significant differences (Kruskal–Wallis test $p > 0.05$) were observed among the three different sites, suggesting that vegetation presence has no significant effects on water conditions (Table 5). However, total suspended solids (TSS) were higher in vegetated sites, although not statistically significant ($p=0.11$). NO_3^- was the nutrient with higher concentrations in all the sampling sites.

Table 5. Mean values (±st.err.) of water parameters at the three sampling sites (E = water entrance; N-V = non-vegetated site; V = vegetated site).

TSS = Total Suspended Solids; ISS = Inorganic Suspended Solids; OSS = Organic Suspended Solids.

2. Discussion

Die-back events of *P. australis* habitats have been well-documented in Europe but the understanding and quantification of their ecological consequences are still challenging (Cerri et al. 2019; L Lastrucci et al. 2017; Van Der Putten 1997). The analysis highlighted the sharp decrease of plant biomass in the Valle Santa wetland over time, mainly due to an extended reduction of *P. australis* habitats. The results revealed that this phenomenon had important implications for carbon cycle. The decrease of aboveground biomass caused the important loss of ecosystem services during the considered period, such as the climate regulation performed by carbon sequestration and storage functions. However, it has to be noted that the estimations of aboveground biomass combine values related to *P. australis* dominated habitats, no vegetated areas (i.e., open water surfaces) and, to a lesser extent, terrestrial habitats with sparse vegetation. Since the majority of the studies measured the productivity of specific freshwater habitats rather than the total biomass amount of wetlands, the comparison with other literature values may be difficult. For instance, measures of biomass production and decomposition rate of common reed beds are available for different brackish environments of the Po river delta. Mean aboveground biomass was found to be approximately 800 g m^{-2} in September-October (Scarton, Day, and Rismondo 2002), while values of annual aboveground production ranged from 876 g dry matter m^{-2} and 1056 g AFDB m^{-2} (Gaglio et al. 2019). However, the higher production values observed in this study are in line with other findings (Brix, Sorrell, and Lorenzen 2001) and can be explained by the higher productivity of *P. australis* in freshwater ecosystems.

The decay rates of P. australis measured in Valle Santa wetland are coherent with other values observed in similar environments. For example, Longhi et al. (2008) (Longhi, Bartoli, and Viaroli 2008) found that about the 40% of whole aboveground part remain undecomposed after one year, which is similar to our estimation (35.4%). As expected, stems decomposed slower than leaves, due to their lower nutrient concentration, high fiber content and highly sclerenchymatous tissues (Gessner 2000). According to the classification of Petersen and Cummins (1974) (Petersen and Cummins 1974), observed k-values of leaves fall into medium range (0.005–0.010), in line with the data of Bertoli et al. (2016) (Bertoli et al. 2016), while stem decay can be classified as slow (<0.005). The lack of significant differences found for mesh sizes, nor for sampling sites, suggests the absence of effects due to macroinvertebrates or local conditions. The first can be explained by the fact that litter bags were abundantly covered by muddy sediment after the first 30 days, thus limiting the action of shredder organisms. The lack of differences between sampling sites can be caused by the homogeneous conditions of the wetland, as also confirmed by water quality descriptors sampled in different wetland zones (Table 5). Other decomposition rates available in literature for Po delta vary largely according abiotic conditions. Scarton et al. (2002) (Scarton et al. 2002) observed 45.4% and 50.4% undecomposed biomass for P. australis leaves and stems, respectively, after one year. Different decomposition rates were found by Gaglio et al. (2019) (Gaglio et al. 2019), equal to the 4.4% and 57.5% of their initial biomass, respectively.

When compared with other ecosystems of the Po delta area, the results demonstrate that aquatic vegetation loss is expected to harm climate regulation capacity more seriously when occuring in freshwater wetlands rather than in brackish environments. This finding has important implications for wetland managers and environmental policy. Under a climate change mitigation perspective, restoring aquatic vegetation in freshwater wetlands can be an efficient solution for sequestering and storing carbon and, because of their high plant biomass productivity, should be considered primarily to other aquatic environments in the Po delta area for improving carbon sequestration. The monetary evaluation of current and past climate mitigation service aims to quantify the economic damages of aquatic vegetation loss in terms of social costs and to inform environmental governance on the potential benefits of environmental restoration.

Although the scope of this work is limited to the consequences of vegetation loss on carbon dynamics, understanding the causes of wetlands deterioration is fundamental to halt this trend and to adopt successful measures for future restoration. In the case of Valle Santa wetland, the factors leading to the observed disappearance of *P. australis* habitats were not clearly demonstrated.

Nonetheless, water level fluctuations may have affected common reed beds, influencing water and nutrient availability, as well as the presence of oxygen in the root zone (Dolinar et al. 2016; Gaberščik et al. 2020). Given the use of the wetland for water regulation purpose, particularly to serve the surrounding croplands, water depth depends on precipitations and agricultural water demand. For this reason, climate change may be an important driver for vegetation loss, decreasing precipitations and increasing the water demand for irrigation. Additionally, further pressures may derive from the grazing activity of common carps, widely abundant in Valle Santa wetland (Lanzoni et al. 2018), that prevent the growth of aquatic vegetation by continuously resuspending sediments from the bottom. The results of this study also contribute to the carbon source-sink dilemma of wetlands (Mitsch and Mander 2018). Wetlands act both as sink of carbon dioxide, by means of sequestration of carbon dioxide from the atmosphere and storage, and as natural sources of greenhouse gases emissions, especially methane. Determining the net results of these processes is a key challenge to determine whether aquatic ecosystems contribute positively or negatively to climate change. In this sense, the results presented in this study provide a quantification of the sink process, fundamental to offset the other processes leading to emissions of greenhouse gases (e.g., methanogenesis). Moreover, aquatic vegetation has a double role of sink for carbon dioxide and to avoid greenhouse gas emissions. In facts, the presence of P. australis in inundated freshwater sediments significantly attenuates methane emissions, both by reducing methanogenesis and promoting methane oxidation (Grünfeld and Brix 1999). On the other hand, the analysis of water chemistry did not highlight any nutrient retention process due to P. australis presence. Even though phytodepuration function of common reed beds are widely documented (Soana et al. 2020), denitrification processes are not supported in absence of constant water fluxes (Castaldelli et al. 2018), such as the case of Valle Santa wetland. The high concentration of NO3- is coherent with the diffuse agricultural pollution which occur in the surrounding land and within the basin of Idice stream. The higher suspended solids observed in vegetated habitats, although not statistically significant, may be due to sediment resuspension performed by common carps and adults of Procambarus clarkii, which are abundantly present in the wetland and other local inland waters (Gavioli et al. 2018; Lanzoni et al. 2018), rather to a trapping effect of vegetation. Therefore, under an ecosystem services perspective, the wetland is used to regulate the timing of water flows but not water quality.

Although the study successfully integrates remote sensing and field measures, some limitations should be considered for the interpretation of results. Satellite images can capture only aboveground biomass. While this will capture vegetation changes over time, the carbon stored in belowground biomass and sediment are omitted. Assessment of carbon sequestration and storage are also affected by the assumption that decomposition processes occur entirely in water and no biomass decay in terrestrial environment are considered.

5. Conclusions

The presence and maintenance of aquatic macrophytes are fundamental for mitigating climate change. In fact, the capacity of inland wetlands to regulate climate relies on the carbon sequestration and storing processes performed by aquatic vegetation, which are necessary to offset and possibly overcome emissions of other greenhouse gases that occur in lentic ecosystems. Therefore, the disappearance of aquatic vegetation represents a serious harm for climate and, more generally, for the provision of ecosystem services. The present study provides a quantification of carbon sequestration and storage over time, both in biophysical and monetary terms, demonstrating that the loss of *P. australis* dominated habitats caused a drastic decrease of climate regulation capacity. This phenomenon may potentially switch the role of inland wetlands from a sink to a source of greenhouse gases.

Monitoring environmental conditions of aquatic macrophytes and assessing the trend of the ecosystem functions and services that depend on their presence are critical aspects for environmental management and sustainable development. The approach adopted in this study also demonstrates the potential values of integrating remote sensing techniques and experimental measures in order to quantify the extent of vegetation loss and its consequences on climate. While remote sensing applications for environmental monitoring and assessment are rapidly evolving, their integration with well-established experimental procedures for the measure of ecological functions can provide a comprehensive understanding of ecological value of ecosystems and address environmental management. This approach finds a successful application in the case of aquatic vegetation, a key component for the functioning of wetlands and the delivery of important ecosystem services, including climate regulation.

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Acknowledgments

Another important chapter in my life ends with this thesis. It has been three full and not exactly simple years, in which the world has changed a lot and we have changed with it. I cannot archive this chapter without thanking the people who have been close to me all this period and they still are. Let's start from the origins, from mom and dad who have always tried to offer, to their daughters, the best they had. You have given us the most important things: a solid and united family, a sense of belonging, the opportunity to study and build our future. A very special thank goes to the most important person in my life: to my sister for her unconditional support… thank you for being you.

A special thank goes to my co-tutor, Mattias, for the patience and professionalism with which he has helped me in these 3 years. I cannot thank General and Applied Ecology group, that has been like may second family. Thank you for all teaches, for your friendship and your collegiality.

Eh, do you want me to not thank my friends? Thank you all for listening to my infinite paranoia, my nonsense but also the most complex and philosophical reasoning.

Last but not least, I want to thank prof. Anna Fano, my tutor. I am grateful to have her as my teacher and my mentor, but also as a friend. She was always there for me, encouraging me to carry on and never give up. Thank you, Annelyse, for your love and care but also for your great advices and suggestions!

Ringraziamenti

Un altro capitolo molto importante della mia vita si chiude con questa tesi. Sono stati tre anni pieni e non proprio semplici, in cui il mondo è cambiato tantissimo e noi con lui. Non posso chiudere questo capitolo senza ringraziare le persone che mi sono state vicino e ancora mi lo so. Iniziamo dalle origini, da mamma e papà che hanno cercato sempre di offrire, alle loro due figlie, il meglio. Ci avete dato le cose più importanti: una famiglia solida e unita, un senso di appartenenza, la possibilità di studiare e di costruirci il nostro futuro. Un ringraziamento speciale va alla persona più importante per me: a mia sorella per il sostegno e l'appoggio incondizionato... Grazie per essere te.

Un grazie particolare va al mio co-tutor, Mattias, per la pazienza e professionalità con la quale mi ha aiutato in questi 3 anni. Non posso non ringraziare quella che per così tanto tempo è stata come una seconda famiglia, il gruppo di Ecologia Generale ed Applicata. Grazie per tutti gli insegnamenti, per l'amicizia e la collegialità.

Eh vuoi che non ringrazi le mie amiche? Grazie a tutte voi, nuove e vecchie, per aver ascoltato le mie infinite pare mentali, le mie stupidaggini ma anche i ragionamenti più complessi e filosofici.

Per ultimo, ma non per importanza, voglio ringraziare la mia tutor, la prof.ssa Anna Fano, alla quale le sono molto grata di averla come insegnante e mentore, ma anche come amica. Mi è sempre stata vicina, incoraggiandomi a continuare e a non arrendermi mai. Grazie, Annelyse, per il tuo amore e la tua cura, ma anche per i tuoi ottimi consigli e suggerimenti!