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Tree planting strategies as Nature-Based Solutions for the provision of ecosystem services

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Tree planting strategies as Nature-Based Solutions for the provision of ecosystem services

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Sommario

Introduction

Climate change, ecosystem degradation, and growing urbanization are among the main global threats today, with significant negative effects on ecological integrity and human health, representing major challenges for the future (IPCC 2022; WHO 2022a). The resulting increase in temperatures, altered precipitation patterns, higher frequency of extreme events, and widespread habitat loss are not only reshaping landscapes but also exacerbating risks for biodiversity, ecosystem functioning, and human well-being. In particular, urban areas concentrate multiple environmental stressors, including air pollution, heat stress, and habitat fragmentation (Grimm et al. 2008), while agricultural landscapes face growing pressures to maintain productivity under increasingly variable climatic conditions (Tilman et al. 2002).

In response to these challenges, at both national and international levels, efforts have increasingly focused on adopting strategies such as biodiversity conservation, climate mitigation, and sustainable land management. Beyond technological and infrastructure-based solutions, ecosystem-based approaches are increasingly considered essential to address environmental problems characterized by interactions across ecological and social dimensions. In this context, the concept of Nature-based Solutions (NBS) has gained increasing attention (Nesshöver et al. 2017). NBS promote the integration of natural processes and ecosystems into land management and policy decisions to provide multiple benefits for both people and nature (Cohen-Shacham et al. 2016; Debele et al. 2023). Within this framework, tree planting practices play a crucial role and are applicable across different spatial scales and socio-ecological contexts (Cortinovis et al. 2022; Hutchins et al. 2024).

This strategic importance is due to the fact that trees and forests play a central role in supporting biodiversity and provide a wide range of ecosystem services (Decocq et al. 2016; Liang et al. 2016; Mori et al. 2017; Thompson et al. 2011). For this reason, their integration into urban and agricultural environments has been widely promoted as a means to counteract environmental degradation while simultaneously delivering benefits to human societies. Among the wide range of tree-based initiatives currently being implemented, increasing attention has been devoted to urban forests and agroforestry systems as two prominent approaches. Urban forests, defined as all trees and wooded areas within and around dense human settlements (Miller et al. 2015), are increasingly integrated into urban planning strategies aimed at improving environmental quality and enhancing urban resilience (Salvatori & Pallante 2021; Jeong & Park 2024). Agroforestry, the practice of growing trees and crops in interacting combinations (Nair et al. 2021), is increasingly promoted as a means of reconciling agricultural production with ecosystem service provision (de Faccio Carvalho et al. 2024; Gupta et al. 2024).

Trees and wooded areas play a crucial role in biodiversity conservation enhancing ecological connectivity and offering suitable habitats for a wide range of plant and animal species (Prevedello et al. 2018). They provide food, shelter, and nesting or breeding sites, supporting different stages of species' life cycles (Rivera et al. 2022; Dueser & Shugart Jr 1978). In addition, tree elements in the landscape, whether continuous forests, riparian buffer strips, hedgerows, or small patches, can function as corridors or stepping stones (Tiang et al. 2021; Fischer & Lindenmayer 2002), facilitating species dispersal and gene flow across fragmented habitats. Several successful afforestation and restoration projects worldwide have demonstrated these benefits, showing how tree-based interventions can effectively strengthen ecological networks and sustain biodiversity (Barzan et al 2015; Navarro-Cerillo et al 2023; Montes-Rojas et al 2024; Menéndez-Miguélez et al 2025). This also applies to urban environments, where urban forests play a crucial role in biodiversity conservation (Alvey 2006): they can support elevated levels of biodiversity and in highly fragmented environments, they can function as stepping stones or ecological corridors, while providing essential habitats for numerous species (Fernández-Juricic & Jokimäki 2001; Werner 2011). Agroforestry systems similarly enhance biodiversity in agricultural landscapes, creating habitats, resources, and corridors for various species, thus mitigating the negative effects of monocultures and habitat fragmentation (Jose 2012; Udawatta et al. 2021).

In addition, trees contribute to several regulating ecosystem services that are particularly relevant in the context of global change. These include climate regulation through carbon sequestration and storage, improvement of air quality through pollutant removal, moderation of local temperatures, and regulation of hydrological processes.

Forest restoration is one of the most effective strategies for climate change mitigation by capturing and storing carbon from the atmosphere (Griscom et al. 2017; Bastin et al. 2019). The amount of carbon that can be sequestered depends on tree species and age, climate, soil type, water and nutrient limitations (Requena Suarez et al. 2019; Salekin et al. 2024). Both the biomass of new trees and the accumulation of carbon in the soil, which is much more long-lived than the aboveground carbon sink, contribute to reduce atmospheric greenhouse gas concentrations. In agroforestry systems, the integration of trees with crops or livestock increases carbon storage both above and below ground, while also providing long-term climate resilience for agricultural production (Kuyah et al. 2019; Eddy & Yang 2022).

Trees and forests can affect local and regional air quality by removing atmospheric pollutants such as O_3, PM_{10}, NO_2, SO_2, and CO (Nowak et al. 2006; Brack 2002) through uptake via leaf stomata (Smith 1990). Air pollution represents a significant threat to public health and WHO data (2022b) show that almost all of the global population (99%) breathes air that exceeds WHO guideline limits (2021) for pollutant concentrations. Several studies establish a correlation between exposure to elevated levels of pollutants and increased incidence of cardiorespiratory diseases and cancer risk (Fajersztajn et al. 2013; Feng et al. 2019), contributing to the number of deaths mainly in children and the elderly (E Almeida et al. 2020). Since the reduction of air pollution is predominantly localized, the impact on public health is particularly significant when tree planting occurs in densely populated areas.

The presence of trees and forests locally influences air and surface temperatures through shading and evapotranspiration (Zhang et al. 2023). Tree cover contributes to the formation of cooler microclimatic conditions, which can support a wide range of plant and animal species (De Frenne et al. 2021). Moreover, in the current context where climate change is driving a global rise in temperatures, a trend further amplified in cities by the urban heat island (UHI) effect (Fini et al. 2017), afforestation and the expansion of tree cover can provide significant health benefits (Wang et al. 2021), particularly for vulnerable populations. Beyond benefits for human health, the cooling effect can additionally benefit a wide range of plant and animal communities (Kotze et al. 2022; De Pauw et al. 2023).

With respect to water regulation, planting trees helps to reduce runoff, intercepting rainfall, alter transpiration rates, increasing the rainfall infiltration and saturated soil hydraulic conductivity, and prolonging the local water storage over time (Buytaert et al. 2007; Silveira et al. 2016; Murphy et al. 2021), representing a key countermeasure in catchments where land cover change has substantially altered hydrological processes (Blöchl et al. 2007; Pattison & Lane 2012). For example, in agroforestry systems, these general hydrological benefits are realized on agricultural land, where trees improve soil structure, reduce erosion, and enhance water availability for crops and livestock, providing an effective solution for climate adaptation (Eddy & Yang 2022). In urban settings, where large portions of land are covered by impervious surfaces, the establishment of new forests can significantly influence hydrological processes by reducing the volume and rate of stormwater runoff, mitigating flood damage, lowering stormwater treatment costs, and alleviating water quality issues (Berland et al. 2017; Dwyer et al. 1992).

However, the effectiveness of tree planting is not universal and the provision of ecosystem services by tree plantations depends on site-specific environmental characteristics, species composition, and plantation design (Baskent et al. 2020; Pérez-Silos et al. 2021). Projects implemented without an appropriate strategy can lead to unfavorable impacts (Farley et al. 2005; Elmarsdottir et al. 2008; Doelman et al. 2020). For this reason a careful planning, management, and context-specific analysis are necessary (Holl & Brancalion 2020). Tree planting that ignores site-specific conditions or the priorities of a given area may be ineffective or even counterproductive. For example, large-scale afforestation on the Chinese Loess Plateau, implemented without considering soil and climatic characteristics, caused water table depletion and increased soil erosion (Feng et al. 2016; Jiang et al. 2016). Similarly, neglecting soil carbon saturation or previous land use can result in soil carbon losses rather than gains (Czimczik et al. 2004). Converting traditionally managed open grasslands into forests can also reduce species richness, decreasing biodiversity-related ecosystem services (Prangel et al. 2023).

Furthermore, it is advisable to define a few primary objectives in terms of ecosystem services, because attempting to maximize all services simultaneously can lead to trade-offs. However, well-planned species selection, plantation design, and long-term management can sometimes deliver multiple benefits simultaneously. For example, Salekin et al. (2024) showed that in New Zealand plantation forests, no single tree species is universally optimal for carbon sequestration, but a mixed-species approach can maximize long-term carbon storage while also enhancing biodiversity. It is fundamental to consider both environmental and social contexts when planning tree planting initiatives. In fact, planning at the landscape scale can help maintain habitat and ecosystem heterogeneity, thereby enhancing connectivity and resilience while reducing trade-offs between ecosystem services (Opdam et al. 2006; Haase et al. 2014a). Comprehending how multiple ecosystem services interact within tree planting initiatives, including the potential trade-offs and synergies among them, remains a major research challenge (Dade et al. 2019; Wang et al. 2024).

The growing interest in tree planting practices is reflected in numerous initiatives promoted at both international, national and local levels, by public as well as private actors. At the global scale, a prominent example is the Bonn Challenge, launched in 2011 and supported by the IUCN, with the goal of restoring 350 million hectares by 2030 (Verdone & Seidl 2017). Similarly, policy frameworks include explicit commitments to increase tree cover, highlighting the role of afforestation and reforestation in addressing climate change and biodiversity loss. An example is the EU Biodiversity Strategy for 2030 (2020), which includes, among its objectives, the commitment to plant at least 3 billion trees by 2030 across the European Union.

While large-scale initiatives play an important political role and help raise and maintain public awareness of global sustainability objectives, such as those embodied in the Sustainable Development Goals (SDGs) (Nilsson et al. 2016), they often rely on broad quantitative targets that provide limited guidance on the spatial allocation and design of interventions. As a result, there is a growing need for approaches capable of informing where tree planting efforts can be most effective, how resources can be allocated efficiently, and how multiple benefits can be maximized across heterogeneous landscapes (Naidoo et al 2006; Holl & Brancalion 2020). In parallel, numerous national and local programs have also included afforestation among their environmental priorities, often in synergy with climate adaptation policies and urban regeneration strategies (John et al. 2025).

As highlighted above, trees in urban environments can provide multiple ecological benefits, including biodiversity support, microclimate regulation, air quality improvement, and water regulation. In addition to their ecological functions, urban forests offer important social ecosystem services, including opportunities for environmental education (Dearborn & Kark 2010), recreational activities, and the promotion of citizens' mental and physical well-being (Nesbitt et al. 2017). Among the numerous local initiatives implemented worldwide, a notable example is the *Forestami* project in the Milan metropolitan area, which aims to increase urban and peri-urban tree cover through coordinated actions involving municipalities, institutions, and citizens, with the objective of improving environmental quality and livability at the metropolitan scale (Forestami 2024). Another relevant case is the *Benjakitti Forest Park* in Bangkok (Thailand), developed within the broader Sponge City framework (TCLF 2025). This is an urban planning approach aimed at reducing flood risk and improving water management by enhancing infiltration, retention, and evapotranspiration through green and blue infrastructure. Established on former industrial brownfield sites, the project integrates urban forestry with water-sensitive design to support flood regulation, stormwater retention, biodiversity enhancement, and recreational functions in a highly dense urban context.

Despite the well-documented benefits of urban forests, important knowledge gaps still limit their effective planning, design, and management. Existing research has largely focused on structural attributes such as tree cover, canopy density, or species richness, while functional ecological processes, including habitat quality, ecological connectivity, species interactions, and the links between functional traits and ecosystem service provision, remain insufficiently understood, particularly in highly fragmented urban landscapes (Goodness et al. 2016; Aronson et al. 2017).

Furthermore, the majority of empirical evidence is based on short-term studies, which constrains our understanding of long-term dynamics in carbon sequestration, biodiversity stabilization, soil development, and ecosystem service provision under ongoing urbanization and climate change (Oldfield et al. 2013). This temporal limitation is particularly critical for processes that unfold slowly, such as forest maturation and the establishment of complex ecological communities. Another major gap concerns the integration of social and cultural dimensions into urban forest research (Haase et al. 2014b). Finally, governance, management, and adaptive planning of urban forests remain weakly addressed in the literature, particularly with respect to participatory approaches, long-term monitoring frameworks, and the integration of ecological objectives into urban planning processes (Andersson at al. 2007).

As explained above, agroforestry systems also provide a wide range of ecological benefits while supporting agricultural production. Agroforestry can reduce the trade-offs often observed between food production and environmental conservation, making it a promising strategy for sustainable land management (Plieninger et al. 2020). These initiatives can make a substantial difference in regions where agriculture exerts strong pressure on ecosystems, contributing to land degradation, soil nutrient depletion, and biodiversity loss. At the international level, agroforestry has been widely promoted by research and development institutions such as World Agroforestry (ICRAF), which coordinates large-scale programs aimed at restoring degraded agricultural landscapes through the integration of trees into farming systems. Initiatives such as Regreening Africa support smallholder farmers across sub-Saharan Africa by promoting context-specific agroforestry practices that enhance ecosystem services while improving food security and climate resilience (CIFOR-ICRAF 2025). In parallel, private initiatives are also contributing to the diffusion of agroforestry practices. Treedom s.r.l. represents an example of a company operating in this field, supporting agroforestry projects in multiple countries by enabling tree planting within agricultural landscapes (Treedom 2025). Through collaborations with local farmers, these initiatives promote tree-based interventions that combine environmental benefits, such as carbon sequestration, soil protection, and biodiversity enhancement, with socio-economic outcomes, including income diversification and improved livelihoods.

Despite growing recognition of agroforestry as a sustainable land-use strategy, important knowledge gaps persist. Much of the existing literature focuses on specific case studies or simplified agroforestry systems, limiting the generalization of results across different biophysical, climatic, and socio-economic contexts (Nair et al. 2021; Jose 2012). A key gap concerns the quantification of trade-offs and synergies between agricultural productivity and ecosystem services. While trees can enhance carbon storage, soil quality, and biodiversity, they may also compete with crops for light, water, and nutrients, with outcomes that depend strongly on species selection, spatial configuration, and management practices (Udawatta et al. 2021). Systematic, comparative assessments of these interactions remain scarce. In addition, socio-economic and institutional dimensions of agroforestry adoption are insufficiently addressed. Factors such as farmers' perceptions, economic incentives, land tenure, policy support, and market access strongly influence the implementation and long-term maintenance of agroforestry systems, yet these aspects are often treated marginally in ecological studies (Plieninger et al., 2020). Finally, long-term empirical studies are particularly lacking, especially with respect to soil carbon accumulation, biodiversity trajectories, and system resilience under future climate scenarios. This limits the development of robust, evidence-based guidelines for agroforestry design and management (Nair et al. 2021).

Taken together, the evidence reviewed above highlights that, despite the growing body of research on urban forests, agroforestry systems, and tree-based interventions more broadly, substantial and cross-cutting knowledge gaps remain. Understanding the conditions under which tree planting interventions deliver desired ecological and social outcomes is therefore one of the central scientific and practical challenges in this field. To make these practices increasingly effective, it is essential to advance our understanding of the ecological and social processes involved, their impacts across different environmental contexts, their potential long-term trajectories under future climate scenarios, and the effects of different choices and management strategies. Despite substantial progress, there are still significant knowledge gaps in these areas (Oldfield et al. 2013; Miller et al. 2020; Li et al. 2021; Scheidel & Gingrich 2020; Nadal-Romero et al. 2023; Dimitrova et al. 2022). These knowledge gaps vary depending on the socio-ecological context (e.g. urban, agricultural, or semi-natural systems) and on the spatial scale of analysis, highlighting the need for integrated and multi-scale research approaches.

Against this background, this thesis seeks to contribute to the advancement of knowledge on tree-based interventions by addressing some of the key gaps identified in the literature. Specifically, the thesis comprises three studies that analyze different tree planting practices and their associated ecosystem services. Each study is distinguished by specific methodological approaches and by the spatial scales at which it was conducted, thus providing a nuanced and comprehensive view of the range of ecosystem services linked to tree-based interventions and identifying potential strategies to improve their effectiveness and sustainability.

This thesis is structured into three chapters, each written in the form of a scientific paper. In detail:

- **CHAPTER 1: Urban afforestation and meadow management as key drivers for wild pollinator conservation**

(Manuscript in draft form, planned for submission to a peer-reviewed journal)

This study aims to advance the understanding of the role of urban afforestation in biodiversity conservation, using wild bees as bioindicators. By adopting a chronosequence approach in forests of different ages and integrating information on forest structure and adjacent grassland management, this study addresses the gap in understanding how forest maturation and management practices jointly influence pollinator communities in highly anthropogenic environments. The results provide insights into functional biodiversity responses to urban afforestation and highlight management strategies capable of enhancing habitat quality and ecological connectivity at the local scale.

- **CHAPTER 2: Assessing the future of coffee agroforestry: carbon dynamics and climate resilience through forest modelling - A Preliminary Study**

(Ongoing study, intended for submission to a peer-reviewed journal once model development is completed. This study was conducted during the doctoral research period abroad at the Helmholtz Centre for Environmental Research – UFZ, Leipzig, Germany, during the third year of the PhD program)

This study represents the initial application of the FORMIND forest model for agroforestry systems, specifically simulating coffee crops to assess how shade trees affect carbon balance and crop resilience under future climate scenarios. Its aim is to contribute to addressing the lack of long-term, mechanistic assessments of agroforestry performance, and to clarifying trade-offs and synergies between shade trees and crop productivity, as well as the limits of agroforestry as a climate adaptation and mitigation strategy under increasing climatic stress. However, some relevant processes still need to be implemented in the model to fully capture the complexity of agroforestry systems.

- **CHAPTER 3: Afforestation priority for multiple objectives at national scale: Italy as a case study**

(Manuscript accepted with minor revisions in Restoration Ecology; the text corresponds to the revised and resubmitted version)

This study proposes a workflow to identify priority areas for afforestation at the national scale, considering four distinct objectives related to different ecosystem services: ecological connectivity, human health, climate mitigation, and water regulation. This approach makes it possible to highlight areas where benefits are maximized, thereby providing a strategic national perspective to support political decision-making in land management and planning. Using Italy as a case study, environmental and social spatial indicators were integrated into a goal-specific multi-criteria analysis, generating priority maps at a 1 km resolution.

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Urban afforestation and meadow management as key drivers for wild pollinator conservation

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CG, RR, PB, FC conceived and designed the research; CG sampled pollinators with contribution of LM, SC; SC monitored forests with contribution of CG, FC; CG, RR, PB analyzed data; CG, RR wrote and edited the manuscript; PB, GV supervised the research.

Abstract

The conservation and enhancement of biodiversity in urban environments have become a crucial priority to mitigate the effects of ecological fragmentation and to safeguard essential ecosystem services. This study investigates the role of urban forests in supporting pollinator communities, with a particular focus on wild bees as key bioindicators. Research was carried out in the city of Milano (Italy), concentrating on Parco Nord, the largest green space in the northern part of the city. The park comprises more than 300 hectares of forests of varying ages and structural conditions, meadows with different mowing regimes, and wetlands. Results show that forest maturity and the resulting structural complexity have a positive influence on the abundance, richness, and diversity of wood-nesting pollinators, while meadow management proved to be a key factor shaping biodiversity in adjacent grasslands: frequent mowing significantly reduces both floral resources and pollinator biodiversity. The study highlights the necessity of integrated management approaches in urban planning to ensure spatial and temporal heterogeneity, including the conservation of mature forest stands with understory and deadwood, together with the maintenance of infrequently mown meadows adjacent to forested areas.

Introduction

Urbanization is rapidly progressing in many regions worldwide, particularly around metropolitan areas (Angel et al. 2011). This process represents one of the major threats to biodiversity (Mcdonald et al. 2008; McKinney 2008). Currently, approximately 55% of the global population lives in urban areas, and this proportion is projected to rise to 68% by 2050 (United Nations 2019). This trend poses significant challenges to biodiversity conservation, highlighting the importance of effective land governance (Huang et al. 2018), which is also a central objective of national and international policies, such as the EU Biodiversity Strategy (2020).

In response to these challenges, the concept of Nature-Based Solutions (NBS) has gained increasing attention (Cohen-Shacham 2016), emphasizing the integration of natural elements into urban landscapes. Urban green infrastructures, and urban forests in particular, represent a strategic component of these solutions (Endreny et al. 2017; Dickinson & Ramalho 2022; Battisti et al. 2024). They provide critical habitats, essential resources, and landscape-scale connectivity, contributing to ecological resilience within otherwise highly fragmented urban matrices (Alvey 2006).

Pollinators are widely recognized as key bioindicators because they are highly sensitive to changes in habitat quality, resource availability, and environmental stressors (Abrol 2012). In addition, they provide crucial ecosystem services essential for biodiversity and ecosystem conservation (Katumo et al. 2022), plant reproduction (Ollerton 2011, 2017; Potts et al. 2016), human health (Garibaldi et al. 2022), and hold significant economic value linked to agricultural production (Smith et al. 2022; Klein et al. 2007). Their decline is now well-documented and primarily driven by direct and indirect human activities (Dicks et al. 2021; Ghisbain et al. 2021; Murphy et al. 2022). Among these, the fragmentation and reduction of natural habitats due to the continuous expansion of intensively cultivated and urbanised areas stand out (Potts et al. 2010; LeBuhn & Vargas Luna 2021). These changes reduce available land and render it impermeable, inaccessible, and resource-poor (Liang et al. 2023). This phenomenon correlates with increased temperatures, particularly in urban centres, directly impacting pollinators while contributing to greater aridity and a decline in food resources (Polidori et al. 2023; Geppert et al. 2023). Other factors negatively affecting pollinator communities include pesticide use (Douglas et al. 2022; Basu et al. 2024), the rise of emerging infectious diseases (Furst et al. 2014; Bailes et al. 2018), and the presence of invasive species (Vanbergen et al. 2018).

Among pollinators, wild bees represent a key group providing pollination services and characterized by diversity, ecological specialization. They include species with different nesting strategies. Families like Megachilidae and some genera of Apidae require mature wood for nesting, most of these species in fact nest in pre-existing wood cavities created by saproxylic organisms, mainly beetles. This cycle highlights the need for mature, medium-to-large-sized wood, typically derived from old-growth forests, and underscores the importance of managing these spaces carefully to avoid removing such resources (Seibold et al. 2015; Staniaszek-kik et al. 2019; Platek et al. 2019; Kotze et al. 2022). Effective conservation must support saproxylic organisms inhabiting the wood, enabling subsequent colonization by wild bees. The proper management of these habitats is delicate but essential for long-term biodiversity conservation. For other genera, such as Bombus, maintaining forest floors with dry leaf litter is crucial to protect overwintering queens and underground nests (Mola et al. 2021). Similarly, sustaining pollinator communities requires ensuring the availability of adequate, diverse, and high-quality food resources in proximity to nesting sites. In this context, forest edges represent critical and biodiverse habitat.

Several studies have shown that urban environments, despite being generally hostile, can support diverse wild bee communities when suitable nesting and foraging resources are available (Theodorou et al., 2020; Silva et al., 2023). The structure of the urban landscape plays a significant role, particularly when it offers habitats heterogeneity and connectivity both within green areas and with surrounding natural spaces (Hrncir 2022; Poole et al. 2024). In particular, forested areas within cities provide essential nesting sites for many terrestrial arthropods, including pollinators that nest in wood, pre-existing cavities, or soil protected by leaf litter (Chase et al. 2023). However, most existing studies have focused on forest presence, canopy cover, or landscape composition at broad spatial scales (Baldock et al. 2015; Aronson et al. 2017), while forest structural maturity and the availability of specific nesting microhabitats are less frequently quantified, particularly in densely urbanized settings (Kotze et al. 2022).

Beyond nesting resources, the availability of adequate, diverse, and temporally continuous floral resources is crucial for sustaining pollinator communities (Ollerton 2017). In urban landscapes, meadow management—particularly mowing frequency and timing—has been identified as a major driver of floral resource availability and pollinator diversity (Noordijk et al. 2010; Lerman et al. 2018). Frequent mowing can drastically reduce flower abundance and the temporal continuity of resources, whereas reduced or staggered mowing regimes can enhance pollinator richness and abundance, even in highly managed urban parks (Wastian et al. 2016). Forest edges represent ecotonal zones of high conservation value, combining nesting opportunities provided by forest structures with abundant trophic resources supplied by herbaceous vegetation (Ye et al. 2021; Perlik et al. 2024; Ulyshen et al. 2024). In urban contexts, these forest–meadow interfaces may play a disproportionate role in supporting pollinators, yet empirical evidence integrating forest structural attributes and grassland management within cities is still scarce (Süle et al. 2025).

While the positive role of forests and habitat heterogeneity has been well documented in agricultural and semi-natural landscapes (Sober et al. 2020; Ulyshen et al. 2023), studies explicitly addressing how urban forest maturity and meadow management jointly shape pollinator communities remain limited (Horák 2018). To address this gap, we adopt a chronosequence approach, using urban forests of different ages as a space-for-time substitution to investigate how increasing forest structural maturity influences wild bee communities and their responses to adjacent meadow management. This study aims to improve our understanding of the role of urban forests in supporting pollinator communities, with a focus on wild bees as key bioindicators. Specifically, we examine how forests at different stages of maturity influence the abundance, richness, and diversity of wood-nesting species, and how meadow management through varying mowing regimes shapes pollinator biodiversity in adjacent grasslands. By integrating these perspectives, the study seeks to evaluate how urban forests and their management contribute to sustaining pollinators and, more broadly, to enhancing biodiversity in metropolitan landscapes.

Materials and methods

Study area

The study was conducted in Lombardy region (northern Italy), in lowland urban areas. We focused on the urban forests of Parco Nord Milano (PNM), the largest green space located in the northern part of Milan, covering a total area of 790 hectares within the city's ring road. Since its establishment in the 1970s, the park has undertaken extensive urban afforestation projects. Currently, forested areas cover more than 300 hectares and include stands of different ages, ranging from recently established woodlands to forests up to 40 years old, thereby representing distinct stages of forest maturity. The park also features meadows with varied mowing regimes to promote biodiversity, as well as recreational spaces and wetlands.

Figure 1: Parco Nord Milano (left); one of the sampling site near forest (right)

The study comprised a total of 15 sampling sites: 12 sites located in meadows adjacent to forested areas in PNM reflecting the diversity in forest stand age, and 3 sites situated in recently planted areas (2022–2024), which served as the control group. Among the 12 forest-adjacent sites, 9 were located at the edge between forested areas and meadows mown twice per year (early summer and autumn), 2 bordered meadows mown monthly from April to October, and 1 was adjacent to a naturally evolving meadow mown every 3–4 years. The 3 control sites were selected to represent meadows not adjacent to mature forest, with management histories and soil conditions similar to forest-adjacent sites: one was within PNM, while the other two were located in nearby municipalities (Legnano and Inveruno; Figure 2).

Forest stands spanning a range of ages were selected as a chronosequence, having been planted between 1984 and 2014. This space-for-time substitution approach was used to evaluate how forest structural development affects wild bee communities. The 12 sampling sites were located adjacent to forest stands established in 11 different years, ranging from the oldest plantation (1984; 39 years old) to the most recent one (2014; 9 years old).

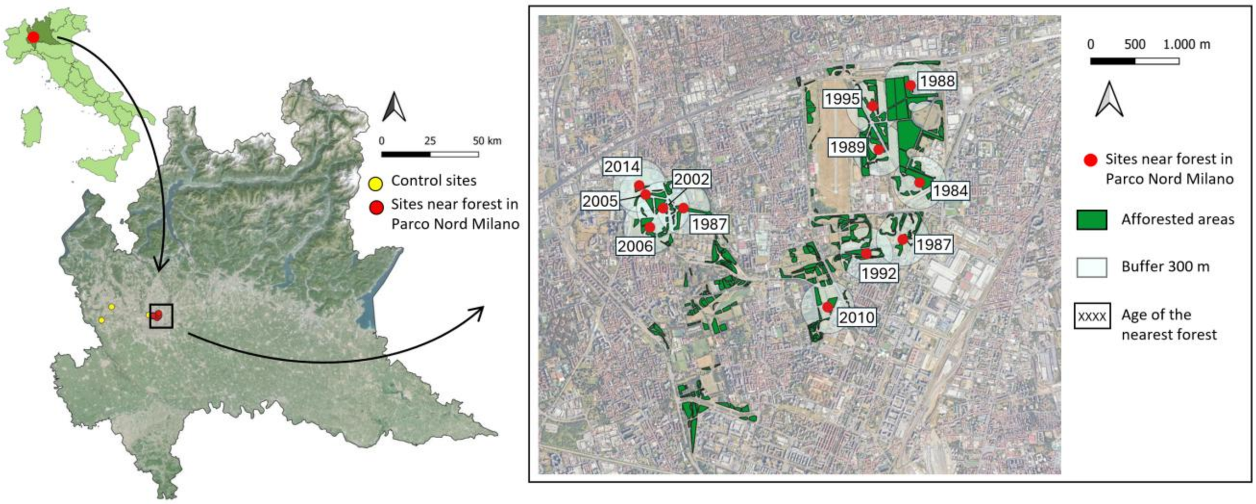

Figure 2: Study area in Lombardy, Italy, and position of sampling sites. On the right, a zoom-in on the sites near forests in Parco Nord, with the age of the closest forest.

Pollinator sampling

Pollinator sampling was conducted monthly from May to September 2023 using a combination of passive and active methods. Passive sampling consisted of three sets of pan traps (yellow, white, and blue), each mounted on stakes, spaced at least 5 m apart, and left in place for 48 h to capture the local pollinator fauna. Active sampling targeted exclusively hymenopterans and was carried out with hand nets for one effective hour within a maximum distance of 30 m from the pan traps. Captured specimens were preserved in ethanol and stored at –20 °C until preparation for morphological identification.

During each monitoring session, vegetation variables were recorded within 1 m² plots, two placed near each pan trap, resulting in six plots per sampling site. These variables included grass height, number of flowering species, heterogeneity of floral abundance, percentage cover of flowering species, and identification of dominant flowering species. In addition, the percentage of bare soil within a 30 m radius and the meadow management condition (mown or unmown) were documented.

Forest data

In each forest stand, forest surveys were carried out in three circular subplots. In each forest-adjacent site, we measured forest structural attributes relevant for wild bee nesting:

- Tree-related microhabitats (TreM), i.e. a morphological feature present on a tree, which highly specialized species can use for at least part of their life cycle (Butler et al. 2022). In each subplot, we recorded the type and number of TreM on every tree with a diameter >7 cm, following the catalogue by Kraus et al. (2016). For the analyses, we selected TreM known to be associated with hymenopterans (Butler et al. 2022). Subplots had a radius of 12.62 m (≈500 m²).

- Deadwood on the ground: within a central circular area of radius 5.64 m (≈100 m²) in each subplot, we measured the central diameter and length of each branch with a diameter >7 cm. We also assessed the decay stage of each piece (1 = sound wood; 2 = intermediate; 3 = rotten, highly decomposed).

Data analysis

All statistical analyses were performed in R. Data analyses were conducted by pooling observations across the entire sampling season for each site, since the sample size (number of sites × repetitions) was not sufficient to reliably support repeated-measures models. Models assumptions (normality and homoscedasticity of residuals) were checked.

Genus-level diversity was calculated using the Shannon–Wiener index. To account for the effect of mowing observed in June, sites were grouped into three categories: (i) control sites, (ii) meadows near forest that were mown at least once immediately before a sampling event, and (iii) meadows near forest where mowing, if it occurred, took place at least three weeks before the subsequent sampling. Differences in genus-level diversity among management categories were tested with one-way ANOVA, followed by Tukey's HSD post-hoc tests.

Relationships between forest age and wood-nesting wild bee communities (abundance, genus-level richness, and diversity) were assessed using linear regressions. In addition, associations between structural forest variables (deadwood volume and TreM) and forest age were tested in the chronosequence plots, including logarithmic transformations where appropriate.

Generalized Linear Models (GLMs) were fitted to test the potential effects of vegetation variables. Generalized Linear Mixed-effects Models (GLMMs) were also tested to account for repeated measures but were not retained due to limited sample size.

Results

We collected and analyzed 1,508 wild bee individuals, representing 22 genera. These included 10 wood-nesting genera (Megachilidae: *Megachile, Osmia, Anthidium, Hoplitis, Lithurgus, Chelostoma, Heriades*; Apidae: *Xylocopa, Ceratina;* Colletidae*: Hylaeus*), 4 brood-parasitic genera (Megachilidae: *Coelioxys, Stelis*; Halictidae: *Sphecodes*; Apidae: *Nomada*), and 8 genera that nest mainly in the ground or use other nesting strategies (Andrenidae: *Andrena, Panurgus;* Apidae*: Bombus, Eucera;* Halictidae*: Halictus, Lasioglossum, Seladonia;* Melittidae*: Dasypoda*).

The ANOVA on genus-level diversity of wild bees, measured using the Shannon–Wiener index, revealed significant differences among the three management categories ($F_{2,12} = 9.30$, $p = 0.0036$). Normality (Shapiro–Wilk test) and homogeneity of variances (Levene's test) were verified and met, allowing the use of ANOVA for comparisons among meadow management treatments. Post-hoc comparisons using Tukey's HSD showed that unmown meadows near the forest had significantly higher bee diversity than mown meadows near the forest (diff = 0.48, $p = 0.0027$), whereas control sites did not differ significantly from either mown or unmown meadows ($p > 0.18$; Figure 3).

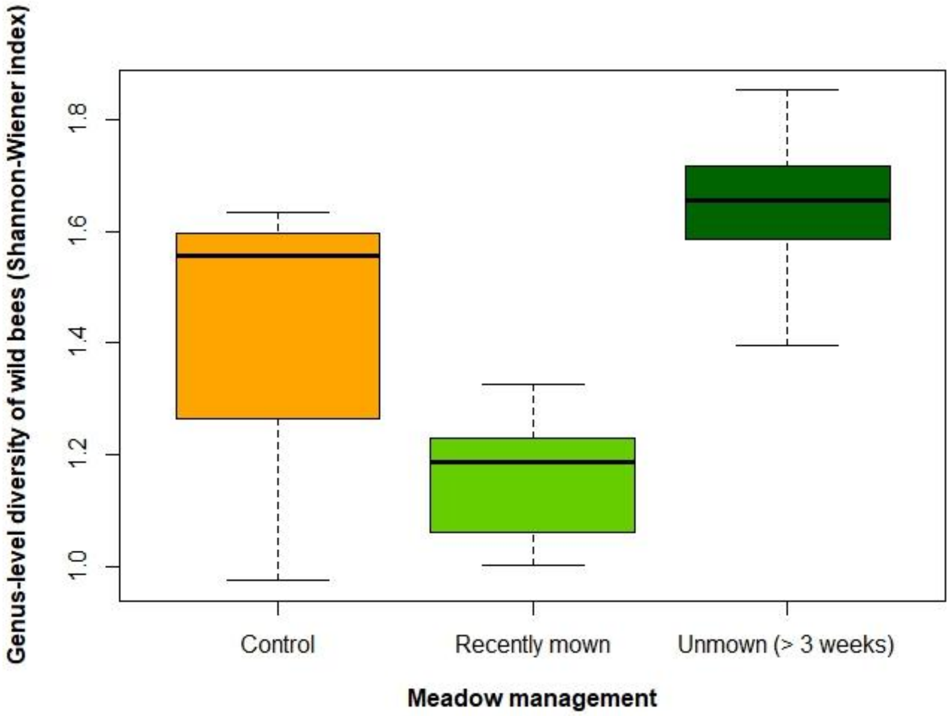

Figure 3: ANOVA test on genus-level diversity of wild bees. The classes are respectively the Control sites (without forest); Recently mown meadow near forest and Unmown (in last 3 weeks) meadow near forest.

Then we restricted the analyses to wild bee genera that nest in wood, with a total number of 118 individuals. Forest age showed a strong positive relationship with multiple descriptors of wood-nesting bee communities (Figure 4). Specifically, the number of individuals increased with forest age ($R^2 = 0.648$, $p = 0.0009$), as did genus richness ($R^2 = 0.619$, $p = 0.0014$), and genus-level diversity (Shannon–Wiener index; $R^2 = 0.469$, $p = 0.0098$). These patterns indicate that older forest stands support more abundant, richer, and more diverse assemblages of wood-nesting wild bees. To reduce potential confounding effects of intensive meadow management, the two sites adjacent to monthly mown meadows were excluded from this analysis (empty circles in Figure 4).

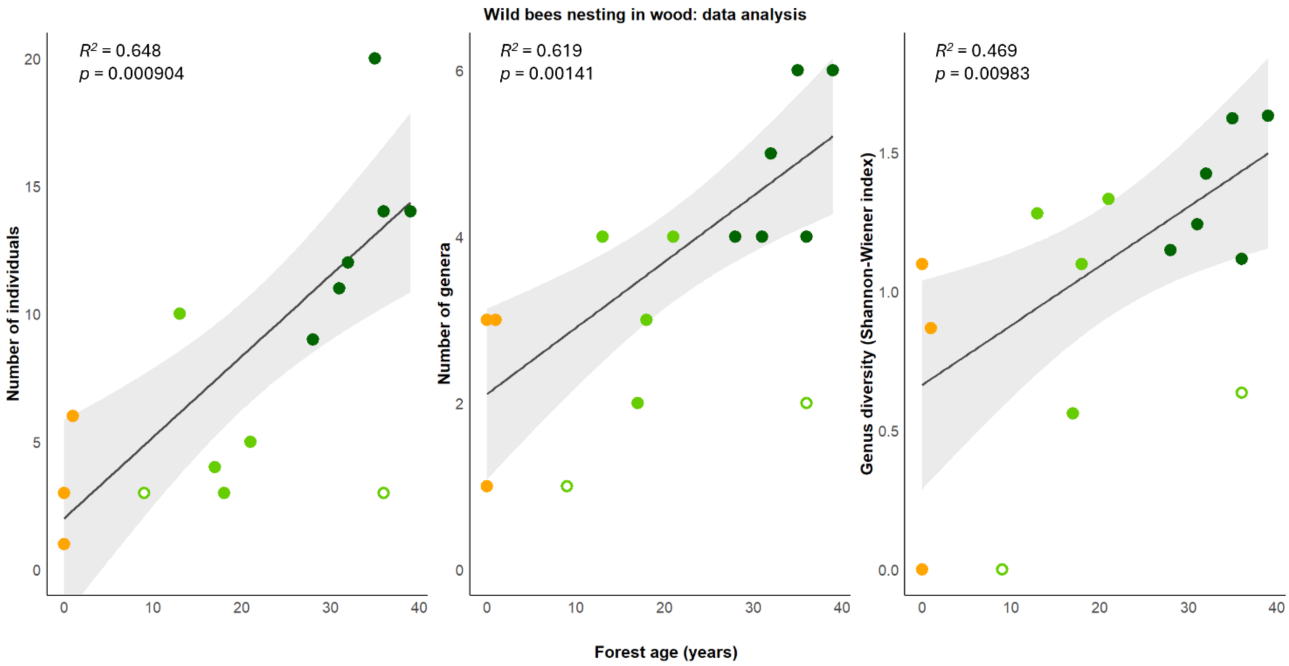

Figure 3: positive linear relationship between forest age and bee communities: abundance, richness and diversity at the genus level.

Structural variables of forest sites along the chronosequence showed clear positive relationships with forest age, particularly for variables directly related to nesting habitat availability (Figure 5). Forest age was positively correlated with both the number of tree-related microhabitats (TreM; r = 0.60) and the total volume of deadwood (r = 0.78; Table 1). Correlations among structural variables indicated that TreM abundance was also positively associated with total deadwood volume (r = 0.72), reflecting the co-development of these features in maturing stands. When deadwood was partitioned by decay stage, intermediate and advanced decay classes showed stronger associations with forest age and TreM abundance than freshly fallen wood (decay class 1), suggesting that biologically available deadwood accumulates with increasing forest maturity.

Regression analyses confirmed these patterns: total deadwood volume increased logarithmically with forest age (R^2 = 0.53, p = 0.003), while the number of TreM increased linearly with forest age (R^2 = 0.36, p = 0.031). Although coefficients of determination were moderate, these results consistently indicate increasing structural complexity and availability of nesting substrates in older urban forests.

Table 1: Pearson correlation coefficients among forest structural variables along the chronosequence. Deadwood volume is partitioned by decay class (cl1 = sound wood; cl2 = intermediate decay; cl3 = advanced decay).

	Age forest	TreM	Total volume of deadwood	volume of deadwood (cl1)	volume of deadwood (cl2)	volume of deadwood (cl3)
Age forest	1.00	0.60	0.78	0.40	0.54	0.47
TreM	0.60	1.00	0.72	0.34	0.02	0.71
Total volume of deadwood	0.78	0.72	1.00	0.61	0.56	0.57
volume of deadwood (cl1)	0.40	0.34	0.61	1.00	-0.06	-0.20
volume of deadwood (cl2)	0.54	0.02	0.56	-0.06	1.00	0.36
volume of deadwood (cl3)	0.47	0.71	0.57	-0.20	0.36	1.00

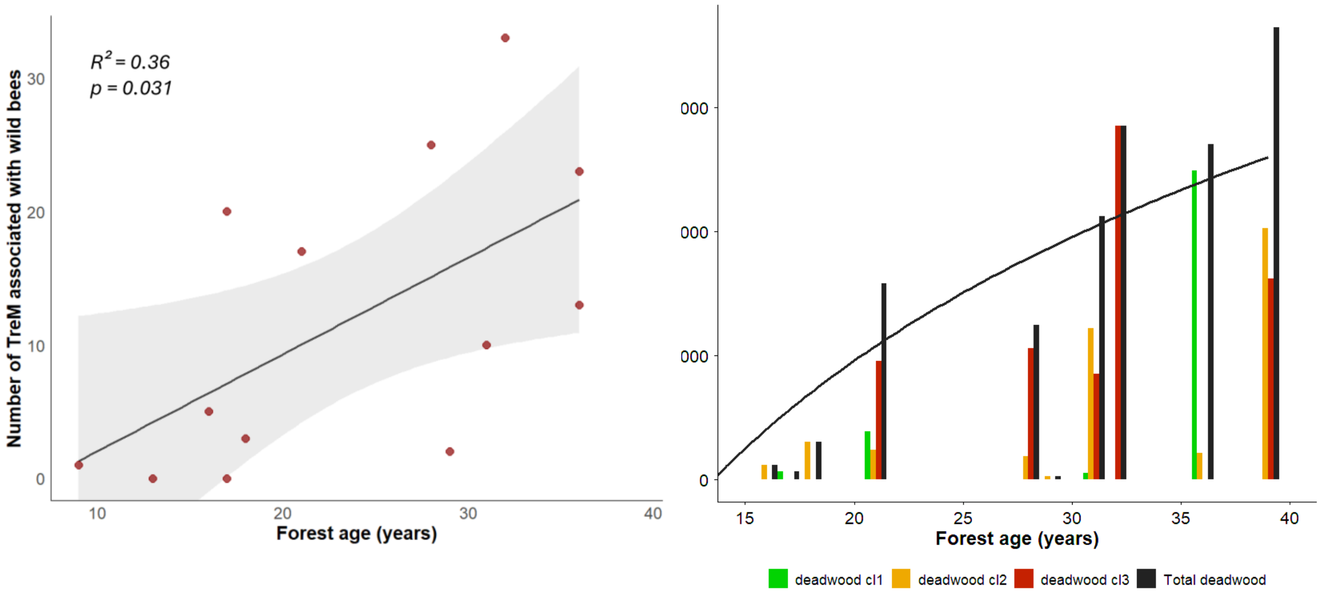

Figure 5: Linear relationship between the number of TreMs and forest age (left), and logarithmic relationship between total deadwood volume and forest age (right). On the right, deadwood volume values for different decay classes are also shown with a barplot.

Generalized Linear Models did not reveal significant effects of local vegetation variables (flower richness and floral cover) on wild bee abundance or genus-level diversity at the site level, with the exception of mowing management, which showed a significant effect consistent with the patterns observed in the ANOVA.

Discussion

In a highly urbanised context, such as the metropolitan area of Milan, where biodiversity is under significant anthropogenic pressure, large urban parks serve as crucial refuges for pollinators (Hall et al. 2016; Silva et al. 2023). Our findings support and extend previous evidence that urban green spaces can sustain diverse pollinator communities when suitable nesting and foraging resources are available. Forested areas, in particular, provide essential nesting sites for saproxylic pollinators and trophic resources along forest edges, which represent habitats rich in plant biodiversity (Perlik et al. 2024; Brockerhoff et al. 2017).

In this study we demonstrate not only that the presence of forests increases pollinator richness and abundance, but also that forest age, structural maturity, and the resulting complexity of TreM exercise a positive influence, in accordance with Ranalli et al. (2025). Nesting opportunities offered by forests are mainly associated with the availability of deadwood in advanced stages of decay, where wild bees either exploit pre-existing cavities created by other saproxylic insects or easily excavate new ones for egg-laying and provisioning (Rappa et al. 2023; Michener 2007). By adopting a chronosequence approach, our study provides empirical insights from a densely urbanised setting and demonstrates the potential of this approach for investigating how forest structural development influences pollinators in urban contexts, as suggested by recent literature (Kotze et al. 2022). Our chronosequence approach provides a framework that is reproducible in other urban contexts where afforestation has occurred at different times, offering a practical tool to guide both the planning of new urban forest interventions and the management of existing green spaces to support pollinators and biodiversity.

As also demonstrated by the meta-analysis of Lassauce et al. (2011), deadwood volume alone is not sufficient to ensure an increase in biodiversity; it is also necessary to include deadwood type and decay class in forest assessments. Our findings highlight how careful management by the urban park authority has allowed the persistence of organic material on the ground, promoting the presence of woody elements at different stages of decomposition. This result is particularly relevant in urban forests, where safety concerns and aesthetic preferences often lead to the systematic removal of deadwood. In line with the objectives of the European Green Deal, deadwood plays a fundamental role in maintaining biodiversity and ecosystem services, as well as in the bioeconomy. It is therefore necessary to integrate the different requirements related to habitat conservation with economic demands and public understanding (Mansuy et al. 2024). The results show how the total volume of deadwood was positively correlated with forest age, confirming that the retention and natural maturation of organic elements favour the creation of suitable TreM for sheltering and nesting across multiple taxa. This study thus underscores the importance of such TreM for wild bees, showing that forest age supports both a higher diversity of TreM (Stokland et al. 2012; Asbeck et al. 2020a,b) and greater nesting opportunities for Hymenoptera (Michener 2007).

Having established the importance of forests, particularly older ones, as nesting and refuge resources, we also tested their efficiency in supporting pollinator communities. Results indicate that mature forests sustain more abundant, richer, and more diverse pollinator assemblages, confirming results of De Groot et al. (2016). All studied forests are located within the same large urban green area, Parco Nord Milano, where they share similar climatic conditions, anthropogenic pressures, and species composition. Nevertheless, forest age proved to be a crucial determinant of nesting site availability and, consequently, of pollinator community support. While stand age itself is influential, it is also necessary to consider the broader landscape context, including the proximity of stands of different ages. For example, the site associated with a relatively young 13-year-old stand (planted in 2010) exhibited unexpectedly high values, likely influenced by older stands (planted in 1994) located within 300 m. More generally, the overall connectivity and configuration of urban green spaces may amplify or limit these local effects, affecting pollinator dispersal, resource access, and community dynamics across the urban landscape (Graffigna et al. 2024; Płaskonka et al. 2024).

Forest habitat also includes edges, which are well known as ecotones of high conservation value, providing abundant trophic resources (Vu Ho et al. 2023; Matlack & Litvaitis 2010). Sampling was carried out in this transitional zone, adjacent to both feeding and nesting resources. Vegetation variables such as flower richness and cover were not significantly correlated with wild bee communities, likely because their effects were masked by management practices, which instead showed strong influence. This result suggests that frequent mowing may override finer-scale vegetation effects, particularly in intensively managed urban parks. Despite being embedded in the urban matrix, Parco Nord Milano represents the largest green area in the city and is easily accessible, attracting large numbers of citizens for daily recreational activities. Park management, however, pursues a dual approach, balancing biodiversity conservation with recreational needs. Furthermore, beyond ecological benefits, the maintenance of wildflower meadows and retention of deadwood also provide educational and cultural ecosystem services, creating opportunities for public engagement, environmental education, and nature-based recreational activities (Paudel & States 2023; Pinto et al. 2022). Our results clearly show that forests adjacent to frequently mown meadows support significantly lower wild bee diversity than those bordering meadows under reduced mowing regimes, consistent with previous findings from urban grasslands (Wastian et al. 2016; Biella et al. 2025). These findings highlight the importance of considering forest and meadow management jointly, rather than as independent components of urban green infrastructure.

Therefore, urban afforestation alone, while necessary, is not sufficient as a biodiversity conservation strategy (Löfroth et al. 2023). Our analyses clearly demonstrate that meadow management plays a key role: frequent mowing leads to a marked loss of biodiversity by reducing both floral resources and refuge opportunities. This study highlights the need for integrated management approaches that ensure heterogeneous urban green spaces encompassing both meadows and forested areas. Forest patches must be conserved over time to allow structural maturity and should be maintained without the removal of understory or natural woody elements (Siitonen et al. 2000). At the same time, for pollinator communities, it is crucial to ensure continuous trophic resources through infrequent mowing in areas adjacent to forest stands.

Finally, this study is based on a single year of sampling, and pollinator communities may vary interannually in response to weather, phenology, and stochastic processes. Long-term monitoring would be valuable to capture temporal variability and to assess the persistence of the observed patterns (Selden et al. 2025). Future multi-year monitoring would be valuable to capture temporal variability and to assess the persistence of the observed patterns. Nevertheless, our findings provide clear, management-relevant evidence that forest maturity and meadow management jointly shape pollinator communities in urban landscapes, offering practical guidance for enhancing biodiversity in metropolitan areas.

Conclusion

Urban reforestation is already known to enhance biodiversity, and this study confirms its effectiveness also for pollinators. In summary, the consistent patterns observed at the forest level indicate that long-term urban reforestation can progressively increase habitat availability for wood-nesting pollinators, with positive spillover effects at the landscape scale. When combined with extensive meadow management, this strategy represents a promising NBS to reconcile ecological and social needs in metropolitan areas. Wild bees, used here as a model group, represent valuable bioindicators of habitat quality and health, suggesting that targeted actions to support pollinators may generate broader benefits for overall biodiversity.

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Assessing the future of coffee agroforestry: carbon dynamics and climate resilience through forest modelling - A Preliminary Study

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CG, FB, AH conceived and designed the research; CG, FB developed the methodology; CG wrote and edited the manuscript; AH, GV supervised the research.

Abstract

Agroforestry represents a key strategy to enhance the resilience of tropical agriculture under climate change, particularly for temperature-sensitive crops such as coffee (*Coffea arabica*). In this study, we present a first attempt to adapt the individual-based forest model FORMIND to simulate coffee agroforestry systems under future climate scenarios. Results show that coffee biomass and productivity are strongly influenced by canopy structure, with shade trees playing a crucial role in mitigating heat stress. Monoculture systems appear unviable under projected temperatures, emphasizing the importance of agroforestry. Model improvements, particularly the inclusion of interactions and specific parameterization, are needed to better represent agroforestry dynamics and carbon balance.

Introduction

In recent decades, the combined effects of climate change, soil degradation, and the growing demand for agricultural production have highlighted the urgent need for more resilient and sustainable production systems (IPCC 2022). In this context, agroforestry—the practice of growing trees and crops in interacting combinations (Nair et al. 2021)—has emerged as a key strategy to reconcile food security, environmental conservation, and adaptation to global changes (Plieninger et al. 2020; Quandt et al. 2023). This approach is recognized as a Nature Based Solution (NBS) (de Faccio Carvalho et al. 2024; Gupta et al 2024), owing to its role in providing important ecosystem services, including the mitigation of environmental stress, the enhancement of biodiversity, and carbon sequestration. Several studies have shown that agroforestry systems possess a higher carbon sequestration potential than monocultures, due to both the accumulation of tree biomass and the greater stability of soil organic matter (Nair et al. 2009; Udawatta & Jose 2012). Beyond carbon storage, these systems contribute to improved soil fertility, microclimatic regulation, and water management (Kanzler et al. 2019; Rolo et al. 2023; Ngaba et al. 2024), thereby supporting agricultural productivity over the long term.

Such practices are particularly strategic in tropical regions, where environmental conditions, often characterized by high levels of habitat fragmentation and deforestation, intersect with fragile socioeconomic contexts (Leakey 2014). In these landscapes, agroforestry not only enhances ecological connectivity and mitigates edge effects near forest fragments (Sistla et al. 2016) but also reduces the long-term degradation associated with monocultures, sustaining soil fertility and productive capacity (Schroth et al. 2000; Muchane et al. 2020). A particularly relevant component is the use of shade trees, which provide canopy cover and a range of services to the crops beneath them (Tscharntke et al. 2011). These practices are already widespread in tropical crops such as coffee and cocoa, where shade trees regulate light availability, reduce evapotranspiration, and contribute to the creation of more stable microenvironments, favorable to both yield and product quality (Niether et al. 2018; Piato et al. 2020). In particular, *Coffea arabica*, which is highly sensitive to temperature, grows best within an optimal range of 18–21 °C and tolerates up to 24 °C, while moderate shading (40–70%) helps maintain microclimatic stability, improving photosynthesis and sustaining fruit development under tropical conditions (Lin 2007).

Despite these well-established benefits, critical questions remain regarding the effectiveness of agroforestry systems under future climate scenarios (Quandt et al. 2023). The capacity of agroforestry systems to buffer climate change impacts is not uniform and strongly depends on species composition, system structure, and future climatic conditions (Abigaba et al. 2024; Dobhal et al. 2024). For example, while shade trees can mitigate heat stress and reduce temperature extremes, their effectiveness may decline under scenarios characterized by increased aridity and altered rainfall regimes, where competition for water resources may become a critical limiting factor (Mbow et al. 2014; Abigaba et al. 2024). Consequently, agroforestry should not be regarded as a universally resilient solution, but rather as a context-dependent strategy whose long-term performance under climate change remains uncertain. Projections indicate an increase in mean global temperature, accompanied by higher climatic variability and more frequent extreme events (IPCC 2022). The resilience of agroforestry to increased aridity, altered rainfall regimes, and shifts in the distribution of both tree and crop species represents one of the main challenges for research and for the design of agricultural and environmental policies (van Noordwijk et al. 2021).

Forest models can provide a useful tool to support this research, as, once adapted to agroforestry contexts, they allow the exploration of long-term ecological and agronomic dynamics under alternative climate scenarios. Despite a growing body of empirical research, significant knowledge gaps remain regarding the long-term dynamics of agroforestry systems under future climate scenarios (Tranchina et al. 2024; Patil et al. 2025a). Most existing studies rely on short-term observations or site-specific experiments and rarely capture the combined effects of climate change, tree–crop interactions, and management decisions over decadal timescales. In particular, quantitative assessments of carbon balance and system resilience under contrasting climate trajectories remain limited (Mahato & Kadaverugu 2024). Process-based and mathematical forest models are increasingly recognized as essential tools to address these gaps, as they allow the explicit representation of growth, competition, and carbon allocation processes under changing environmental conditions. However, although numerous models have been developed to simulate physiological processes and forest dynamics across a wide range of spatial scales, their application to agroforestry systems remains limited, highlighting the need to adapt and extend existing modelling frameworks to capture the structural complexity and functional interactions characteristic of agroforestry systems (Mahato & Kadaverugu 2024; Giannitsopoulos et al. 2025).

The present study aims to initiate the development of agroforestry simulations using FORMIND, an individual-based, process-based forest model previously applied to both natural and managed forest systems. We focus on coffee agroforestry to quantify carbon balance and assess whether the use of shade trees may remain a viable strategy under future climate scenarios.

Methods

<u>FORMIND Forest Model</u>

FORMIND is an individual-based and process-based vegetation model that simulates the growth of forests (Fisher 2016). It describe forest succession in small-scale forest patches (patch: 20 m × 20 m) and the total area can range from 1 ha up to several km^2 (Armstrong et al. 2020; Rödig et al. 2020). The four main processes considered are: tree growth, mortality, regeneration and competition.

Tree growth in FORMIND is driven by light availability, which is calculated as incoming irradiance at the top of each tree crown and depends on vertical canopy structure and shading by neighbouring trees (competition for light). Growth can be constrained by environmental limitations, including temperature and soil water availability, as well as by competition for light. Mortality in FORMIND occurs through multiple mechanisms, applied sequentially. These include background mortality, size-dependent mortality, space limitation, and growth-dependent mortality. Additional mortality processes, such as treefall-induced mortality and fragmentation effects, can be activated depending on the simulation setup. Regeneration is simulated through the production of seeds by adult individuals. Seeds enter a seed pool and can germinate and establish only if local light and space conditions are suitable (FORMIND documentation 2025).

FORMIND simplifies tree shape by representing stems and crowns as cylinders. In addition, it groups species into Plant Functional Types (PFTs) based on their functional traits, such as maximum height, growth rate, or light requirements. This simplification is particularly useful and effective in tropical forests, where the high diversity of species would otherwise make modeling extremely challenging (Hiltner at el. 2018).

Parameterization

So far, FORMIND has been applied to natural or managed forests, with site-specific parameterizations. Since no comprehensive database with field measurements from coffee agroforestry systems was suitable, we initiated the adaptation of the model by building upon an existing, well-tested parameterization (Hiltner et al. 2018, 2021) developed for the Paracou test site in French Guiana (Location: 5°16′28″N, 52°55′25″W). This choice was motivated by the fact that Paracou represents a tropical environment with temperature ranges comparable to those found in areas where coffee is cultivated under shade-tree systems.

Starting from this Baseline Parameterization (BP), we applied the following modifications:

- Temperature and precipitation data: daily temperature and precipitation data correspond to the Paracou site, used for BP. Precipitation data were reduced by 50% to fall within a range suitable for coffee cultivation (La et al. 2019, Muñoz-Villers et al. 2020). Therefore, while this study is not tied to a specific site, it aims to implement the model for coffee agroforestry systems and to provide a framework that can be applied to real coffee-growing sites.
- Climate scenarios and simulation period: we implemented climate change scenarios based on four RCP pathways (RCP 2.6, 4.5, 6.0, and 8.5 (Dunne et al. 2014)) covering the period 2001–2100, and used by Hiltner et al. (2021). Since coffee plantations typically do not exceed 30 years of productive lifespan (Bunn et al. 2015; Moat et al. 2019), simulations were conducted over 30-year intervals. The 100-year period was therefore divided into three equal intervals: 2001–2033, 2034–2066, and 2067–2100, each of which was used separately in the simulations to represent the dynamics of agroforestry systems under both current and future conditions (respectively defined here as RCPX.Xa, RCPX.Xb, RCPX.Xc).
- Area size: the simulation area covers 9 ha, divided into 225 patches of 20 × 20 m each.
- Selection of PFTs: coffee agroforestry systems can vary widely in terms of management intensity and structural complexity. Among shaded coffee systems, strategies range from shaded monocultures to the introduction of coffee into native forest ecosystems (Moguel & Toledo 1999). In this study, we simulated different scenarios using either two or three PFTs. In the two-PFT scenarios, one PFT represents coffee plants and the other represents medium-height shade trees, reflecting a polyculture system with selected shade trees, such as leguminous species that contribute to soil nitrogen or are used for commercial purposes. In the three-PFT scenarios, an additional PFT represents tall-growing trees, adding structural complexity. Since shade trees are not fully mature at the initial stage, this simulation represents a tree-planting intervention in agroforestry systems moving toward a traditional polyculture system with native forest species. For the two shade tree PFTs, we selected PFTs 4 and 7 from the BP, both fast-growing.

Considering that traditional agroforests typically have an average canopy height of 20–30 m, whereas in commercial polycultures generally the average doesn't exceed 15 m (Moguel & Toledo 1999), we limited the potential maximum heights to 20 m for PFT 4 and 35 m for PFT 7.

- Management of the plantation: assuming a managed system, we deactivated the options for treefall of dying trees, which would have caused the mortality of neighboring plants, and for natural regeneration.
- Coffee parametrization: to model coffee plants, with a medium growth rate (Useful Tropical Plants Database 2014), we selected and adapted PFT 3. Coffee plants, in particular *Coffea arabica*, are shrubs or small trees. Coffee plants were assigned a maximum height of 3 m, reflecting regular pruning practices in cultivated systems. Although this approach does not capture short-term biomass fluctuations due to pruning, it ensures realistic plant dimensions. We set the Leaf Area Index (LAI) to 4.2 (Costa et al. 2019) and the maximum leaf photosynthesis rate to 4.0 CO_2 m^{-2} s^{-1} (Ramalho et al. 1997). Based on a bibliographic survey of dendrometric data (Negash et al. 2013; Tesfay et al. 2022), we modified the plant geometry parameters as follows. Crown diameter (CD) is computed from DBH using the allometry compiled by Jucker et al. (2017); for our simulations we set the crown coefficients to cd0 = 3.5 and cd1 = 0.2. To calculate plant height as a function of diameter, FORMIND model uses Michaelis–Menten equation (Molto et al. 2014): $H = \frac{\alpha \cdot Dbh}{\beta \cdot Dbh}$ [1] where we modify the setting for PFT 3 with α = 5.8 and β = 0.046.
- Initial setup: since we disabled natural regeneration and we required both a defined plant density and a initial situation with shade trees already providing canopy cover, we created inventory files to use as input. In particular, this approach allowed us to configure the two-PFT and three-PFT scenarios by selecting the proportion of PFT 7 trees relative to the total number of shade trees. We fixed the number of coffee plants at 1000 individuals per hectare. For each scenario type, we assumed two different canopy cover levels for shade trees by setting the mean total crown area per patch to two values: 33 m² and 66 m². This, in turn, determined the initial number of shade trees, enabling comparisons between different planting densities. We called the configurations respectively mca33_2PFTs, mca33_3PFTs, mca66_2PFTs, and mca66_3PFTs.

Number of repetitions and analysis

We simulated the different scenarios obtained from different initial setups, under each climate scenario with 20 replicates. We then observed the growth trends of the system across the different scenarios. In particular, we considered total biomass values to evaluate overall growth and carbon accumulation, while NPP (Net Primary Productivity) indicates the annual productivity rate of the system. To compare results across future climate scenarios, we analyzed biomass values from the final year of each simulation (year 30) for each configuration scenario. For NPP, we considered the average of the last ten years, since annual values are subject to fluctuations and a single-year measurement would be unrepresentative.

Shade-trees effect on temperature reduction

Although the FORMIND model does not explicitly calculate temperature at different canopy heights, it provides LAI values above the coffee plants. Using these outputs, we estimated the reduction in air temperature experienced by coffee trees due to shade from overstory trees by applying Equation 2 (Van Oijen et al. 2010):

$$T_C = T - \Delta T(1 - e^{-k\, LAI_t}) \qquad [2]$$

where T_C is air temperature amongst the coffee trees (°C), T is ambient temperature (°C), ΔT is the maximum reduction in daily average temperature because of shading (°C), k is the global radiation extinction coefficient of the trees (m^2 m^{-2}), and LAI_t is tree leaf area index (m^2 m^{-2}). We then analyzed the mean air temperature during the final year of simulation, accounting for the shade-tree effect. Following values reported for tropical forest canopies in FORMIND we set $k = 0.7$, and we set $\Delta T = 5$ °C (Beer et al. 1997).

The percentage of radiation reduced due to shading by trees was calculated using the Beer-Lambert law: $Irradiance\ reduction\ (\%) = \left(1 - e^{-k\ LAI_t}\right) \cdot 100$.

Results

The model was run for 30 years under the different scenarios. The 20 replicates for each scenario resulted in limited variability overall; examples of this are shown for total biomass and NPP within the agroforestry system (e.g., simulation mca66_3PFTs_RCP2.6a, Figure 1). Biomass trends show that, over time, tree growth leads to increased shading by the shade trees, demonstrating that the model accurately simulates this aspect (Figure 1).

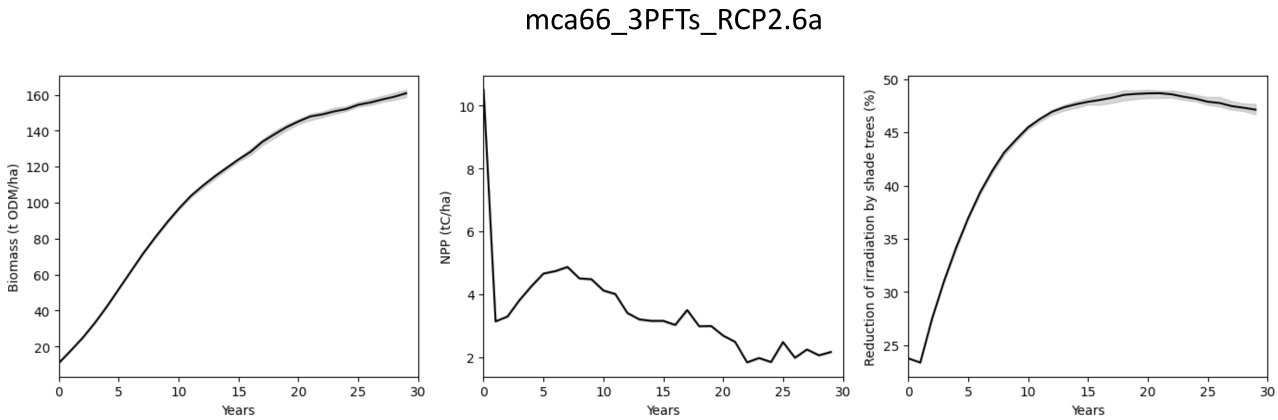


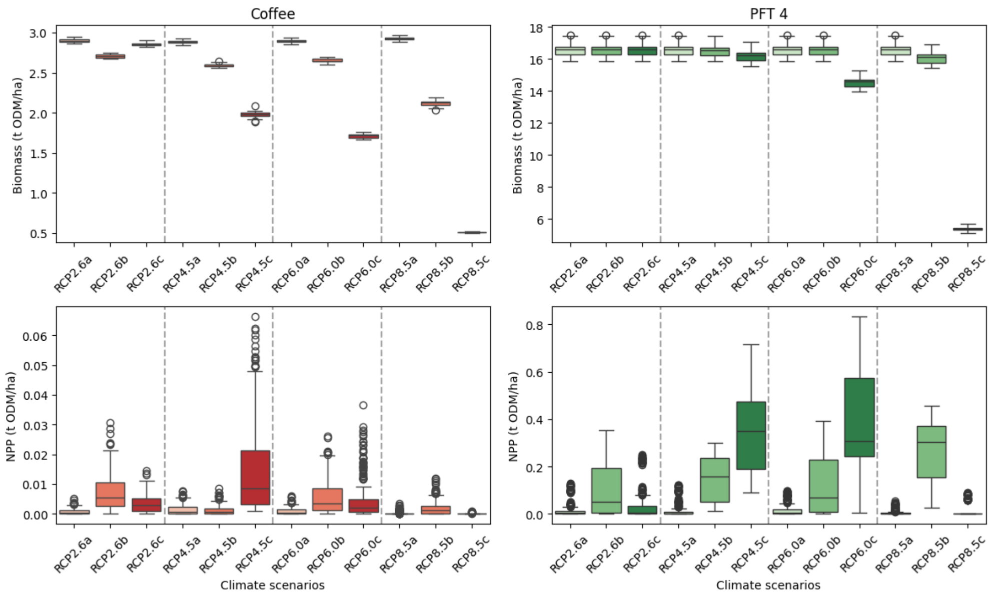
Figure 1: Trends of biomass (left), NPP (in the middle) and irradiation reduction by shade trees (right) in the agroforestry system with 20 repetitions (grey lines) and the mean (black line), under the configuration mca66_3PFTs and climate scenario RCP2.6a.

The distribution of total biomass per hectare for each PFT aligns with temperature patterns across the climate trajectories. In RCP2.6, a slight temperature increase is observed during the second 30-year period, followed by a return to values similar to current conditions in the final period. In the other four RCP scenarios, progressive warming leads to increased thermal stress and corresponding decreases in biomass. Biomass at year 30 clearly reflects these temperature trends, with particularly critical conditions under RCP8.5c (Figure 2). Coffee biomass at year 30 declined progressively from RCP2.6 to RCP8.5, with median reductions ranging from approximately 76% to 85% under RCP8.5c relative to the corresponding baseline scenario (RCP8.5a). The magnitude of the reduction varied among configurations, but the direction of the response was consistent across all simulations. Fluctuations in NPP are reflected in the spread of the boxplots (Figure 2), showing the distribution of values over the last 10 years for each climate scenario and configuration. Shade trees, particularly large forest species, are the main contributors to carbon assimilation in the system. NPP patterns correspond to temperature distributions for PFT 7 in three-PFT configurations, while coffee and PFT 4 exhibit no clear trend and higher variability, suggesting lower resilience to climatic fluctuations. Coffee biomass is higher in configurations with fewer shade trees (mca33_XPFTs) compared to higher shade coverage (mca66_XPFTs), reflecting light competition rather than temperature-related benefits. Shading in all scenarios remains below 60%, which is within the optimal range for coffee growth.

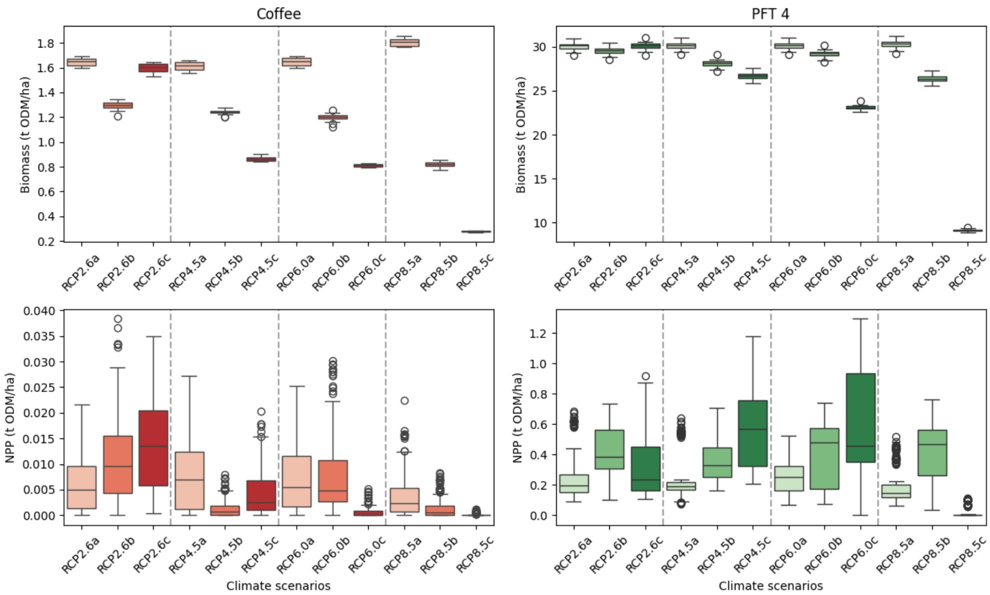
Biomass values for the two-PFT configurations are comparable to those reported in agroforestry systems (Tesfay et al. 2022). With the addition of PFT 7, biomass values approach those observed in secondary tropical forests (Gonçalves et al. 2017). However, the model still presents limitations. For example, coffee biomass decreases with increasing shade, a counterintuitive result that likely reflects the model's current treatment of light competition, which does not fully account for the potential benefits of temperature reduction and microclimatic buffering provided by shade trees.

Temperature variations among the different climate scenarios, ranging from 0 to approximately +6.3°C, have a stronger impact on the outcomes than precipitation changes, which are limited to reductions of 0–10%. For coffee cultivation, precipitation remains largely within the optimal range.

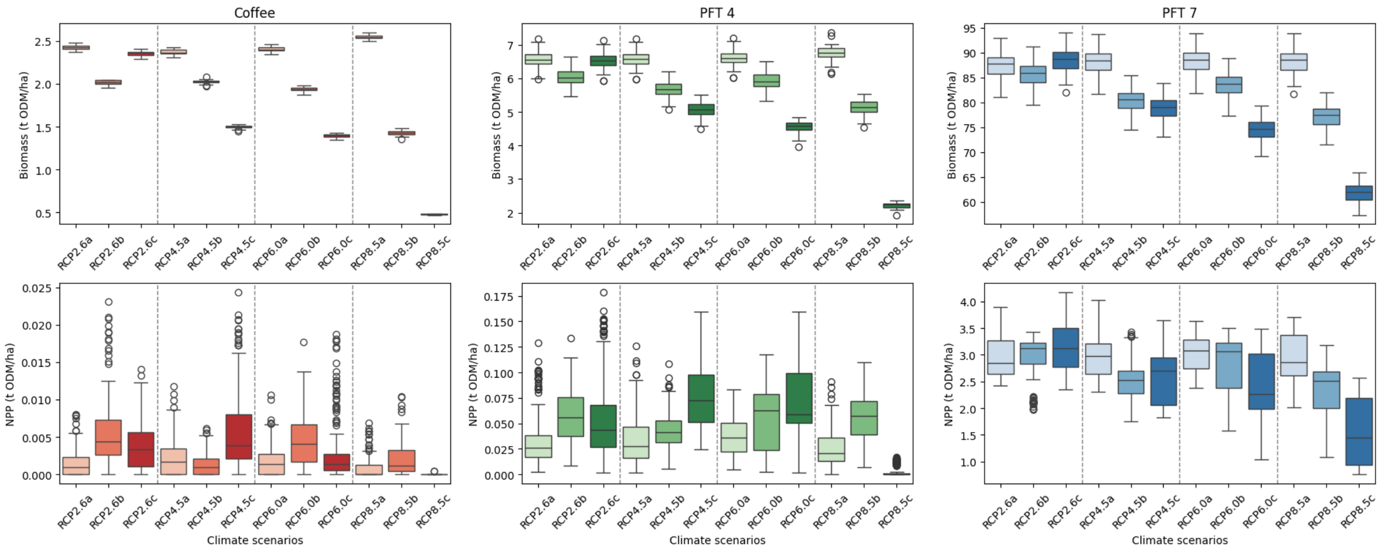
mca33_2PFTs



mca66_2PFTs



mca33_3PFTs



mca66_3PFTs

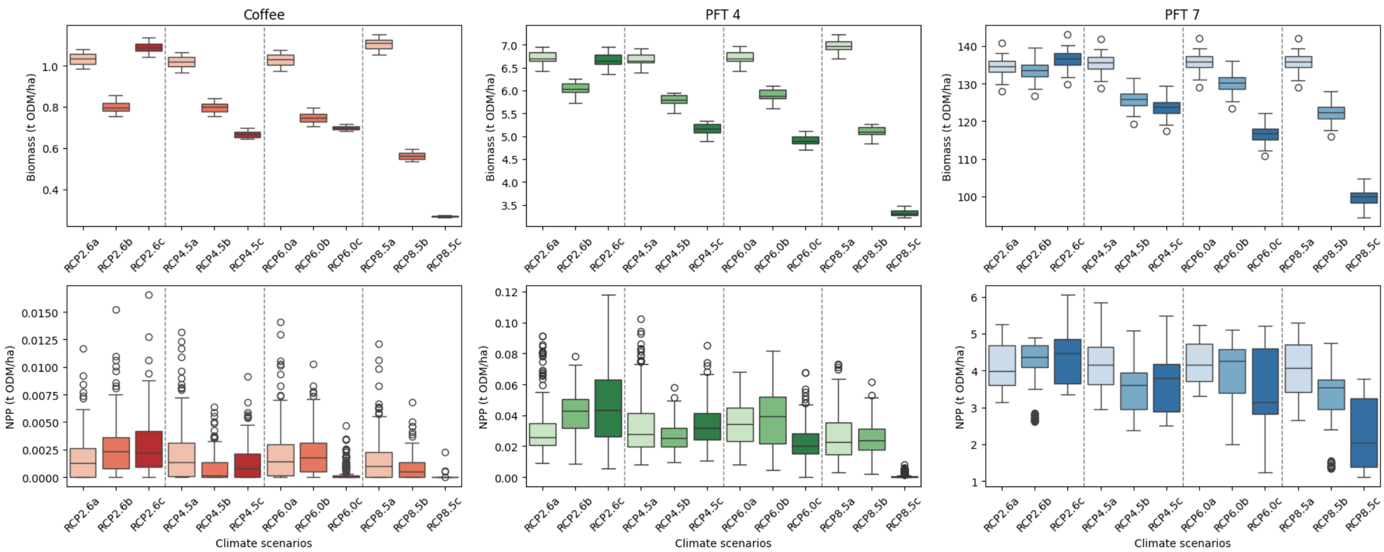


Figure 2: For each PFT (Coffee = red, PFT4 = green, PFT7 = blue), boxplots show the distribution of total biomass values in the final year of the simulation and NPP values over the last 10 years across different configurations and climate scenarios.

Temperature reductions due to shading, calculated using Equation 2, range between 0.5 and 2°C. These results indicate that, under the climatic conditions used and the set configurations, coffee cultivation can be highly critical, as T_C — the temperature among the coffee trees — remains above 24°C in future scenarios. Among the different options, the recommended configuration in this context is mca66_3PFTs, which represents a more complex agroforestry system with higher canopy cover.

Table 1: values of mean di T_C (air temperature among coffee trees, °C) for the different configurations and climate scenarios at year 30 of the simulations.

	RCP2.6a	RCP2.6b	RCP2.6c	RCP4.5a	RCP4.5b	RCP4.5c	RCP6.0a	RCP6.0b	RCP6.0c	RCP8.5a	RCP8.5b	RCP8.5c
mca33_2PFTs	25.2	25.3	25.3	25.4	26.2	25.9	24.8	25.8	26.7	25.4	26.7	28.0
mca66_2PFTs	24.5	24.8	24.8	24.8	25.7	25.4	24.3	25.3	26.2	24.8	26.3	27.7
mca33_3PFTs	24.9	25.0	25-0	25.0	25.9	25.5	24.5	25.4	26.3	25.1	26.4	27.6
mca66_3PFTs	24.1	24.2	24.2	24.3	25.2	24.8	23.7	24.7	25.5	24.3	25.7	26.9

Discussion

This study represents an initial attempt to apply the FORMIND model to agroforestry systems, but the results highlight that further model development is needed. Despite increasing empirical studies on shaded coffee systems, long-term quantitative projections integrating climate change, tree–crop interactions, and management remain scarce, highlighting the need for process-based modeling approaches. Coffee biomass values are consistent with literature reports, considering the varying planting densities across different study sites (Tesfay et al. 2022). However, the model is not yet fully realistic and requires further refinement. In the climate scenarios considered, temperatures exceed 24°C, which is typically above the optimal range for coffee cultivation in the absence of shade-tree protection (Lin 2007).

The FORMIND model currently includes light competition as a key process. Consequently, as canopy cover from shade trees increases, the growth rate and resulting coffee biomass decrease. An important future development for simulating agroforestry systems is therefore to incorporate not only competitive mechanisms but also facilitative interactions. Specifically, at elevated temperatures, increased shading within the optimal range should reduce temperature stress for coffee plants, potentially enhancing productivity, as suggested by empirical studies (Niether et al. 2018; Piato et al. 2020). The first step towards this is implementing temperature estimation for each canopy layer as a function of LAI and modeling tree growth accordingly, allowing the system to capture both competitive and beneficial interactions between shade trees and crops. With this future implementation, the model will also be able to represent potential trade-offs caused by shade trees, balancing the benefits of shading (e.g., temperature reduction) with competition for light, water, and space. Although the model is not yet able to simulate the benefits of shading, from a multi-ecosystem service perspective it clearly demonstrates the potential of agroforestry for carbon sequestration consistent with several studies (Lugo-Pérez et al. 2023; Vallejos-Torres et al. 2024).

This limitation currently makes it difficult to directly compare our results with empirical studies of coffee plantations (e.g., Andrade et al. 2021; Asanok et al 2024; Patil et al. 2025b). Once facilitative effects and temperature interactions are implemented, such comparisons will become crucial for validating model predictions and evaluating trade-offs and multi-ecosystem service supply between productivity, microclimate regulation, and carbon sequestration.

The current model approximates pruning by setting a maximum coffee height, which ensures realistic plant dimensions but does not capture biomass fluctuations caused by pruning or management. Similarly, fruit development and harvest were not modeled, preventing the direct assessment of coffee yield. Including these processes in future FORMIND simulations would allow for a more comprehensive analysis of both ecological and agronomic outcomes.

Further improvements could be achieved by compiling a database of dendrometric measurements to refine model parameterization and by using site-specific climatic data, bringing the simulations closer to reality. For these reasons, we did not simulate a coffee monoculture scenario, which would likely fail under the temperatures considered, highlighting the importance of agroforestry practices.

Comparing the idea of using FORMIND with the use of other models applied in agroforestry reveals several potential strengths and interesting aspects, as well as areas that require further development and implementation. For example, APSIM (Agricultural Production Systems Simulator) stands out as a widely used modular framework, capable of simulating tree–crop interactions and predicting crop productivity under different management regimes and climate scenarios (Holzworth DP et al. 2014). However, tree representation in APSIM remains simplified (e.g., proxy trees or limited tree modules) and does not capture individual tree dynamics, multi-layer canopy structure, or long-term carbon allocation. In contrast, FORMIND is an individual-based, process-oriented model that explicitly simulates tree establishment, growth, mortality, and competition, allowing multiple PFTs to interact within the same stand. This PFT-based approach enables

FORMIND to capture structural complexity, canopy stratification, shading effects, and NPP dynamics over time, providing unique insights into ecological and carbon dynamics in complex agroforestry systems, features that remain largely unexplored by crop-oriented models. Nevertheless, limitations remain, such as the absence of modules for fruit production and harvest, and for facilitative interactions, which would be important to assess agronomic productivity under climate change.

Simulations across the different climate scenarios confirm that future conditions may be highly challenging, imposing significant stress on both vegetation and crops. The RCP2.6 scenario, which represents a relatively stable climate, is unfortunately considered unlikely without rapid and drastic global interventions. Indeed, the results for mean temperatures and shading-induced reductions (Table 1) in the final year indicate that, under this parameterization, the effect of shade trees would not be sufficient to sustain coffee cultivation in areas where temperatures currently exceed 25°C. This highlights the importance of being able to anticipate the feasibility of future crops in order to avoid investing in cultivations that may become unsuitable over time, and emphasizes that models represent a valuable approach to develop for this purpose.

Conclusion

This study demonstrates the potential of using the FORMIND model to simulate coffee agroforestry systems under future climate scenarios. Our results show that coffee biomass and NPP are highly sensitive to canopy structure and shading, with higher canopy cover (mca66_3PFTs) providing better resilience under elevated temperatures. Monoculture coffee systems are unlikely to be viable under projected climate conditions, highlighting the importance of diverse agroforestry configurations.

Although the model currently emphasizes light competition, incorporating facilitative effects of shade and improving parameterization with species-specific dendrometric data would enhance its predictive capacity. Such improvements could allow a more accurate assessment of carbon balance and better representation of agroforestry dynamics.

Overall, forest models like FORMIND are valuable tools for guiding climate-adaptive agricultural practices, informing management strategies, and supporting sustainable agroforestry design in tropical regions.

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Afforestation priority for multiple objectives at national scale: Italy as a case study

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CG, LMWR, SM, MCP, LP, FS, LS, GV conceived and designed the research; CG developed the methodology with contributions from MM (number tropical nights), SG (fragmentation map), AC (water regulation), LMWR (climate mitigation); CG created the maps; CG, LMWR, AC, GV wrote and edited the manuscript; GV supervised the research.

Abstract

Afforestation is increasingly recognized as a key strategy to address climate change and ecological degradation, offering multiple ecosystem services. However, strategic planning is needed to ensure that afforestation actions are ecologically effective and economically efficient by targeting areas where ecosystem service provision is most significant.

In this study we provide a workflow to prioritize afforestation areas at national scale tailored to four distinct ecosystem service goals: ecological connectivity, human health, climate mitigation, and water regulation.

Using Italy as case study, we applied goal-specific analysis to areas considered suitable for afforestation, integrating geospatial datasets of environmental and social factors. For each objective, spatial indicators were combined into a priority map at 1 km resolution, subsequently merged into a composite map. The top 20% priority areas for each objective were identified to inform national and regional planning.

Approximately 55% of Italy's land was deemed suitable for afforestation, including urban and agricultural zones, through opportunities for urban forestry and agroforestry systems. High-priority areas to improve ecological connectivity were focused around urban centers and extensive agricultural plains. Water regulation benefits were most relevant in urbanized and vineyard landscapes, while urban areas also emerged as priorities for human health. Climate mitigation potential was highest in moist temperate and mountain regions. Overlap among goals was limited, revealing significant trade-offs across ecosystem services.

The workflow provides a replicable approach to identify goal-specific afforestation priorities across diverse landscapes. It supports coordinated strategies to enhance ecosystem service provision and restoration effectiveness at national and regional levels.

Goal-specific afforestation planning can maximize ecological benefits, cost-effectiveness, and policy impact. By identifying priority areas for different ecosystem services, the workflow helps practitioners target interventions where they yield the greatest benefits. Urban and agricultural areas emerge as important opportunities for multifunctional interventions, including urban forestry and agroforestry. The limited overlap among goals underscores the need to explicitly balance trade-offs when designing multi-objective restoration programs. The workflow is readily adaptable to other European countries, thanks to the availability of large-scale spatial datasets, offering a practical tool to guide strategic forest landscape restoration.

Introduction

Forests provide numerous ecosystem services (ESs) supporting human well-being and environmental sustainability. Alongside more traditional provisioning ESs, such as wood and non-timber forest products (Clason et al. 2008; Baskent et al. 2020) and cultural services (Paracchini et al. 2014), increasing attention has been given recently to regulating services such as carbon sequestration, improvement of air quality, temperature reduction, water regulation and conservation of biodiversity (Thompson et al. 2011; Decocq et al. 2016; Liang et al. 2016; Mori et al. 2017). Afforestation can play a crucial role contributing to such goals. In recent years, afforestation has sparked the interest of public and private sectors worldwide, both promoting ambitious tree planting initiatives (e.g., IUCN 2016) with the aim of restoring biodiversity and improving the provision of multiple ESs (Brockerhoff et al. 2013; Chazdon et al. 2016; Waring et al. 2020). These objectives are in line with European policies, among which the EU Biodiversity Strategy (2020), EU Forest Strategy (2021), and EU Climate Law (2021). For instance, the EU Biodiversity Strategy includes a commitment to plant at least 3 billion trees in the EU by 2030.

Forest ecosystem service provision is influenced not only by species choice, plantation design, and management, but also by site-specific environmental conditions (Baskent et al. 2020; Pérez-Silos et al. 2021). Several studies highlighted how plantations without an appropriate strategy can lead to unfavorable impacts (Farley et al. 2005; Elmarsdottir et al. 2008; Doelman et al. 2020). Afforestation without considering site-specific characteristics or the needs and priorities of a certain area, can be ineffectual or even counterproductive for ESs. For example, the massive afforestation carried out on the Chinese Loess Plateau, ignoring the soil and climate characteristics, caused water table decline and soil erosion (Jiang et al. 2016). Furthermore, afforestation without considering soil carbon saturation or the previous land use, could lead to soil carbon losses instead of gains (Czimczik et al. 2004). A further example concerns places where afforestation implies losses of biodiversity, such as in traditionally managed open grasslands, where transition to forested areas can lead to a reduction in species richness and correlated provision of ESs (Prangel et al. 2023). For these reasons and given the limited funding, efficient planning requires identifying areas to be afforested to maximize benefits, avoid unintended consequences, and optimize costs. To identify priority areas, the first step should lay in defining one or more goals for afforestation; an effective approach then involves the use of analysis that considers both environmental and socio-economic factors (Sun et al. 2022; Fahrudin et al. 2024; Güngör & Şen 2024) to maximize the pursuit of such goals.

Several recent studies have proposed national or large-scale assessments of afforestation potential, focusing either on single objectives or on multiple ecosystem services (Bradfer-Lawrence et al. 2021; Baggio-Compagnucci et al. 2022; Burke et al. 2023; Fahrudin et al. 2024), and emphasizing the need for spatially explicit evaluations to support strategic planning. In this context, our study advances this field by integrating four distinct afforestation goals within a reproducible methodology based on openly available spatial data, enabling scalable and transferable applications to other European countries.

The main objective of this research is to prioritize afforestation areas based on four specific ESs (ecological connectivity, human health, climate mitigation, and water regulation), aligned with global and national sustainable development goals, thereby offering a more comprehensive and targeted approach to afforestation planning. Our findings allow us to generate goal-specific maps of afforestation priorities at the national scale, as well as synthesis maps to provide an integrated overview of afforestation priorities considering all the goals together. The specific goals we consider herein are:

a. Ecological connectivity
Ecological networks are a strategic tool for biodiversity conservation (Jongman 1995). The importance of ecological connectivity and urban challenges highlight the need for large-scale planning to define a targeted planting plan, assessing where it is most strategic to invest efforts and resources (Kremer et al. 2016; Di Pirro et al. 2021).

b. Human health
The compound effects of climate warming and Urban Heat Island effect (Fini et al. 2017), pose a significant threat to human health, especially for urban vulnerable populations (Heaviside et al. 2017). Air pollution also represents a significant threat to public health, as WHO data (2022) indicate that 99% of the global population breathes air with pollutant concentrations exceeding WHO recommended limits. Urban forests can improve air quality by removing atmospheric pollutants, and reduce air temperatures through shading and evapotranspiration (Brack 2002; Nowak et al. 2006; Yan et al. 2018).

c. Climate mitigation
Forest restoration, thanks to carbon sequestration, is one of the most effective strategies for climate change mitigation (Griscom et al. 2017; Bastin et al. 2019). Carbon sink potential depends on tree species and age, climate, soil type, water and nutrient limitations (Requena Suarez et al. 2019). Both the biomass of new trees and the accumulation of carbon in the soil, which is much more long-lived that the aboveground carbon sink, contribute to reduce atmospheric greenhouse gas concentrations.

d. Water regulation
Afforestation is a key countermeasure adopted where the land cover change exerted a strong alteration on catchment hydrology (Blöchl et al. 2007; Pattison & Lane 2012). Planting trees helps reduce runoff, intercepting rainfall, alter transpiration rates, increasing the rainfall infiltration and saturated soil hydraulic conductivity, and prolonging the local water storage over time (Buytaert et al. 2007; Silveira et al. 2016; Murphy et al. 2021). With these perspectives, the

current challenge is to identify location of afforestation to maximize the mitigation of negative consequences of land use change and climate change.

We selected Italy as a representative and complex case study, given its unique combination of geographical, socio-economic, and environmental characteristics.

Case study

Italy's territory (Figure1), spanning approximately 300,000 km^2, is 36.7% covered by forests (Gasparini et al. 2022). Similar to other European countries, Italy exhibits a contrasting spatial pattern: forest expansion in mountainous and marginal zones due to the abandonment of traditional agricultural and pastoral activities (Sallustio et al. 2015; Malandra et al. 2018), alongside increasing urbanization of lowlands and coastal zones (Romano & Zullo, 2014).

Today, urban areas in Italy cover 7% of the national territory -among the highest in the EU- and continue to expand despite the demographic decline observed in the country (ISPRA 2016). This increasing urbanization poses a serious threat to biodiversity (McKinney 2006; Mcdonald et al. 2008), which in Italy is exceptionally rich thanks to its high climatic, orographic, and geological variability (Falcucci et al. 2007). The network of protected areas covers 22% of the country (ISPRA 2024), including over 2,600 sites.

Urbanization also has significant implications for public health. Although air pollutant concentrations have declined in recent years, they still exceed the WHO recommended limits particularly in urbanized areas (SNPA 2024a). Climate change is marked by a mean temperature increase of 0.4°C per decade since 1981 (SNPA 2024b) and a decline in average precipitation since 1961-1990, alongside more frequent extreme rainfall and prolonged dry spells (Brunetti et al. 2004; Iannuccilli et al. 2021) with significant socio-economic and environmental impacts (Maxwell et al. 2019; Mateos et al. 2023).

In agreement with European strategies, Italy planned large-scale actions targeting its 14 Metropolitan Cities to plant 6.6 million trees by 2026 (National Recovery and Resilience Plan, PNRR, 2021), aiming to combat air pollution, climate change, and biodiversity loss.

These dynamics make Italy an ideal setting to test an afforestation planning approach that integrates multiple objectives and considers specific territorial features. Di Pirro et al. (2022) explored afforestation priorities in Italy based on the analysis of some of their expected benefits, i.e., air pollution, thermal stress, and hydraulic vulnerability, but their analysis should be expanded to a broader suite of forests' contributions.

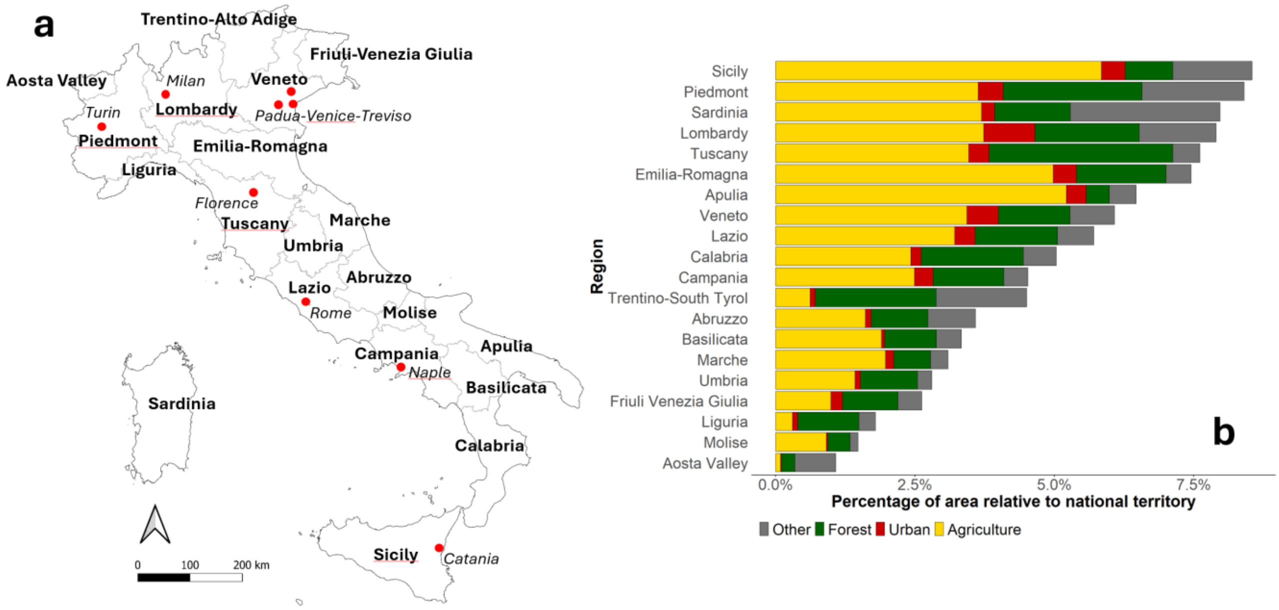

Figure 1 a) Regions of Italy and some cities mentioned in the text (using MAP01); b) The bar chart details each region's percentage of surface area relative to the national total. Colors indicate land use categories derived from the Corine Land Cover dataset (MAP02): urban areas (artificial surfaces), agricultural areas, forests, and other land uses.

Methods

The analysis and spatial data processing were executed using Geographic Information System (GIS) and Google Earth Engine programming. This study was conducted at a national scale with a spatial resolution of 1 km. All input layers were resampled, when necessary, to a common 1 km² grid using bilinear interpolation for continuous variables and mode assignment for categorical ones. Each resulting map, called Priority Map (PMAP), presents afforestation priority as a score from 0 to 1.

First, we selected suitable areas for afforestation. To do this, we applied a spatial prioritization analysis specific to each of the four afforestation target goals, based on the integration of multiple indicators relevant to each objective (Figure2). Each factor is spatially explicit, with data available at the national scale (Table1).

To identify potentially suitable areas for afforestation, we excluded the following categories: (i) existing forests (from Corine Land Cover 2018, CLC, MAP02 in Table1); (ii) zones above 1600 m a.s.l. (from Digital Elevation Model, DEM, MAP03), in order to exclude areas beyond the upper tree line or with unfavorable climatic conditions; (iii) wetlands (MAP02) and protected areas (MAP04), that require site-specific consideration; (iv) land uses (MAP02) unsuitable for forest growth, either natural conformation (water bodies, beaches and dunes, rocky surfaces, heaths, glaciers and perennial snow) or anthropogenic (airports, port areas, road and railway networks, and technical infrastructures); (v) areas with potential non-forest natural vegetation (MAP05). Agricultural areas were not excluded because they could be restored though the creation of hedgerows and the adoption of alternative agroforestry systems as afforestation actions. Urbanized areas were also considered as potentially suitable, as they may contain green spaces that are afforestable but could be undetected at the given resolution, or areas that could be converted due to land-use changes, and are therefore crucial for achieving the study objectives.

The values resulting from the four composite indicators were all rescaled to a range [0–1]. In the resulting PMAPs, afforestation priority is assigned based on score percentiles. This approach supports territorial decision-making in cases where a specific afforestation target (in terms of area) must be met. As an illustrative metric, we also analyzed the spatial distribution of High-Priority Areas (HPAs)—defined here as those exceeding the 80th percentile of the suitability distribution—for each afforestation goal.

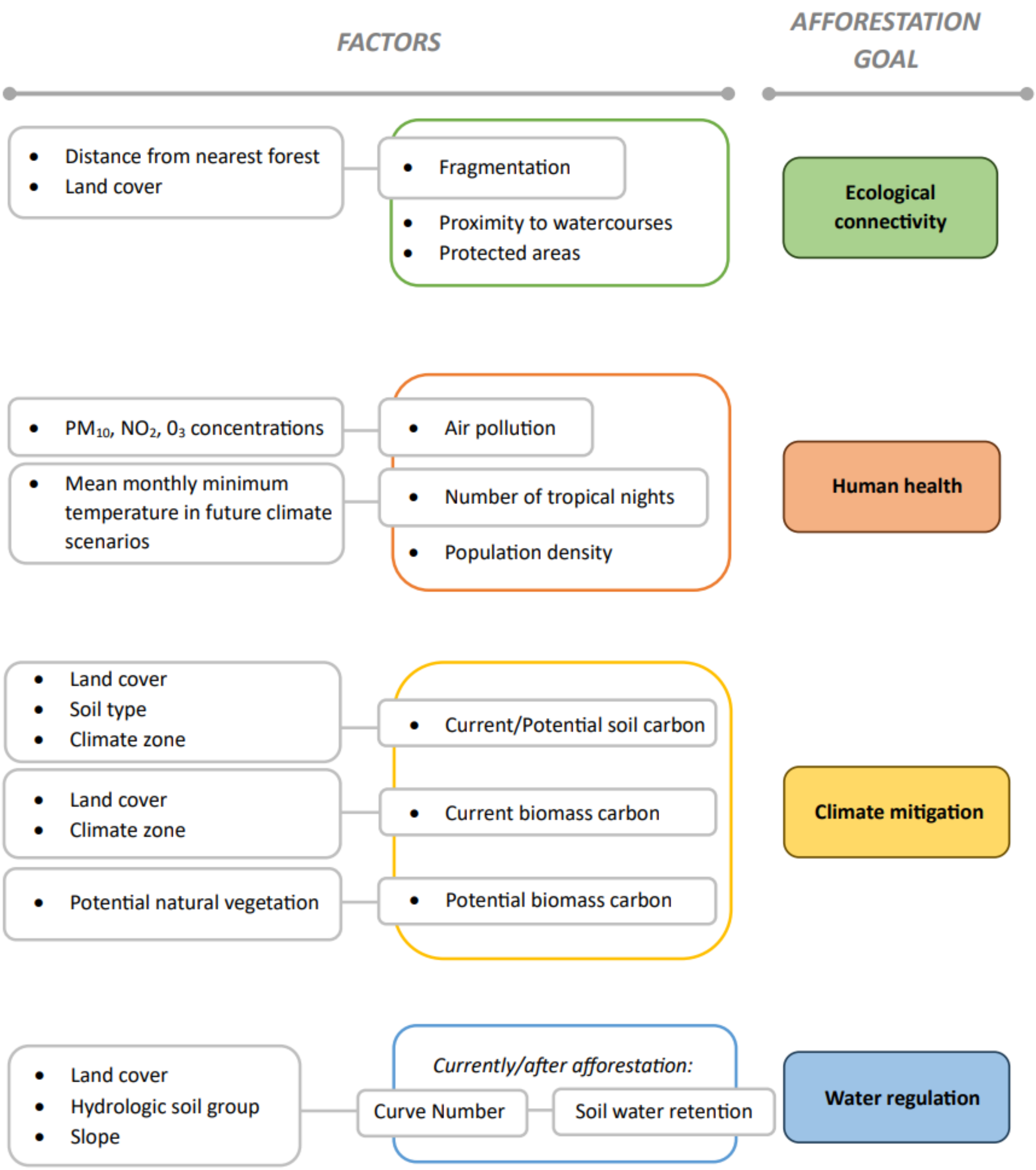

Figure 2 Overview of the factors considered in each goal-specific methodology.

a. Ecological connectivity

Under this goal, we prioritized areas where afforestation can establish or improve ecological corridors or provide stepping stones in highly fragmented zones, thereby strengthening the national ecological network of forest areas. Specifically, we focused on three indicators, analyzed through dedicated maps:

- Fragmentation (F): we assessed the ecological connectivity at each 1 km pixel based on a Least Cost Path (LCP) algorithm (Adriaensen et al. 2003), i.e., considering both the shortest distance from existing forests and the degree of ecological resistance ("cost") to animal movement played by the land uses (MAP02) occurring along the path. We assigned the minimum cost per pixel to forest areas and the maximum cost value to highly anthropized areas such as urban fabric, infrastructures, and intensive agriculture (TableS1). The subsequent step involved identifying the least-cost path connecting each pixel with the nearest forest land (MAP02). Pixels with higher cumulative LCP values correspond to areas with lower forest connectivity and higher priority for afforestation. The result was mapped in MAP12.
- Protected areas (PA): we assigned higher priority to sites that have the potential to improve connectivity between protected forest patches. To do so, for each pixel we calculated the sum of the squared Euclidean distances from the centroids of the forested patches in the two nearest protected areas (intersection of MAP02 and MAP04). Since smaller sums imply stronger corridor potential, we multiplied all values by -1 so that larger (i.e., less negative) values would denote higher priority.
- Proximity to watercourses (PW): riparian buffers of variable width (Fischer & Fischenich 2000) act as significant corridors for movement and dispersal (Machtans et al. 1996; Burbrink et al. 1998; Santos et al. 2011). To assign higher priority to ecosystems along watercourse (MAP06), we assigned to all pixels within 1 km buffer from rivers a value of 1, and 0 to all other pixels.

We rescaled all three indices to a range [0, 1] and calculated the afforestation priority P by applying Equation 1.

$$P = F\left(1 + PA + \frac{PW}{2}\right) \quad [1]$$

where F serves as primary driver, scaling the combined influence of the other factors, prioritizing afforestation in heavily anthropized areas and penalizing areas close to existing forests, where natural afforestation could occur. PW was weighted at 50% to balance its binary scale against the continuous nature of PA. The result was mapped in PMAP1.

b. Human health

In this case the methodology prioritizes areas where a large number of people are exposed to higher health risks, with a focus on temperature and air quality:

- Population density (d): number of inhabitants per km^2 (MAP07).
- Air quality: we considered the concentrations of PM_{10}, NO_2, and O_3, three air pollutants harmful to human health that trees can partially absorb (Yang et al. 2005). Pollutant concentrations were assessed using the maps from Horálek et al. (2021) (M8), which provide the annual mean concentration of PM_{10} and NO_2 ($a1$, and $a2$, respectively), and the 93.2nd

percentile of maximum daily 8-hour mean concentration of O_3, ($a3$) which is associated with a higher mortality rate compared to the 24-hour average concentration (Yang et al. 2012).

- Temperature: we considered the number of tropical nights (NTN) per year in future climate scenarios as indicator of health risk (Yavaşlı & Erlat 2024). Since daily data were not available at national scale as raster maps, we estimated NTN using a nonlinear equation. This approach is largely used in literature for many climatological parameters, such as the Growing degree days above 5°C (Wang et al. 2011; Wang et al. 2016). For every pixel, we calculated NTN as a function of monthly minimum temperature using Equation 2:

$$NTN = \sum_{i=1}^{12} \left(\frac{nd_i}{1+1.382\times10^6 \cdot e^{-0.7162\, Tmin_i}} \right) \qquad [2]$$

  where nd_i is the number of days in i-th month; $Tmin_i$ is the mean monthly minimum temperature for i-th month. This equation was calibrated using daily temperature time series from 5395 stations across Europe provided by ECA&D (Klein Tank et al. 2002) and then applied to minimum temperature expected for 2041-2070. Future temperature values were averaged from those predicted by five global climate models (GFDL-ESM4, IPSL-CM6A-LR, MPI-ESMAP1-2-HR, MRI-ESM2-0, UKESMAP1-0-LL) under CMIP6 scenario SSP-370, using datasets provided by CHELSA (MAP09). The result was mapped in MAP13.

All variables were rescaled to a range [0, 1]. Priority P was calculated by assigning equal weight to temperature and air quality, and multiplying the result by d by applying Equation 3.

$$P = d\left(NTN + \frac{a1+a2+a3}{3}\right) \qquad [3]$$

This is consistent with a risk assessment framework (risk = hazard × vulnerability). The result was mapped in PMAP2.

c. Climate mitigation

We assessed potential carbon stock increases after afforestation using IPCC methodologies (2006), relying on Tier 2 and Tier 3 data to account for the heterogeneity of soil properties and drivers.

First, we estimated the carbon stock in non-forest soils under their current land uses, by assigning a value for soil (SC_0) and biomass (BC_0) carbon from the IPCC database (IPCC 2019) to each unique combination (TableS2) of land use (MAP02), climate zone, and soil type (MAP10). Climate zones map (MAP14) at 1 km resolution was created based on the IPCC classification (IPCC 2019) using climatic data from the CHELSA dataset (MAP09). We associated each land use category in the CLC dataset with the most similar one in the IPCC database.

Second, we calculated potential soil (SC_p) and biomass (BC_p) carbon stock attained post-afforestation. We derived SC_p (TableS2) as above, assuming "Forest" as land use. We assigned BC_p (TableS3) corresponding to the natural potential vegetation type (MAP05), using the mean above- and below-ground values reported by the National Forest Inventory (Gasparini et al. 2022). Crucially, the use of mean values ensures that forests at all stages of development are captured, not just fully mature stands. Because of this, our results are not linked to a specific time horizon; rather, they represent the carbon pool expected when a forest has reached the current average conditions of Italian forests.

We calculated carbon gain or losses (ΔC) following afforestation subtracting from the potential carbon stock after afforestation the current carbon stock as follows:

$$\Delta C = (SC_p + BC_p) - (SC_0 + BC_0) \quad [4]$$

Negative values, indicating potential carbon losses from afforestation, were set to 0. The results were mapped in PMAP3

d. Water regulation

To detect the priority areas where tree planting can improve the water storage, we applied one of the most popular hydrological methods, the Curve Number (CN) methodology (Ponce & Hawkins 1996). This hydrological parameter serves as a fundamental empirical approach to estimate the surface runoff, considering both the soil properties and the land cover characteristics. CN is widely used, therefore, for the determining the spatial distribution of runoff generation (Mockus 1964). The CN values range from 0 to 100.

We compiled different spatial datasets, including CLC (MAP02), hydrologic soil group (HSG, MAP11), and the DEM (MAP03). First, we assigned an initial CN value (CN_{raw}) combining land cover category and HSG (TableS4, Cislaghi et al. 2024). Second, we employed a slope-adjusted formulations of CN_{raw} to include the terrain steepness variability (Wu et al. 2024) as follows:

$$CN_{adj} = CN_{raw} \cdot \frac{322.79 + 15.63 \cdot \alpha}{\alpha + 323.52} \quad [5]$$

Where α is the slope (in mm^{-1}), considered valid within the range [0.14-1.4]. Values outside this range were assigned the corresponding boundary values. CN_{adj} was mapped in MAP15.

Then, we calculated the current potential soil water retention S_0 (Mishra and Singh, 2003; Mishra et al, 2014) using the standard formula:

$$S_0 = \frac{25400}{CN_{adj}} - 254 \quad [6]$$

where S_0 is expressed in mm. Afforestation impact was calculated by assuming that all the land cover converts to a healthy forest, with CN in function of the HSG (TableS4). We repeated the procedure (Equations 5 and 6) providing the future potential soil water retention S_p. Then, we computed the difference ΔS between S_p and S_0 representing the hydrological improvements (more water yield and less surface runoff). The final gridded map of ΔS was normalized and the results generate PM4.

e. Combining priority maps

We overlaid the four individual Priority Maps (PMAP1, PMAP2, PMAP3, and PMAP4) to generate two synthetic maps, representing priority areas from a multifunctional perspective. Since the score distributions for the four afforestation goals are markedly different—due to the specific methodologies applied for each objective—the use of absolute mean scores would be misleading. Therefore, we considered the average of the corresponding percentile classes (PMAP5). In the second map (PMAP6), we visualized the number of goals (0 to 4) under which each pixel is classified as a HPA. This highlights areas where afforestation can simultaneously maximize benefits for multiple ESs.

Table 1 Summary and references of the maps and datasets used in the analyses (the last access to all links is 21 July 2025)

Map Code	Name	Source
MAP01	Administrative boundaries of Italia and its regions	ISTAT (2025) https://www.istat.it/notizia/confini-delle-unita-amministrative-a-fini-statistici-al-1-gennaio-2018-2/
MAP02	CORINE Land Cover 2018 (CLC)	European Union's Copernicus Land Monitoring Service information. CORINE Land Cover 2018; Vector: https://doi.org/10.2909/71c95a07-e296-44fc-b22b-415f42acfdf0; Raster: https://doi.org/10.2909/960998c1-1870-4e82-8051-6485205ebbac
MAP03	DEM TINITALY	Tarquini S, Isola I, Favalli M, Battistini A, Dotta G (2023) TINITALY, a C of Italy with a 10 meters cell size (Version 1.1). Istituto Nazionale di Geofisica e Vulcanologia (INGV). https://doi.org/10.13127/tinitaly/1.1
MAP04	Protected Areas - WDPA	UNEP-WCMC and IUCN (2025) Protected Planet: The World Database on Protected Areas (WDPA). UNEP-WCMC and IUCN, Cambridge, UK. www.protectedplanet.net.
MAP05	Potential Natural Vegetation Map of Italy	Blasi C, Capotorti G, Alós Ortí MM, Anzellotti I, Attorre F, Azzella MM, Carli E, Copiz R, Garfì V, Manes F, Marando F, Marchetti M, Mollo B, Zavattero L (2017) Ecosystem mapping for the implementation of the European Biodiversity Strategy at the national level: The case of Italy. Environmental Science & Policy, 78:173-184.
MAP06	National Hydrographic Network - Dataset	ISPRA (2010) Reticolo Idrografico Nazionale - Dataset. https://geodati.gov.it/geoportale/visualizzazione-metadati/scheda-metadati/?uuid=ispra_rm:01Idro250N_DT
MAP07	Population density	CIESIN (Center for International Earth Science Information Network) - Columbia University (2018) Gridded Population of the World, Version 4 (GPWv4): Population Density, Revision 11 (Version 4.11). NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H4F47M65
MAP08	Air quality	Horálek J, Schreiberová M, Vlasáková L, Marková J, Kurfürst P, Schovánková J, Schneider P, & Tognet F (2021) ETC/ATNI Report 2020/10: European air quality maps for 2018. PM10, PM2.5, Ozone, NO2 and NOx Spatial estimates and their uncertainties. Zenodo. https://doi.org/10.5281/zenodo.4638651
MAP09	Climate dataset - CHELSA V.2.1	Karger DN, Conrad O, Böhner J, Kawohl T, Kreft H, Soria-Auza RW, Zimmermann NE, Linder P, Kessler M (2017) Climatologies at high resolution for the Earth land surface areas. Scientific Data 4: 170122. https://chelsa-climate.org/
MAP10	IPCC Soil Classes	FAO & IIASA (2023) Harmonized World Soil Database Version 2.0. Rome and Laxenburg. https://earthmap.org/?aoi=it&boundary=level0&feature&layers=%7B%22HWSD2_IPCC_Soils%22%3A%7B%22opacity%22%3A1%7D%7D&map=%7B%22center%22%3A%7B%22lat%22%3A41.553311480972795%2C%22lng%22%3A12.56417627922847%7D%2C%22zoom%22%3A5%2C%22mapType%22%3A%22roadmap%22%7D&statisticsOpen=true
MAP11	Hydrologic Soil Groups	Ross CW, Prihodko L, Anchang JY, Kumar SS, Ji W, and Hanan NP (2018) Global Hydrologic Soil Groups (HYSOGs250m) for Curve Number-Based Runoff Modeling. ORNL DAAC, Oak Ridge, Tennessee, USA. https://doi.org/10.3334/ORNLDAAC/1566

Results

About 55% of the Italian territory has been found to be potentially suitable for afforestation (Figure3a). A large part of the area (84%) is currently occupied by agricultural land, and a very small proportion is covered by urbanized zones (9%). Sicily exhibits the largest relative suitable area for afforestation, accounting for 6% of the national territory, followed by Emilia-Romagna and Sardinia, each with 5% (Figure3b).

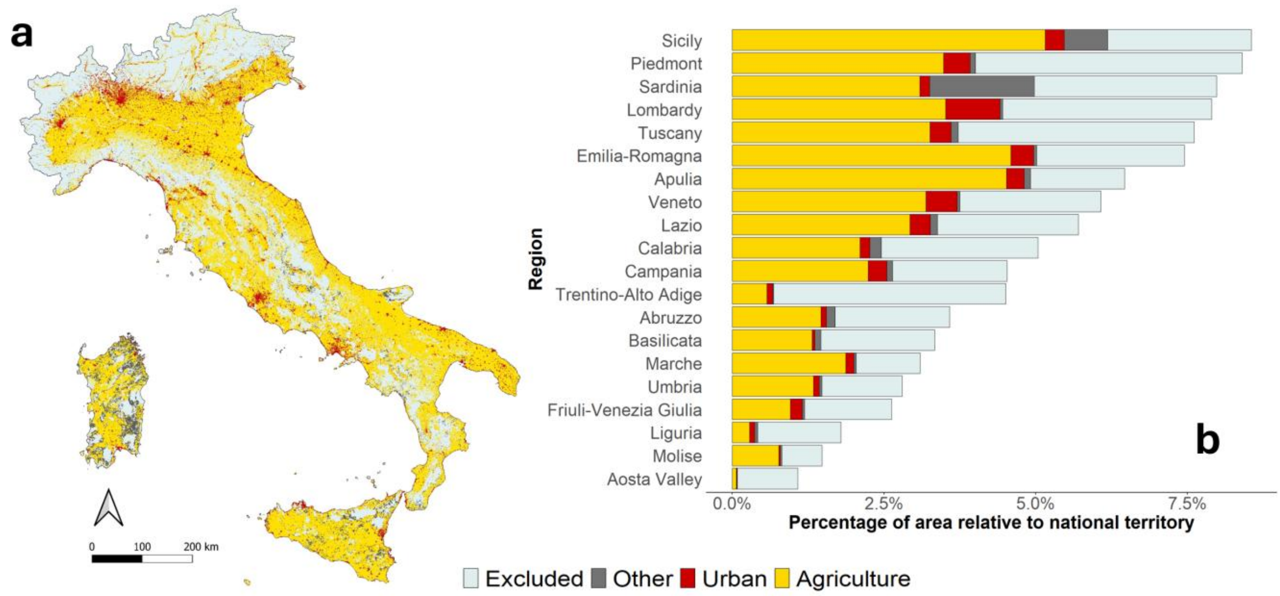

Figure 3 a) The map visualizes the areas identified as suitable for afforestation. b) The bar chart details each region's percentage of surface area relative to the national total. Colors indicate land use categories derived from the CLC dataset (MAP02): urban areas (artificial surfaces), agricultural areas, other land use, and areas excluded from afforestation

The F coefficient (MAP12, FigureS1) ranges from 0 to 64,198, although more than 88% of the assessed areas fall below a value of 10,000. Its distribution mirrors in the score indicating afforestation priority (Figure4b), which shows a pronounced skewness with values heavily concentrated near zero. Since F is the main driver in the model, MAP12 closely resembles the priority afforestation map (PMAP1-Figure4a). However, the inclusion of the two additional factors, PA and PW, introduces local-scale variations (e.g., Figure4c).

Priorities for afforestation aimed at enhancing ecological connectivity (Figure4a) are mainly concentrated in the major lowlands: the Po Valley (extending from Piedmont to Veneto), and the Apulian plains (with 20% and 18% of the HPAs, respectively, TableS5). Other notable concentrations are found along the Adriatic coast (west coast) and in peri-urban areas surrounding the metropolitan cities of Turin, Rome, Naples, and Catania. Specifically, HPAs correspond to areas with a score above 0.09 (Figure4b). Sardinia, generally characterized by lower F, exhibits the lowest priority for such interventions (48% of suitable areas has score 0).

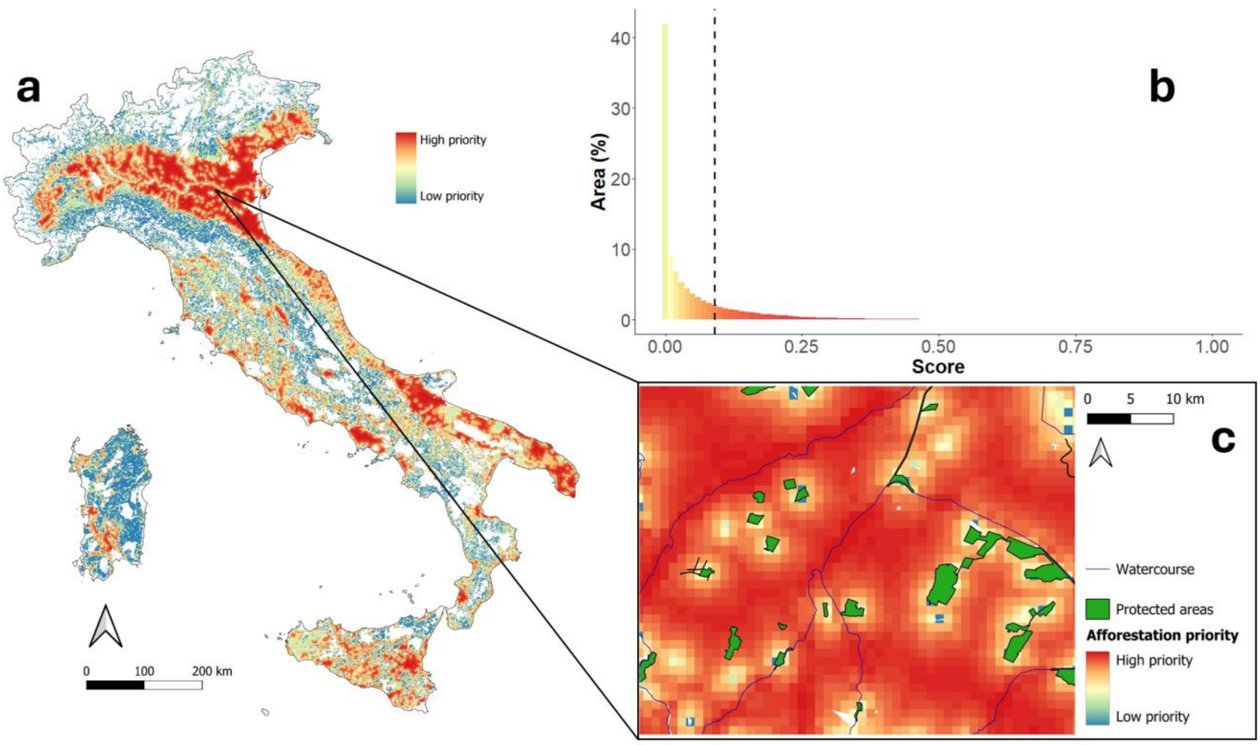

Figure 4 a) The map PMAP1 illustrates the spatial distribution of afforestation priority, specifically for the ecological connectivity goal. b) Histogram showing the distribution of afforestation scores across the areas considered; the black dashed line indicates the 80th percentile threshold. c) Zoomed-in view illustrating an example how afforestation priority varies within a highly fragmented landscape that includes rivers and protected areas.

Population density shows a wide but skewed distribution toward zero, with vast sparsely populated areas and a few urban centers—such as Rome, Naples, Milan, and Turin—with densities that exceed 30,000 inhabitants per km². This pattern strongly influences the final score: HPAs include areas with a score above 0.03 (Figure5b). The PMAP aimed at human health (PMAP2, Figure5a) show a significant overlap with the most urbanized zones, notably the Po Valley, characterized by both high levels of atmospheric pollution and high population density. Areas bordering the main industrial centers as Milan, Venice-Treviso-Padua, and Turin stand out with high priority (e.g., Figure5c). Similarly, high priority is found in correspondence with major cities such as Rome, Naples, and Florence. The NTN projected for the future (MAP13, FigureS2) raises priorities along densely populated coastal zones, notably the Adriatic coast. Sparsely populated areas are characterized by very low priority, as only a small portion of the population would benefit from the public health-related ESs resulting from new forests. This is evident in the regional distribution (TableS5), with Lombardy and Veneto showing large shares of HPAs (18% and 15% of HPAs, respectively), while Sicily and Sardinia, with vast low-population density zones, have large shares of low priority zones.

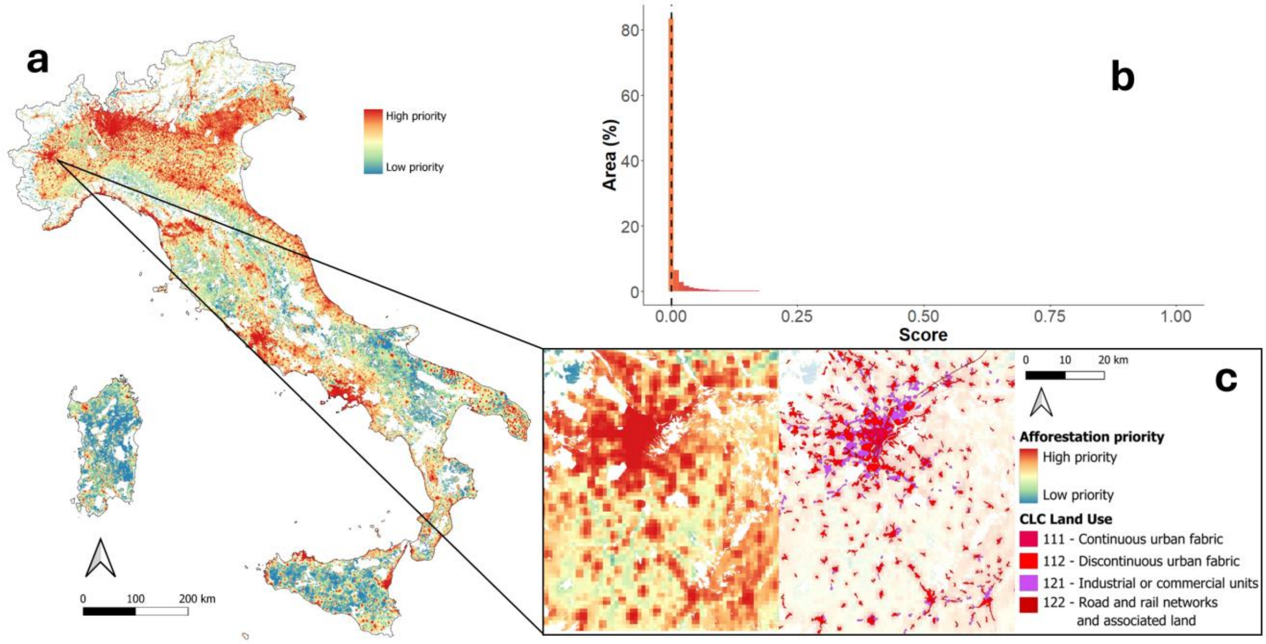

Figure 5 a) The map illustrates the spatial distribution of afforestation priority, specifically for the human health goal. b) Histogram showing the distribution of afforestation scores across the areas considered; the black dashed line indicates the 80th percentile threshold. c) Zoomed-in view illustrating an example how afforestation priority is strongly associated with urban land use.

The climate zone map (MAP14, FigureS3) developed for this study provided a suitable spatial resolution for the analysis. Carbon-related goals reveal a stark dichotomy between lowland areas, characterized by low priority, and mountainous zones, which instead exhibit high priority (PMAP3, Figure6a). This spatial distribution exhibits a strong correlation with climate. Those areas with a significant ΔC are located in climatic zones classified by the IPCC as Warm and Cool Temperate Moist (e.g., Figure6c). Soil type in Italy does not vary significantly, as 95% of the national territory is classified as High Activity Clay Soils; therefore, this variable has a limited influence on the overall results. Notably, zero carbon gains or losses were observed in rice paddies (e.g. Figure6c), which currently store large quantities of soil carbon thanks to the temporary waterlogging. Some orchard areas also exhibited negative ΔC. This outcome is explained by the fact that, based on the available data, the existing biomass in these areas exceeds that of the potential natural vegetation (TableS3). While the highest prioritization scores are concentrated along the Alpine and Apennine mountains, Sicily stands out as the region encompassing the largest extent of HPAs (more than 18% of the national total). In this case, HPAs are the areas with a score above 0.28 (Figure6b).

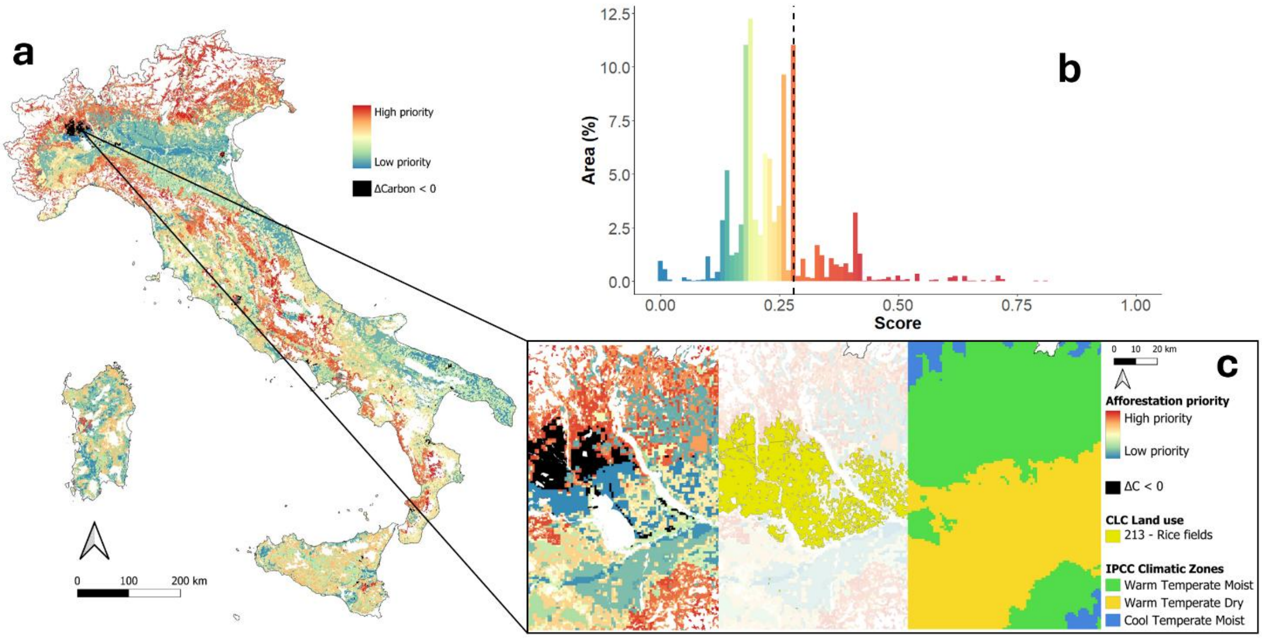

Figure 6 a) The map illustrates the spatial distribution of afforestation priority, specifically for the climate mitigation goal. b) Histogram showing the distribution of afforestation scores across the areas considered; the black dashed line indicates the 80th percentile threshold. c) Zoom-view showing both lack of priority in rice field areas, and the correlation between HPAs spatial distribution and climatic zones.

The highest priorities for afforestation aimed at water regulation (Figure7a) are concentrated in vineyard areas as well as in heavily urbanized areas (e.g., Figure7c), particularly around Milan, Florence, Rome, and Naples. In these areas, ΔS values generally range between 35 and 65 mm. The regions that would benefit most from afforestation targeting the water regulation are those lying in the Po Valley, such as Lombardy and Veneto (16% and 14% of HPAs, respectively). HPAs for water regulation are areas with a score above 0.27. Low priority was found in mountainous areas, where steeper slopes translates to modest potential gains in soil water infiltration.

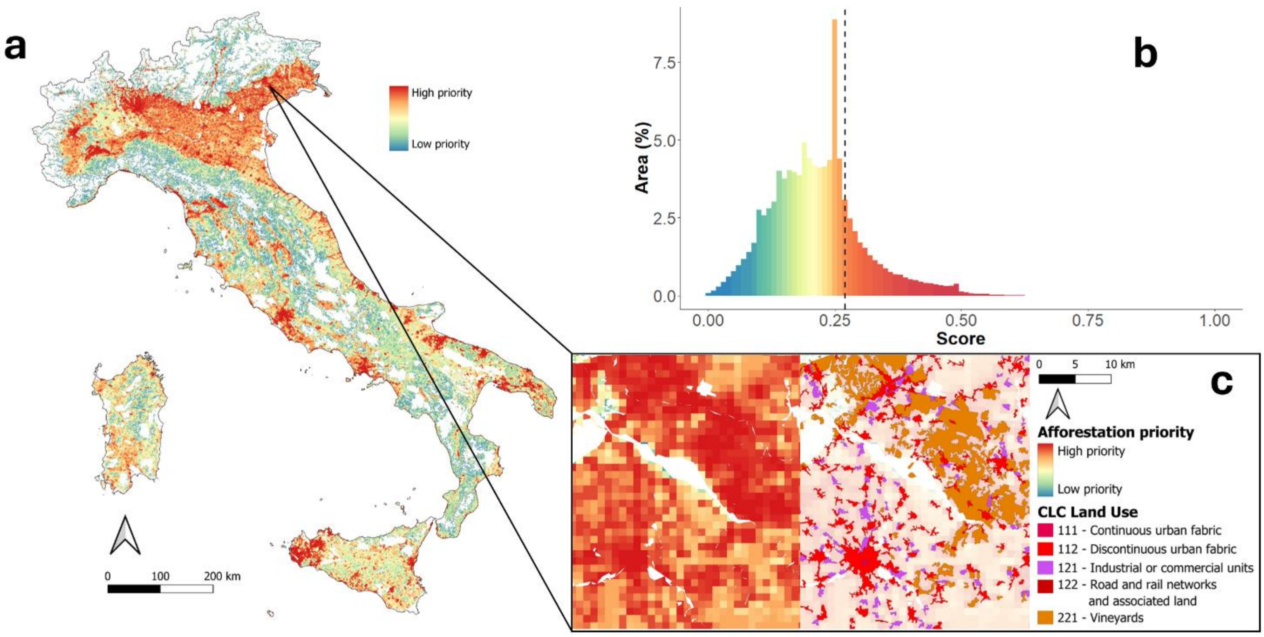

Figure 7 a) The map illustrates the spatial distribution of afforestation priority, specifically for the water regulation goal. b) Histogram shows the distribution of afforestation scores across the areas considered; the black dashed line indicates the 80th percentile threshold. c) Zoomed-in view highlighting the relevance of vineyards and urban areas in the spatial distribution of afforestation priority.

From the analysis of the composite maps, we observed that higher average priority was concentrated at densely urbanized areas (PMAP5, Figure8a). This is further confirmed in PMAP6 (Figure8b): only 13% of the total suitable area exhibits high priority for at least two of the objectives considered here, and this happened predominantly around the most urbanized contexts. This also highlighted the existence of significant trade-offs when trying to establish priority locations for afforestation aimed at different goals.

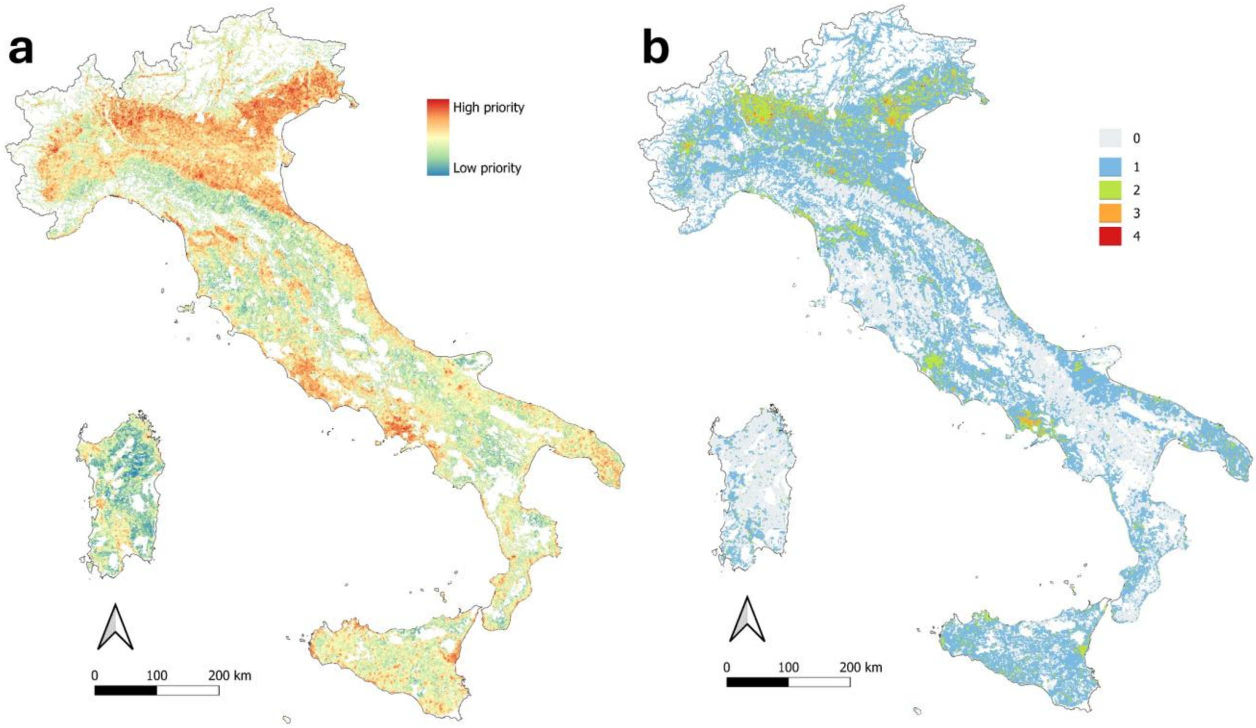

Figure 8 a) The displays the average afforestation priority, calculated by averaging the percentile ranks of the scores obtained for each of the four afforestation goals, and highlights areas with higher overall priority. b) The map indicates the number of objectives (out of four) for which each point is considered HPA. Both maps reveal a predominant concentration of higher priority areas within densely urbanized contexts.

Discussion

The flexible approach to each goal proved effective in identifying priority areas for Ess improvement, using diverse inputs and assessment criteria. Its strength lies in accounting for diverse factors and assessment criteria. Notably, the identification of priority areas and maximum benefits followed different logics: for ecological connectivity and human health, priorities corresponded to areas with the most severe issues; conversely, for water regulation and climate mitigation, priorities aligned with areas showing the greatest projected improvement under future forest cover.

It is crucial to consider that the areas identified as suitable for afforestation are not entirely convertible to forest. In urban areas, for instance, usable patches may be small, below the analysis resolution, yet significant for achieving specific ESs objectives. The prevalence of agricultural areas (84%) highlights the need to adopt agroforestry approaches or careful land-use planning that integrates increased tree cover with sustainable agricultural practices. This study provides a national-level overview, offering a basis for more detailed analyses at the local scale, focused on actually available areas.

High values of F observed in lowland and coastal zones, driven by intensive agriculture and extensive urbanization, makes them priority zones for interventions aimed at enhancing ecological connectivity. This aligns with global trends where urban expansion leads to significant habitat loss and fragmentation, negatively impacting biodiversity by reducing species richness, altering community composition, and disrupting ecological processes (McKinney 2006; McDonald et al. 2008). Recognizing the crucial role of urban environments for biodiversity conservation (Dearborn & Kark 2010), investing resources in these areas is essential. Specifically, urban forests can exhibit high biological richness (Alvey 2006) and serve as ecological corridors or stepping-stones (Lapin et al. 2024). Similarly, the conversion from conventional agriculture to agroforestry or the introduction of ecological infrastructure (e.g., hedgerows, riparian buffers) promotes an increase in biodiversity, by creating diverse microhabitats (Santos et al. 2022). Careful planning can maximize connectivity between protected areas, a crucial strategy for effective population conservation (Coetzee et al. 2014). However, in planning afforestation interventions, it is vital to safeguard specific priority habitats, such as open grasslands and clearings. In Italy, these habitats are particularly vulnerable due to forest expansion and the abandonment of traditional land management practices (Pallotta et al. 2022). Therefore, a balanced approach is essential to avoid inadvertently reducing biodiversity by converting valuable open habitats.

As suggested by the resulting map and given the high percentage of the Italian population residing in urban areas, urban forests play a crucial role for public health and quality of life in the cities. This focus is globally pertinent, as the share of the world's population living in urban areas is projected to rise from 55% in 2019 to 68% by 2050 (UN 2019). The urban areas identified as suitable in this study are not entirely afforestable, and the actual available surface will be smaller. It is therefore essential to rethink and redesign green infrastructures to maximize their ecological functionality and social accessibility within the existing urban fabric. Urban areas present both constraints, such as historic or densely built neighborhoods, and opportunities, like former industrial sites, which can be repurposed for urban greening (Rink & Schmidt 2021). Including urban areas in our methodology departs from many approaches and underscores the need for context-specific priorities balancing ecological, social, and cultural considerations (Resemini et al. 2025).

The results for climate mitigation differ from others, suggesting that afforestation strategies specifically oriented towards climate mitigation may require a distinct spatial approach. ΔC was higher in the Warm and Cool Temperate Moist regions, where wet climatic conditions and higher precipitation enhance soil carbon storage (Ogle et al. 2005) through decreased decomposition rates (Schuur et al. 2001), increased mineral-associated organic matter (Heckman et al. 2023) and hydroclimatic influence (Huang et al. 2023). This underscores water availability as a key driver for forests in the Mediterranean area. Under comparable conditions of soil and land use, climate will play a decisive role in shaping additional carbon storage. Also urban areas present a high potential carbon storage (Teo et al. 2021), although our methodology might not be refined enough to properly capture it. Conversely, results showed a low or negative ΔC in rice paddies, consistent with existing literature (Schwarz et al. 2024).

Regarding water infiltration potential, low values of the S index, associated with high values of CN (TableS4, FigureS4), are found in urbanized areas, characterized by a high percentage of impervious surfaces. The conversion of impervious surfaces into forests leads to the most significant increases in water infiltration into the soil. This practice aligns with the "sponge city" concept and falls within

Nature-Based Solutions (NBS) for sustainable urban drainage, representing a strategic approach to mitigate frequent flooding and water management issues in urban areas. Among agricultural areas, cultivating crops in straight rows without interrow grass cover causes a significant runoff generation and low capacity of water storage. Most vineyard show these features, exacerbated by frequent management practices that cause soil compaction. In these areas, agroforestry techniques can be implemented to improve hydrological functions, through enhanced soil water retention and microclimate regulation (Favor & Udawatta 2021). This is particularly important under climate change, highlighting agroforestry as a multifunctional ES provider, also supporting agricultural productivity.

Consistent with earlier findings that highlight the inherent trade-offs among ecosystem services (Chisholm 2010; Doelman et al. 2019), the limited spatial overlap of HPAs across multiple goals suggests that afforestation projects should focus on one or two key objectives rather than aiming to maximize all benefits simultaneously. The concentration of multifunctional priorities in densely populated areas underscores the strategic value of urban and forests in enhancing environmental quality, biodiversity, and well-being. This reinforces the importance of integrating multifunctional green infrastructure into urban planning.

The PMAPs can be used in several ways: i) observing the maps individually, to align projects with specific ESs goals by targeting priority areas at NS; ii) to evaluate whether pre-selected sites, due to land tenure or availability, are relevant for any ESs and tailor interventions accordingly; iii) to visualize overlapping HPAs and plan for multifunctionality.

Given Italy's ambitious goal of planting 6.6 million trees by 2026, under the PNRR, our study provides a spatially explicit information framework at national scale, identifying priority areas for reforestation based on multiple ESs objectives. The resulting maps can therefore represent a fundamental decision-support tool to direct planting efforts towards areas where the investment will generate the greatest environmental and social benefits. The spatial differentiation of priorities highlighted by our analysis underscores the need for a strategic and targeted approach to achieve the national goal, optimizing resources and maximizing afforestation impact.

This study could be further expanded by incorporating additional ecosystem services as explicit targets, such as provisioning or cultural services (Burke et al. 2023), and by developing specific methodologies for each. Moreover, future work could integrate species-selection models to evaluate how different tree species or planting designs may influence ecosystem service provision, thereby supporting even more realistic and operational planning (Baggio-Compagnucci et al. 2022; Lautenbach et al. 2017).

The 1 km resolution is appropriate for strategic national planning and can also be relevant at macro-regional or regional scales . However, finer-scale local analyses remain advisable for operational planning, stakeholder engagement, and integration into local development policies. This enables translating national strategies into actionable, context-specific interventions.

An important aspect to consider is the temporal evolution of territorial conditions (Łągiewska & Bartold 2025) and, consequently, the need to regularly update the results in line with newly released datasets. For example, the outcomes of this study could be updated when the new CLC 2024 dataset becomes available (Copernicus n.d.). Another promising direction concerns the emerging availability of do-it-yourself tools and automated platforms able to update land cover information and detect changes over user-defined time ranges (Bartold et al. 2025). Such systems, which rely on automated satellite-image processing, provide an opportunity to complement traditional maps. Incorporating these approaches could enhance the transferability of our methodology, enabling more frequent updates and supporting dynamic applications and territorial management.

The visualization of priority in percentiles significantly aids administrative and policy decisions by providing a relative ranking of suitability. This approach allows to set pragmatic targets (e.g., top 10% or 20%) aligned with available funding and facilitating more efficient resource allocation. Furthermore, the distribution analysis of HPAs in the regions (TableS5) is instrumental for national planning. It provides a strategic overview, enabling authorities to understand the spatial distribution of diverse afforestation opportunities across the country. This macroscopic perspective is crucial for designing a comprehensive, nation-wide afforestation program that effectively distributes efforts and maximizes benefits by tailoring interventions to regional characteristics and needs. National scale results should inform regional forest programs, which translate national strategies into regional priorities and guide both strategic and operational decisions at municipal level. Such multi-level integration enhances coherence between national goals and local actions, boosting the impact and feasibility of afforestation.

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Table S1 Degree of ecological resistance ("cost") to animal movement played by the land uses (classified with Corine Land Cover 2018)

CLC code	CLC description	COST
141	Green urban areas	1
243	Land principally occupied by agriculture, with significant areas of natural vegetation	1
244	Agro-forestry areas	1
311	Broad-leaved forest	1
312	Coniferous forest	1
313	Mixed forest	1
321	Natural grasslands	1
322	Moors and heathland	1
323	Sclerophyllous vegetation	1
324	Transitional woodland-shrub	1
411	Inland marshes	1
412	Peat bogs	1
522	Estuaries	1
112	Discontinuous urban fabric	2
333	Sparsely vegetated areas	2
334	Burnt areas	2
511	Water courses	2
512	Water bodies	2
121	Industrial or commercial units	3
131	Mineral extraction sites	3
132	Dump sites	3
111	Continuous urban fabric	5
122	Road and rail networks and associated land	5
123	Port areas	5
124	Airports	5
133	Construction sites	5
142	Sport and leisure facilities	5
211	Non-irrigated arable land	5
212	Permanently irrigated land	5
213	Rice fields	5
222	Fruit trees and berry plantations	5
223	Olive groves	5
241	Annual crops associated with permanent crops	5
242	Complex cultivation patterns	5
331	Beaches, dunes, sands	5
332	Bare rocks	5
335	Glaciers and perpetual snow	5
421	Salt marshes	5
422	Salines	5
423	Intertidal flats	5
521	Coastal lagoons	5
523	Sea and ocean	5

Table S2 Values for current soil (SC_0) and biomass (BC_0) carbon stock, and potential soil carbon stock post-afforestation (SC_p) to each unique combination of land u (Corine Land Cover 2018), IPCC climate zone, and IPCC soil class.

SPODIC SOIL

	Cool temperate Moist			Warm temperate Moist		
CLC Land Use category	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})
Mineral extraction sites	42.2	1.0	128.0	47.2	1.0	143.0
Dump sites	42.2	1.0	128.0	47.2	1.0	143.0
Construcion sites	42.2	0.0	128.0	47.2	0.0	143.0
Green urban areas	105.0	20.6	128.0	117.3	20.6	143.0
Sport and leisure facilities	105.0	13.1	128.0	117.3	13.1	143.0
Non-irrigated arable land	89.6	4.7	128.0	98.7	4.7	143.0
Permanently irrigated land	89.6	4.7	128.0	98.7	4.7	143.0
Rice fields	172.8	4.7	128.0	193.1	4.7	143.0
Vineyards	92.2	2.8	128.0	103.0	2.8	143.0
Fruit trees and berry plantations	92.2	63.0	128.0	103.0	63.0	143.0
Olive groves	92.2	6.9	128.0	103.0	6.9	143.0
Pastures	128.0	6.4	128.0	143.0	6.3	143.0
Annual crops associated with permanent crops	128.0	6.4	128.0	143.0	6.3	143.0
Complex cultivation patterns	92.2	13.1	128.0	103.0	13.1	143.0
Land principally occupied by agriculture, with significant areas of natural vegetation	105.0	13.7	128.0	117.3	13.7	143.0
Agro-forestry areas	92.2	20.6	128.0	103.0	20.6	143.0
Natural grassland	128.0	6.4	128.0	143.0	6.3	143.0
Moors and heathland	128.0	13.7	128.0	143.0	13.7	143.0
Sclerophyllous vegetation	128.0	13.7	128.0	143.0	13.7	143.0
Sparsely vegetated areas	105.0	1.0	128.0	117.3	1.0	143.0
Burnt areas **	108.8	BCₚ*0.26	128.0	121.6	BCₚ*0.26	143.0

VOLCANIC SOIL

	Cool temperate Moist			Warm temperate Dry			Warm temperate Moist			Tropical Dry		
CLC Land Use category	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})	**SC₀** (Mg* ha^{-1})	**BC₀** (Mg* ha^{-1})	**SCₚ** (Mg* ha^{-1})
Mineral extraction sites	44.9	1.0	136.0	27.7	1.0	84.0	45.5	1.0	138.0	16.5	1,0	50,0
Dump sites	44.9	1.0	136.0	27.7	1.0	84.0	45.5	1.0	138.0	16.5	1,0	50,0
Construcion sites	44.9	0.0	136.0	27.7	0.0	84.0	45.5	0.0	138.0	16.5	0,0	50,0
Green urban areas	111.5	20.6	136.0	78.1	20.6	84.0	113.2	20.6	138.0	46.5	20,9	50,0
Sport and leisure facilities	111.5	13.1	136.0	78.1	13.1	84.0	113.2	13.1	138.0	46.5	4,7	50,0

Non-irrigated arable land	95.2	4.7	136.0	63.8	4.7	84.0	95.2	4.7	138.0	46.0	4,7	50,0
Permanently irrigated land	95.2	4.7	136.0	63.8	4.7	84.0	95.2	4.7	138.0	46.0	4,7	50,0
Rice fields	183.6	4.7	136.0	113.4	4.7	84.0	186.3	4.7	138.0	67.5	4,7	50,0
Vineyards	97.9	2.8	136.0	60.5	2.8	84.0	99.4	2.8	138.0	50.5	2,8	50,0
Fruit trees and berry plantations	97.9	63.0	136.0	60.5	63.0	84.0	99.4	63.0	138.0	50.5	9,0	50,0
Olive groves	97.9	6.9	136.0	60.5	6.9	84.0	99.4	6.9	138.0	50.5	6,9	50,0
Pastures	136.0	6.4	136.0	84.0	2.9	84.0	138.0	6.3	138.0	50.0	4,1	50,0
Annual crops associated with permanent crops	136.0	6.4	136.0	84.0	2.9	84.0	138.0	6.3	138.0	50.0	4,1	50,0
Complex cultivation patterns	97.9	13.1	136.0	60.5	13.1	84.0	99.4	13.1	138.0	50.5	36,1	50,0
Land principally occupied by agriculture, with significant areas of natural vegetation	111.5	13.7	136.0	78.1	13.7	84.0	113.2	13.7	138.0	46.5	36,1	50,0
Agro-forestry areas	97.9	20.6	136.0	60.5	20.6	84.0	99.4	20.6	138.0	50.5	20,9	50,0
Natural grassland	136.0	6.4	136.0	84.0	2.9	84.0	138.0	6.3	138.0	50.0	4,1	50,0
Moors and heathland	136.0	13.7	136.0	84.0	13.7	84.0	138.0	13.7	138.0	50.0	36,1	50,0
Sclerophyllous vegetation	136.0	13.7	136.0	84.0	13.7	84.0	138.0	13.7	138.0	50.0	36,1	50,0
Sparsely vegetated areas	111.5	1.0	136.0	78.1	1.0	84.0	113.2	1.0	138.0	46.5	1,0	50,0
Burnt areas **	115.6	BC_p*0.26	136.0	71.4	BC_p*0.26	84.0	117.3	BC_p*0.26	138.0	42.5	BC_p*0.26	50,0
SANDY SOIL												
	Cool temperate Moist			**Warm temperate Dry**			**Warm temperate Moist**			**Tropical Dry**		
CLC Land Use category	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})
Mineral extraction sites	16.8	1.0	51.0	3.3	1.0	10.0	11.9	1.0	36.0	3.0	1,0	9,0
Dump sites	16.8	1.0	51.0	3.3	1.0	10.0	11.9	1.0	36.0	3.0	1,0	9,0
Construcion sites	16.8	0.0	51.0	3.3	0.0	10.0	11.9	0.0	36.0	3.0	0,0	9,0
Green urban areas	41.8	20.6	51.0	9.3	20.6	10.0	29.5	20.6	36.0	8.4	20,9	9,0
Sport and leisure facilities	41.8	13.1	51.0	9.3	13.1	10.0	29.5	13.1	36.0	8.4	4,7	9,0
Non-irrigated arable land	35.7	4.7	51.0	7.6	4.7	10.0	24.8	4.7	36.0	8.3	4,7	9,0
Permanently irrigated land	35.7	4.7	51.0	7.6	4.7	10.0	24.8	4.7	36.0	8.3	4,7	9,0
Rice fields	68.9	4.7	51.0	13.5	4.7	10.0	48.6	4.7	36.0	12.2	4,7	9,0
Vineyards	36.7	2.8	51.0	7.2	2.8	10.0	25.9	2.8	36.0	9.1	2,8	9,0
Fruit trees and berry plantations	36.7	63.0	51.0	7.2	63.0	10.0	25.9	63.0	36.0	9.1	9,0	9,0
Olive groves	36.7	6.9	51.0	7.2	6.9	10.0	25.9	6.9	36.0	9.1	6,9	9,0
Pastures	51.0	6.4	51.0	10.0	2.9	10.0	36.0	6.3	36.0	9.0	4,1	9,0
Annual crops associated with permanent crops	51.0	6.4	51.0	10.0	2.9	10.0	36.0	6.3	36.0	9.0	4,1	9,0
Complex cultivation patterns	36.7	13.1	51.0	7.2	13.1	10.0	25.9	13.1	36.0	9.1	36,1	9,0
Land principally occupied by agriculture, with significant areas of natural vegetation	41.8	13.7	51.0	9.3	13.7	10.0	29.5	13.7	36.0	8.4	36,1	9,0

Agro-forestry areas	36.7	20.6	51.0	7.2	20.6	10.0	25.9	20.6	36.0	9.1	20,9	9,0
Natural grassland	51.0	6.4	51.0	10.0	2.9	10.0	36.0	6.3	36.0	9.0	4,1	9,0
Moors and heathland	51.0	13.7	51.0	10.0	13.7	10.0	36.0	13.7	36.0	9.0	36,1	9,0
Sclerophyllous vegetation	51.0	13.7	51.0	10.0	13.7	10.0	36.0	13.7	36.0	9.0	36,1	9,0
Sparsely vegetated areas	41.8	1.0	51.0	9.3	1.0	10.0	29.5	1.0	36.0	8.4	1,0	9,0
Burnt areas **	43.4	BC_p*0.26	51.0	8.5	BC_p*0.26	10.0	30.6	BC_p*0.26	36.0	7.7	BC_p*0.26	9,0
LOW ACTIVITY CLAY SOIL (LAC)												
	Cool temperate Moist			**Warm temperate Dry**			**Warm temperate Moist**			**Tropical Dry**		
CLC Land Use category	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})
Mineral extraction sites	25.1	1.0	76.0	6.3	1.0	19.0	18.2	1.0	55.0	6.3	1,0	19,0
Dump sites	25.1	1.0	76.0	6.3	1.0	19.0	18.2	1.0	55.0	6.3	1,0	19,0
Construcion sites	25.1	0.0	76.0	6.3	0.0	19.0	18.2	0.0	55.0	6.3	0,0	19,0
Green urban areas	62.3	20.6	76.0	17.7	20.6	19.0	45.1	20.6	55.0	17.7	20,9	19,0
Sport and leisure facilities	62.3	13.1	76.0	17.7	13.1	19.0	45.1	13.1	55.0	17.7	4,7	19,0
Non-irrigated arable land	53.2	4.7	76.0	14.4	4.7	19.0	38.0	4.7	55.0	17.5	4,7	19,0
Permanently irrigated land	53.2	4.7	76.0	14.4	4.7	19.0	38.0	4.7	55.0	17.5	4,7	19,0
Rice fields	102.6	4.7	76.0	25.7	4.7	19.0	74.3	4.7	55.0	25.7	4,7	19,0
Vineyards	54.7	2.8	76.0	13.7	2.8	19.0	39.6	2.8	55.0	19.2	2,8	19,0
Fruit trees and berry plantations	54.7	63.0	76.0	13.7	63.0	19.0	39.6	63.0	55.0	19.2	9,0	19,0
Olive groves	54.7	6.9	76.0	13.7	6.9	19.0	39.6	6.9	55.0	19.2	6,9	19,0
Pastures	76.0	6.4	76.0	19.0	2.9	19.0	55.0	6.3	55.0	19.0	4,1	19,0
Annual crops associated with permanent crops	76.0	6.4	76.0	19.0	2.9	19.0	55.0	6.3	55.0	19.0	4,1	19,0
Complex cultivation patterns	54.7	13.1	76.0	13.7	13.1	19.0	39.6	13.1	55.0	19.2	36,1	19,0
Land principally occupied by agriculture, with significant areas of natural vegetation	62.3	13.7	76.0	17.7	13.7	19.0	45.1	13.7	55.0	17.7	36,1	19,0
Agro-forestry areas	54.7	20.6	76.0	13.7	20.6	19.0	39.6	20.6	55.0	19.2	20,9	19,0
Natural grassland	76.0	6.4	76.0	19.0	2.9	19.0	55.0	6.3	55.0	19.0	4,1	19,0
Moors and heathland	76.0	13.7	76.0	19.0	13.7	19.0	55.0	13.7	55.0	19.0	36,1	19,0
Sclerophyllous vegetation	76.0	13.7	76.0	19.0	13.7	19.0	55.0	13.7	55.0	19.0	36,1	19,0
Sparsely vegetated areas	62.3	1.0	76.0	17.7	1.0	19.0	45.1	1.0	55.0	17.7	1,0	19,0
Burnt areas **	64.6	BC_p*0.26	76.0	16.2	BC_p*0.26	19.0	46.8	BC_p*0.26	55.0	16.2	BC_p*0.26	19,0
LIGH ACTIVITY CLAY SOIL (HAC)												
	Cool temperate Moist			**Warm temperate Dry**			**Warm temperate Moist**			**Tropical Dry**		
CLC Land Use category	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})
Mineral extraction sites	26.7	1.0	81.0	7.9	1.0	24.0	21.1	1.0	64.0	6.9	1,0	21,0
Dump sites	26.7	1.0	81.0	7.9	1.0	24.0	21.1	1.0	64.0	6.9	1,0	21,0

Construcion sites	26.7	0.0	81.0	7.9	0.0	24.0	21.1	0.0	64.0	6.9	0,0	21,0
Green urban areas	66.4	20.6	81.0	22.3	20.6	24.0	52.5	20.6	64.0	19.5	20,9	21,0
Sport and leisure facilities	66.4	13.1	81.0	22.3	13.1	24.0	52.5	13.1	64.0	19.5	4,7	21,0
Non-irrigated arable land	56.7	4.7	81.0	18.2	4.7	24.0	44.2	4.7	64.0	19.3	4,7	21,0
Permanently irrigated land	56.7	4.7	81.0	18.2	4.7	24.0	44.2	4.7	64.0	19.3	4,7	21,0
Rice fields	109.4	4.7	81.0	32.4	4.7	24.0	86.4	4.7	64.0	28.4	4,7	21,0
Vineyards	58.3	2.8	81.0	17.3	2.8	24.0	46.1	2.8	64.0	21.2	2,8	21,0
Fruit trees and berry plantations	58.3	63.0	81.0	17.3	63.0	24.0	46.1	63.0	64.0	21.2	9,0	21,0
Olive groves	58.3	6.9	81.0	17.3	6.9	24.0	46.1	6.9	64.0	21.2	6,9	21,0
Pastures	81.0	6.4	81.0	24.0	2.9	24.0	64.0	6.3	64.0	21.0	4,1	21,0
Annual crops associated with permanent crops	81.0	6.4	81.0	24.0	2.9	24.0	64.0	6.3	64.0	21.0	4,1	21,0
Complex cultivation patterns	58.3	13.1	81.0	17.3	13.1	24.0	46.1	13.1	64.0	21.2	36,1	21,0
Land principally occupied by agriculture, with significant areas of natural vegetation	66.4	13.7	81.0	22.3	13.7	24.0	52.5	13.7	64.0	19.5	36,1	21,0
Agro-forestry areas	58.3	20.6	81.0	17.3	20.6	24.0	46.1	20.6	64.0	21.2	20,9	21,0
Natural grassland	81.0	6.4	81.0	24.0	2.9	24.0	64.0	6.3	64.0	21.0	4,1	21,0
Moors and heathland	81.0	13.7	81.0	24.0	13.7	24.0	64.0	13.7	64.0	21.0	36,1	21,0
Sclerophyllous vegetation	81.0	13.7	81.0	24.0	13.7	24.0	64.0	13.7	64.0	21.0	36,1	21,0
Sparsely vegetated areas	66.4	1.0	81.0	22.3	1.0	24.0	52.5	1.0	64.0	19.5	1,0	21,0
Burnt areas **	68.9	BC_p*0.26	81.0	20.4	BC_p*0.26	24.0	54.4	BC_p*0.26	64.0	17.9	BC_p*0.26	21,0

WETLAND SOIL (HAC)												
	Cool temperate Moist			**Warm temperate Dry**			**Warm temperate Moist**			**Tropical Dry**		
CLC Land Use category	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})
Mineral extraction sites	42.2	1.0	128.0	24.4	1.0	74.0	44.6	1.0	135.0	7.3	1,0	22,0
Dump sites	42.2	1.0	128.0	24.4	1.0	74.0	44.6	1.0	135.0	7.3	1,0	22,0
Construcion sites	42.2	0.0	128.0	24.4	0.0	74.0	44.6	0.0	135.0	7.3	0,0	22,0
Green urban areas	105.0	20.6	128.0	68.8	20.6	74.0	110.7	20.6	135.0	20.5	20,9	22,0
Sport and leisure facilities	105.0	13.1	128.0	68.8	13.1	74.0	110.7	13.1	135.0	20.5	4,7	22,0
Non-irrigated arable land	89.6	4.7	128.0	56.2	4.7	74.0	93.2	4.7	135.0	20.2	4,7	22,0
Permanently irrigated land	89.6	4.7	128.0	56.2	4.7	74.0	93.2	4.7	135.0	20.2	4,7	22,0
Rice fields	172.8	4.7	128.0	99.9	4.7	74.0	182.3	4.7	135.0	29.7	4,7	22,0
Vineyards	92.2	2.8	128.0	53.3	2.8	74.0	97.2	2.8	135.0	22.2	2,8	22,0
Fruit trees and berry plantations	92.2	63.0	128.0	53.3	63.0	74.0	97.2	63.0	135.0	22.2	9,0	22,0
Olive groves	92.2	6.9	128.0	53.3	6.9	74.0	97.2	6.9	135.0	22.2	6,9	22,0
Pastures	128.0	6.4	128.0	74.0	2.9	74.0	135.0	6.3	135.0	22.0	4,1	22,0
Annual crops associated with permanent crops	128.0	6.4	128.0	74.0	2.9	74.0	135.0	6.3	135.0	22.0	4,1	22,0
Complex cultivation patterns	92.2	13.1	128.0	53.3	13.1	74.0	97.2	13.1	135.0	22.2	36,1	22,0

Land principally occupied by agriculture, with significant areas of natural vegetation	105.0	13.7	128.0	68.8	13.7	74.0	110.7	13.7	135.0	20.5	36,1	22,0
Agro-forestry areas	92.2	20.6	128.0	53.3	20.6	74.0	97.2	20.6	135.0	22.2	20,9	22,0
Natural grassland	128.0	6.4	128.0	74.0	2.9	74.0	135.0	6.3	135.0	22.0	4,1	22,0
Moors and heathland	128.0	13.7	128.0	74.0	13.7	74.0	135.0	13.7	135.0	22.0	36,1	22,0
Sclerophyllous vegetation	128.0	13.7	128.0	74.0	13.7	74.0	135.0	13.7	135.0	22.0	36,1	22,0
Sparsely vegetated areas	105.0	1.0	128.0	68.8	1.0	74.0	110.7	1.0	135.0	20.5	1,0	22,0
Burnt areas **	108.8	BC_p*0.26	128.0	62.9	BC_p*0.26	74.0	114.75	BC_p*0.26	135.0	18.7	BC_p*0.26	22,0
ORGANIC SOIL (HAC) *												
	Cool temperate Moist			**Warm temperate Dry**			**Warm temperate Moist**			**Tropical Dry**		
CLC Land Use category	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})	**SC_0** (Mg* ha^{-1})	**BC_0** (Mg* ha^{-1})	**SC_p** (Mg* ha^{-1})
Mineral extraction sites	28.7	1.0	87.0	29.0	1.0	88.0	29.0	1.0	88.0	28.4	1,0	86,0
Dump sites	28.7	1.0	87.0	29.0	1.0	88.0	29.0	1.0	88.0	28.4	1,0	86,0
Construcion sites	28.7	0.0	87.0	29.0	0.0	88.0	29.0	0.0	88.0	28.4	0,0	86,0
Green urban areas	71.3	20.6	87.0	81.8	20.6	88.0	72.2	20.6	88.0	80.0	20,9	86,0
Sport and leisure facilities	71.3	13.1	87.0	81.8	13.1	88.0	72.2	13.1	88.0	80.0	4,7	86,0
Non-irrigated arable land	60.9	4.7	87.0	66.9	4.7	88.0	60.7	4.7	88.0	79.1	4,7	86,0
Permanently irrigated land	60.9	4.7	87.0	66.9	4.7	88.0	60.7	4.7	88.0	79.1	4,7	86,0
Rice fields	117.5	4.7	87.0	118.8	4.7	88.0	118.8	4.7	88.0	116.1	4,7	86,0
Vineyards	62.6	2.8	87.0	63.4	2.8	88.0	63.4	2.8	88.0	86.9	2,8	86,0
Fruit trees and berry plantations	62.6	63.0	87.0	63.4	63.0	88.0	63.4	63.0	88.0	86.9	9,0	86,0
Olive groves	62.6	6.9	87.0	63.4	6.9	88.0	63.4	6.9	88.0	86.9	6,9	86,0
Pastures	87.0	6.4	87.0	88.0	2.9	88.0	88.0	6.3	88.0	86.0	4,1	86,0
Annual crops associated with permanent crops	87.0	6.4	87.0	88.0	2.9	88.0	88.0	6.3	88.0	86.0	4,1	86,0
Complex cultivation patterns	62.6	13.1	87.0	63.4	13.1	88.0	63.4	13.1	88.0	86.9	36,1	86,0
Land principally occupied by agriculture, with significant areas of natural vegetation	71.3	13.7	87.0	81.8	13.7	88.0	72.2	13.7	88.0	80.0	36,1	86,0
Agro-forestry areas	62.6	20.6	87.0	63.4	20.6	88.0	63.4	20.6	88.0	86.9	20,9	86,0
Natural grassland	87.0	6.4	87.0	88.0	2.9	88.0	88.0	6.3	88.0	86.0	4,1	86,0
Moors and heathland	87.0	13.7	87.0	88.0	13.7	88.0	88.0	13.7	88.0	86.0	36,1	86,0
Sclerophyllous vegetation	87.0	13.7	87.0	88.0	13.7	88.0	88.0	13.7	88.0	86.0	36,1	86,0
Sparsely vegetated areas	71.3	1.0	87.0	81.8	1.0	88.0	72.2	1.0	88.0	80.0	1,0	86,0
Burnt areas **	74.0	BC_p*0.26	87.0	74.8	BC_p*0.26	88.0	74.8	BC_p*0.26	88.0	73.1	BC_p*0.26	86,0

Table S3 Values for potential biomass carbon stock post-afforesttion (BC_p) corresponding to the natural potential vegetation type (Blasi et al. 2017)

Code (Blasi et al.)	Class of Natural potential vegetation (Blasi et al.)	Class National inventory	Region of interest	**BC_p** Potential C stock (Mg * ha^{-1})
10a	Mediterranean maquis and garrigues	Mediterranean pines	Campania, Basilicata, Calabria, Sicily, Sardinia, Apulia, Molise	38.0
11a	Hygrophilous and hydrophytic freshwater vegetation of the Alps	Hygrophilous forest	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	46.0
11b	Continental hygrophilous and hydrophytic freshwater vegetation of the Po Plain	Hygrophilous forest	Piedmont, Emilia Romagna, Lombardy, Veneto	37.4
11c	Hygrophylous and hydrophytic freshwater vegetation of the Italian peninsula and islands	Hygrophilous forest	Italy	38.9
12a	Halo-hygrophylous vegetation of the northern Adriatic coasts	Hygrophilous forest	Friuli VG, Veneto, Emilia - Romagnba	39.1
12b	Halophilous and sub-halophilous vegetation of the Italian peninsula and islands	-		
13a	Psammophilous vegetation of the northern Adriatic coasts	-		
13b	Psammophilous vegetation of the Italian peninsula and islands	-		
14a	Chasmophytic vegetation of coastal cliffs	-		
15a	Pioneer vegetation on recent vulcanites on Mt. Etna and Vesuvio	-		
1a	Nival and sub-nival glareicolous and chasmophytic vegetation of the Alps	-		
2a	High-altitude grasslands with *Kobresia myosuroides*, *Carex sp.pl.*, *Festuca sp.pl.*, *Sesleria sphaerocephala* of the Alps	-		
2b	High-altitude grasslands (locally chasmophytic or glareicolous) with *Sesleria sp.pl.*, *Festuca macrathera*, *Nardus stricta*, *Carex kitaibeliana* of the Apennines and Apuan Alps	-		
3a	High-altitude shrubs and forests with *Pinus cembra, P. mugo, Larix decidua, Rhododendron sp.pl., Vaccinium sp.pl*. of the Alps	Black pines	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	58.4
3b	High-altitude shrub vegetation with *Juniperus communis subsp. alpina, Pinus mugo, Vaccinium myrtillus, Rhamnus alpina subsp. fallax* of the Apennines	Black pines	Liguria, Emilia-Romagna, Tuscany, Marche, Umbria, Lazio, Abruzzo, Molise, Campania, Basilicata, Calabria	73.0
3c	Oro-Mediterranean cushion-like shrub vegetation of Mt. Etna and Sardinian mountains	-		
4a	*Picea abies* and/or *Abies alba* forests of the Alps	Norwey spruce or Fir	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	117.0
5a	*Pinus sylvestris* and/or *P. nigra* forests of the Alps	Scots pine and Mountain pine	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	60.5
5b	Oro-Mediterranean and Mediterranean mountain forests with *Pinus leucodermis* or *P. laricio subsp. Calabrica*	Black pines	Campania, Basilicata, Calabria, Sicily, Sardinia, Apulia, Molise	65.8

5c	Mediterranean pine forests with *Pinus halepensis*, *P. pinaster* and/or *P. pinea*	Mediterranean pines	Campania, Basilicata, Calabria, Sicily, Sardinia, Apulia, Molise	38.0
6a	*Fagus sylvatica* forests with Abies *alba, Picea abies, Sorbus aucuparia, Carpinus betulus* of the Alps	Beech	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	95.0
6b	*Fagus sylvatica* forests with *Acer pseudoplatanus, Abies alba*, *Cardamine enneaphyllos, Geranium nodosum* of the Apennines	Beech	Liguria, Emilia-Romagna, Tuscany, Marche, Umbria, Lazio, Abruzzo, Molise, Campania, Basilicata, Calabria	114.8
6c	Mediterranean mountain *Fagus sylvatica* forests locally with *Ilex aquifolium, Taxus baccata, Acer cappadocicum subsp. lobelii*, *Abies nebrodensis*	Beech	Campania, Basilicata, Calabria, Sicily, Sardinia, Apulia, Molise	134.1
7a	Forests with *Ostrya carpinifolia, Fraxinus excelsior* and/or *Carpinus betulus* of the Alps, pre-Alps and Karst	Hornbeam and Hophornbeam	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	48.0
7b	Forests with *Ostrya carpinifolia* of the Apennines	Hornbeam and Hophornbeam	Liguria, Emilia-Romagna, Tuscany, Marche, Umbria, Lazio, Abruzzo, Molise, Campania, Basilicata, Calabria	41.6
8a	Mesophilous forests with *Quercus petraea, Q. robur* and/or *Carpinus betulus* of the Alps and Pre-Alps	Temperate oaks	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	50.7
8b	Thermophilous forests with *Quercus pubescens* and/or *Ostrya carpinifolia* of the Alps and pre-Alps	Temperate oaks	Liguria, Piedmont, Aosta Valley, Lombardy, Trentino Alto Adige, Friuli VG, Veneto	50.7
8c	Continental forests with *Quercus robur*, *Q. petraea* and/or *Carpinus betulus* of the Po Plain	Temperate oaks	Piedmont, Emilia Romagna, Lombardy, Veneto	49.1
8d	Forests with *Quercus petraea* and/or *Q. cerris* of the central and northern Apennines and sub-Apennines	Mediterranean oaks	Liguria, Emilia-Romagna, Tuscany, Marche, Umbria, Lazio, Abruzzo, Molise, Campania, Basilicata, Calabria	57.7
8e	Mesophilous forests with *Quercus pubescens* and/or *Ostrya carpinifolia* of the Apennines and sub-Apennines	Temperate oaks	Liguria, Emilia-Romagna, Tuscany, Marche, Umbria, Lazio, Abruzzo, Molise, Campania, Basilicata, Calabria	35.9
8f	Mesophilous forests with *Quercus cerris* of the Italian peninsula	Mediterranean oaks	Italy	52.2
8g	Thermophilous forests with *Quercus cerris* (locally with *Q. frainetto*) of the Italian peninsula	Mediterranean oaks	Italy	52.2
8h	Thermophilous forests with *Quercus virgiliana* of the Italian peninsula	Holm oak	Italy	48.9
8i	Eastern thermophilous forests with *Quercus virgiliana, Q. trojana, Q. macrolepis* or *Q. frainetto* of Murge and Salento	Holm oak	Apulia	37.8
8l	Mediterranean deciduous and semideciduous forests with *Quercus virgiliana, Q. congesta*, *Q. ichnusa, Q. gussoni* of Sicily and Sardinia	Holm oak	Sicily, Sardinia	53.2
9a	Evergreen forests with *Quercus ilex* of the Italian peninsula (and local occurrences in Insubria)	Holm oak	Italy	48.9
9b	Evergreen forests with *Quercus ilex* and/or *Q. calliprinos* of the Apulia region	Holm oak	Apulia	37.8
9c	Evergreen forests with *Quercus ilex* (and local occurences of *Q. calliprinos* on sands) of Sicily and Sardinia	Holm oak	Sicily, Sardinia	53.2
9d	Evergreen forests with *Quercus suber*	Coark oak	Italy	28.3

Table S4 Values of CN parameter values in function of land use (Corine Land Cover 2018) and hydrologic soil group (A, B, C and D)

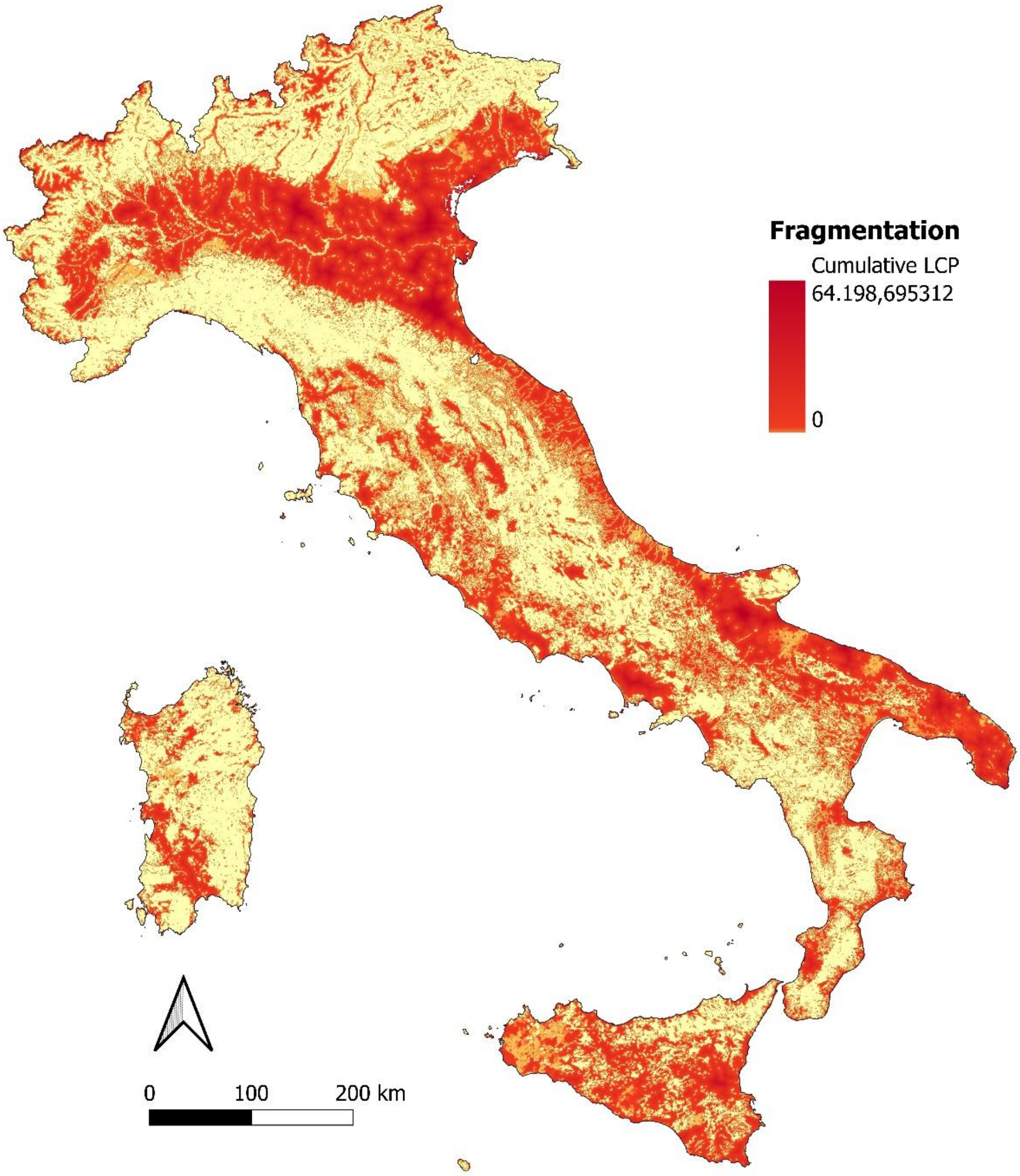
Corine Land Cover: description	**HSG A**	**HSG B**	**HSG C**	**HSG D**
111 - Continuous urban fabric	89	92	94	95
112 - Discontinuous urban fabric	77	85	90	92
121 - Industrial or commercial units	81	88	91	93
122 - Road and rail networks and associated land	98	98	98	98
123 - Port areas	98	98	98	98
124 - Airports	98	98	98	98
131 - Mineral extraction sites	76	85	89	91
132 - Dump sites	81	88	91	93
133 - Construction sites	77	86	91	94
134 - Aree degradate non utilizzate e non vegetate	72	82	87	90
141 - Green urban spaces	49	69	79	84
142 - Sport and leisure facilities	68	79	86	89
211 - Not-irrigated arable land	61	73	81	84
212 - Permanently irrigated land	62	71	78	81
213 - Rice fields	62	71	78	81
221 - Vineyards	76	85	90	93
222 - Fruit trees and berry plantations	43	65	76	82
223 - Olive groves	43	65	76	82
224 - Wood arboriculture plantations	67	78	85	89
231 - Pastures	49	69	79	84
241 - Annual crops associated with permanent crops	61	73	81	84
242 - Complex cultivation patterns	61	73	81	84
243 - Land principally occupied by agriculture with significant areas of natural vegetation	61	73	81	84
244 - Agro-forestry areas	67	78	85	89
311 - Broad-leaved forest	36	60	73	79
312 - Coniferous forest	36	60	73	79
313 - Mixed forests	36	60	73	79
321 - Natural grassland	49	69	79	84
322 - Moors and heathland	49	69	79	84
323 - Sclerophyllous vegetation	35	56	70	77
324 - Transitional woodland-shrub	35	56	70	77
331 - Beaches dunes sands	46	65	77	82
332 - Bare rocks	96	96	96	96
333 - Sparsely vegetated areas	63	77	85	88
334 - Burnt areas	81	88	91	93
335 - Glaciers and perpetual snow	36	60	73	79
411 - Inland marshes	98	98	98	98
412 - Peat bogs	98	98	98	98
421 - Salt marshes	98	98	98	98
422 - Salines	98	98	98	98
423 - Intertidal flats	98	98	98	98
511 - Water courses	98	98	98	98
512 - Water bodies	98	98	98	98
521 - Coastal lagoons	98	98	98	98
522 - Estuaries	98	98	98	98
Afforestation	35	56	70	77

Table S5 Percentage of High-Priority Areas (HPAs) per region relative to the total HPAs

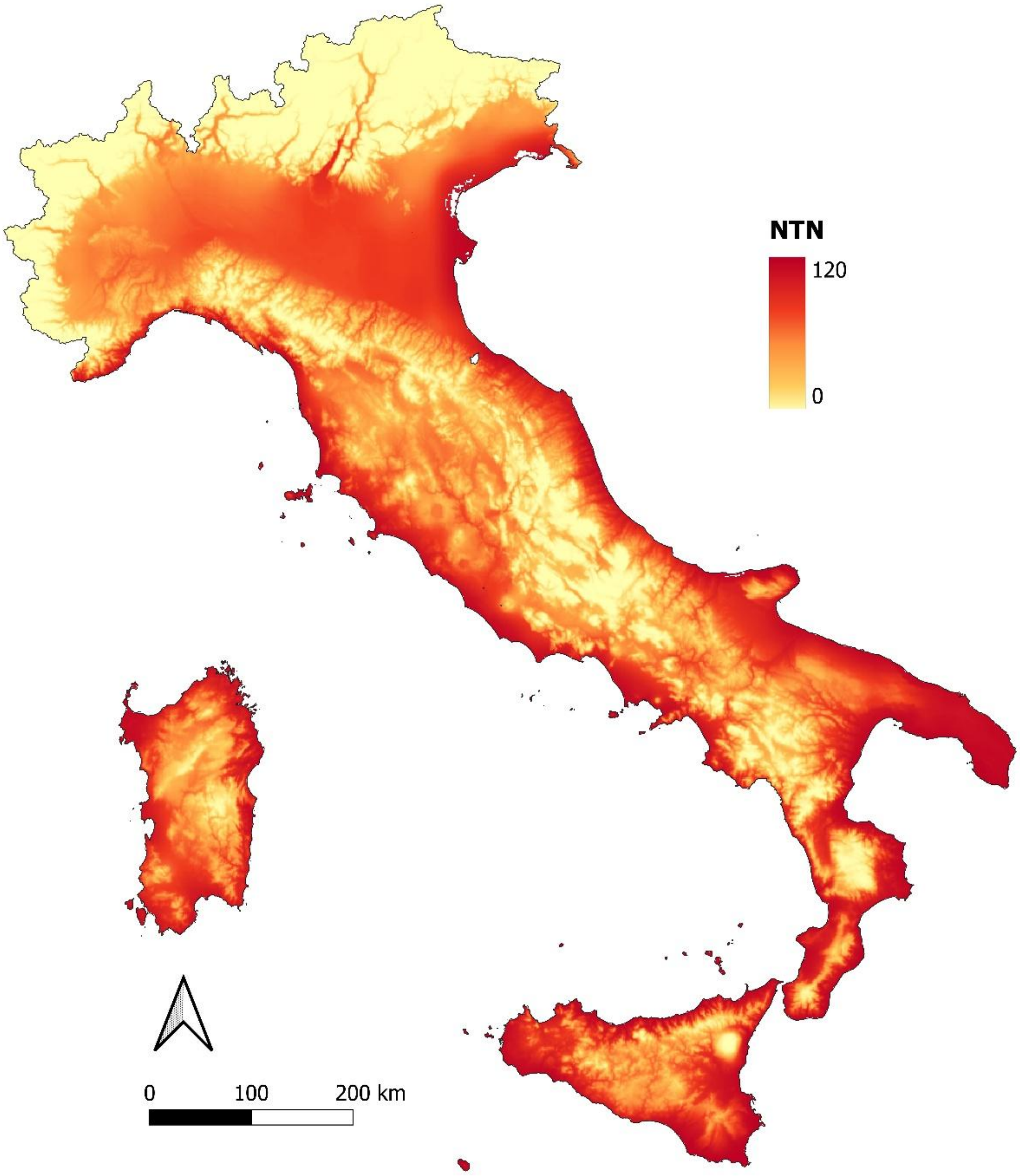
Region	HPAs for ecological connectivity goal (%)	HPAs for human health goal (%)	HPAs for climate mitigation goal (%)	HPAs for regulation of the water cycle goal (%)
Abruzzo	0.9	2.2	2.2	1.8
Basilicata	0.7	0.6	0.8	0.4
Calabria	1.5	3.7	4.6	1.5
Campania	3.8	6.3	4.6	3.5
Emilia-Romagna	19.6	9.6	5.7	8.2
Friuli-Venezia Giulia	2.1	2.5	6.6	4.4
Lazio	2.8	7.8	7.6	5.8
Liguria	0.0	1.6	1.8	0.5
Lombardy	14.7	18.4	8.4	15.8
Marche	1.5	3.1	1.9	1.8
Molise	1.1	0.4	1.0	0.4
Piedmont	5.4	6.4	10.4	8.3
Apulia	17.6	5.6	0.0	9.5
Sardinia	1.8	2.1	2.1	3.1
Sicily	7.4	6.4	18.4	11.5
Tuscany	1.8	6.0	7.0	6.3
Trentino-Alto Adige	0.0	1.1	4.9	0.7
Umbria	0.9	1.5	2.9	1.9
Aosta Valley	0.0	0.1	0.5	0.1
Veneto	16.4	14.6	8.5	14.2

All Maps M12-15 created for this study are available in Raster format (.tif). PMs maps are available in vectorial format (.shp) on request

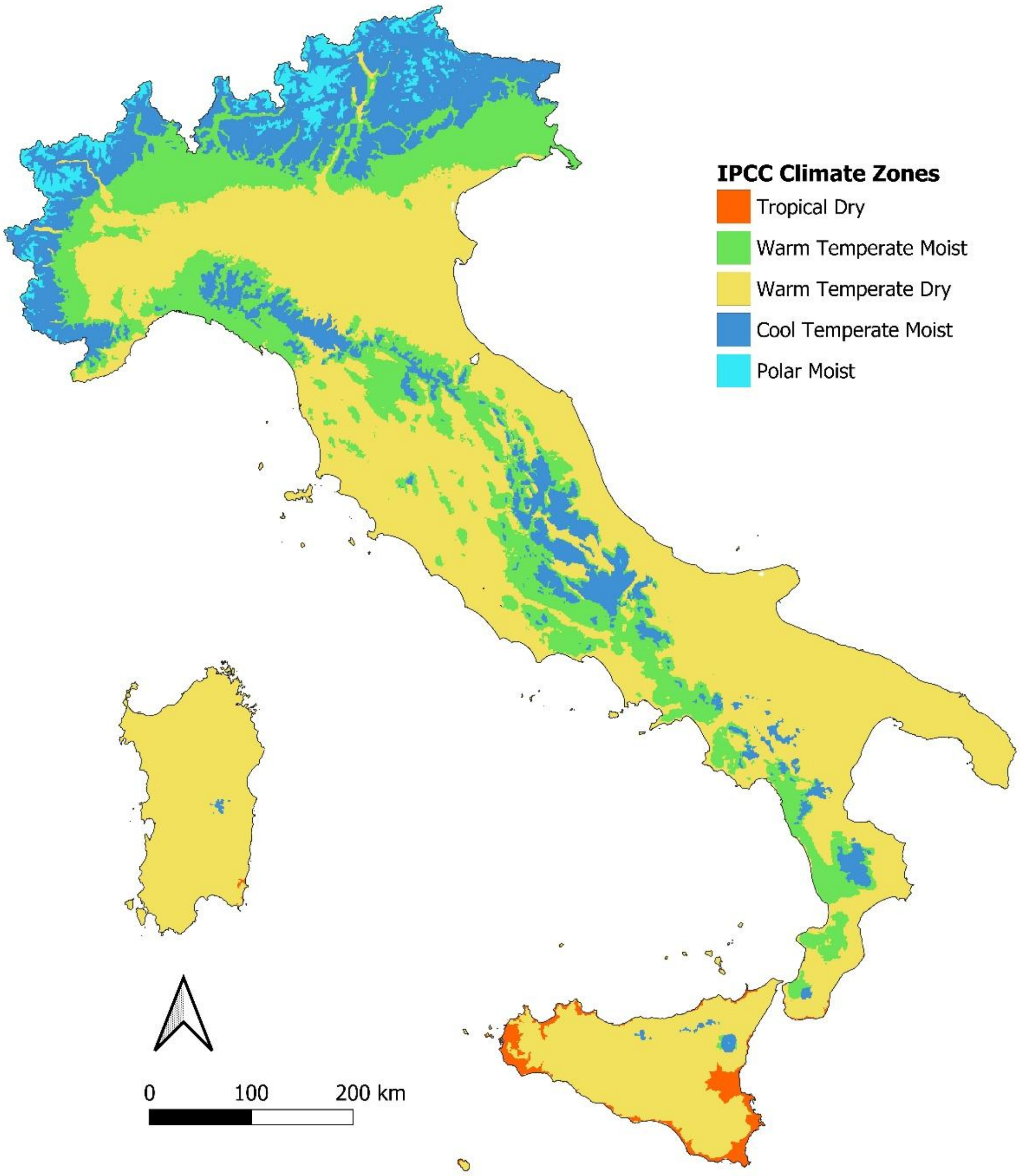
FigureS1 M12 – Fragmentation (F) map, colored by quantiles



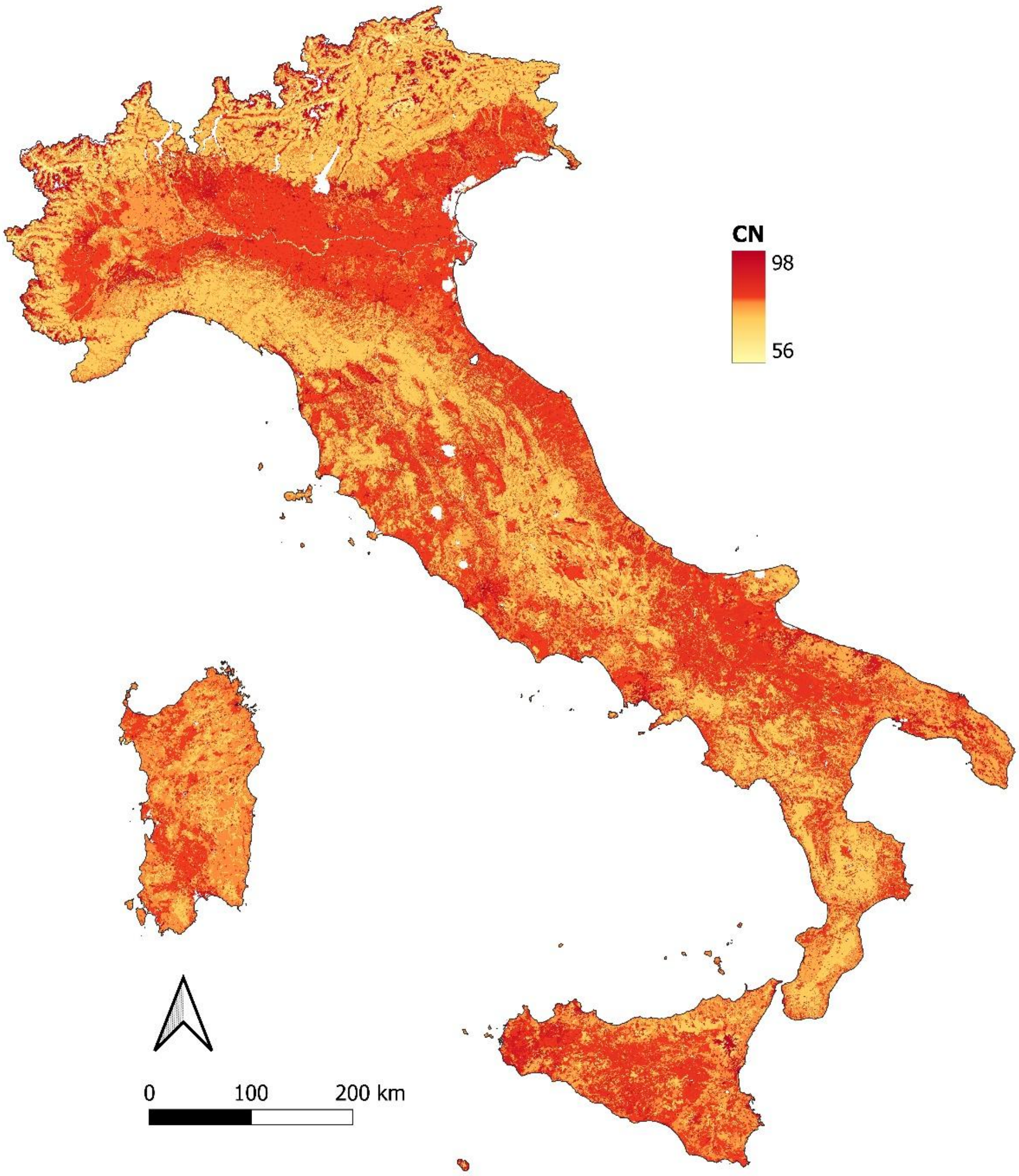
FigureS2 M13 – Map of Number of Tropical Nights (NTN) per year in future climate scenarios (period 2041-2070), colored by quantiles



FigureS3 M14 – IPCC Climatic Zones map



FigureS4 M15 –Map of CN_{adj} (slope-adjusted values of CN), colored by quantiles



Conclusion

The three studies presented in this thesis differ in both objectives and methodological approaches, yet they all highlight a fundamental insight: tree planting can provide substantial ecosystem service benefits, although poorly planned interventions may be counterproductive or involve trade-offs. These findings underscore the importance of carefully considering both environmental and social contexts when designing and implementing tree planting initiatives.

In the first study, we demonstrated that urban afforestation can be an effective long-term strategy for wild pollinator conservation. Urban forests create microhabitats and provide deadwood suitable for nesting. However, their success also depends on the management of surrounding grasslands: maintaining habitat heterogeneity and implementing staggered mowing schedules ensures adequate trophic resources. It is also important a landscape-scale planning and public communication to raise awareness of the ecological value of unmown areas, suggesting a zoning strategy that balances recreational spaces and naturally managed habitats.

The second study confirmed the significant contribution of the tree component in agroforestry systems to carbon sequestration, highlighting their advantage over intensive monoculture agriculture. It also supports previous evidence of role of shade trees in coffee cultivation for regulating local microclimates and reducing temperature stress. Converting monoculture coffee plantations into diversified agroforestry systems may become crucial in many tropical regions to maintain productivity under rising temperatures while contributing to climate mitigation. Our results show that tree species selection, canopy density, and proper management are key factors determining the success of tree planting in agroforestry.

The third study identified priority areas for afforestation at the national scale, demonstrating that some regions offer higher potential to maximize ecosystem service provision, while afforesting others may lead to negative outcomes—for example, converting rice paddies can result in soil carbon losses. We also found that some areas can deliver multiple services, such as urban zones where afforestation may enhance biodiversity, water regulation, and human health. However, significant trade-offs exist among different services, highlighting the importance of defining clear objectives for tree planting interventions to ensure effectiveness.

Overall, these studies support the view that tree planting can play a crucial role in addressing global environmental challenges by contributing to climate mitigation, biodiversity conservation, and ecosystem restoration. We also provide practical insights and recommendations to enhance the effectiveness of such interventions, while acknowledging the limitations imposed by available economic and management resources. Continued research on ecological and social dynamics is essential to inform sustainable land management decisions.

These studies advance the understanding of the relationships between tree planting and regulating ecosystem services, yet they can be further developed. For instance, multi-year sampling of wild bees or inclusion of additional invertebrate groups would strengthen ecological analyses. The agroforestry study represents the first application of the FORMIND model to these systems, and future work should incorporate facilitative effects of shade trees to capture more complex interactions. Finally, the national-scale prioritization framework could be adapted to finer spatial resolutions, providing actionable insights for regional or municipal land-use planning.

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