Ph.D. in Environmental and Evolutionary Biology XXXVIII Cycle

Analyses of environmental factors affecting Potential Natural Vegetation in different types of urban ecosystems: a tool for supporting effective forestation

Village of Gildone (CB), Molise Region Italy (photo by Alessandro Montaldi)

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Abstract

Urbanisation represents one of the most transformative forces shaping contemporary ecosystems, with important impacts on biodiversity and ecological processes. Ongoing urbanisation trends are expected to worldwide affect the sustainability and resilience of ecosystems in terms of biodiversity, ecosystem services, and human well-being, as citizens are often exposed to environmental conditions detrimental to their health. In Europe, the Nature Restoration Regulation (1991/2024) emphasises the role of cities in biodiversity recovery, given that urban areas cover 22% of the EU's land and host most of its population. Key measures include achieving No Net Loss of urban green spaces (e.g., parks, green roofs, urban forests) and tree canopy cover by 2030, as compared to 2024 levels, and their increase from 2031 onward until satisfactory levels are met. Ground-based knowledge of urban ecosystems at the local level is essential for implementing restoration actions, including Nature-based Solutions (NbS) and Green Infrastructure (GI), to support and conserve local biodiversity in cities. Within the framework of the National Biodiversity Future Center (Spoke 5 - Urban Biodiversity), this research aimed at developing an integrated framework to define and map Potential Natural Vegetation (PNV) as a knowledge tool viable in urban and peri-urban landscapes, by combining new methodological insights with field observations. Through three interconnected research activities conducted in the Italian Functional Urban Areas of Rome and Campobasso, this work provided a holistic understanding of how anthropogenic pressures can alter vegetation dynamics while proposing actionable solutions for sustainable urban planning. The research builds upon a hierarchical methodology for PNV mapping (first research topic), which integrates bioclimatic, lithological, and geomorphological spatial analyses with phytosociological surveys. By employing multivariate analyses to identify diagnostic environmental thresholds, this approach produced high-resolution ecological maps that improved existing regional and national cartographic products in both precision and thematic detail. This methodological advancement is not only able to properly support ecological restoration, by providing reference models, but also serves as a critical baseline for detecting anthropogenic impacts on vegetation systems, especially when other variables typical of highly urbanised areas are considered.

Focusing on these impacts, the second research topic was directed towards the understanding of how the urban landscape may disrupt PNV character and mature successional stages, with a focus on anthropogenic landform modifications impacts in the case study of Rome. Through comparative vegetation analyses across a disturbance gradient, the study revealed that anthropogenic landforms caused a shift in woodland composition, favouring invasive and nitrophilous species while altering edaphic and microclimatic conditions. These findings demonstrated that urban forests comprise distinct floristic assemblages, necessitating revised frameworks that also recognise the environmental and dynamic profile of novel ecosystems. Differences in species composition between woodlands in different conditions were thus quantified, in terms of ecological distances from PNV, modelled and mapped across the city. The identification and spatialisation of areas that shifted furthest from reference systems, with an unlikely reversible deviation from PNV, represent a useful finding for the practical implementation of the research. Moving towards the urban - periurban interface, the third research topic examined how landscape fragmentation and edge effects facilitate biological invasions in transitional shrubland ecosystems. Combining field surveys with landscape metrics analysis, the research quantified invasion drivers and developed a predictive model of invasibility. Based on these ecological insights and the availability of PNV maps, the study also proposed a practical solution

through the design of a targeted GI network that simultaneously addresses invasion containment and habitat restoration—a model adaptable to other peri-urban regions. This work achieved three key contributions: (1) a standardised yet flexible protocol for PNV typification and mapping in urban ecosystems; (2) empirical evidence of urbanisation's cascading effects on environmental characters, landscape mosaics, vegetation systems and dynamics; (3) science-based strategies for managing novel urban ecosystems and integrating PNV knowledge into spatial planning processes. Management and planning tools were therefore provided to reconcile urban ecosystems with biodiversity conservation and restoration, particularly relevant for Mediterranean regions and other global biodiversity hotspots undergoing rapid urbanisation. The integrated approach developed here provides a template for addressing one of the critical challenges of actual time: designing cities that behave as self-sustaining socio-ecological systems. Future applications could extend this framework to other biogeographical contexts, incorporating additional dimensions such as climate change resilience and ecosystem service optimisation.

Sintesi

L'urbanizzazione è una delle forze principali che modellano gli ecosistemi, con effetti significativi sulla biodiversità e sui processi ecologici. Si prevede che le attuali tendenze di urbanizzazione influenzeranno, a livello mondiale, la sostenibilità e la resilienza degli ecosistemi in termini di biodiversità, servizi ecosistemici erogati e benessere umano, poiché i cittadini sono spesso esposti a condizioni ambientali dannose per la loro salute. In Europa, il Regolamento sul Ripristino della Natura (1991/2024) sottolinea il ruolo delle città nel recupero della biodiversità, dato che le aree urbane coprono il 22% del territorio dell'UE e ospitano la maggior parte della sua popolazione. Le misure chiave includono il raggiungimento del “No Net Loss” di spazi verdi urbani (es. parchi, tetti verdi, foreste urbane) e di copertura arborea entro il 2030, rispetto ai livelli del 2024, e il loro incremento dal 2031 in poi, fino al raggiungimento di livelli soddisfacenti.

Una conoscenza approfondita degli ecosistemi urbani a livello locale è essenziale per implementare azioni di ripristino, tra cui le Nature-based solutions (NbS) e le Infrastrutture Verdi (IV), al fine di supportare e conservare la biodiversità locale nelle città. Nell'ambito del National Biodiversity Future Center (Spoke 5 - Biodiversità Urbana), questa ricerca ha mirato a sviluppare un framework integrato per definire e mappare la Vegetazione Naturale Potenziale (VNP) come strumento di conoscenza applicabile ai paesaggi urbani e peri-urbani, combinando nuove prospettive metodologiche con osservazioni sul campo. Attraverso tre attività di ricerca condotte nelle Functional Urban Areas italiane di Roma e Campobasso, questo lavoro ha fornito una comprensione olistica di come le pressioni antropiche possano alterare le dinamiche della vegetazione, proponendo al contempo soluzioni operative per una pianificazione urbana sostenibile.

Il primo lavoro si basa su una metodologia gerarchica per la mappatura della VNP che integra la divisione del territorio su basi bioclimatiche, litologiche e geomorfologiche con l’esecuzione di rilievi fitosociologici. Utilizzando analisi multivariate per identificare soglie ambientali diagnostiche, questo approccio ha prodotto mappe ecologiche ad alta risoluzione che hanno migliorato i prodotti cartografici regionali e nazionali esistenti, sia in precisione sia nel dettaglio tematico. Tale avanzamento metodologico non solo è in grado di supportare adeguatamente il ripristino ecologico fornendo modelli di riferimento, ma costituisce anche una base di riferimento per rilevare gli impatti antropici sui sistemi vegetazionali, soprattutto quando si tengono conto di altre variabili tipiche delle aree altamente urbanizzate.

Focalizzandosi su tali impatti, il secondo lavoro è stato condotto per comprendere come il paesaggio urbano possa alterare la VNP e gli stadi successionali maturi, con particolare focus sull'impatto delle alterazioni geomorfologiche di origine antropica nell’area studio di Roma. Attraverso analisi comparative della vegetazione lungo un gradiente di disturbo, lo studio ha rivelato che le forme del paesaggio antropogeniche determinano un cambiamento nella composizione delle comunità boschive, favorendo specie invasive e nitrofile e alterando le condizioni edafiche e microclimatiche. Questi risultati hanno dimostrato che le foreste urbane presentano associazioni floristiche distinte, rendendo necessarie nuove cornici interpretative che riconoscano anche il profilo ambientale e dinamico dei nuovi ecosistemi (*novel ecosystems*). Le differenze in composizione di specie tra boschi in condizioni diverse, sono state quantificate in termini di distanze ecologiche dalla VNP, e quindi modellizzate e mappate in tutta la città. L'identificazione e la spazializzazione delle aree che si allontanano maggiormente dai sistemi di riferimento, con una deviazione dal PNV difficilmente reversibile, rappresentano un risultato utile per l'attuazione pratica della ricerca. Spostandosi verso l'interfaccia urbano-periurbana, il terzo lavoro ha esaminato come la frammentazione del paesaggio e gli effetti margine facilitino le invasioni biologiche negli ecosistemi arbustivi di transizione. Combinando rilievi di campo con l'analisi delle metriche di paesaggio, la ricerca ha quantificato i driver di invasione e ha sviluppato un modello predittivo di invasibilità. Sulla base di queste conoscenze ecologiche e della disponibilità di mappe della VNP, lo studio ha anche proposto una soluzione pratica, progettando una rete di IV mirata che affronta simultaneamente il contenimento delle invasioni e il ripristino degli habitat, un modello adattabile ad altre aree periurbane.

Questo lavoro ha prodotto tre contributi principali: (1) un protocollo standardizzato ma flessibile per la tipificazione e la mappatura della VNP negli ecosistemi urbani; (2) prove empiriche sugli effetti a cascata dell'urbanizzazione sull'ambiente, sui mosaici paesaggistici, sui sistemi vegetazionali e sulle loro dinamiche; (3) strategie basate su evidenze scientifiche per la gestione degli ecosistemi urbani e per l'integrazione della conoscenza sulla VNP nei processi di pianificazione spaziale. Sono stati quindi forniti strumenti gestionali e di pianificazione per conciliare gli ecosistemi urbani con la conservazione e il ripristino della biodiversità, particolarmente rilevanti per le regioni mediterranee e per altri hotspot di biodiversità globale soggetti a rapida urbanizzazione. L'approccio integrato qui sviluppato fornisce un modello per affrontare una delle sfide cruciali del nostro tempo: progettare città che funzionino come sistemi socio-ecologici autosufficienti. Le applicazioni future potrebbero estendere questo framework ad altri contesti biogeografici, incorporando dimensioni aggiuntive come la resilienza ai cambiamenti climatici e l'ottimizzazione dei servizi ecosistemici.

Introduction

Within the policy framework concerning nature conservation and restoration, biodiversity in urban areas has become an increasingly important topic in recent years (UN, 2015; EC, 2020; CBD, 2022). Urbanisation is indeed an inexorable global process that is reshaping planet's ecosystems, concentrating the human population and its activities in highly impacted areas (Grimm et al., 2008). Transformation of the landscape is recognised as a primary driver of biodiversity loss, alteration of biogeochemical cycles, and degradation of ecosystem services, with profound implications for environmental sustainability and human well-being (Grimm et al., 2008; Frondoni et al., 2011; Theodorou, 2022). These issues cannot be counterbalanced solely by protecting natural areas outside the urban and peri-urban environment. Part of our attention should also be directed towards the improvement and re-naturalisation of most impacted ecosystems within cities (EC, 2020; Rosenzweig, 2003)

In response to this challenge, the international community and institutions have developed policies and strategic frameworks to reverse this trend and promote urban sustainability. The United Nations Sustainable Development Goals (SDGs), particularly SDG 11 ("Make cities and human settlements inclusive, safe, resilient and sustainable") and SDG 15 ("Protect, restore and promote sustainable use of terrestrial ecosystems"), have laid the groundwork for global action (UN, 2015). At the European level, the EU Biodiversity Strategy for 2030 (EC, 2020) and, more recently, the pioneering Nature Restoration Regulation (Reg. EU 2024/1991), have translated these goals into concrete obligations. The latter, in particular, requires Member States to halt the net loss of urban green spaces and increase tree canopy cover, explicitly acknowledging the vital role of cities in nature conservation. These policy directives create a clear imperative: not only to protect existing natural areas but also to actively implement ecological restoration and afforestation interventions on a large scale within the urban and peri-urban fabric. These targets can be effectively achieved through land management and urban planning, comprising the design of Green Infrastructure (GI), defined as a “strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services” and the consequent implementation of Nature-based Solutions (EC, 2020). However, the effectiveness of such interventions is far from being guaranteed. Planting trees or creating green spaces without a solid ecological basis can lead to failure, waste of resources, and, in some cases, even exacerbate existing problems, such as the spread of invasive alien species (Hulme et al., 2008; Kowarik, 2011; Palmer et al., 2014). Urban and landscape planning, therefore, urgently need robust scientific tools to provide species ecologically suited to a given site and design interventions that are resilient, self-sustaining, and capable of maximising ecological benefits (Capotorti et al., 2019). Effective biodiversity conservation, management and restoration rely on a thorough comprehension of ecosystems, which makes it necessary to describe, characterise, and spatially locate them at an appropriate scale, thereby providing valuable information for local administration and urban planning (Dickson et al., 2014; Maes et al., 2020). Consequently, the definition of urban ecosystems and their geographic delimitation are crucial aspects. An urban ecosystem can be considered as a complex landscape mosaic comprising both artificial (built) surfaces and natural or semi-natural green and blue spaces closely related to them, which also encompass peri-urban areas (Maes et al., 2020). Multiple approaches exist to delineate urban ecosystems, and at the European level, two main alternatives have been indicated for addressing the regulations and ecosystems accounting aims (Babi-Almenar et al., 2023):

- Local Administrative Units (LAU), i.e. cities (>50 000 inhabitants) and towns and suburbs classified considering the degree of urbanisation (EUROSTAT, 2018).
- Functional Urban Areas (FUA), considering cities as core areas plus the surrounding commuting zone, representing the metropolitan level (Maes et al., 2016; EUROSTAT, 2018).

These typologies present pros and cons: FUA do not encompass the entire urban population of the EU27 (leaving out 20.7%), whereas LAU cover the total population, and are aligned with administrative limits, thereby offering greater adaptability for local policy and management. Conversely, FUA account for a higher proportion of semi-natural ecosystems (including Natura2000 sites) and can be disaggregated using LAU as sub-territorial units (Zulian et al., 2022). The selection between these frameworks is contingent upon the spatial extent and compositional characteristics of the study area, but mainly the specific objectives and the detail of investigation. In this context, the Ecological Classification of Land (ELC) is a valid approach for studying ecosystems across diverse urban settings. Thanks to its modularity ELC can help ecosystem typology and distribution to be delineated with a hierarchical definition of portions of the territory, or Environmental Land Units (ELU), that are homogeneous in physical and biological terms (Bailey et al., 2004). ELUs are capable of sustaining adapted species and communities belonging to a specific vegetation series, which culminates with the establishment of the mature stage or Potential Natural Vegetation (PNV) (Rivas-Martinez, 2005; Capotorti et al., 2012). PNV is defined as the mature stage of a vegetation series that would establish in an area under certain ecological conditions, without human disturbance, remaining stable over time. PNV is therefore bound to a ‘particular ecological zone’, which can be seen as the ELU where the combination of environmental factors (climate, lithology and geomorphology) is capable of sustaining a specific vegetation type (Tüxen, 1956; Zerbe, 1998). PNV-based landscape classifications promote the understanding of landscape patterns, and the concept of PNV emerges as a tool of fundamental importance in restoration ecology, since its definition coincides with the concept of reference ecosystems (Fischer et al., 2019; Nelson et al., 2024). While a return to a "pristine nature" is unrealistic in deeply and permanently altered urban contexts, the value of PNV lies in its use as an ecological baseline. PNV helps define the ecological potential of a site, and guides the selection of suitable native species for restoration projects (Capotorti et al., 2013). However, in urban environments, several factors can significantly impact PNV besides the actual vegetation. Urban and peri-urban landscapes are profoundly altered, presenting a mosaic of diverse land uses and cover types that disrupt the original functioning of ecosystems and their potential (Sanderson et al., 2002). Local species extinction due to landscape-level transformations, alteration of biogeochemical cycles due to land use and management, and introduction of new species related to artificial land cover determine the establishment of “novel” communities, assembled as a result of a biotic response to human-induced abiotic conditions (Hobbs et al., 2006; Boscutti et al., 2018; Frondoni et al., 2011). All this, therefore, leads to a departure from the PNV or reference model known for an area. This change must be taken into account to ensure that any restoration actions are more effective by selecting species that are truly adapted to the different types of urban ecosystems (Capotorti et al., 2012). Once all the basic knowledge of an area is collected, it is possible to proceed with the necessary restoration and conservation interventions and these actions should be brought together within GIs designed and planned to address specific problems. Within the framework of Urban Biodiversity research of the National Biodiversity Future Center, the present work is positioned precisely at the intersection of the demands dictated by the policy framework and the need for operational scientific tools. The overall objective is to develop and validate a multi-scale approach for analysing vegetation in urban ecosystems, providing concrete support for planning ecological restoration and afforestation interventions. The research has been structured into three steps, ranging from FUAs to the municipal and sub-municipal level and from methodological to analytical and applicative approaches (Figure 1):

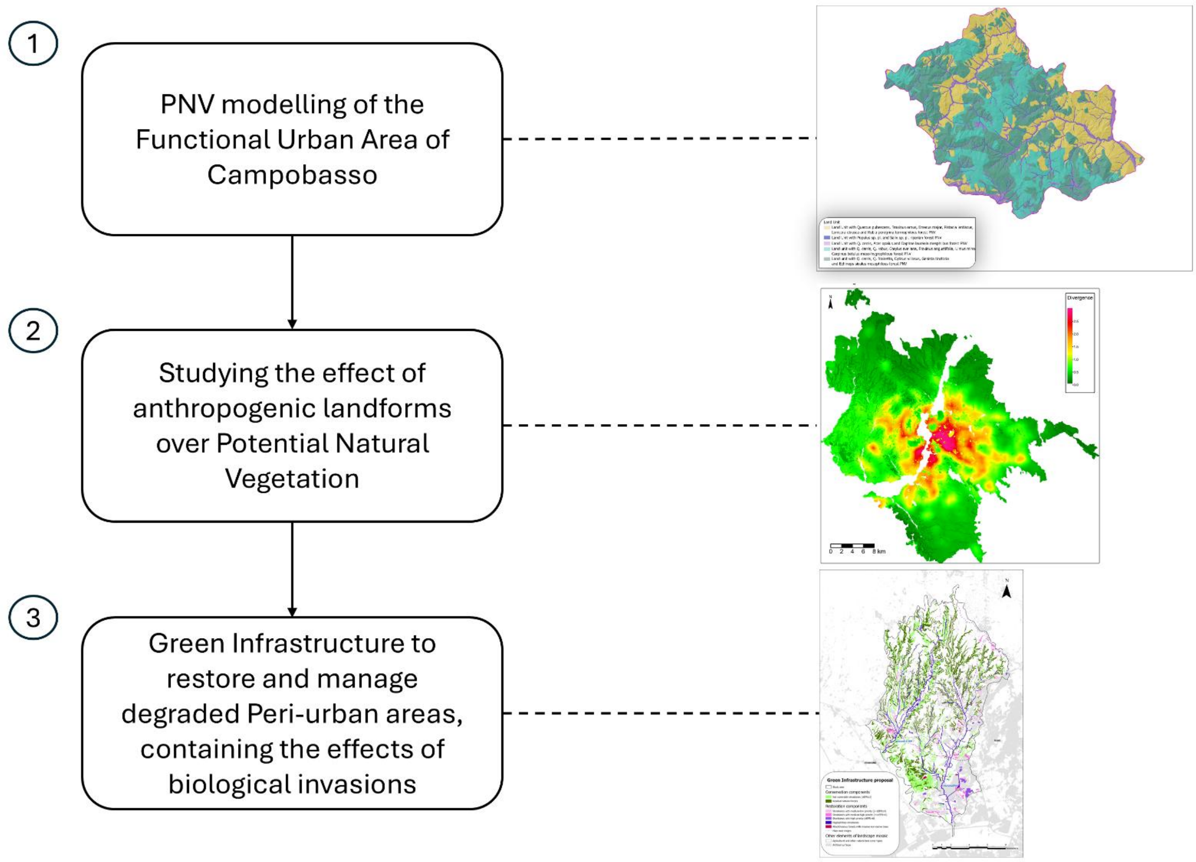

Figure 1: multi-level framework of the research

1- The first essential step was to develop a quantitative, replicable, and fine-scale protocol to produce or upgrade PNV maps in different typologies of urban ecosystems, overcoming the limitations of traditional approaches, which were often qualitative and too coarse for the heterogeneity of the urban mosaic. The aim was to develop a repeatable, robust and scalable methodology for identifying appropriate reference ecosystems and generating high-resolution PNV maps under a hierarchical and semi-automated process.
   This methodology was refined at the FUA level Campobasso (southern Italy), which include the connected peri-urban areas (comprising many Natura2000 sites) otherwise excluded, considering only the LAU/municipality. The area exhibits a moderate anthropogenic impact and where environmental factors determining ecological potential are still clearly discernible (**Part I**).
2- Subsequently, the research shifted to a more populated urban system, Rome, to investigate how the legacy of a long-standing urbanisation affects PNV. In particular, the focus was placed on a pervasive yet often overlooked factor, i.e. the impact of anthropogenic landforms.

This phenomenon has been observed principally at the municipal level, thus the study considered only the LAU. While widespread landscape transformations, like land-use changes, are well-documented drivers of biodiversity loss, other less evident human activities can also reshape ecosystems on a more fundamental level, significantly altering the determinant environmental features of PNV. The extent to which profound morphological alterations (e.g., quarries, landfills, embankments) create permanent environmental conditions that cause real vegetation communities to deviate from their natural potential was defined and quantified, thus contributing to fill an important knowledge gap (**Part II**).

3- Finally, the research brought to an applied synthesis. The knowledge on PNV information and disturbance factors was used to address one of the major threats to biodiversity, that is biological invasions. Using PNV as a reference for selecting resilient native species to be planted and/or facilitated, a GI network was designed. The addressed ecological theme and the applicative purpose of this study made it necessary to be performed at the sub-municipal level. This tool is not conceived as a simple increase in green surface but as a strategic and multifunctional solution, capable of enhancing ecological connectivity for native species while simultaneously serving as a barrier to contain the pressure of the invasive alien ones (**Part III**).

Overall, this work aims to build a bridge between the challenging goals of current environmental policies and the operational issues of spatial planning, providing a scientific framework that allows the goal of "restoring nature" to be translated into concrete, effective, and ecologically coherent actions within and around urban areas.

Part I: Mapping Potential Natural Vegetation through a quantitative and multivariate approach. The case study of Campobasso Functional Urban Area (Italy)

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Abstract

To provide a methodology useful for fine-scale ecological research on urban and peri-urban areas, we propose an updated approach for the Ecological Land Classification (ELC) based on Potential Natural Vegetation, tested in the Functional Urban Area (FUA) of Campobasso (Italy). We applied a divisive process, hierarchically dissecting the territory from broader to narrower land units (LU) based on environmental discontinuities with respect to bioclimate, lithology and geomorphology. Within these LU, phytosociological surveys of woodland patches representative of the current vegetation potential were homogeneously stratified. Sampled data were thus investigated by statistical clustering and Indicator Species Analyses, allowing different mature vegetation typologies to be recognised and respective relationships with environmental variables to be detected. In particular, threshold values for each environmental diagnostic feature with respect to PNV discontinuities were estimated through multivariate analyses. Finally, a detailed PNV/Environmental Land Unit map of the study area was obtained, enhancing not only the geometric but also the typological detail of the cartographic information already available at the regional and national level. By improving knowledge on mature forest ecosystems and respective environments, the approach may effectively support restoration actions in the specific case study, but can also be transferred and generalised for broader biodiversity research on urban systems, such as that on the overall Italian FUA promoted by the National Biodiversity Future Center.

Keywords: Potential Natural Vegetation, Ecological Land Classification, Ecological modelling, Indicator species, Reference ecosystems

Highlights:

- Defining reference ecosystems is crucial for effective restoration
- Ecological land units and PNV represent useful reference models
- An updated and semi-automatic process is proposed here for their spatialisation
- The process may be easily attuned to other urban context types

1. Introduction

Nature and biodiversity make life possible, provide health and social benefits and support our economy (WHO, 2023). In Europe, as in many other regions worldwide, ecosystems and services they provide are under pressure from urban sprawl, intensive agriculture, pollution, biological invasion and climate change, resulting in widespread degradation since more than 80% of habitats are in poor condition (EC, 2020). In keeping with the UN Decade on Ecosystem Restoration strategy (UN, 2019), the EU Nature Restoration Regulation (1991/2924) aims to put biodiversity on the path to recovery by 2030 to benefit people, climate and the planet and set targets concerning different types of natural ecosystems, along with those that are strictly dependent on human activities, such as farmlands and cities.

Effective biodiversity conservation, management and restoration rely on a thorough comprehension of ecosystems, which makes it necessary to describe, characterise, and spatially locate them (Dickson et al., 2014; Maes et al., 2020). The ELC can help delineate ecosystem typology and distribution through a hierarchical approach that defines homogeneous portions of territory, or environmental Land Units (LU), based on homogeneity of physical and biological features at various scales (Bailey et al., 2004). The outcome of the classification process should be easy to read but also informative, and a hierarchical setup may aid better understanding of mutual influence between components and contexts (Klijn & de Haes, 1994). Thus, hierarchically arranged LU provide a geographical framework that can be used to address several environmental issues according to ecological boundaries (Gallant et al., 2004; Omernik, 2004) and, more specifically, they represent a surrogate for the manifold aspects of biodiversity. In Italy, some examples of ELC are available that have been performed at the national (Blasi et al., 2010; Capotorti et al., 2012; Smiraglia et al., 2013; Blasi et al., 2014) and also at the local level (Vuerich et al., 2001; Blasi et al., 2005a; Casavecchia et al., 2007; Facioni et al., 2015), and the same occurred throughout Europe (Metzger et al., 2005; EEA, 2017) and worldwide (Sayre et al., 2009; Danby & Slocombe, 2005; Sanchez & Priego-Santander, 2011).

If, on the one hand, LU are determined by the ensemble of physical factors characterising a territory, the biological component emerging from the interaction with these environmental variables completes the biotope existing in that portion of land, representing the overall ecosystem (Munier et al., 2001; Qiu et al., 2010). However, European landscapes have been heavily transformed by anthropogenic activities, even in their surface and subsurface abiotic features (Pica et al., 2024; Burnelli et al., 2025), so that several biotopes could coexist in one single LU due to different land cover/land use types (agricultural, artificial and natural). A valid approach to classify LU is associating the physical factors with the corresponding Potential Natural Vegetation (PNV), that is, the mature stage of the vegetation series that could develop in a site under undisturbed conditions (Bioret et al., 2019; Konrad et al., 2022). PNV is a quite established concept, originally defined by Tüxen (1956) as 'the vegetation that would develop in a particular ecological zone or environment, assuming the conditions of flora and fauna to be natural, if the action of man on the vegetation mantle stopped and in the absence of substantial alteration in present climatic conditions', as well as a helpful tool which is gaining importance in recent ecological and vegetation science researches (Somodi et al., 2021; Bourdouxhe et al., 2023; Wang et al., 2023; Qu et al., 2024). This definition bound the distribution of PNV to a 'particular ecological zone', which can be seen as the LU where the combination between environmental factors is capable of sustaining a defined and specific mature vegetation type. Therefore, PNV-based landscape classifications promote the understanding of landscape patterns, showing areas with uniformity in the abiotic conditions that shape vegetation potential (Fischer et al., 2019). The PNV concept is often confused with the earlier concept of climax (of which it overcomes the problems associated with the necessary definition of a persistence time scale) or as pre-human vegetation, leading to biased criticism already faced and debated by precedent researches (Farris et al., 2010; Somodi et al., 2012; Fischer et al., 2013).

In the last few years, different works have explored the potential of these approaches for ecological restoration, considering local PNV, adjacent remnant ecosystems, or environmentally similar sites as models to recognise reference ecosystems (Durbecq et al., 2020; Toma et al., 2023; Nelson et al., 2024). However, notwithstanding the reference model is always expected to share similar environmental conditions with restoration sites, the concept of LU and hierarchical classification was not directly addressed, even though intrinsically respected. In many cases, PNV mapping relied on expert knowledge (Bohn et al., 2004; Vuerich et al., 2001; Blasi et al., 2005a), which clearly led to some subjective decisions. However, automatic estimates of PNV distribution are also being developed (Fischer et al., 2013 et 2019; Somodi et al., 2017), which could form the basis for more formalised PNV-based classification of land.

In environments profoundly transformed by anthropogenic activities, restoration actions based on PNV-based landscape classification are crucial. The analysis of environmental variables enables the definition of local vegetation potential despite broad and radical land transformations impacting physical factors and biological dynamics. Potential vegetation approaches represent a fundamental tool for selecting species suited for these environments in ecosystem restoration actions or as a guide for nature-oriented agricultural and silvicultural management (Keersmaeker et al., 2013; Capotorti et al., 2019; Somodi et al., 2024; Valeri et al., 2024).

Recent studies on PNV, species distribution, and habitat mapping have predominantly focused on large-scale analyses, utilising existing vegetation databases, remote-sensing data, and machine learning techniques with broad environmental layers (Hengl et al., 2018; Konrad et al., 2022; Bourdouxhe et al., 2023). While these approaches provide valuable insights into regional and global vegetation patterns, they often overlook hierarchical site stratification, limiting their applicability for fine-scale restoration planning. Thus, high-resolution PNV maps at the local scale are paramount as a reference for land management and development plans (Blasi et al., 2008; Facioni et al., 2015; Aronson et al., 2017; Sturbois et al., 2023).

Particularly in urban and peri-urban landscapes, effective ecosystem restoration thus necessitates a robust and scalable methodology for identifying appropriate reference ecosystems. As highlighted, PNV, along with hierarchical classification of land, offers a comprehensive framework for guiding such restoration efforts, together with other tools potentially available in a territory, such as local floristic databases (Muratet et al., 2008; de Francesco et al., 2025). The challenge lies in developing a repeatable process, whereby automatically produce or upgrade PNV maps, in different areas and at different scales, readapting the classification process to the characteristics of the territory also based on available data. Accordingly, the main goal of this work is to develop a repeatable method for generating high-resolution PNV maps under a hierarchical and semi-automated process. The method, tested in the Functional Urban Area (FUA) of Campobasso (southern Italy), is expected to overcome the limits of available broad-scale analyses by providing more detailed and site-specific information.

2. Material and Methods

2.1 Study Area

The Functional Urban Area (FUA) of Campobasso (Figure 1) was selected as a case study for testing the land classification process. According to the EU-OECD definition, FUA comprise cities and their commuting zones and therefore consist of a densely inhabited center (a core area) and a less densely populated surrounding zone, whose labour market is highly integrated with the city (Dijkstra et al., 2019). Campobasso is located in Molise Region (southern Italy), between the Apennine Range westward and the Adriatic Sea eastward, and is representative of medium-sized historical urban systems, with relatively contained artificial surfaces embedded into an agricultural matrix with substantial persistence of forest cover (Varricchione et al., 2024). Within the Apennine ecoregional province (Blasi et al., 2014 et 2018), it falls under temperate sub-continental climatic conditions, at the limit with the Mediterranean zone, characterised by an average annual temperature of 13.3°C and an average annual precipitation of 806 mm (Pesaresi et al., 2017; Martinelli and Matzarakis, 2017). The elevation range is between 174 and 1078 m a.s.l., with a predominantly hilly and mountainous physiographic context characterised by sedimentary formations, and a prevalence of arenaceous-sandy and sandy clay outcrops, alongside extensive carbonate, conglomeratic, and marly substrata (Bucci et al., 2022). The highest elevations are found westward, as a part of the Apennines Range situated further inland on the Italian peninsula. The elevations gradually descend eastward, following the drainage directions.

The heterogeneous ecological conditions allow the establishment of different forest PNV types, which is interesting from an ecological modelling perspective. In comparison, actual forest cover represents a quarter of the territory (24.7 %), principally represented by *Quercus cerris*, *Q. frainetto*, *Q. pubescens*, and *Populus alba* forests (D'Angeli et al., 2024; Varricchione et al., 2024). The FUA also includes (fully or partially) 29 Special Areas of Importance (SAC) of the European Natura2000 network (EC, 2022; MASE, 2024).

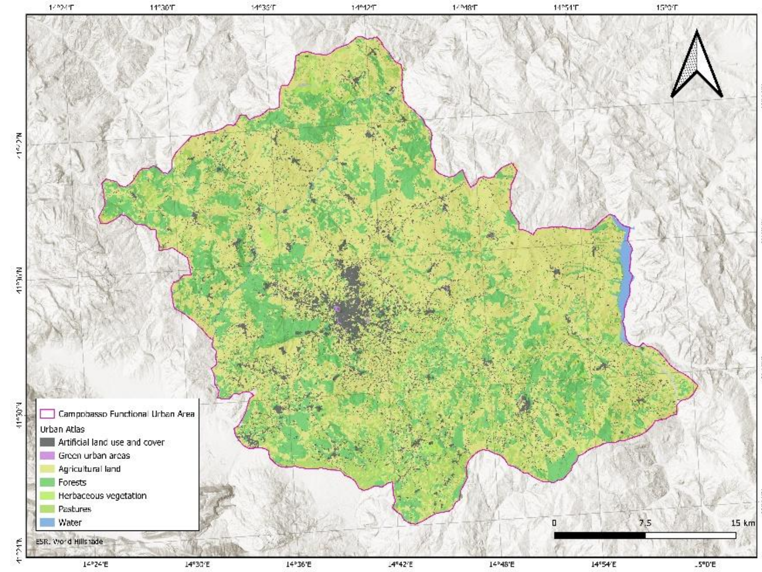

Figure 1: current land cover of the study area (retrieved from urban atlas, eea, 2018). Coordinate system is wgs 84 (epsg 4326)

2.2 Methodological framework

The methodological process followed a multi-step approach, as shown in Figure 2. With a first phase of ecological classification of land, the study area was hierarchically subdivided according to the process proposed by Blasi et al. (2005a), which identifies at progressively finer geographic scales, land regions, land systems and land subsystems based on bioclimate, lithology, and morphology, respectively. Second, a sampling design was set for the investigation of mature vegetation stages across the different land subsystems and FUA sectors by means of a stratified random selection of forest patches to be surveyed. Vegetation surveys were conducted in each selected patch by recording all occurring species and respective cover/abundance according to the phytosociological method (Braun-Blanquet, 1932). Sampled data were analysed using clustering, dimensionality reduction techniques and indicator species analysis to identify and characterise mature forest communities. Third, different statistical tests were used to determine which environmental variables were significantly associated with the recognised vegetation types, thus acting as indicators for plant communities distribution. Dimensionality reduction was used to visualise the identified clusters, as well as the considered environmental variables, onto common components obtained as linear combinations of the latter. Finally, two alternative PNV maps were derived from modelling techniques, thus validated and synthesised into a final cartographic outcome. Basic data used throughout the process are listed in the following sections and more in-depth described in the Supplementary Material (Section 1). Mapping and statistical analyses were performed using RStudio software (version 4.4.2), QGIS 3.34.4, SAGA GIS 9.3.1 and MATLAB-based platform Topotoolbox (R Core Team, 2024; Schwanghart & Scherler, 2014).

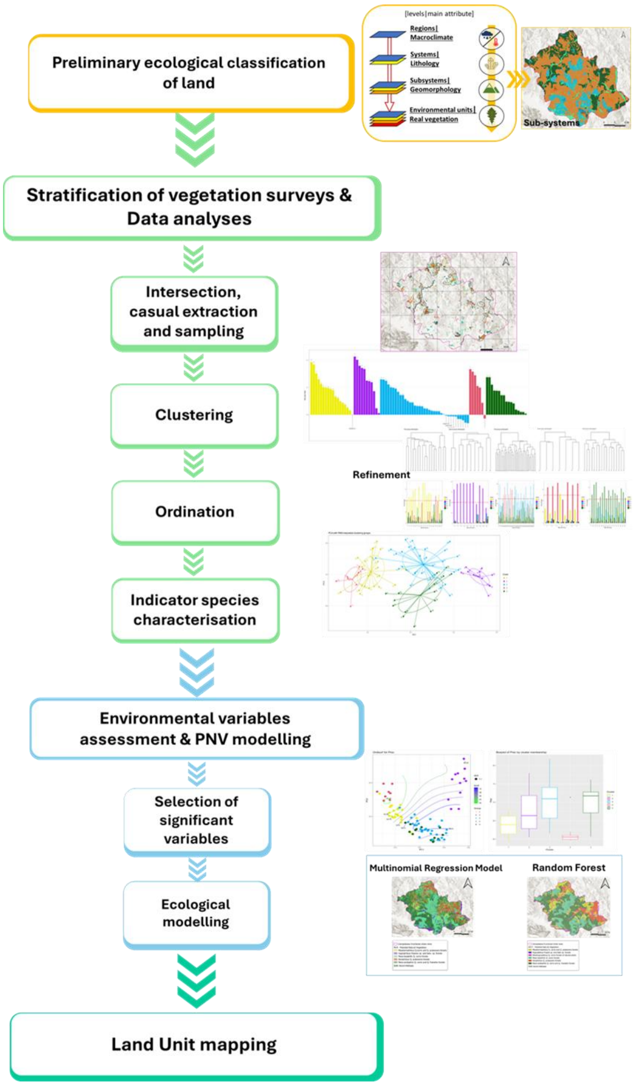

Figure 2: Flowchart shows the methodological approach used for the land unit mapping

2.2.1 Preliminary ecological land classification

A hierarchical ELC that is consistent with the approach proposed for Italian landscapes (Blasi et al., 2000), was first carried out up to the subsystem level to ensure an exhaustive sampling of forest types across the study area. At a coarse level, Land Regions were defined considering macroclimate features retrieved from the Phytoclimatic map of Italy (Smiraglia et al., 2013). Within Land Regions, Land Systems were thus defined according to a lithological thematic layer obtained by collecting, homogenising and synthesising geological and geolithological maps at both the national and regional scales. Namely, the digital national lithological map (Bucci et al., 2022) was reclassified in terms of lithological properties by considering the information and documentation from the geological sheets of the National Geological Service (scale 1:100.000) and from the Geolithological map of Italy (scale 1:500 000) (ISPRA, 2009). Within Land Systems, Land Subsystems were finally defined by reclassifying Geomorphons landforms, a technique using a machine vision approach (Jasiewicz & Stepinski, 2013). The method identifies terrain features such as pits, flats, footslopes, hollows, ridges, shoulders, slopes, spurs, summits, and valleys. From these classes, three main morphological features were considered, i.e.valley-fluvial domain landforms, sub-flat/mild slope, and steep slope (> 30-degree slope), and further refined according to important altitudinal thresholds (as for the limit between hilly and sub-mountain vegetation belts; Blasi et al., 2005b; Taffetani et al., 2012) and hydrographic features (as for the valley-fluvial domain; ISPRA, 2008).

2.2.2 Vegetation surveys and analyses

For ensuring a sound representativity of field surveys, a sampling design was set for the investigation of mature vegetation stages across the different sectors of the FUA and occurring land subsystems. First, actual occurrence of forests was retrieved from the pertinent regional map (Chirici, 2010). The study area was thus divided into 16 regular grid cells of 10 km x 10 km and forest patches to be sampled randomly extracted by subsystems in each cell. A sampling priority was suddenly assigned to patches with a wide extent, to avoid edge effects and better guaranteeing the occurrence of a core habitat, and occurring into SAC, since forests in Natura2000 sites are expected be in better conservation status and more sustainably managed than those occurring outside the network (Regione Molise, 2017; EU, 1992).

A phytosociological relevé was conducted at each selected polygon according to the approach of Braun-Blanquet (Braun-Blanquet, 1932; Dengler, 2016). Plant species were recognised following the Flora of Italy (Pignatti, 2017) and respective nomenclature was standardised according to Bartolucci et al. (2024) and Galasso et al. (2024). The resulting vegetation dataset was transformed based on the method proposed by Van der Maarel (1979) scale for subsequent statistical analyses, which included:

- Clustering of the surveys: a Partitioning Around Medoids (PAM) non-hierarchical clustering was applied to a Hellinger matrix distance partitioning subsets of surveys to identify plant communities characterised by similar species composition (Kaufman & Roesseeuw, 1987; Borcard et al., 2011). The optimal number of clusters was obtained by maximising the average silhouette value of PAM clustering. To improve the ecological interpretation of the communities, some sites have been reassigned to the relative neighbour groups denoted by PAM itself based on their site-specific silhouette values. To further refine the partitioning, other clustering techniques were considered. Fuzzy memberships of a fuzzy k-medoid applied considering PAM medoids and the inside-group hierarchical structures of a hierarchical clustering applied to the sites served as tools to support the assignment of surveys (Ward, 1963; Dunn, 1973; Murtagh & Contreras, 2011; Ruspini et al., 2019).
- Dimensionality reduction: Principal Component Analysis (PCA) was employed to clearly visualise the distribution of the surveys built upon species data in a two-dimensional space, also projecting relevant species (Pearson, 1901; Greenacre et al., 2022).
- Indicator Species Analysis: IndVal was performed to identify indicator species of each cluster and ecologically characterise plant communities along with the physiognomic dominant trees (Dufrêne & Legendre, 1997; De Cáceres et al., 2010).

2.2.3 PNV ecological modelling and mapping

To reach the finest level of LU classification, an additional and comprehensive set of environmental variables was considered (Table 1) that are more detailed than those previously used to define subsystems and allow the ecological niche for mature vegetation communities to be more precisely reconstructed. The influence of different environmental variables (Table 1) on mature forests was assessed using Analysis Of Variance (ANOVA), Kruskal-Wallis test and Chi-square test (Pearson, 1900; Fisher, 1922; Kruskal & Wallis, 1952; Tabachnick, 2007; McKight & Najab, 2010; Pandis, 2016; Wuensch, 2011) and graphically visualised through PCA. Significant variables resulting from the aforementioned tests were then used to model the distribution of PNV types by means of two different tools. The first model was built according to a Random Forest Classification Tree (RFCT) algorithm, used to identify threshold values of environmental variables able to distinguish vegetation groups, define the ecological niche of each forest type, and generate a predictive map (Breiman, 2001). The second model was built according to the coefficients of a Multinomial Logistic Regression

(MLR), used to estimate the effect of each environmental variable on the definition of vegetation groups, the membership of each sample to each group, and the potential distribution of the different forest types. For MLR, Variance Inflation Factor (VIF) and pseudo-R^2 were considered for the best model selection (Theil, 1969; Akinwande, 2015). The resulting two outputs were compared, validated, and combined to produce a final PNV map as robust as possible.

Table 1: Environmental variables considered for the ecological modelling of PNV distribution

Environmental variable	Description	Source
Lithology	Lithological types classified into five classes. Qualitative data [alluvial deposits, sandy clay, limestone-marly, sandy-arenaceous, varicoloured clay]	Digital lithological map of Italy (Bucci et al., 2022)
Elevation	Altitude retrieved from the digital elevation model (DEM) with a grid cell of 5 meters. Quantitative data [m a.s.l.]	Regional technical map (RTM) of Molise Region 1:5000
Mean Temperature	Mean monthly temperature for the period 2011-2020 downscaled at a resolution of 5m x 5m. Quantitative data [°C]	Mean monthly temperature mapped at a resolution of 1km x 1 km (ISPRA)
Mean Precipitation	Mean monthly precipitation for the period 2011-2020 downscaled at a resolution of 5m x 5m. Quantitative data [mm]	Mean monthly precipitation mapped at a resolution of 1km x 1 km (ISPRA)
Aspect	DEM-derived classes that serve as a proxy for seasonal wetting and drying cycles of soils (Auslander et al., 2003). Qualitative aspect, [N,E,W,S,NW,NE,SW,SE]	RTM of Molise 1:5000
Heatload	DEM-derived data on aspect and slope for quantifying insolation (McCune & Keon, 2002). Quantitative data [dimensionless]	RTM of Molise 1:5000
Solar radiation aspect index (Solarad)	DEM-derived data on aspect for quantifying insolation (Roberts & Cooper, 1989). Quantitative data [dimensionless]	RTM of Molise 1:5000
Slope	DEM-derived data measuring the steepness of terrain. Quantitative data [degrees]	RTM of Molise 1:5000
Slope Area Domains	DEM-derived geomorphological reclassification of the landscape into process domains, based on the widely recognised functional relationship between drainage area (A, in m^2) and slope (S, in m m^{-1}). The relationship enables the delineation of local geomorphic domains and supports the	RTM of Molise 1:5000

	interpretation of process interactions (Delchiaro et al., 2023, 2025; Iacobucci et al., 2024; Valiante et al., 2024; Vergari et al., 2019; Wang, 2024). Qualitative data [I,II,III,IV,V]	
SAGA Wetness Index (TWI)	DEM-derived TWI, representing potential infiltration capacity and used as a proxy for estimating the presence and thickness of saturated soil horizons (e.g., Martinello et al., 2024). Quantitative data [dimensionless]	CTR of Molise 1:5000
Flow accumulation	DEM-derived flow accumulation, quantified as the number of pixels that flow into every pixel of the DEM. Quantitative data [sqm]	RTM of Molise 1:5000
Tangential curvature	DEM-derived tangential curvature that describes the horizontal slope geometry and serves as a proxy for shallow gravitational stress convergence/divergence and surface water flow patterns. Quantitative data [dimensionless]	RTM of Molise 1:5000
Profile curvature	DEM-derived profile curvature that refers to the vertical slope profile and serves as a proxy for shallow gravitational stress convergence/divergence and surface water flow patterns. Quantitative data [dimensionless]	RTM of Molise 1:5000
Height above nearest drainage (HAND) or vertical distance to channel network	DEM-derived parameter that quantifies the vertical distance of each DEM cell above the nearest stream cell along flow paths. Quantitative data [m]	RTM of Molise 1:5000
Horizontal distance to channel network	DEM-derived parameter that quantifies the horizontal distance of each DEM cell to the stream network along flow paths. Quantitative data [m]	RTM of Molise 1:5000

3. Results

3.1 Land subsystems map of the FUA of Campobasso

The preliminary ELC resulted into 2 Land Regions, 8 Land Systems, and 20 Land Subsystems (Figure 3). The main part of the territory (88 000 ha, 85%) falls into the Temperate Region, which includes the most diffused land systems, i.e. the Sandy-arenaceous, conglomeratic and arenaceous-marly (42% of the territory, 42 000 ha), the Limestone-marly (16%, 16 000 ha), the Clayey (15%, 15 000 ha) and the Alluvial deposits one (11%, 11 000 ha). In the Mediterranean Region, the Sandy arenaceous Land System still prevails (7.3%, 7500 ha), followed by the Clayey (3.3%, 3400 ha), the Alluvial deposits (3.2%, 3300 ha) and the Limestone-marly ones (0.35%, 360 ha).

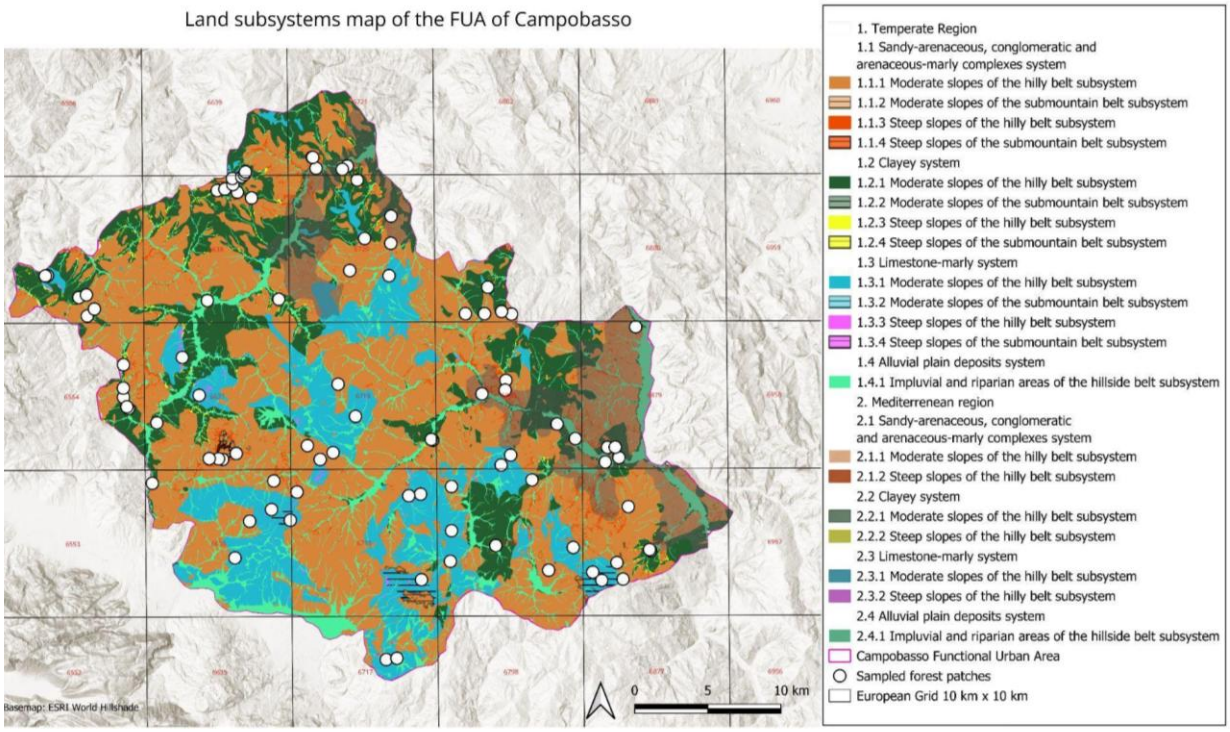

Figure 3: Land Subsystem map of the FUA of Campobasso, represented along with the European Grid 10 km x 10 km. White dots show the location of forest surveys.

3.2 Mature forest communities

A total of 90 forest vegetation surveys, including 245 species of vascular plants, were performed all over the 16 10 km x 10 km cells and 20 Land Subsystems therein, with few exceptions due to accessibility constraints. PAM clustering assigned the 90 observations to a set of five medoids, thus defining five clusters (optimal number according to silhouette values, Figure 4A). The projection of these clusters onto two components identified by the PCA, built upon species data, is shown in Figure 4B (see also Supplementary material, section 2), while the results of Species Indicator Analysis are shown in Table 2.

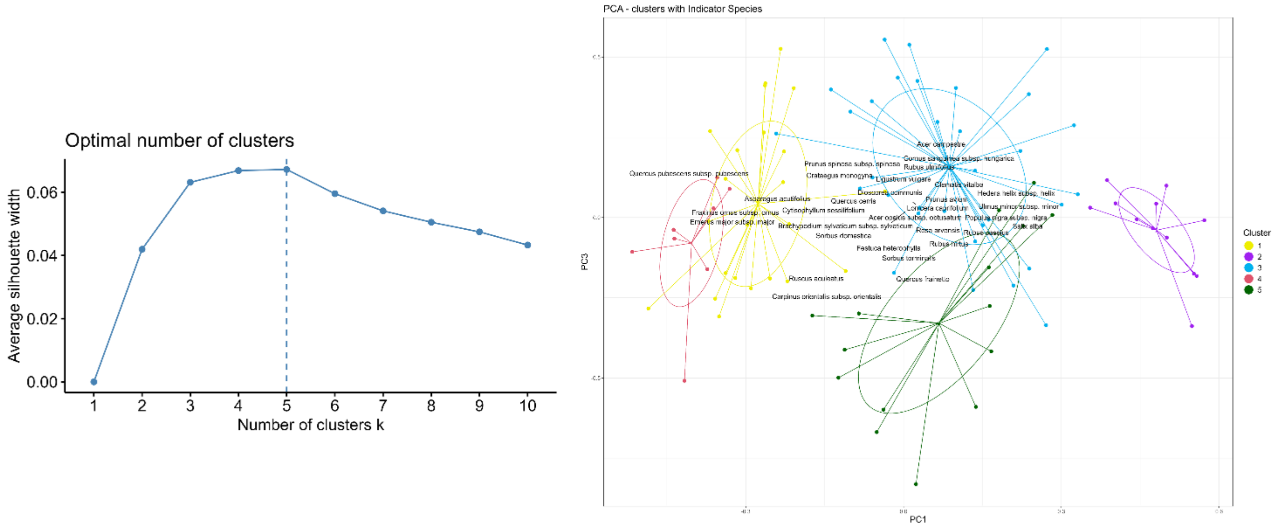

Figure 4: Panel A, Optimal number of clusters and Panel B, PCA with cluster classification and projected species with score >0.08 on PC1 and PC3, scaling 2. Clusters: 1 (*Quercus cerris* and *Q. pubescens* forests), 2 (*Populus* sp. pl. and *Salix* sp. pl. forests), 3 (*Q. cerris* forests), 4 (*Q. pubescens* forests), 5 (*Q. cerris* and *Q. frainetto* forests).

Table 2: Indicator species for each cluster obtained by the IndVal analysis.

	Cluster	IndVal	P-value
Emerus major subsp. *major*	1	0.34	0.00
Cytisophyllum sessilifolium	1	0.31	0.00
Fraxinus ornus subsp. *ornus*	1	0.30	0.02
Ruscus aculeatus	1	0.27	0.04
Pseudoturritis turrita	1	0.22	0.01
Lonicera etrusca	1	0.18	0.06*
Salix alba	2	0.92	0.00
Populus nigra subsp. *nigra*	2	0.75	0.00
Rubus caesius	2	0.75	0.00
Populus alba	2	0.57	0.00
Carex pendula	2	0.50	0.00
Equisetum telmateia	2	0.50	0.00
Urtica dioica	2	0.50	0.00
Ulmus minor subsp. *minor*	2	0.40	0.00
Eupatorium cannabinum subsp. *cannabinum*	2	0.33	0.00
Arundo plinii	2	0.31	0.00
Corylus avellana	2	0.30	0.00
Humulus lupulus	2	0.25	0.00
Petasites hybridus subsp. *hybridus*	2	0.25	0.00
Chaerophyllum temulum	2	0.25	0.01
Ranunculus lanuginosus	2	0.23	0.00
Sambucus nigra	2	0.20	0.01
Euphorbia amygdaloides	2	0.18	0.02
Amorpha fruticosa	2	0.17	0.02
Salix caprea	2	0.17	0.02
Silybum marianum	2	0.17	0.03
Stachys sylvatica	2	0.17	0.03

Tamarix africana	2	0.17	0.02
Juglans regia	2	0.14	0.04
Orobanche hederae	2	0.13	0.06*
Daphne laureola	3	0.37	0.00
Acer campestre	3	0.37	0.00
Quercus cerris	3	0.31	0.00
Hedera helix subsp. *helix*	3	0.30	0.01
Acer opalus subsp. *obtusatum*	3	0.24	0.03
Ligustrum vulgare	3	0.30	0.01
Quercus pubescens subsp. *pubescens*	4	0.47	0.00
Clematis flammula	4	0.32	0.00
Cistus salviifolius	4	0.25	0.01
Brachypodium rupestre	4	0.68	0.00
Pistacia lentiscus	4	0.50	0.00
Teucrium chamaedrys subsp. *chamaedrys*	4	0.44	0.00
Silene italica subsp. *italica*	4	0.43	0.00
Asparagus acutifolius	4	0.41	0.00
Spartium junceum	4	0.39	0.00
Astragalus monspessulanus subsp. *monspessulanus*	4	0.25	0.01
Melica ciliata	4	0.25	0.01
Rubia peregrina	4	0.24	0.02
Phleum pratense	4	0.21	0.01
Muscari comosum	4	0.18	0.03
Geranium purpureum	4	0.16	0.06*
Quercus frainetto	5	0.83	0.00
Luzula forsteri	5	0.51	0.00
Echinops siculus	5	0.17	0.05
Genista tinctoria	5	0.38	0.00
Cytisus villosus	5	0.33	0.00
Rosa arvensis	5	0.41	0.00
Carpinus orientalis subsp. *orientalis*	5	0.32	0.01
Lathyrus niger	5	0.29	0.00
Sorbus domestica	5	0.29	0.02
Sorbus torminalis	5	0.26	0.02
Rubus hirtus group	5	0.26	0.02
Potentilla reptans	5	0.25	0.00
Prunus avium	5	0.24	0.03
Dactylis glomerata subsp. *glomerata*	5	0.24	0.03
Lathyrus venetus	5	0.22	0.03
Cyclamen repandum subsp. *repandum*	5	0.20	0.01
Cruciata laevipes	5	0.19	0.04
Betonica officinalis	5	0.13	0.05

** species with p-value approximated to excess at 0.06.*

The indicator species, along with the physiognomic dominant trees, allowed the five clusters to be confirmed as meaningful forest types and ecologically characterised as follows:

- Forest type 1: *Quercus cerris* and *Q. pubescens* forests characterised by *Emerus major* subsp. *major*, *Fraxinus ornus* subsp. *ornus, Cytisophyllum sessilifolium* and *Lonicera etrusca,* typical of meso-thermophilous conditions principally located over calcareous, or at least at basic reaction, substrata;
- Forest type 2: *Populus* sp. pl. and *Salix* sp. pl. forests characterised by *Carex pendula*, *Petasites hybridus* subsp. *hybridus, Rubus caesius, Eupatorium cannabinum* subsp. *cannabinum* and *Humulus lupulus,* typical of meso-hygrophilous conditions in valleys and alluvial plains;
- Forest type 3: *Quercus cerris* forests characterised by *Daphne laureola*, *Acer opalus* subsp. *obtusatum* and *Acer campestre,* typical of mesophilous conditions at high elevation (over 700 m a.s.l.) over calcareous substrata;
- Forest type 4: *Quercus pubescens* forests, characterised by *Clematis flammula*, *Pistacia lentiscus*, *Rubia peregrina* and *Brachypodium rupestre*, typical of thermophilous condition given by low elevation (under 500 m a.s.l.) on south facing slope;
- Forest type 5: *Quercus cerris* and *Q. frainetto* forests characterised by *Echinops siculus*, *Cytisus villosus*, *Genista tinctoria* and *Sorbus torminalis,* typical of mesophilous conditions (between 500 and 900 m a.s.l.) and acidophilous sandy-areneaceous substrata.

3.3 Significant environmental variables and PNV modelling

Among the overall considered environmental variables (Table 1), those assessed as statistically significant (Table 3) were retained for the subsequent PNV modelling steps (complete results are available in the Supplementary material, section 3).

Table 3: Significant variables selected for PNV modelling

Variable	Test	P-value
Precipitation	Kruskal-Wallis	5.34 E-06
Temperature	Kruskal-Wallis	8.35 E-08
Elevation	ANOVA	2.77 E-11
Topographic Wetness Index	Kruskal-Wallis	1.33E-06
Slope	Kruskal-Wallis	1.53 E-05
Heatload	Kruskal-Wallis	0.04 E-00
Vertical distance from the channel network	Kruskal-Wallis	2.01 E-06
Horizontal distance from the channel network	Kruskal-Wallis	4.52 E-07
Slope Area Domains	Chi-squared	0.01 E-01

Lithology	Chi-squared	0.05 E-02

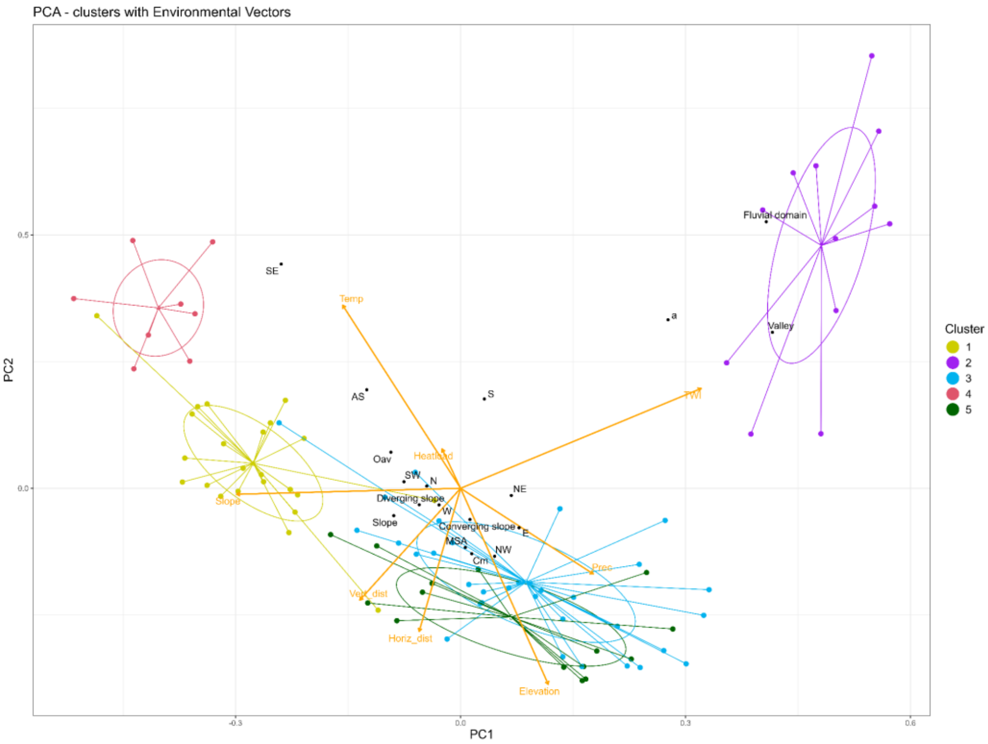

Figure 5: PCA with environmental variables; vectors for quantitative variables with envfit p-value<0.05, and points for qualitative variable classes. Lithological classes: a (alluvial deposits), AS (sandy clay), Cm (Carbonate and marly), MSA (sandy-arenaceous), Oav (varicoloured Clay). Aspect classes: N (North), E (East), W (west), S (south), SE (South-east), SW (South-west), NE (North-east), NW (North-west). Clusters: 1 (*Quercus cerris* and *Q. pubescens* forests), 2 (*Populus* sp. pl. and *Salix* sp. pl. forests), 3 (*Q. cerris* forests), 4 (*Q. pubescens* forests), 5 (*Q. cerris and Q. frainetto* forests).

The PCA revealed that the five forest types are indeed differently related to the significant environmental variables. *Q. cerris* and *Q. pubescens* forests and *Q. pubescens* forests (types 1 and 4) are related to higher temperature and lower precipitation values, steeper and diverging slopes, higher heat load values, warmer aspects, and mainly occur on calcareous, clayey and sandy-clayey lithologies (Figure 5; Supplementary material, section 2). By contrast, *Q. cerris* forests and *Q. cerris* and *Q. frainetto* forests (types 3 and 5) are both associated with higher elevation and precipitation values and lower temperatures, with the former prevailing on marly-carbonate substrates and the latter on sandy-arenaceous substrata. Lastly, *Populus* sp. pl. and *Salix* sp. pl. forests (type 2) are associated with valleys and fluvial domains, characterised by high values of TWI nearby the channel network in both vertical and horizontal dimension (Supplementary material, section 2).

Based on the selected environmental variables, MLR and RFCT were employed to model the PNV distribution, with the second algorithm run considering 9 variables at each node and 5 minimum samples per node. The estimated MLR coefficients and the RFCT used to generate the two PNV maps (Figure 7) are reported in Table 4 and Figure 6, respectively.

Table 4: Multinomial Logistic regression used to model the PNV distribution. ^{NS} stands for non-significant coefficient. ^{NA} missing P-value

	Cluster 1-3	Cluster 2-3	Cluster 4-3	Cluster 5-3

(Intercept)	397.97	150.59	-147.56	-74.60
Slope	-1.86	0.79^{NS}	1.69	-0.42
Temperature	23.35	-36.52	101.10	8.88
Topographic Wetness Index	-78.58	45.56	-101.13	-37.42
Precipitation	-6.35	0.43^{NS}	-17.44	3.25
Slope area: Diverging slope	120.13	123.56	-35.52	-87.19
Slope area: Slope	121.15	104.08	-191.88	-54.19
Slope area: Converging slope	218.02	76.55	-33.46	-19.68
Slope area: Valley	-140.08	88.10	2.80	48.47
Lithology AS	70.33	86.89^{NA}	242.90	-259.84
Lithology Cm	180.17	-80.74	105.43	-204.98
Lithology MSA	139.34	76.34	309.58	252.77
Lithology Oav	146.85	-113.21	408.70	209.13
Vertical distance	0.50	-5.03	2.99	0.06^{NS}
Horizontal distance	-0.12	0.23^{NS}	-0.50	0.01^{NS}
Heatload	-26.51	-106.84	227.17	-269.76
Residual deviance:	20.10			
AIC:	148.10			
Pseudo R-square	0.87			

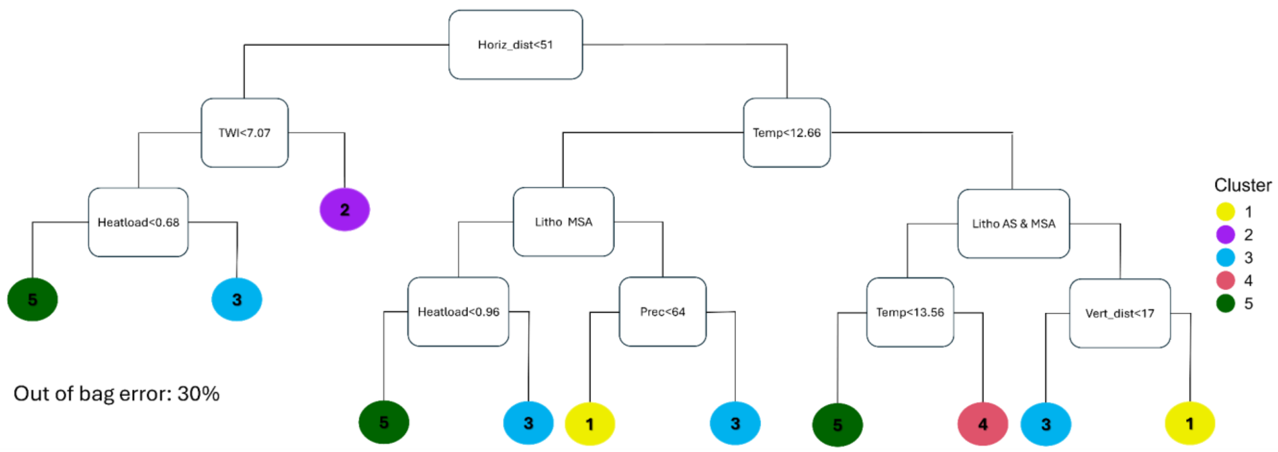

Figure 6: Random forest classification tree used to model the PNV distribution. Horiz_dist (Horizontal distance from channel network), TWI (Topographic wetness index), Litho MSA & AS (Lithology Sandy-arenaceous and Sandy-clay), Heatload, Temp (Temperature), Prec (Precipitation). Clusters: 1 (*Quercus cerris* and *Q. pubescens* forests), 2 (*Populus* sp. pl. and *Salix* sp. pl. forests), 3 (*Q. cerris* forests), 4 (*Q. pubescens* forests), 5 (*Q. cerris* and *Q. frainetto* forests).

Both predictive maps show a clear pattern along the climatic gradient. The two thermophilous *Quercus pubescens* PNV types (types 1 and 4) which distinguish each other by aspect and slope factors, distribute at low altitude and trace the Mediterranean wedge entering the valleys (Figure 7), while the rest of the territory is vocated for the two mesophilous *Quercus cerris* PNV types (types 3 and 5), with a disjunct distribution determined by lithological features, and for the hygrophilous *Populus* sp. pl. and *Salix* sp. pl. PNV type (type 2) in wetter sites nearby the channel network (Figure 7). Main differences between the maps rely instead on the resolution in representing these general behaviours. The MLR derived map shows a fine-grained differential distribution of PNV types, based on detailed aspect and topographic position, whilst the RFCT derived map gave more importance to climatic and lithological variables identifying homogeneous units with a more marked geographic meaning but approximate spatial scope for distinctive PNV types. Nevertheless, the RFCT allowed PNV variants to be detected and mapped, as in the case of *Quercus cerris* forests (type 3) that, when either occurring nearby the hydrographic network (Horiz_dist < 51) but with contained soil moisture (TWI<7.07) or beyond the horizontal limit but still below the 17 meters of vertical distance from the channel network (Vert_dist <17), become enriched in meso-hygrophilous species, such as *Corylus avellana*, *Fraxinus angustifolia* subsp. *oxycarpa*, *Ulmus minor* subsp. *minor* and *Carpinus betulus.*

By combining the models, a final comprehensive LU map was derived leveraging the different strengths of the two PNV maps. It spatially defines:

- the potential sites for *Quercus cerris* and *Q. pubescens* meso-thermophilous forests (type 1) and those for *Q. pubescens* thermophilous forests (type 4), merged into a unique LU to avoid the excessive fragmentation provided by the MLR and the alternative under- and over-estimation provided by the RFCT. This "Land Unit with *Quercus pubescens* subsp. *pubescens, Fraxinus ornus* subsp. *ornus, Emerus major, Pistacia lentiscus, Lonicera etrusca* and *Rubia peregrina* termophilous forest PNV" is mainly located between 200 and 500 m a.s.l. in the Mediterranean Region;
- the "Land Unit with *Populus* sp. pl. and *Salix* sp. pl. riparian forest PNV", obtained by retaining the spatialisation from RFCT (that follows both the thresholds of TWI>7.07 and horizontal distance from the channel network<51 meters), but refined according to the MLR (that limits this type to sites with an horizontal distance from the channel network<51 meters) for enhancing geographic contiguity and precision of representation of the respective PNV (type 2) ;
- the "Land unit with *Quercus cerris*, *Acer opalus* subsp. *obtusatum* and *Daphne laureola* mesophilous forest PNV", obtained by retaining the spatialisation of the respective mature forest type (type 3) from RFCT. Within this unit, the ancillary occurrence of meso-thermophilous forests of *Q. cerris* and *Q. pubescens* (type 1) on steep slopes and warm expositions, represented by the MLR, was considered as an internal thermophilous-xeroedaphic variant;
- the "Land unit with *Quercus cerris, Quercus robur* subsp. *robur, Corylus avellana, Fraxinus angustifolia* subsp. *oxycarpa, Ulmus minor* subsp. *minor* and *Carpinus betulus* meso-hygrophilous forest PNV", detected as a distinct sub-type from the previous one by the RFCT (either for TWI<7.07 and horizontal distance from the channel network<51 meters or horizontal distance from the channel network>51 meters and vertical distance<17 meters). The boundaries suggested by the RFCT were however refined according to MLR (type 2, but just constrained by an horizontal distance from the channel network>51 meters), allowing the colluvial environments and alluvial plains that make up this unit to be better delimited;
- the "Land unit with *Quercus cerris, Q. frainetto, Cytisus villosus, Genista tinctoria* and *Echinops siculus* mesophilous forest PNV", obtained by retaining the spatialisation of the respective mature forest type (type 5) from RFCT. Within this unit, the ancillary occurrence of meso-thermophilous forests of *Q. cerris* and *Q. pubescens* (type 1) on steep slopes (>25

degrees), represented by the MLR, was considered as an internal thermophilous-xeroedaphic variant.

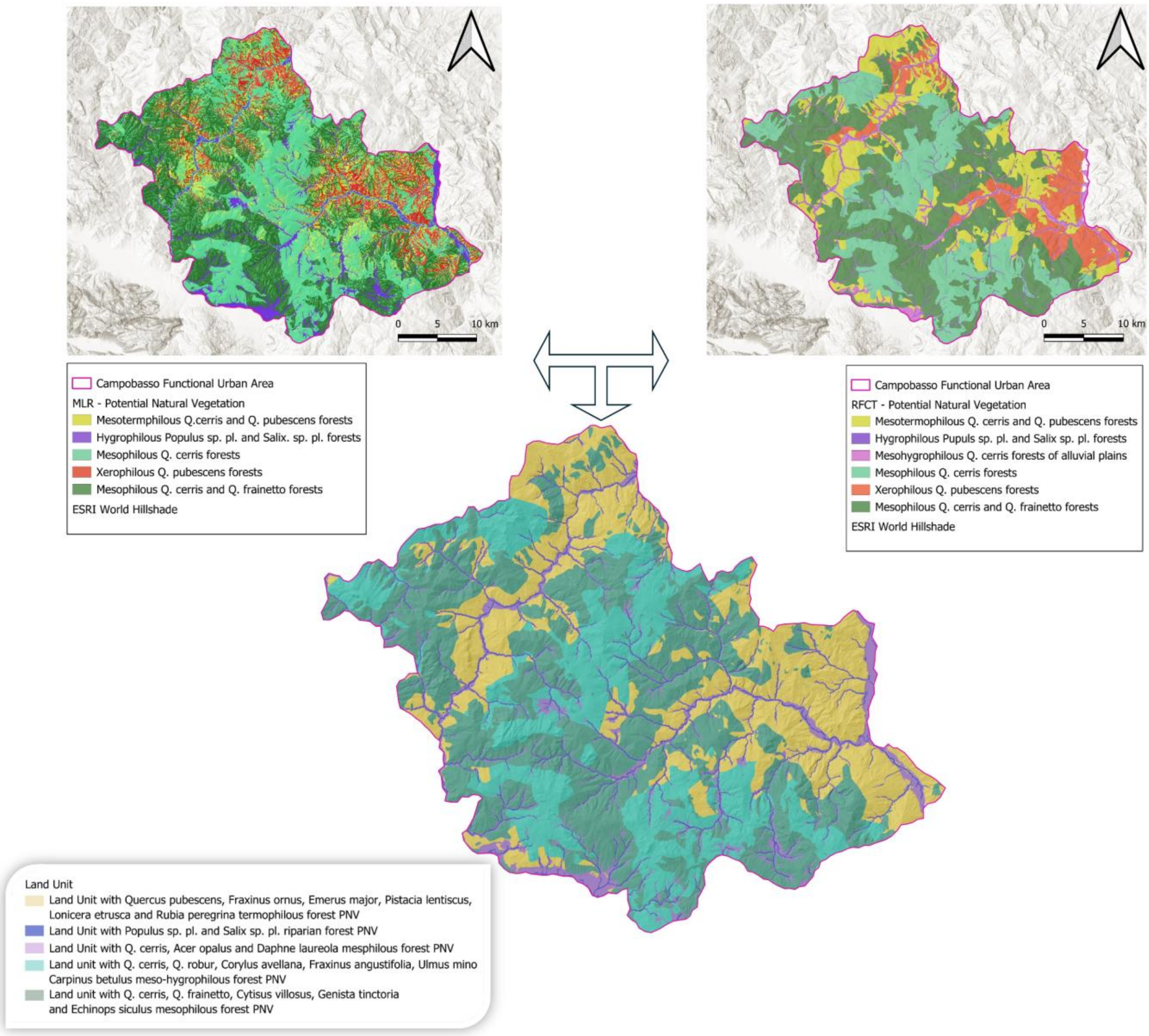

Figure 7: Potential Natural Vegetation maps; the first two are obtained by modelling with MLR and RF. The last is the LU map obtained thanks to the synthesis of the two maps, reassuming the most valuable information for potential natural vegetation.

4. Discussion

The method herein proposed for fine-scale ELC and PNV mapping resulted to be robust, since it is based on models showing a high confidence level in describing mature forest distribution. The outputs derived for the test case study, in the FUA of Campobasso, provided in turn a useful tool for ecosystem monitoring, management, and restoration in a medium-size urban system. Recent examples of similar applications actually include PNV guided local-scale restoration of industrial sites (Török et al., 2018), but also a nation-wide programme for urban and periurban reforestation in metropolitan cities (MASE, 2021). Given the usefulness and potential of these products, it is therefore necessary to incorporate high-resolution maps and gather precise ecological information across various types of urban ecosystems to address the growing need for effective restoration actions (EC, 1991/2024). The adopted process is very flexible and repeatable in other urban or peri-urban areas, with environmental or structural conditions that may differ from those in the present case study, owing to the possibility of adapting all the phases, from the design of sampling campaign to the choice and selection of environmental variables, calibration of models and final ELC, to peculiar characters of the target area (Klijn & de Haes, 1994; Guisan & Zimmerman, 2000; Cadenasso et al., 2007).

The stratified random sampling of different forest types, across coarse land units and geographic sectors, is actually able to ensure an ecologically and spatially representative investigation of mature vegetation communities. This approach has been adopted to balance the need for a representative sample with the practical limitations of field research, including time constraints and accessibility—a common challenge in ecological fieldwork (De Cáceres et al., 2015; Roleček et al., 2007). In particular, the stratification according to land subsystems ensured that the most relevant aspects of the relationship between plant communities and physical environmental features were captured, a particularly important aspect in ecological modelling studies (Elith & Leathwick, 2009; Attorre et al., 2014). Thus, subsequent analyses of vegetation relevés (with PAM, ISA and PCA) confirmed that sampled communities share similar environmental conditions and composition with mature stages associations reported in the territory (Paura & Abbate, 1995), a relationship also testified by the correspondence between forest type indicator species and diagnostic species of corresponding phytosociological associations/alliances. Namely: i) *Quercus cerris* and *Q. pubescens* meso-thermophilous forests can be referred to the association *Lonicero etruscae-Quercetum cerridis* subass. *festucetosum exaltatae* (suballiance *Lauro nobilis-Quercenion pubescentis*; alliance *Carpinion orientalis*) reported for the territory as *Lonicero xilostei-Quercetum cerridis* (Taffetani et al., 2012) and subsequently proposed and validated with this new name (Terzi et al., 2020, 2021); ii) hygrophilous forests can be referred to the alliances *Populion albae* and *Salicion albae* which characterise the riparian geosigmetum across the Italian peninsula (Blasi, 2010); iii) mesophilous *Q. cerris* forests are mainly referable to the *Daphno laureolae-Quercetum cerridis* subass. *rosetosum arvensis* as described for the territory in Taffetani et al., 2012, although in wetter conditions, species related to the suballiance *Pulmonario apenninae-Carpinenion betuli* may appear (Blasi, 2010); iv) thermophilous *Q. pubescens* forests are referable to the association *Roso sempervirentis-Quercetum pubescentis* described for central and southern Italian regions (Biondi, 1986; Pirone et al., 2004); and v) mixed *Q. cerris* and *Q. frainetto* mesophilous forests are referable to the association *Echinopo siculi-Quercetum frainetto* (alliance *Crataego laevigatae-Quercion cerridis*) described in Molise by Blasi and Paura (1995). Such correspondences also allowed for an expert-based interpretation of the statistical analyses, resulting in the adjustment of survey membership to main clusters, always supported by PAM nearest neighbour values and additional fuzzy and inside-group hierarchical clustering. Nevertheless, a more in-depth check and updating of syntaxonomic attribution for sampled forest types goes beyond the aims of the present work and has been postponed to future research advances, eventually hypothesising sub-associations and variants joined to local ecological and biogeographic conditions.

As regards environmental variables, they were initially selected based on their potential influence on plant community development, but then filtered according to statistical significance and autocorrelation, facilitating the identification of indicators for different mature vegetation types. Moreover, even though commonly considered in vegetation ecology, some of these variables have been originally adopted and/or combined for the present case study. For example, precipitation and temperature values, combined with the distribution of mesophilous and thermophilous vegetation types, guided a local scale refinement of the boundary between Temperate and Mediterranean bioclimates posed at the national level (Smiraglia et al., 2013). Other common variables included TWI, which represents a widely used indicator for assessing soil moisture potential and infiltration capacity (Kopecký & Čížková, 2010), and vertical and horizontal distances from the drainage network, which provide complementary insights into the hydrological connectivity of a given point to the stream system and are very useful for understanding respective vegetation responses (Stephan & Korban, 2025). Together, these metrics were adopted to identify flood-prone or water-saturated areas, such as alluvial plains and colluvial areas, as ecological niches for meso-hygrophilous forests that may be difficult to be spatially distinguished from the strictly hygrophilous ones. Besides morphometric variables, it is also known that aspect significantly influences potential direct solar radiation and temperature, affecting in turn meso-bioclimatic conditions (Bennie et al., 2008). Herein we adopted instead the heatload, which combines the effect of aspect and slope as a proxy for evapotranspiration and water stress of plants and resulted to perform better than other variables considering only exposure (such as aspect and Solar radiation index). On the other hand, the slope-area (S–A) relationship represents an innovative approach for geomorphological classification of the landscape into functional process domains. This method was applied to characterise landforms based on "water availability", expressed through the interplay between water accumulation potential (drainage area) and its dissipation (slope). Since it enables an objective delineation of geomorphic domains, including convex hillslopes, unchanneled zones, debris-flow-prone areas, and well-developed fluvial channels (Delchiaro et al., 2023, 2025), in the present study it was explored as a geomorphological proxy for vegetation distribution, under the hypothesis that water availability and erosion regimes significantly influence vegetation cover. This method is potentially more effective than traditional terrain descriptors (such as slope or elevation alone), as it integrates dynamic flow accumulation and dissipation processes at the basin scale.

As regards PNV modeling, both the algorithms performed well and produced comparable results. Notwithstanding the limited size of the input dataset, comprising 90 vegetation surveys, the explained variance for the regression and out-of-bag error for the random forest classification tree were satisfactory, probably also due to the optimised representativity of the sampling design (Hirzel & Guisan, 2002). For the same reason a strict validation was not performed, but the selected models showed a satisfactory interpretability (Lou et al., 2012; Lipton, 2018). In fact, thanks to MLR coefficients and RFCT, it is possible to interpret and understand the resulting spatialisation of vegetation types, allowing non-expert readers to discern the results and the mechanism behind the construction of the maps. In particular, the RF map resulted to be more easily readable and ecologically interpretable, whilst the MLR cartographic product resulted in a finer delineation of ecological niches for different PNV types.

Finally, the synthetic LU map was intended to practically combine the required local scale accuracy in PNV spatialisation, mainly provided by MLR output, with the more easily employable geographical scope of the RFCT output. In order to retain the information on fine-scale ecological heterogeneity, due for example to slope, aspect, and local lithology variation, the LU map reported dominant PNV types alongside their variants. Even though not exactly aligned, the proposed approach is similar to that of Multiple Potential Natural Vegetation, aimed at accounting dual potentialities (e.g., grassland vs forest) in analysed contexts (Somodi, 2012, 2017; Bourdouxhe et al., 2023), and also to that of novel or potential replacement vegetation (Chytrý, 1998; Kowarik & von der Lippe 2018). Especially the latter could be interesting for attuning the classification process in more urbanised areas, where geomorphological alterations and urban heat island effects may significantly disrupt and jeopardise the natural ecological potential.

5. Conclusions

Proper management and environmental restoration requires knowledge, classification, and mapping of ecosystems and respective reference systems, such as PNV models. Here, a robust and repeatable methodology is proposed to identify and map PNV using a semi-automatic approach, leveraging stratified field observations for optimising an ELC up to the land unit level.

Given its flexibility, the process may be applied to contexts other than a medium sized FUA, after site specific adjustments especially in the case of very anthropised urban systems. These estimated PNV maps and the other connected tools, such as local species databases, represent fundamental information for carrying out restoration actions and implementing Nature-Based Solutions that are coherent with local ecological potential and biogeographic settings.

References

1. Akinwande, M. O., Dikko, H. G., & Samson, A., 2015. Variance inflation factor: as a condition for the inclusion of suppressor variable (s) in regression analysis. *Open journal of statistics*, *5*(07), 754. 10.4236/ojs.2015.57075.
2. Aronson, M. F., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., & Vargo, T., 2017. Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment*, *15*(4), 189-196.https://doi.org/10.1002/fee.1480
3. Attorre, F., Francesconi, F., De Sanctis, M., Alfò, M., Martella, F., Valenti, R., & Vitale, M., 2014. Classifying and Mapping Potential Distribution of Forest Types Using a Finite Mixture Model. Folia Geobot 49, 313–335. https://doi.org/10.1007/s12224-012-9139-8.
4. Auslander, M., Nevo, E., & Inbar, M., 2003. The effects of slope orientation on plant growth, developmental instability and susceptibility to herbivores. *Journal of Arid Environments*, *55*(3), 405-416. https://doi.org/10.1016/S0140-1963(02)00281-1
5. Bailey, R. G., 2004. Identifying ecoregion boundaries. *Environmental management*, *34*(Suppl 1), S14-S26. https://doi.org/10.1007/s00267-003-0163-6.
6. Bartolucci, F., Peruzzi, L., Galasso, G., Alessandrini, A., Ardenghi, N. M. G., Bacchetta, G., Banfi, E., Barberis, G., Bernardo, L., Bouvet, D., Bovio, M., Calvia, G., Castello, M., Cecchi, L., Del Guacchio, E., Domina, G., Fascetti, S., Gallo, L., Gottschlich, G., Guarino, R., Gubellini, L., Hofmann, N., Iberite, M., Jiménez-Mejías, P., Longo, D., Marchetti, D., Martini, F., Masin, R. R., Medagli, P., Peccenini, S., Prosser, F., Roma-Marzio, F., Rosati, L., Santangelo, A., Scoppola, A., Selvaggi, A., Selvi, F., Soldano, A., Stinca, A., Wagensommer, R. P., Wilhalm, T., & Conti, F., 2024. A second update to the checklist of the vascular flora native to Italy. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(2), 219–296. https://doi.org/10.1080/11263504.2024.2320126.
7. Bennie, J., Huntley, B., Wiltshire, A., Hill, M. O., & Baxter, R., 2008. Slope, aspect and climate: Spatially explicit and implicit models of topographic microclimate in chalk grassland. Ecological modelling, 216(1), 47-59. https://doi.org/10.1016/j.ecolmodel.2008.04.010.
8. Biondi E., 1986. La vegetazione del Monte Conero (con carta della vegetazione al la scala 1:10.000). Regione Marche, Ancona.
9. Bioret, F., Boullet, V., Choisnet, G., Roux, C., Thébaud, G., Panaïotis, C., Chalumeau, A., Delbosc, P., Demartini, C., Gauberville, C., Cianfaglione, K., Bensettiti, F., & Lalanne, A., 2019. Landscape phytosociology concepts and definitions applied to serial and catenal vegetation mapping. *Contribuţii Botanice*, *LIV*, 47–53. 10.24193/Contrib.Bot.54.3.
10. Blasi C. (ed.), 2010 - La Vegetazione d'Italia. Palombi & Partner S.r.l Roma.
11. Blasi C., Paura B., 1995. Su alcune stazioni a Quercus frainetto Ten. in Campania e in Molise; analisi fitosociologica e fitogeografica. Ann. Bot. (Roma), Studi sul territorio, 51 (10) (1993): 353-366. https://hdl.handle.net/11695/7346.
12. Blasi, C., & Frondoni, R., 2011. Modern perspectives for plant sociology: The case of ecological land classification and the ecoregions of Italy. *Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology*, *145*(sup1), 30–37. https://doi.org/10.1080/11263504.2011.602747.
13. Blasi, C., Capotorti, G., & Frondoni, R., 2005a: Defining and mapping typological models at the landscape scale, Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology: Official Journal of the Societa Botanica Italiana, 139:2, 155-163. https://doi.org/10.1080/11263500500163629.
14. Blasi, C., Capotorti, G., Copiz, R., & Mollo, B., 2018. A first revision of the Italian Ecoregion map. *Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology*, *152*(6), 1201–1204. https://doi.org/10.1080/11263504.2018.1492996
15. Blasi, C., Capotorti, G., Copiz, R., Frondoni, R., Guida, D., Mollo, B., Smiraglia, L., & Zavattero, L., 2010. The ecoregional approach for landscape classification: an Italian example. In *Living Landscape, The European Landscape Convention in research perspective (Florence, 18-19 October 2010), Conference Materials* (Vol. 1, pp. 178-193).
16. Blasi, C., Capotorti, G., Copiz, R., Guida, D., Mollo, B., Smiraglia, D., & Zavattero, L., 2014. Classification and mapping of the ecoregions of Italy. *Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology*, *148*(6), 1255-1345. https://doi.org/10.1080/11263504.2014.985756.
17. Blasi, C., Carranza, M. L., Frondoni, R., & Rosati, L., 2000. Ecosystem classification and mapping: a proposal for Italian landscapes. Applied Vegetation Science, 3(2), 233-242. https://doi.org/10.2307/1479002.
18. Blasi, C., Fortini, P., Grossi, G., & Presti, G., 2005b. Faggete e cerrete mesofile nell'Alto Molise. *Fitosociologia*, *42*(2), 67-81.

19. Blasi, C., Zavattero, L., Marignani, M., Smiraglia, D., Copiz, R., Rosati, L., & Del Vico, E., 2008. The concept of land ecological network and its design using a land unit approach. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 142(3), 540–549. https://doi.org/10.1080/11263500802410892.
20. Borcard, D., Gillet, F., & Legendre, P., 2011. *Numerical ecology with R* (Vol. 2, p. 688). New York: springer. https://doi.org/10.1007/978-3-319-71404-2.
21. Bourdouxhe, A., Wibail, L., Claessens, H., & Dufrêne, M., 2023. Modeling potential natural vegetation: A new light on an old concept to guide nature conservation in fragmented and degraded landscapes. Ecological Modelling, 481, 110382. https://doi.org/10.1016/j.ecolmodel.2023.110382.
22. Braun-Blanquet, J., 1932. Plant sociology. The study of plant communities. 10.1007/978-3-7091-8110-2.
23. Breiman, L. Random Forests. *Machine Learning* 45, 5–32, 2001. https://doi.org/10.1023/A:1010933404324.
24. Bucci, F., Santangelo, M., Fongo, L., Alvioli, M., Cardinali, M., Melelli, L., & Marchesini, I., 2022. A new digital lithological map of Italy at the 1: 100 000 scale for geomechanical modelling. *Earth System Science Data*, *14*(9), 4129-4151. https://doi.org/10.1594/PANGAEA.935673.
25. Burnelli, M., Melelli, L., & Alvioli, M., 2025. Geomorphodiversity and anthropization indices for Italian urban areas. *Geomorphology*, *471*, 109532. https://doi.org/10.1016/j.geomorph.2024.109532.
26. Cadenasso, M. L., Pickett, S. T., & Schwarz, K., 2007. Spatial heterogeneity in urban ecosystems: reconceptualizing land cover and a framework for classification. *Frontiers in Ecology and the Environment*, *5*(2), 80-88. https://doi.org/10.1890/1540-9295(2007)5[80:SHIUER]2.0.CO;2.
27. Capotorti, G., De Lazzari, V., & Alós Ortí, M., 2019. Local scale prioritisation of green infrastructure for enhancing biodiversity in Peri-Urban agroecosystems: A multi-step process applied in the Metropolitan City of Rome (Italy). *Sustainability*, *11*(12), 3322. https://doi.org/10.3390/su11123322. (a)
28. Capotorti, G., Guida, D., Siervo, V., Smiraglia, D., & Blasi, C., 2012. Ecological classification of land and conservation of biodiversity at the national level: The case of Italy. *Biological Conservation*, *147*(1), 174-183. https://doi.org/10.1016/j.biocon.2011.12.028.
29. Capotorti, G., Ortí, M. M. A., Copiz, R., Fusaro, L., Mollo, B., Salvatori, E., & Zavattero, L., 2019. Biodiversity and ecosystem services in urban green infrastructure planning: A case study from the metropolitan area of Rome (Italy). *Urban Forestry & Urban Greening*, *37*, 87-96. https://doi.org/10.1016/j.ufug.2017.12.014. (b)
30. Casavecchia, S., Biondi, E., Catorci, A., Pesaresi, S., Cesaretti, S., & Vitanzi, A., 2007. La regionalizzazione biogeografica quale elemento per una migliore comprensione del valore degli habitat: il caso della Regione Marche. *Fitosociologia*, *44*(2-Suppl. 1), 103-113. Umbria_Umidi.
31. Chirici G., Carta forestale su base Tipologica 1:10 000 del Molise, 2010.
32. Chytrý, M., 1998. Potential replacement vegetation: an approach to vegetation mapping of cultural landscapes. Applied Vegetation Science, 1(2), 177-188. https://doi.org/10.2307/1478947.
33. D'Angeli, C., Ceralli, D., Varricchione, M., Carranza, M. L., de Francesco, M. C., Innangi, M., Santoianni, L., A., & Stanisci, A., 2024. Carta della Natura del Comune di Campobasso: Carta degli habitat alla scala 1: 5000. *ISPRA. https://www.isprambiente.gov.it/it/servizi/sistema-carta-della-natura/cartografia/carta-della-natura-a-scala-locale/carta-della-natura-del-comune-di-campobasso*.
34. Danby, R.K. and Slocombe, D.S., 2005. Regional ecology, ecosystem geography, and transboundary protected areas in the st. elias mountains. Ecological Applications, 15: 405-422. https://doi.org/10.1890/04-0043.
35. De Cáceres, M., Chytrý, M., Agrillo, E., Attorre, F., Botta-Dukát, Z., Capelo, J., Czúcz, B., Dengler, J., Ewald, J., Faber-Langendoen, D., Feoli, E., Franklin, S.B., Gavilán, R., Gillet, F., Jansen, F., Jiménez-Alfaro, B., Krestov, P., Landucci, F., Lengyel, A., Loidi, J., Mucina, L., Peet, R.K., Roberts, D.W., Roleček, J., Schaminée, J.H.J., Schmidtlein, S., Theurillat, J.-P., Tichý, L., Walker, D.A., Wildi, O., Willner, W. and Wiser, S.K., 2015. A comparative framework for broad-scale plot-based vegetation classification. Appl Veg Sci, 18: 543-560. https://doi.org/10.1111/avsc.12179.
36. De Cáceres, M., Legendre, P., & Moretti, M., 2010. Improving indicator species analysis by combining groups of sites. *Oikos*, *119*(10), 1674-1684. https://doi.org/10.1111/j.1600-0706.2010.18334.x.
37. de Francesco M.C., Carranza M.L., Capotorti G., Del Vico E., D'Angeli C., Montaldi A., Paura B., Santoianni L.A., Varricchione M., Stanisci A., 2025. DALIA, a relational DAtabase of trees, shrubs and LIAnas taxa recorded in the Functional Urban Area of Campobasso (Italy). Vegetation Ecology and Diversity, 62, e155222. https://doi.org/10.3897/ved.155222.
38. De Keersmaeker, L., Rogiers, N., Vandekerkhove, K., De Vos, B., Roelandt, B., Cornelis, J., De Schrijver, A., Onkelinx, T., Thomaes, A., Hermy, M., & Verheyen, K*.,* 2013. Application of the Ancient Forest Concept to Potential Natural Vegetation Mapping in Flanders, A Strongly Altered Landscape in Northern Belgium. *Folia Geobot* 48, 137–162. https://doi.org/10.1007/s12224-012-9135-z.

39. Delchiaro, M., Della Seta, M., Martino, S., Nozaem, R., & Moumeni, M., 2023. Tectonic deformation and landscape evolution inducing mass rock creep driven landslides: the Loumar case-study (Zagros Fold and Thrust Belt, Iran). *Tectonophysics*, *846*, 229655. https://doi.org/10.1016/j.tecto.2022.229655.
40. Delchiaro, M., Vergari, F., Esposito, C., & Del Monte, M., 2025. The influence of anthropogenic topographic changes on geomorphological processes in the city of Rome (Italy): A case study of the Malagrotta area. *Earth Surface Processes and Landforms*, *50*(3), e70033. https://doi.org/10.1002/esp.70033.
41. Dengler, J., 2016. Phytosociology. *International Encyclopedia of Geography: People, the Earth, Environment and Technology: People, the Earth, Environment and Technology*, 1-6. https://doi.org/10.1002/9781118786352.wbieg0136.
42. Dickson, B., Blaney, R. Miles, L., Regan, E., van Soesbergen, A.,Väänänen, E., Blyth, S., Harfoot, M. Martin, C.S., McOwen, C., Newbold, T., van Bochove, J., 2014. Towards a global map of natural capital: key ecosystem assets. UNEP, Nairobi, Kenya.
43. Dijkstra, L., Poelman, H., Veneri, P., 2019. The EU-OECD definition of a functional urban area. OECD Regional Development Working Papers, No. 2019/11, OECD Publishing, Paris, doi: https://doi.org/10.1787/d58cb34d-en.
44. Dufrêne, M., & Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological monographs*, *67*(3), 345-366. https://doi.org/10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2.
45. Dunn, J. C., 1973. A Fuzzy Relative of the ISODATA Process and Its Use in Detecting Compact Well-Separated Clusters. *Journal of Cybernetics*, *3*(3), 32–57. https://doi.org/10.1080/01969727308546046
46. Durbecq, A., Jaunatre, R., Buisson, E., Cluchier, A. and Bischoff, A, 2020, Identifying reference communities in ecological restoration: the use of environmental conditions driving vegetation composition. Restor Ecol, 28: 1445-1453. https://doi.org/10.1111/rec.13232
47. EC (European Commission), 2020. https://environment.ec.europa.eu/topics/nature-and-biodiversity_mt.
48. EC (European Commission). Natura 2000—Environment. 2022. Available online: https://ec.europa.eu/environment/nature/natura2000/index_en.htm#:~:text=Stretching%20over%2018%25%20of%20the,and%20threatened%20species%20and%20habitats (accessed on 18 June 2025)
49. EC 1991/2024. Regulation (EU) 2024/1991 of the European Parliament and of the Council of 24 June 2024 on nature restoration and amending Regulation (EU) 2022/869. http://data.europa.eu/eli/reg/2024/1991/oj.
50. EEA, 2017. Biogeographical regions in Europe | European Environment Agency's home page last access 15/04/2025.
51. Elith, J., & Leathwick, J. R., 2009. Species distribution models: ecological explanation and prediction across space and time. *Annual review of ecology, evolution, and systematics*, *40*(1), 677-697.https://doi.org/10.1146/annurev.ecolsys.110308.120159.
52. EU, 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. http://data.europa.eu/eli/dir/1992/43/oj.
53. Europe shapefile 10 km 100 km — European Environment Agency.
54. European Environment Agency (EEA), 2018. Urban Atlas, Copernicus Land Monitoring Service. https://land.copernicus.eu/local/urban-atlas.
55. Facioni L., Burrascano S., Del Vico E., Rosati L., Tilia A., & Blasi C., 2015. Phytosociological analysis of white oak (Quercus pubescens s.l.) woodlands and related successional stages: spatial patterns and their drivers. Phytocoenologia, 45(4):325–364. . https://doi.org/10.1127/phyto/2015/0008.
56. Farris, E., Filibeck, G., Marignani, M., & Rosati, L., 2010. The power of potential natural vegetation (and of spatial-temporal scale): A response to Carrión & Fernández (2009). *Journal of Biogeography*, *37*(11), 2211-2213. https://doi.org/10.1111/j.1365-2699.2010.02323.x
57. Fischer, H. S., Winter, S., Lohberger, E., Jehl, H., & Fischer, A., 2013. Improving transboundary maps of potential natural vegetation using statistical modeling based on environmental predictors. *Folia Geobotanica*, *48*, 115-135. https://doi.org/10.1007/s12224-012-9150-0.
58. Fischer, S., Michler, B., & Fischer, A., 2019. High resolution predictive modelling of potential natural vegetation under recent site conditions and future climate scenarios: Case study Bavaria. *Tuexenia*, *39*. 10.14471/2018.39.001.
59. Fisher, R. A., 1922. On the interpretation of χ^2 from contingency tables, and the calculation of P. Journal of the Royal Statistical Society, 85(1), 87–94.DOI: 10.2307/2340521.
60. Galasso, G., Conti, F., Peruzzi, L., Alessandrini, A., Ardenghi, N. M. G., Bacchetta, G., Banfi, E., Barberis, G., Bernardo, L., Bouvet, D., Bovio, M., Castello, M., Cecchi, L., Del Guacchio, E., Domina, G., Fascetti, S., Gallo, L., Guarino, R., Gubellini, L., Guiggi, A., Hofmann, N., Iberite, M., Jiménez-Mejías, P., Longo, D., Marchetti, D., Martini, F., Masin, R. R., Medagli, P., Musarella, C. M., Peccenini, S., Podda, L., Prosser, F., Roma-Marzio, F., Rosati, L., Santangelo, A., Scoppola, A., Selvaggi, A., Selvi, F., Soldano, A., Stinca, A., Wagensommer, R. P., Wilhalm, T., & Bartolucci, F., 2024. A second update to the checklist of the vascular flora alien to Italy. Plant

Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(2), 297–340. https://doi.org/10.1080/11263504.2024.2320129.
61. Gallant, A.L., Loveland, T.R., Sohl, T.L., Napton, D.E., 2004. Using an Ecoregion Framework to Analyze Land-Cover and Land-Use Dynamics. Environmental Management 34 (Suppl. 1), S89–S110. https://doi.org/10.1007/s00267-003-0145-3.
62. Gallizia Vuerich, L., Poldini, L., & Feoli, E., 2001. Model for the potential natural vegetation mapping of Friuli Venezia-Giulia (NE Italy) and its application for a biogeographic classification of the region.
63. Greenacre, M., Groenen, P. J., Hastie, T., d'Enza, A. I., Markos, A., & Tuzhilina, E., 2022. Principal component analysis. *Nature Reviews Methods Primers*, *2*(1), 100. https://doi.org/10.1038/s43586-022-00184-w.
64. Guisan, A., & Zimmermann, N. E., 2000. Predictive habitat distribution models in ecology. *Ecological modelling*, *135*(2-3), 147-186. https://doi.org/10.1016/S0304-3800(00)00354-9.
65. Hengl, T., Walsh, MG., Sanderman, J., Wheeler, I., Harrison, SP., Prentice, IC., 2018. Global mapping of potential natural vegetation: an assessment of machine learning algorithms for estimating land potential. PeerJ 6:e5457 https://doi.org/10.7717/peerj.5457.
66. Hirzel, A., & Guisan, A., 2002. Which is the optimal sampling strategy for habitat suitability modelling. *Ecological modelling*, *157*(2-3), 331-341. https://doi.org/10.1016/S0304-3800(02)00203-X.
67. Iacobucci, G., Delchiaro, M., Troiani, F., & Nadali, D., 2024. Land-surface quantitative analysis for mapping and deciphering the construction processes of piedmont alluvial fans in the Anti-Lebanon Mountains. *Geomorphology*, *453*, 109148. https://doi.org/10.1016/j.geomorph.2024.109148.
68. ISPRA Sinanet, SINAnet Groupware, last access on 13/05/2025
69. ISPRA, 2008. Reticolo Idrografico, 2008. https://gn.mase.gov.it.
70. ISPRA, 2009. Carta Geolitologica d'Italia. https://gn.mase.gov.it.
71. Jasiewicz, J., & Stepinski, T. F., 2013. Geomorphons—a pattern recognition approach to classification and mapping of landforms. *Geomorphology*, *182*, 147-156. https://doi.org/10.1016/j.geomorph.2012.11.005.
72. Klijn, F., & de Haes, H. A. U., 1994. A hierarchical approach to ecosystems and its implications for ecological land classification. *Landscape ecology*, *9*, 89-104. https://doi.org/10.1007/BF00124376.
73. Konrád, K. D., Bede-Fazekas, Á., Molnár, Z., & Somodi, I., 2022. Multilayer landscape classification based on potential vegetation. *Preslia*, *94*(4), 631-650. 10.23855/preslia.2022.631.
74. Kopecký, M., & Čížková, Š., 2010. Using topographic wetness index in vegetation ecology: does the algorithm matter?. *Applied Vegetation Science*, *13*(4), 450-459. https://doi.org/10.1111/j.1654-109X.2010.01083.x.
75. Kowarik, I., & von der Lippe, M., 2018. Plant population success across urban ecosystems: A framework to inform biodiversity conservation in cities. Journal of Applied Ecology, 55(5), 2354-2361. https://doi.org/10.1111/1365-2664.13144.
76. Kraufman, L., & Rousseeuw, P., 2005. Finding groups in data: An introduction to cluster analysis.–Hoboken.
77. Kruskal, W. H., & Wallis, W. A., 1952. Use of Ranks in One-Criterion Variance Analysis. *Journal of the American Statistical Association*, *47*(260), 583–621. https://doi.org/10.1080/01621459.1952.10483441.
78. Lipton, Z. C., 2018. The mythos of model interpretability: In machine learning, the concept of interpretability is both important and slippery. *Queue*, *16*(3), 31-57. https://doi.org/10.1145/3233231.
79. Lou, Y., Caruana, R., & Gehrke, J. 2012. Intelligible models for classification and regression. In *Proceedings of the 18th ACM SIGKDD international conference on Knowledge discovery and data mining* (pp. 150-158). https://doi.org/10.1145/2339530.233955.
80. Maes, J., Teller, A., Erhard, M., Conde, S., Vallecillo Rodriguez, S., Barredo Cano, J. I., Paracchini, M.-L., Abdul Malak, D., Trombetti, M., Vigiak, O., Zulian, G., Addamo, A., Grizzetti, B., Somma, F., Hagyo, A., Vogt, P., Polce, C., Jones, A., Marin, A., Ivits, E., Mauri, A., Rega, C., Czucz, B., Ceccherini, G., Pisoni, E., Ceglar, A., De Palma, P., Cerrani, I., Meroni, M., Caudullo, G., Lugato, E., Vogt, J., Spinoni, J., Cammalleri, C., Bastrup-Birk, A., San-Miguel-Ayanz, J., San Román, S., Kristensen, P., Christiansen, T., Zal, N., De Roo, A., De Jesus Cardoso, A., Pistocchi, A., Del Barrio Alvarellos, I., Tsiamis, K., Gervasini, E., Deriu, I., La Notte, A., Abad Viñas, R., Vizzarri, M., Camia, A., Robert, N., Kakoulaki, G., Garcia Bendito, E., Panagos, P., Ballabio, C., Scarpa, S., Montanarella, L., Orgiazzi, A., Fernandez Ugalde, O., & Santos-Martín, F., 2020. Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment, Publications Office of the European Union, Luxembourg. JRC120383. https://data.europa.eu/doi/10.2760/757183.
81. Martinelli, L., & Matzarakis, A., 2017. Influence of height/width proportions on the thermal comfort of courtyard typology for Italian climate zones. *Sustainable Cities and Society*, *29*, 97-106. https://doi.org/10.1016/j.scs.2016.12.004.

82. Martinello, C., Delchiaro, M., Iacobucci, G., Cappadonia, C., Rotigliano, E., & Piacentini, D. (2024). Exploring the geomorphological adequacy of the landslide susceptibility maps: A test for different types of landslides in the Bidente river basin (northern Italy). *Catena*, *238*, 107835. https://doi.org/10.1016/j.catena.2024.107835
83. MASE (Italian Ministry of Environment and Energy Security), 2021. Piano di riforestazione urbana ed extraurbana. PNRR Misura 2 – Componente 4. PNRR_ INVESTIMENTO_3.1_Avviso_pubblico.pdf. https://www.mase.gov.it/sites/defaul t/files/archivio/bandi/PNRR_INVESTIMENTO_3.1_Avviso_pubblico.pdf.
84. MASE, 2024. Rete Natura 2000 (SIC/ZSC e ZPS). https://gn.mase.gov.it.
85. McCune, B., and D. Keon, 2002. Equations for potential annual direct incident radiation and heat load index. Journal of Vegetation Science. 13:603-606. https://doi.org/10.1111/j.1654-1103.2002.tb02087.x.
86. McKight, P. E., & Najab, J., 2010. Kruskal‐wallis test. *The corsini encyclopedia of psychology*, 1-1. https://doi.org/10.1002/9780470479216.corpsy0491.
87. Metzger, M.J., Bunce, R.G.H., Jongman, R.H.G., Mücher, C.A. and Watkins, J.W., 2005, A climatic stratification of the environment of Europe. Global Ecology and Biogeography, 14: 549-563. https://doi.org/10.1111/j.1466-822X.2005.00190.x.
88. Münier, B., Nygaard, B., Ejrnæs, R., & Bruun, H. G., 2001. A biotope landscape model for prediction of semi-natural vegetation in Denmark. *Ecological Modelling*, *139*(2-3), 221-233. https://doi.org/10.1016/S0304-3800(01)00244-7.
89. Muratet, A., Porcher, E., Devictor, V., Arnal, G., Moret, J., Wright, S., & Machon, N., 2008. Evaluation of floristic diversity in urban areas as a basis for habitat management. Applied vegetation science, 11(4), 451-460. https://doi.org/10.3170/2008-7-18530.
90. Murtagh, F., & Contreras, P., 2012. Algorithms for hierarchical clustering: an overview. *Wiley Interdisciplinary Reviews: Data Mining and Knowledge Discovery*, *2*(1), 86-97. https://doi.org/10.1002/widm.53.
91. Nelson, C.R., Hallett, J.G., Romero Montoya, A.E., Andrade, A., Besacier, C., Boerger, V., Bouazza, K., Chazdon, R., Cohen-Shacham, E., Danano, D., Diederichsen, A., Fernandez, Y., Gann, G.D., Gonzales, E.K., Gruca, M., Guariguata, M.R., Gutierrez, V., Hancock, B., Innecken, P., Katz, S.M., McCormick, R., Moraes, L.F.D., Murcia, C., Nagabhatla, N., Pouaty Nzembialela, D., Rosado-May, F.J., Shaw, K., Swiderska, K., Vasseur, L., Venkataraman, R., Walder, B., Wang, Z., & Weidlich, E.W.A. 2024. Standards of practice to guide ecosystem restoration – A contribution to the United Nations Decade on Ecosystem Restoration 2021-2030. Rome, FAO, Washington, DC, SER & Gland, Switzerland, IUCN CEM. https://doi.org/10.4060/cc9106e.
92. Omernik, J.M., 2004. Perspectives on the nature and definition of ecological regions. Environmental Management 34 (Suppl. 1), S27–S38. https://doi.org/10.1007/s00267-003-5197-2.
93. Pandis, N., 2016. The chi-square test. American journal of orthodontics and dentofacial orthopedics, 150(5), 898-899. The chi-square test - American Journal of Orthodontics and Dentofacial Orthopedics.
94. Paura, B., & Abbate, G., 1995. I querceti a caducifoglie del Molise: primo contributo sulla sintassonomia e corologia. ANNALI DI BOTANICA, 51, 325-340. https://hdl.handle.net/11695/5942.
95. Pearson, K., 1900. X. *On the criterion that a given system of deviations from the probable in the case of a correlated system of variables is such that it can be reasonably supposed to have arisen from random sampling*. *The London, Edinburgh, and Dublin Philosophical Magazine and Journal of Science*, *50*(302), 157–175. https://doi.org/10.1080/14786440009463897.
96. Pearson, K., 1901. LIII. *On lines and planes of closest fit to systems of points in space*. *The London, Edinburgh, and Dublin Philosophical Magazine and Journal of Science*, *2*(11), 559–572. https://doi.org/10.1080/14786440109462720.
97. Pesaresi, S., Biondi, E., & Casavecchia, S., 2017. Bioclimates of Italy. *Journal of Maps*, *13*(2), 955–960. https://doi.org/10.1080/17445647.2017.1413017
98. Pica, A., Lämmle, L., Burnelli, M., Del Monte, M., Donadio, C., Faccini, F., Lazzari, M., Mandarino, A., Melelli, L., Perez Filho, A., Russo, F., Stamatopoulos, L., Stanislao, C., Brandolini, P., 2024. Urban Geomorphology Methods and Applications as a Guideline for Understanding the City Environment. Land, 13(7), 907; https://doi.org/10.3390/land13070907.
99. Pignatti, S., Guarino, R., & La Rosa, M., 2017. Flora d'italia (Vol. 1).
100. Pirone, G., Ciaschetti, G., & Frattaroli, A. R., 2004. Appunti sulla vegetazione della Valle del Trigno (Abruzzo meridionale). *Inform. Bot. Ital*, *36*(1), 13-27.
101. Qiu, L., Gao, T., Gunnarsson, A., Hammer, M., & von Bothmer, R., 2010. *A methodological study of biotope mapping in nature conservation*. *Urban Forestry & Urban Greening*, *9*(2), 161–166. https://doi.org/10.1016/j.ufug.2010.01.003.
102. Qu, J., Liu, Q., Gui, D., Feng, X., Wang, H., Zhao, J., Guangyan W., & Wei, G., 2024. Precise vegetation restoration in arid regions based on potential natural vegetation and potential normalized difference vegetation index. *Restoration Ecology*, *32*(2), e13967. https://doi.org/10.1111/rec.13967.

103. Regione Molise - Pianificazione forestale., 2017. Last access 16/04/2025.
104. Roberts. D.W., and Cooper, S.V., 1989. Concepts and techniques of vegetation mapping. In Land Classifications Based on Vegetation: Applications for Resource Management. USDA Forest Service GTR INT-257, Ogden, UT, pp 90-96
105. Roleček, J., Chytrý, M., Hájek, M., Lvončík, S., & Tichý L., 2007. Sampling design in large-scale vegetation studies: Do not sacrifice ecological thinking to statistical purism!. *Folia Geobot* 42, 199–208. https://doi.org/10.1007/BF02893886.
106. Rousseeuw, L. K. P. J., & Kaufman, P., 1987. Clustering by means of medoids. In *Proceedings of the statistical data analysis based on the L1 norm conference, neuchatel, switzerland* (Vol. 31, p. 28).
107. Ruspini, E. H., Bezdek, J. C., & Keller, J. M., 2019. Fuzzy clustering: A historical perspective. *IEEE Computational Intelligence Magazine*, *14*(1), 45-55. 10.1109/MCI.2018.2881643.
108. Sánchez, M. C., & Priego-Santander, Á. G., 2011. Biophysical landscapes of a coastal area of Michoacán state in Mexico. Journal of Maps, 7(1), 42–50. https://doi.org/10.4113/jom.2011.1098.
109. Sayre, R., Comer, P., Warner, H., & Cress, J., 2009. A new map of standardized terrestrial ecosystems of the conterminous United States. Reston, VA: U.S. Geological Survey, "Professional Paper 1768", 17 pp. https://pubs.usgs.gov/pp/1768.
110. Schwanghart, W., & Scherler, D., 2014. TopoToolbox 2–MATLAB-based software for topographic analysis and modeling in Earth surface sciences. *Earth Surface Dynamics*, *2*(1), 1-7. https://doi.org/10.5194/esurf-2-1-2014.
111. 'Servizio WMS - CTR 5000', accessed 2025-06-10, http://data.europa.eu/88u/dataset/r_molise-00013-20120413-094513.
112. Smiraglia, D., Capotorti, G., Guida, D., Mollo, B., Siervo, V., & Blasi, C., 2013. Land units map of Italy. *Journal of Maps*, *9*(2), 239-244.https://doi.org/10.1080/17445647.2013.771290.
113. Somodi, I., Ewald, J., Bede-Fazekas, Á., & Molnár, Z., 2021. The relevance of the concept of potential natural vegetation in the Anthropocene. *Plant Ecology & Diversity*, *14*(1-2), 13-22.https://doi.org/10.1080/17550874.2021.1984600.
114. Somodi, I., Konrád, K. D., Vizi, D. B., Tallósi, B., Samu, A., Kajner, P., & Bede-Fazekas, Á.. 2024. Modelling of potential vegetation identifies diverging expectable outcomes of river floodplain widening. Global Ecology and Conservation, 50, e02859. https://doi.org/10.1016/j.gecco.2024.e02859.
115. Somodi, I., Molnár, Z., & Ewald, J., 2012. Towards a more transparent use of the potential natural vegetation concept–an answer to Chiarucci et al. *Journal of Vegetation Science*, *23*(3), 590-595. https://doi.org/10.1111/j.1654-1103.2011.01378.x.
116. Somodi, I., Molnár, Z., Czúcz, B., Bede-Fazekas, Á., Bölöni, J., Pásztor, L., Annamária L., & Zimmermann, N. E., 2017. Implementation and application of multiple potential natural vegetation models–a case study of Hungary. *Journal of Vegetation Science*, *28*(6), 1260-1269. https://doi.org/10.1111/jvs.12564.
117. Stephan, J., & Korban, M., 2025. Ecological niche modelling using MaxEnt for riparian species in a Mediterranean context. *Ecological Indicators*, *171*, 113167. https://doi.org/10.1016/j.ecolind.2025.113167.
118. Sturbois, A., De Cáceres, M., Bifolchi, A., Bioret, F., Boyé, A., Gauthier, O., Grall, J., Grémare, A., Labrune, C., Robert, A., Schaal, G., & Desroy, N., 2023. Ecological quality assessment: A framework to report ecosystems quality and their dynamics from reference conditions. Ecosphere, 14(12), e4726. https://doi.org/10.1002/ecs2.4726.
119. Tabachnick, B. G., & Fidell, L. S., 2007. *Experimental designs using ANOVA* (Vol. 724). Belmont, CA: Thomson/Brooks/Cole. 05142_17_IFC.qxd.
120. Taffetani, F., Catorci, A., Ciaschetti, G., Cutini, M., Di Martino, L., Frattaroli, A. R., Paura B., Pirone G., Rismondo M. & Zitti, S., 2012. The Quercus cerris woods of the alliance Carpinion orientalis Horvat 1958 in Italy. *Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology*, *146*(4), 918-953. https://doi.org/10.1080/11263504.2012.682613.
121. Terzi, M., Ciaschetti, G., Fortini, P., Rosati, L., Viciani, D. & Di Pietro, R. 2021. New syntaxon names for the Italian Quercus cerris woods. Mediterr. Bot. 42, e75592. https://dx.doi.org/10.5209/mbot.75592.
122. Terzi, M., Ciaschetti, G., Fortini, P., Rosati, L., Viciani, D., & Di Pietro, R., 2020. A revised phytosociological nomenclature for the Italian Quercus cerris woods. Mediterranean Botany, 41(1), 101-120. https://hdl.handle.net/11573/1413441.
123. Theil, H., 1969. A multinomial extension of the linear logit model. *International Economic Review, 10*(3), 251–259. https://doi.org/10.2307/2525642.
124. Toma, T. S. P., Overbeck, G. E., de Souza Mendonça Jr, M., & Fernandes, G. W., 2023. Optimal references for ecological restoration: the need to protect references in the tropics. Perspectives in Ecology and Conservation, 21(1), 25-32. https://doi.org/10.1016/j.pecon.2023.01.003

125. Török, K., Csecserits, A., Somodi, I., Kövendi-Jakó, A., Halász, K., Rédei, T., & Halassy, M., 2018. Restoration prioritization for industrial area applying multiple potential natural vegetation modeling. Restoration Ecology, 26(3), 476-488. https://doi.org/10.1111/rec.12584.
126. Tuxen, R., 1956. Die heutige potentille natuerliche vegetation als Gegenst vegetation skartierung. Angew Pflanzensoziologie, 13, 5-42.
127. U. Bohn, G. Gollub, C. Hettwer, Z. Neuhäuslová, Th. Raus, H. Schlüter, H. Weber. [Bearb./Eds.] Karte der natürlichenVegetation Europas/Map of the Natural Vegetation of Europe, Maßstab/Scale 1:2.500.000, Interaktive/Interactive CD-ROM- Erläuterungstext, Legende, Karten / Explanatory Text, Legend, Maps. Landwirtschaftsverlag, Münster, 2004.
128. United Nations, 2019. Resolution adopted by the General Assembly: United Nations Decade on Ecosystem Restoration (2021–2030). Document Viewer
129. Valeri, S., Schmitz, M. F., Acosta-Gallo, B., Iamonico, D., Villodre, M., Arnáiz-Schmitz, C., & Capotorti, G., 2024. Condition of linear landscape elements improves with contiguity to protected habitats: Empirical evidence useful for agroecosystem accounting and restoration. doi.org/10.1016/j.agee.2024.109199.
130. Valiante, M., Di Benedetto, A., & Aloia, A., 2024. A Comparison of Landforms and Processes Detection Using Multisource Remote Sensing Data: The Case Study of the Palinuro Pine Grove (Cilento, Vallo di Diano and Alburni National Park, Southern Italy). *Remote Sensing*, *16*(15), 2771. https://doi.org/10.3390/rs16152771.
131. van der Maarel, E., 1979. Transformation of cover-abundance values in phytosociology and its effects on community similarity. *Vegetatio*, *39*(2), 97–114. 10.1007/BF00052021.
132. Varricchione, M., Laura Carranza, M., D'Angeli, C., Carla de Francesco, M., Innangi, M., Santoianni, L. A., & Stanisci, A., 2024. Exploring the distribution pattern of native and alien forests and their woody species diversity in a small Mediterranean city. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(6), 1335–1346. https://doi.org/10.1080/11263504.2024.2415613.
133. Vergari, F., Troiani, F., Faulkner, H., Del Monte, M., Della Seta, M., Ciccacci, S., & Fredi, P., 2019. The use of the slope–area function to analyse process domains in complex badland landscapes. *Earth Surface Processes and Landforms*, *44*(1), 273-286. https://doi.org/10.1002/esp.4496.
134. Wang, H., Liu, Q., Gui, D., Feng, X., Qu, J., Zhao, J., Guangyan W., & Wei, G., 2023. Simulation of potential vegetation distribution in arid areas at regional scale. *Catena*, *232*, 107428. https://doi.org/10.1016/j.catena.2023.107428.
135. Wang, Y., 2024. Identifying geomorphic domains using hierarchically clustered drainage area-slope scaling. *Earth Surface Processes and Landforms*, *49*(6), 1762-1777. https://doi.org/10.1002/esp.5796.
136. Ward, J. H., 1963. Hierarchical Grouping to Optimize an Objective Function. *Journal of the American Statistical Association*, *58*(301), 236–244. https://doi.org/10.1080/01621459.1963.10500845.
137. World Health Organization, 2023. One Health, last access 07/07/2025.
138. Wuensch, K.L., 2011. Chi-Square Tests. In: Lovric, M. (eds) International Encyclopedia of Statistical Science. Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-642-04898-2_173.

Part II: How far from naturalness: anthropogenic landform effects on potential vegetation in an urban ecosystem (Capital City of Rome, Italy)

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Abstract

The urban landscape of Rome resulted from a century-old stratification of human activities, which also changed natural morphology and, in turn, ecosystem supporting environment and dynamics. However, the specific impact of geomorphological alterations on woodland successional stages has remained poorly explored. This study aims at filling this gap and concomitantly providing useful knowledge for practical applications in urban restoration ecology. On the basis of identification and mapping of anthropogenic landforms, vegetation surveys have been conducted to assess the intensity of eventual shifts in spontaneous woodland dynamics across a disturbance gradient. Vegetation surveys were performed both inside (altered) and outside (low naturalness and high naturalness controls) anthropogenic landforms to qualify and quantify site-level shifts from Potential Natural Vegetation/PNV (local reference systems) due to geomorphological alteration and/or alternative human disturbances. Woodlands on anthropogenic landforms resulted to be significantly divergent in species composition from PNV, being characterised by invasive, nitrophilous, and light-demanding species, and the Redundancy Analysis showed this type of disturbance alters woodland composition much more than other impacts of the urban context. Differences in species composition between woodlands in different condition was thus quantified, in terms of ecological distances from PNV, modelled and mapped across the city, by applying a Generalised Additive Model (GAM). The identification and spatialisation of areas that shifted furthest from reference systems, with an unlikely reversible deviation from PNV, represent a useful finding for practical implementation of the research, especially in the field of urban ecological restoration.

Keywords: Potential Natural Vegetation, Ecological Land Units, Urban ecology, Ecological distance, Restoration actions, Anthropogenic landforms

Highlights:

- Actual vegetation shift from reference systems was qualified and quantified in Rome
- Anthropogenic landform effect on woodland communities was distinctively assessed
- Ecological distances from PNV were spatialised across the study area
- Research findings are useful to inform ecological restoration in urban ecosystems

1. Introduction

As the world population increasingly inhabits cities, biodiversity in urban areas becomes a crucial element to be conserved and restored (CBD, 2022; Grimm et al., 2008; UN, 2015). Urbanisation actually exert negative impacts on sustainability and resilience of urban ecosystems in terms of species survival, ecological functions, provision of ecosystem services, and human well-being, as citizens are often exposed to detrimental conditions due to natural habitat loss, resource depletion and pollution (Manes et al., 2012; McDonald et al., 2019; Adla et al., 2022). These pressures cannot be solely counterbalanced by protecting residual natural areas around cities, but a shift towards active enhancement and renaturalisation of human-modified habitats is also needed (EC, 2020; Perring et al., 2015; Rosenzweig, 2003). In Europe, the Nature Restoration Regulation/NRR (1991/2024) emphasises the role of cities in biodiversity recovery and climate change adaptation and mitigation, given that urban areas cover 22% of the EU land and host most of its population. Key NRR targets include No Net Loss by 2030, ensuring no reduction in urban green spaces (e.g., parks, green roofs, urban forests) and tree canopy cover as compared to 2024 levels, and increasing their share from 2031 onward, until satisfactory levels are met. These targets can be achieved through land management and urban planning, including the design of green infrastructure and related implementation of restoration actions and nature-based solutions (EC, 2020). The general aim is therefore to bring back nature into cities, reconciling the urban environment with natural elements, and enhancing biodiversity along with human well-being (Lundholm & Richardson, 2010; Capotorti et al., 2020).

Restoration actions, such as reforestation and tree planting, should be supported by ground-based knowledge to ensure a proper selection of species that are effectively adapted to local site conditions and capable of establishing self-sustaining ecosystems in the medium and long term (Capotorti et al., 2019; Blasi et al., 2025). It is a foundational concept in ecology that, besides the biogeographic setting and biotic interactions, the abiotic features of the environment determine the composition, richness, and distribution of species (Kraft et al., 2014). Thus, according to climate and physiography, including geology and geomorphology, different Environmental Land Units (ELU) can be defined within a landscape that represent homogeneous zones capable of hosting a specific type of mature ecosystem (Smiraglia et al., 2013). ELU define the territorial extent of potential natural vegetation (PNV), i.e. the mature vegetation that would develop if human disturbance is removed, and may be adopted as a reference model for ecosystem restoration (Tuxen, 1956; Zerbe, 1998; Nelson et al., 2024). However, especially within and around cities, several factors may contribute to a shift from original PNV, as the different urban land uses and land cover types directly or indirectly alter the composition, structure and functions of ecosystems and their dynamics (Sanderson et al., 2002). For example, the extent of residual natural habitats determines the pool of species available, the ecological connectivity among habitats affects their dispersal (Valeri et al., 2021), the high proportion of artificial land cover facilitates the introduction and abundance of exotic, sometimes invasive, species (Boscutti et al., 2018; Montaldi et al., 2024) capable of affecting not only the composition but also the dynamics of vegetation communities (Celesti-Grapow et al., 2001), and land use and management legacy influences the spontaneous renaturalisation of disturbed green spaces, also given the frequent chemical, physical and biological degradation of soils (McLauchlan, 2006; Bart & Davenport, 2015; Zapolska et al., 2023). Altogether, local species extinction, alteration of biogeochemical cycles and introduction of exotic species may determine the assemblage of "novel" or "emerging" communities, even representing long-standing woodland stages (Hobbs et al., 2006; Kowarik, 2011). Even though the effect was well-documented with respect to land-use change drivers, the shift from local PNV due

to anthropogenic alteration of geomorphology was poorly investigated. Nevertheless, urban development-related activities profoundly reshape natural topography in cities, with humans acting as a genuine physical process that determines persistent environmental changes (Somodi et al., 2021). Anthropogenic geomorphological processes, like accumulation (e.g., valley filling, slope terracing, dumps, or artificial hills) and erosion (e.g., excavation, mining, or extraction activities), can even overtake natural processes, resulting in a deep transformation of original landforms (Del Monte et al., 2016; Vergari et al., 2021), up to the creation of entirely artificial ones (Szabò et al., 2010; Thornbush & Allen, 2018). In very short timeframes, these processes lead to changes in slope shape, drainage patterns, and type of outcropping substrate (Stephens et al., 2021; Mandarino et al., 2021; Pica et al., 2024), as well as to complex changes in physiography and edaphic conditions (Del Monte et al., 2016; La Vigna et al., 2016; Delchiaro et al., 2025), which in turn accommodate late-successional vegetation types that differ from natural ones, at least at the site scale.
Accordingly, the present research was aimed at qualifying and quantifying the site level impacts of human-induced geomorphological alterations on potential vegetation, so that either the hypothetical conservation value (Kowarik & Von der Lippe, 2018) or the priority restoration/requalification needs for related novel woodlands could be alternatively assessed. The Municipality of Rome Capital was selected as a case study, due to the occurring historical and spatially widespread geomorphological alteration. An in-depth understanding of such divergencies from local PNV models could be useful for planning effective restoration of self-sustaining ecosystems in the area, as well as for complying with current strategic and policy requirements (EC, 1991/2024) also in other cities.

2. Materials and methods

2.1 Study Area

The study area (Figure 1) corresponds to the municipality of Rome Capital, extending over 1 287 km^2 and having a population of 2.9 million inhabitants (Roma Capitale, 2022). The area has a notable ecosystem heterogeneity due to combined environmental and anthropic determinant factors (Celesti-Grapow et al, 2013; Blasi et al., 2005). Namely, the central location in the Mediterranean Basin, moulded by both Western and Eastern biogeographic influences, the transitional ecoregional setting between the Tyrrhenian sea and Apennine mountain range, and a long-lasting interaction with human activities determined an outstanding variety of species and communities (Blasi & Frondoni, 2011; Frondoni et al., 2011; Capotorti et al., 2012). Morphology is characterised by lowland and gentle slopes, including wide volcanic plateaux, sedimentary hills and coastal plains, and the alluvial plain of the Tiber River (Funiciello & Giordano, 2008), with an average altitude of 20 m and a maximum of 139 m a.s.l. A huge urban development, which shaped the current form of the city, prevalently occurred during the last 70 years with notable effects on agricultural and (semi-)natural systems (Frondoni et al., 2011; Figure 1a).

An ecological classification of land, carried out at the municipal level on the basis of differential bioclimatic, lithological and morphological arrangements, brought to the identification and mapping of 16 different ELU, which support the development of as many types of local PNV (Blasi et al., 2005). Most widespread ELU / PNV types, especially considered for the herein research, are pyroclastic slopes and plateaux with *Q. cerris* forests and sandy hills with *Q. pubescens* and *Q. suber* forests (Figure 1b). Alluvial valleys, including the Tiber River and the Aniene ones, as well as the coastal and subcoastal ELU, were excluded because of their more limited extent, few residuals of woodland cover and the lack of anthropogenic landforms.

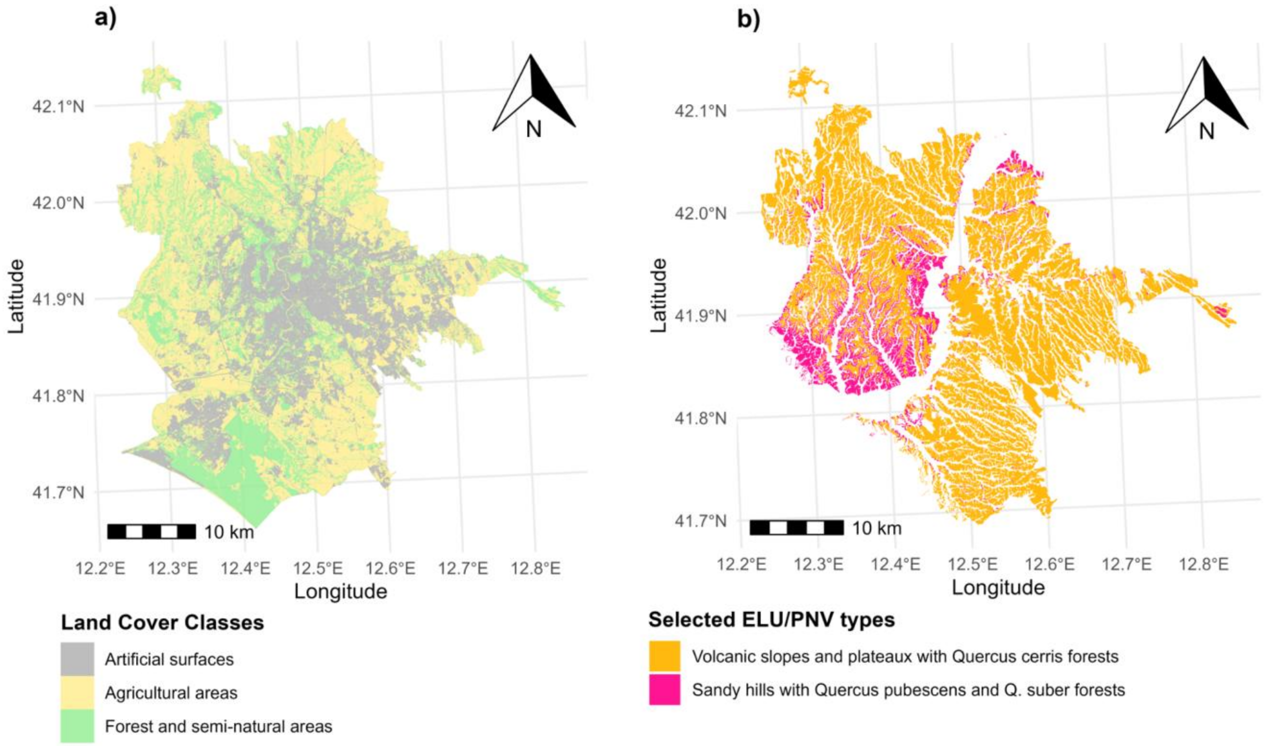

Figure 1: Study area: main land cover types in the overall Municipality of Rome Capital (a) and prevailing ELU /PNV types selected for the research (b).

The case of Rome is an illustrative example of how, in urban and peri-urban areas, natural landforms and landforms altered and created by human activities coexist (Burnelli et al., 2025). Fluvial processes and landforms are prevalent, both erosional and constructional, along with widespread litho-structural and volcanic landforms and landslides, mainly on the hydrographic right bank of the Tiber River (Del Monte et al., 2016). Human activities in the three millennia of increasing urbanisation transformed the natural landscape with anthropogenic morphogenesis (Vergari et al., 2021), such as accumulation (infilling, dumping, channelising, embanking) and erosional processes (mining, quarrying, or excavating) (Delchiaro et al., 2025). Site-level effects of geomorphological alteration on local ELU and PNV were however not detached in the available documents.

2.2 Methodological framework

A stratified sampling design was employed to investigate site-level impacts of anthropogenic landforms and/or of other effects of the urban context on local PNV. Targeting the two most widespread ELU in the municipality of Rome Capital, a sample of wooded areas was investigated along a human disturbance gradient, within and around the city center. Surveys were conducted in 10m x 10m square plots subdivided into quadrants, with complete floristic inventories and plant species frequency assessment. Topsoils (0-30 cm) were also sampled from a subset of plots, for a

preliminary investigation of concurrent effects of geomorphological alterations on the physico-chemical characteristics of woodland surface substrate.
Differential shifts from PNV between high naturalness, low naturalness and altered site woodlands were thus investigated by canonical ordination of species against environmental variables, indicator species analysis, chorological, native status and life form composition, and α- and β- diversity. Finally, a linear model was set for quantifying and mapping the degree of ecological distance from local reference PNV.
The overall arrangement of the research is shown in Figure 2 and more specifically described in the following sections.

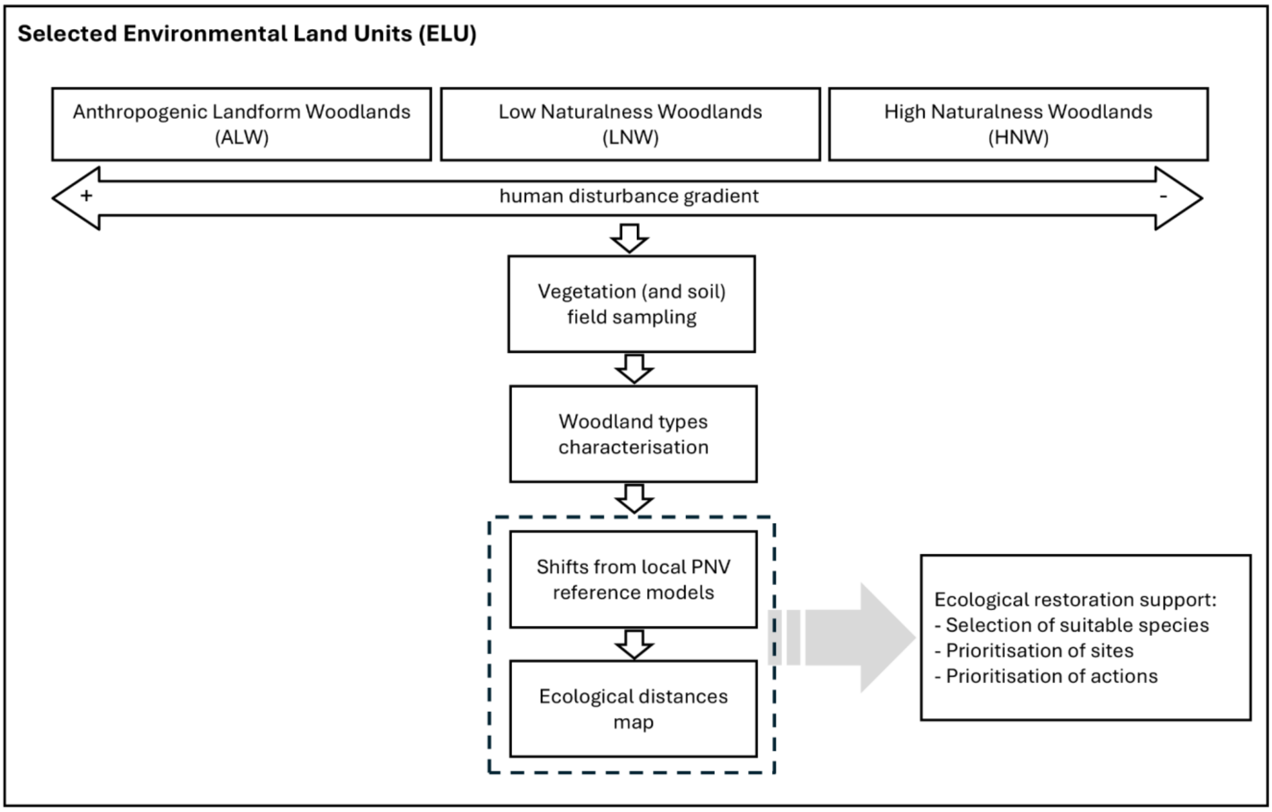

Figure 2: Research workflow and potential application of the outcomes in urban restoration planning and design.

2.2.1 Sampling design and data collection

Sites for vegetation (80 samples) and soil surveys (10 samples) were selected across the two most widespread ELU (40 vegetation samples each) in the Municipality of Rome Capital from the overall wooded areas with an extent of at least 20m×20m. Wooded areas were recognised in a GIS environment (QGIS, version 3.34.4) by overlapping a real vegetation map (1:25 000, MMU, 0.1 ha; CIRBFEP, 2013) with the European high-resolution layers on Forest Type (FTY) and Small Woody Features (SWF) (EEA, 2018, 2021). Among this initial pool, three different woodland types were thus distinguished according to documented anthropogenic alteration of geomorphology, surrounding landscape mosaic features, degree of naturalness, and importance for landscape ecological connectivity:

- Anthropogenic Landform Woodlands (ALW): Woodlands on anthropogenic landforms represented according to the different categories of human-induced processes outlined in the Italian national guidelines for geomorphological survey and mapping (e.g., fill, terraced slope, quarry, etc.) (Campobasso et al., 2018). The map of anthropogenic landforms was drawn in a GIS environment on the basis of multi-temporal datasets over the past two centuries (including historical topography, orthophotos, satellite imagery, and geognostic data), further refined with ancillary sources (such as archaeological reports, historical archives, and iconographical/artistic materials) and field observations (Del Monte et al., 2016).
- Low Naturalness Woodland controls (LNW): Woodlands located in landscape contexts similar to those of ALW as for urban green spaces extent and fragmentation, but on sites without evidence of geomorphological alterations. Attributes for surrounding landscape composition and fragmentation were retrieved from a classification of 1 km x 1 km grid cells, proposed for the investigation of biodiversity across national urban systems in Italy (Dondina et al., 2025).
- High Naturalness Woodland controls (HNW): Woodlands in well preserved condition were identified by considering compositional aspects and by assessing structural and functional features of ecological connectivity. Namely, a high naturalness status was assigned to those woodlands already recognised as mature-stage forest patches in the Real Vegetation Map of the Province of Rome (CIRBFEP, 2013), and also representing core areas under a structural perspective and important nodes for the functional connectivity. Core areas, i.e. focal habitats composed of forest pixels whose distance from non-forested areas is greater than a specified edge width (Saura et al., 2011), were identified with a Morphological Spatial Pattern Analysis, performed using the GUIDOS toolbox software (Vogt, 2022). The respective importance as ecological network nodes was instead assessed using the Conefor 2.6 software, by calculating the reduction in the Probability of Connectivity index (PC) due to node removal, expressed as a percentage (Saura et al., 2007). The PC was previously calculated for the entire municipal territory using a threshold dispersal distance of 115 meters, which corresponds to the average dispersal capacity of dominant mature-stage species in the study area (i.e. *Quercus* spp.; Kurek et al., 2019).

The three types were expected to guarantee a representative sample for the investigation of eventual shift in woodland successional stages due to geomorphological alteration (ALW; 20 surveys), with respect to potential effects of other disturbances related to the urban context (LNW; 20 surveys) and to near-natural conditions (HNW; 40 surveys). Plantations and managed green urban areas were instead excluded from the analysis, as supposed to fall under direct human control of vegetation assemblages and dynamics. Surveys were performed with the square method, so that 10m x 10m (100 m^2) plots were marked in the field, at least 10 meters apart from the edge, divided into four quadrants of 5m x 5m, and all occurring plants listed for each quadrant. Species were named according to Bartolucci et al. (2024) and Galasso et al. (2024), except for those of the genus *Quercus*, which followed Pignatti et al. (2017). Given the recognised influence of definite environmental and landscape features on plant distribution, a set of additional variables was also retrieved/recorded for each sampling plot. Environmental variables included: mean monthly temperature and precipitation for the period 2011-2020, mapped at a resolution of 1km x 1km (ISPRA, 2024); lithological type (Regione Lazio, 2012); elevation, derived from a 5 meter Digital Elevation Model (DEM) (Regione Lazio, 2002; Hutchinson et al., 2011); heatload exposition index (McCune & Keon, 2002); and slope and geographic coordinates (measured in the field). Geomorphological alteration variables included:

An_landform, reporting whether the woodland was set on anthropogenic or natural landform; and Landform type (according to Szabò et al., 2010), specifying if the landform was i) natural, ii) anthropogenic for accumulation (positive), iii) anthropogenic for erosion (negative), or iv) mixed (negative/positive, i.e. terracing). Landscape variables included: percentage of landscape (PLAND) covered by either artificial or natural (forest) surfaces at three different radii from the square plot (250 m, 500 m, 1000 m); overall patch area containing the sampled plot and its Fractal dimension index, Shape-area and Perimeter-area ratio (McGarigal, 2015); and age of vegetation cover since 1950, 1980, or 2001, based on multi-temporal land cover data retrieved from Frondoni et al., (2011). For a representative subset of woodland plots within a same ELU, ranging from HNW and LNW up to some ALW across a range of different age of vegetation cover, soils were also sampled according to the Lucas method (EC, 2022), by using a gouge to collect the top 30 cm after removing the litter. Samples were subsequently analysed to explore main pedological features thus considering for soil texture, pH, electrical conductivity, total calcium carbonate, active calcium carbonate, organic matter, total nitrogen, phosphorus (ppm), calcium (ppm), magnesium (ppm), potassium (ppm), sodium (ppm), C.E.C (meq), base saturation (%), and C/N ratio (Sabatini et al., 2015; Bart et al., 2014). Resulting parameters were thus individually graphically represented for a preliminary interpretation of relationships with different woodland types.

2.2.2 Characterisation of woodland types and quantification of ecological distances from local PNV

Woodland communities were characterised through indicator species analysis using the IndVal index, with a search for indicator species of pooled groups, along with α- and β- diversity, chorological types, native status, and life forms (Borcard et al., 2011; Midolo et al., 2024; Abbate et al., 2012). The hypothesis that anthropogenic landforms may significantly impact on the structure and naturalness of woodland successional stages was tested using Redundancy Analysis (RDA), performed on Hellinger-transformed indicator species data and their relationships with environmental and landscape features, considered as explanatory variables (Bisigato et al., 2016). A forward stepwise procedure was adopted to select only significant variables, while a Variance Inflation Factor (VIF) analysis was performed to evaluate multicollinearity and remove the highly correlated ones. Specifically, the effect of anthropogenic landforms was assessed by conditioning it in the RDA, i.e. by removing its contribution from the model and observing the decrease in total explained variance (Borcard et al., 2011).

To quantitatively measure the differential distance between the three woodland types and local PNV, a three-stage analysis was employed. First, a Principal Component Analysis (PCA) was performed on the Hellinger-transformed species data filtered for indicator species. Second, the Principal Components scores of HNW, for separate surveys between volcanic and sandy ELU, were averaged to generate reference PNV centroids, and the Euclidean distance from PNV centroids calculated for each survey. Third, a Generalised Additive Model (GAM) was fitted to explain these distances using environmental explanatory variables. Akaike Information Criterion (AIC), R^2, Root Mean Square Error (RMSE) and Leave-One-Out Cross-Validation (LOOCV) were employed for model selection.

In turn, estimated GAM coefficients were used to predict and map the ecological distance from PNV across the overall study area, with a spatial resolution of 30m.

All the analyses were performed using RStudio software (version 4.4.2) (R Core Team, 2024).

3. Results

3.1 Characteristics of woodland types on anthropogenic landforms with respect to low- and high-naturalness controls

The 80 vegetation surveys are represented in Figure 3. The comprehensive database with species lists and environmental and landscape variables for each vegetation survey is provided in Supplementary material A. Soils were preliminarily investigated for the sandy ELU, with two samples collected from non-altered substrata (one corresponding to a HNW and another to a LNW), and eight from substrata under anthropogenic landform alteration (Figure 3).

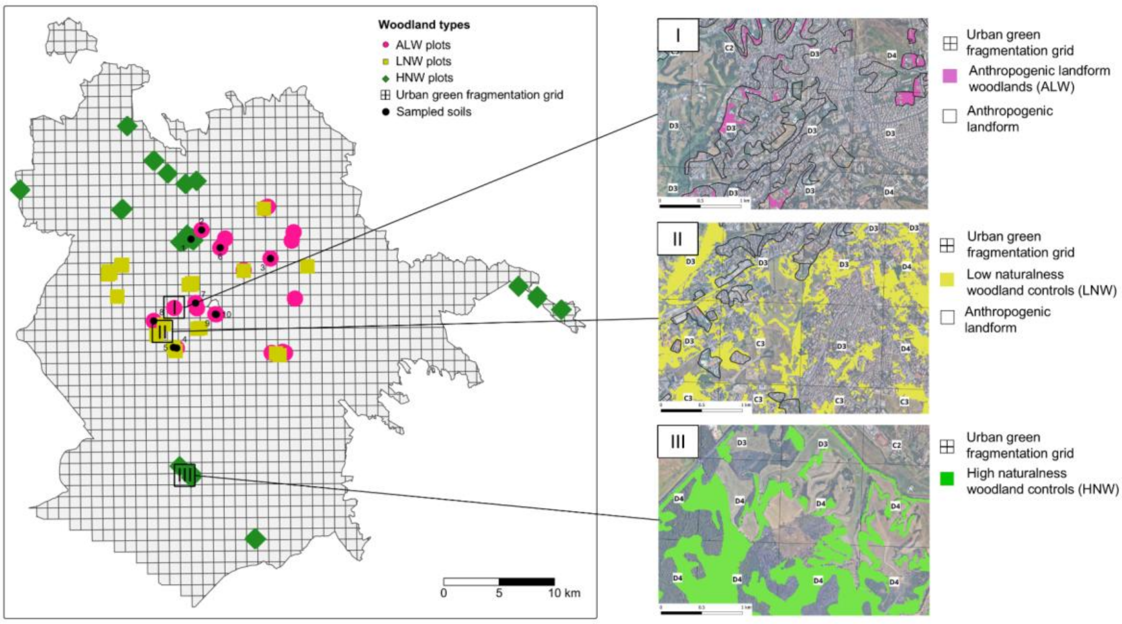

Figure 3: Left side: sampled plots in the study area, coloured by woodland type (ALW, LNW, and HNW). The reference 1km x 1km grid is also represented. Right side: illustrative satellite views of woodland types, along with surrounding landscape mosaic and grid cell categories (coded from A to D according to increasing extent of green spaces, and from 1 to 4 according to decreasing degree of fragmentation; Dondina et al., 2025)

Anthropogenic landform characteristics

For the 20 ALW sampled sites, anthropogenic landform characteristics (Table 1) resulted to be driven by three main processes:

A. Accumulation of anthropogenic deposits and techno-soils. Human activities, such as filling surfaces, embankments, and ridges (often created for motorways or railways), lead to the deposition of a diverse and heterogeneous mix of materials. This material includes artifacts, broken brick, concrete, reworked or redeposited natural sediment, and other debris. The resulting layer adds significant thickness, thus modifying the local outcropping lithology and the associated soils.

E. Anthropogenic erosion. This process involves large-scale removal of terrain due to activities like industrial building excavations, quarrying/mining, and surface collapse over underground artificial cavities. These movements modify the type and continuity of the outcropping lithology and can significantly alter the distance to the water table.

M. Mixed Processes, due to combined erosion and accumulation, typically resulting in slope terracing for construction or agricultural purposes.

All these transformations to natural morphology directly modified slope incline, curvature, and orientation at the site level. Transformation processes were registered for both sub-horizontal summit surfaces (plateaus) and slopes, and affected both the considered ELU, either on volcanic or sandy substrata.

Table 1: Characterisation of individual anthropogenic landform types for the 20 ALW surveys. Process type E corresponds to erosion landforms, A to constructional landforms, and M to mixed erosion/constructional landforms. Artificialisation period indicates the time span during which the morphological transformation occurred.

Process type	Landform type	Area [hm^2]	Artificialisation period [year]
M	Terraced slope by: building industry	3,01	1970-1999
M	Terraced slope by: building industry	54,34	1970-1990
E	Excavation surface	3,10	1920-1950
M	Terraced slope by: building industry	41,18	1970-1990
M	Terraced slope by: building industry	25,84	1970-1990
M	Terraced slope by: building industry	153,87	1970-1990
A	Ridge created for: motorway / railway / dumping	6,56	1872-1924
E	Abandoned quarry/mine	3,13	1924-1934
A	Embankment/infill	3,52	1936-1949
A	Filling surface in: ancient quarry	0,82	>1924
E	Abandoned quarry/mine	7,31	<1924

M	Terraced slope by: building industry	7,04	1949-1970
A	Filling surface in: ancient valley	8,05	1970
A	Filling surface in: ancient valley	3,29	1970-1990
A	Filling surface in: ancient valley	3,14	2009
A	Filling surface in: ancient valley	3,01	1990-1999
A	Filling surface in: ancient valley	2,03	1924-1936
E	Urbanised area potentially subject to subsidence	7,07	>1894; <1924
E	Urbanised area potentially subject to subsidence	2,55	>1894; <1924
E	Urbanised area potentially subject to subsidence	1,74	>1894; <1924

Indicator species

Indicator species analysis (Table 2) revealed that physiognomic oak species of the local PNV, such as *Quercus cerris* and *Q. ilex* for volcanic ELU, and *Q. suber* and *Q. pubescens* for sandy ELU, are typical for both LNW and LNW&HNW, along with a remarkable contingent of nemoral species. HNW and LNW are however distinguished from each other by ruderal and earlier successional species prevailing in the latter type, such as *Prunus spinosa*, *Torilis africana, Silene latifolia*, *Avena barbata* and *Hordeum murinum*. Among indicator species of ALW instead, no physiognomic species of the PNV mature stages occur, but a rich contingent of exotic and often invasive, such as *Robinia pseudoacacia, Ailanthus altissima, Yucca gloriosa, Ligustrum lucidum,* followed by ruderal and nitrophilous ones, such as *Parietaria judaica, Euphorbia peplus* and *Oxalis stricta.* Moreover, HNW and ALW, at the two ends of the considered disturbance scale, have no indicator species in common. Complete results, for the analysis also performed by the two distinct ELU, are reported in the Supplementary material B.

Table 2: IndVal multipatt analysis for the three woodland types ALW, LNW, and HNW.

Woodland types	Species	indval	p-value
ALW			
	Robinia pseudoacacia	0.68	0.00
	Ligustrum lucidum	0.66	0.00
	Parietaria judaica	0.54	0.00
	Ailanthus altissima	0.51	0.00
	Anisantha sterilis	0.50	0.00
	Bryonia dioica	0.47	0.00
	Fumaria capreolata subsp. *capreolata*	0.46	0.01

	Oxalis stricta	0.45	0.01
	Oloptum miliaceum	0.40	0.02
	Chamaerops humilis subsp. *humilis*	0.39	0.02
	Euphorbia peplus	0.39	0.02
	Yucca gloriosa	0.39	0.02
	Anthriscus sylvestris subsp. *sylvestris*	0.38	0.04
	Rubus hirtus	0.37	0.02
LNW			
	Quercus pubescens subsp. *pubescens*	0.54	0.02
	Prunus spinosa subsp. *spinosa*	0.53	0.00
	Torilis africana	0.50	0.00
	Quercus suber	0.49	0.03
	Rosa arvensis	0.39	0.03
HNW			
	Ruscus aculeatus	0.92	0.00
	Fraxinus ornus subsp. *ornus*	0.80	0.00
	Asplenium onopteris	0.73	0.00
	Stellaria media subsp. *media*	0.69	0.00
	Cyclamen hederifolium subsp. *hederifolium*	0.69	0.00
	Alliaria petiolata	0.68	0.00
	Phillyrea latifolia	0.65	0.00
	Cornus mas	0.58	0.00
	Ranunculus lanuginosus	0.55	0.00
	Anemone apennina	0.54	0.00
	Ligustrum vulgare	0.53	0.00
	Luzula forsteri	0.52	0.00
	Lamium purpureum	0.45	0.02
	Vicia incana	0.45	0.02
	Stachys sylvatica	0.45	0.03
	Chaerophyllum temulum	0.42	0.03
	Lonicera japonica	0.42	0.03
LNW & ALW			
	Silene latifolia	0.69	0.00
	Ulmus minor subsp. *minor*	0.66	0.00
	Urtica membranacea	0.65	0.02
	Arum italicum subsp. *italicum*	0.53	0.01
	Hordeum murinum subsp. *murinum*	0.49	0.01
	Avena barbata	0.45	0.02
HNW & LNW			
	Rubia peregrina	0.73	0.01
	Dioscorea communis	0.68	0.01
	Smilax aspera	0.66	0.00
	Quercus ilex	0.65	0.00
	Quercus robur subsp. *robur*	0.64	0.01
	Quercus cerris	0.60	0.04
	Acer campestre	0.56	0.01

Chorological types, native status and life forms

Chorological spectra for the three woodland types in the two different ELU are illustrated in Figure 4. HNW and LNW show a greater occurrence of Atlantic, Mediterranean and Paleotemperate species. LNW and ALW are mainly characterised instead by a higher proportion of exotic (as also confirmed by the analysis of native status; Table 3) and Turanian-Mediterranean species. The composition of woodlands of the same type does not significantly differ between the two environmental units, except for Steno-Mediterranean species, which are less abundant in HNW than in LNW and ALW on sandy substrates compared to volcanic ones.

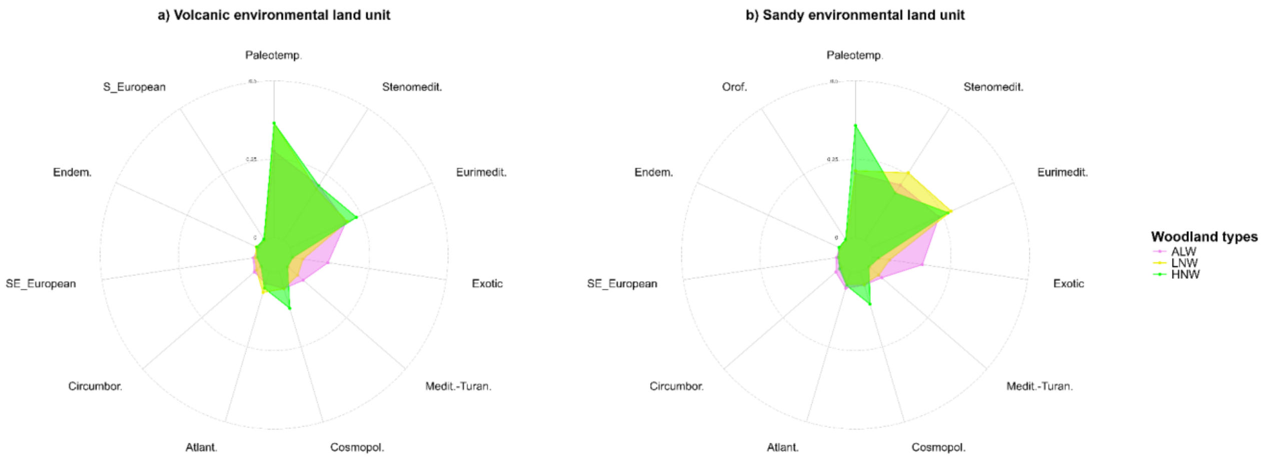

Figure 4: Radar chart of chorotypes, based on percentage composition within the three woodland types HNW, LNW and ALW on volcanic (a) and sandy ELU (b).

Table 3: Percentage of Native, Exotic and Archaeophyte species in the three woodland types per ELU.

	Native (%)	Exotic (%)	Archaeophyte (%)
Volcanic ELU			
ALW	87	12	1
LNW	96	3	1
HNW	100	0	0
Sandy ELU			
ALW	84	15	1
LNW	94	6	1
HNW	98	1	1

Life form spectra show a higher proportion of geophytes in HNW and of therophytes in LNW and ALW (Figure 5), with no marked differences between the two ELU except for geophytes (more frequent on sandy substrates than on volcanic counterparts). The proportion of therophytes was generally lower across all forest types on sandy substrates. However, the relative distribution among the three vegetation groups remained similar to that observed in volcanic ELU.

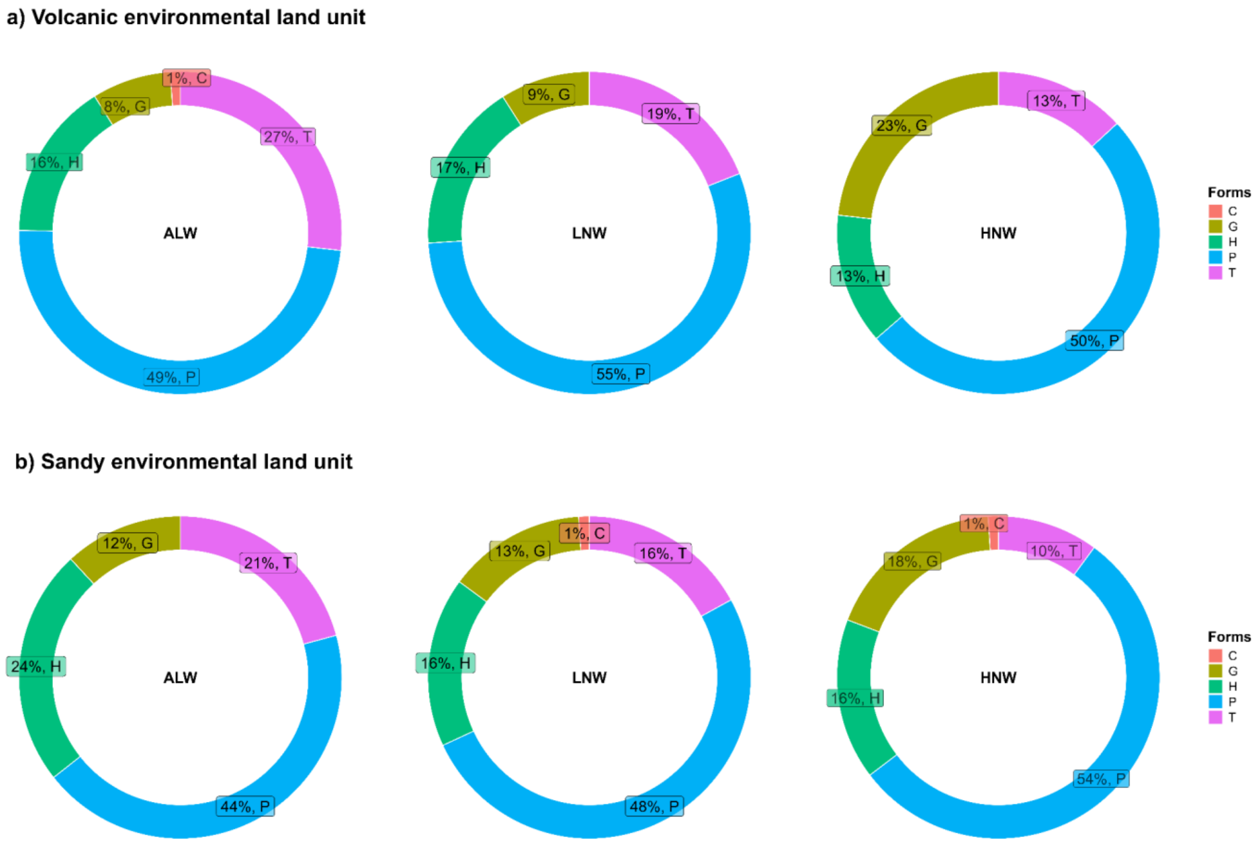

Figure 5: Doughnut chart of life forms, based on percentage composition within the three woodland types ALW, LNW, and HNW on volcanic (a) and sandy ELU (b).

Diversity metrics

The α-diversity, measured as species richness, evenness and Shannon diversity, is higher in HNW, followed by ALW and LNW, as shown in Table 4. Given the higher standard deviation in all α-diversity metrics, ALW exhibit greater variability than the other woodland types. On the other hand, β-diversity shows a higher species turnover in LNW than in ALW and HNW, also in this case with no significant differences between the two ELU.

Table 4: α-diversity (Species richness, Eveness, and Shannon index) and β-diversity measures of the three woodland types HNW, LNW and ALW, in each of the two ELU.

Type	Species richness (mean)	Evenness (mean)	Shannon index (mean)	β diversity
Volcanic ELU				
ALW	19.00 ± 7.48	0.65 ± 0.08	1.89 ± 0.48	11.94
LNW	16.90 ± 4.98	0.59 ± 0.07	1.67 ± 0.33	13.43
HNW	22.00 ± 5.53	0.94 ± 0.03	2.87 ± 0.25	10.31
Sandy ELU				
ALW	18.90 ± 7.98	0.60 ± 0.12	1.73 ± 0.56	12.01
LNW	16.40 ± 6.38	0.59 ± 0.07	1.64 ± 0.41	13.84
HNW	21.80 ± 5.17	0.80 ± 0.04	2.46 ± 0.29	10.41

3.2 Ecological distance from local PNV in altered and not altered site woodlands

Anthropogenic landforms and environmental drivers effects on woodland successional stages

A recognisable effect of anthropogenic landforms on woodland vegetation was confirmed by the RDA model, which explained 32% of the constrained variance, corresponding to an adjusted R^2 of 0.24. When the *An_landform* variable was removed from the model, the adjusted R^2 decreased to 0.14, representing a 42 % loss of the overall explained variance. The Global test of significance with P-value <0.05, rejects the null hypothesis according to which no relationships exist between the response data (site's species composition) and the explanatory variables, confirming that the measured ecological variables represent statistically meaningful drivers of community assembly. The RDA biplot confirms a disturbance pattern among the three groups, which is even more evident when observing the first and third axis biplot (Figure 6). The first axis is characterised by landscape variables, i.e. proportion of land cover classes (*PLAND_art_1000* in opposition to *PLAND_wood_250*), legacy (*Veg_age*), perimeter-area ratio (*PARA*), along with environmental variables, i.e. precipitation and temperature, and geomorphological variables, i.e. by the two classes of *An_landform* ("Natural landforms" and "Anthropogenic landforms"). Higher values of artificial land cover and perimeter-area ratio, as well as lower values of forest land cover and of *Veg_age*, are associated with LNW and ALW. The only variable that seems to distinguish the ALW from LNW is *An_landform*, since ALW are distributed around the "Anthropogenic landform" class. The second and third axis are instead mainly characterised by environmental variables alone, such as Heatload, indicating the effect of topographic aspect on species assemblages, and lithology, with Sands and volcanic rocks resembling the effects of the two local ELU.

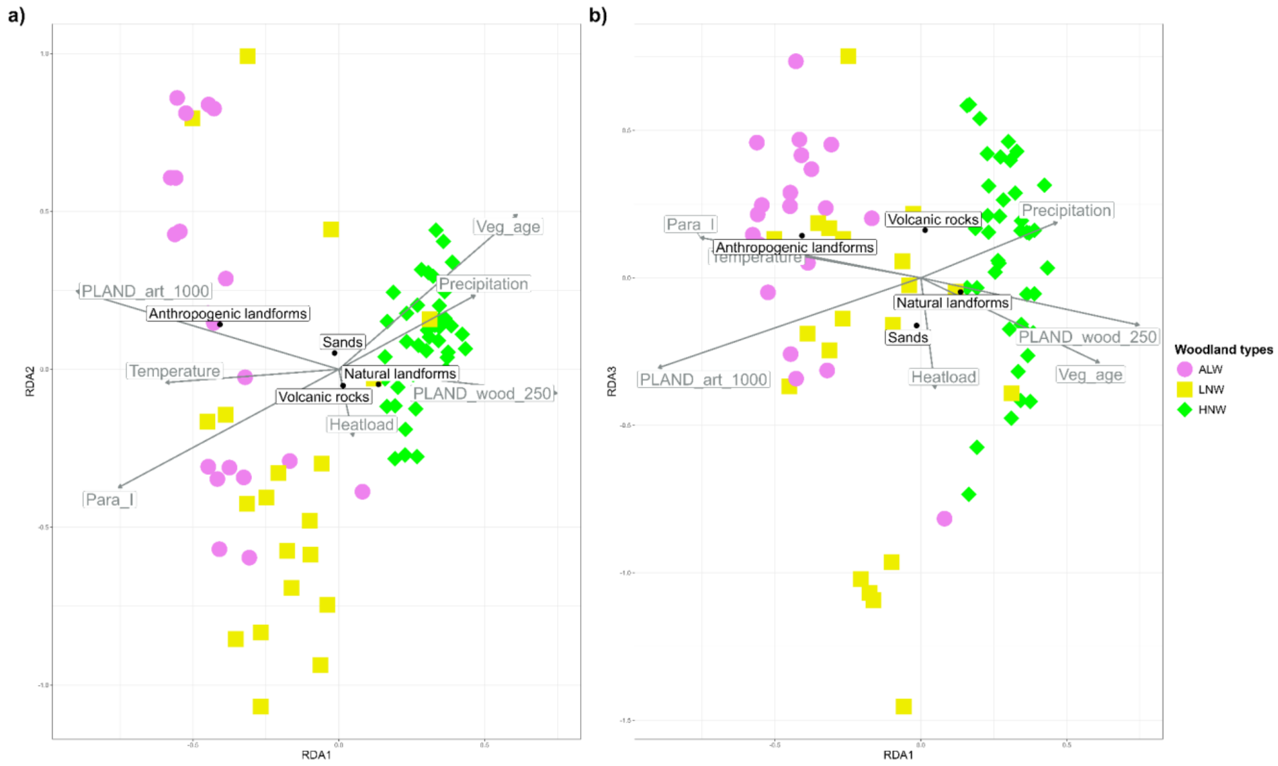

Figure 6: RDA biplot with environmental variables (vectors for quantitative variables and points for qualitative variable classes) over first and second axes (a) and the first and third axes (b). Woodland samples are coloured by type: ALW, LNW, and HNW.

Measurement and spatial modelling of ecological distances from reference PNV

Ecological distances of sampled woodlands from reference PNV centroids, adopted as mature stage models for each ELU, are shown with radar plots in Figure 7a and 7b. The average ecological distance of the three woodland types from the reference baseline is also exemplified with linear plots in Figure 7c and 7d. Both the woodland types associated with the urban context, i.e. LNW and ANW, resulted to be much more distant from reference PNV than HNV and quite close to each other. The effect of geomorphological alteration (distance between ALW and LNW) is however discernable and more marked for volcanic than for sandy ELU.

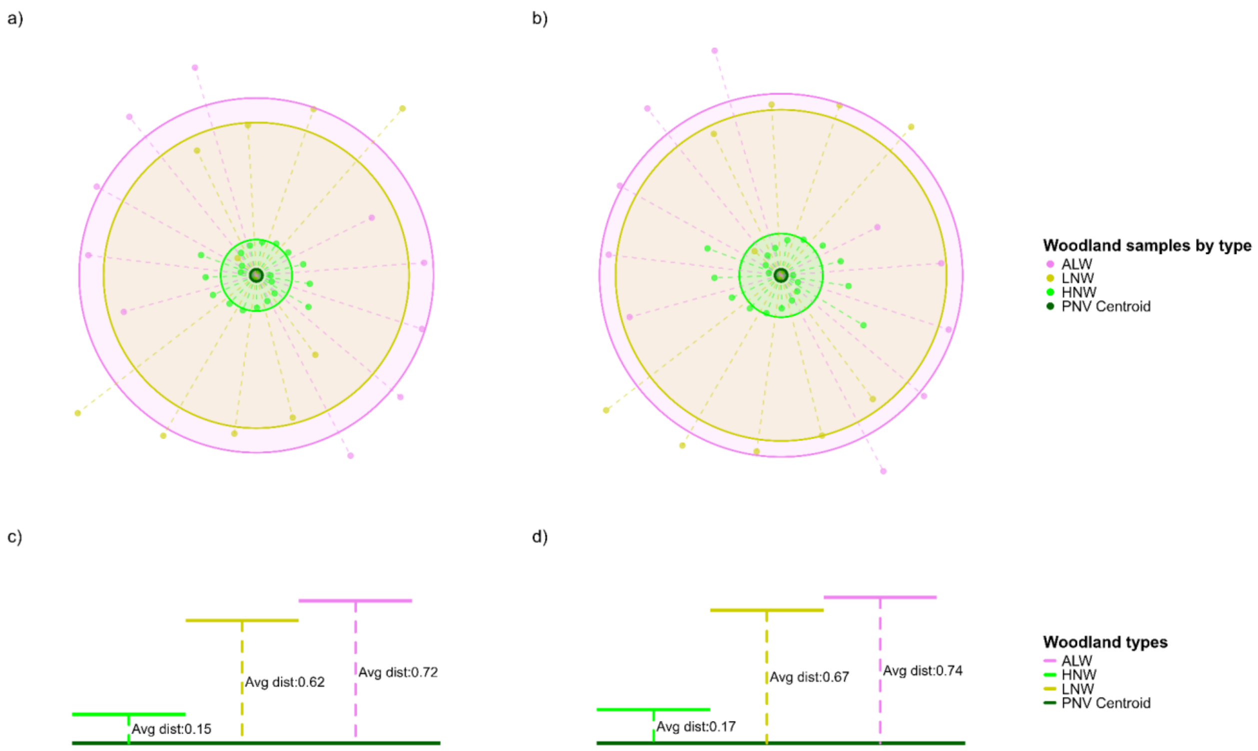

Figure 7: Radial and linear plot of ecological distances from PNV centroids for HNW, LNW and ALW types in each analysed ELU. Radial plots for volcanic ELU (a) and sandy ELU (b) show the distance between each sampled woodland and the respective PNV centroid, with average distances coloured by woodland type represented as circles around the centroid. Linear plots for volcanic ELU (c) and sandy ELU (d) report the average distances per woodland type from the reference model.

The selected GAM can be expressed by the formulas:

$$PNV\ distance\ \sim Gamma(\mu_i, \theta)$$

$$log(\mu_i) = \beta_0 + \beta_1 \cdot years_i + \beta_2 \cdot PLAND\_art\_1000_i + s(latitude_i, longitude_i, df = 29)$$

Where μ_i represents the expected value of PNV distance for survey *i,* modeled through a log link, and θ is the dispersion (shape) parameter of the Gamma distribution, estimated from the data and assumed constant across observations. $years_i$ indicates the years of naturalisation of the site *i*, $PLAND\_art\_1000_i$ is the percentage of artificial land cover and *s* indicates a smooth nonlinear function to include spatial variability in the model.

The model explains 74.0% of the deviance (pseudo R^2 = 0.67) and indicates that the years of naturalisation lead to a decrease in the distance from the reference model, while an increase in the percentage of surrounding artificial land cover is associated with higher ecological distances (Table 5; Supplementary material B). In Figure 8, the prediction is extended to the overall extent of the investigated ELU, allowing the expected ecological distance from PNV to be spatially modelled across the city, independently from actual land cover at each site. Higher ecological distances were primarily predicted at locations characterised by recent naturalisation and very artificialised landscape context, such as the city centre. Conversely, lower ecological distances were modelled for residual natural and agricultural zones, especially occurring in suburban sectors of the municipality but also at some central locations. These areas consistently exhibit low levels of artificialisation of the surrounding landscape mosaic, though age of vegetation cover may vary so that also agricultural surfaces were estimated to maintain moderate ecological distances from natural baselines.

Table 5: Final GAM obtained by supervised selection.

Predictor	Estimate	Std. Error	t value	p-value
(Intercept)	-0.843	0.193	-4.347	4.71E-05
PLAND_art_1000	0.015	0.003	4.705	1.28E-05
Veg_age	-0.012	0.003	-4.180	8.48E-05
Smooth terms	edf	Ref.df	f value	p-value
s(x,y)	8.68	11.60	3.150	1.1E-03
Deviance explained	74.0 %			
AIC	-75.2			
R^2	0.67			
RSME	0.16			
LOOCV error	0.04			

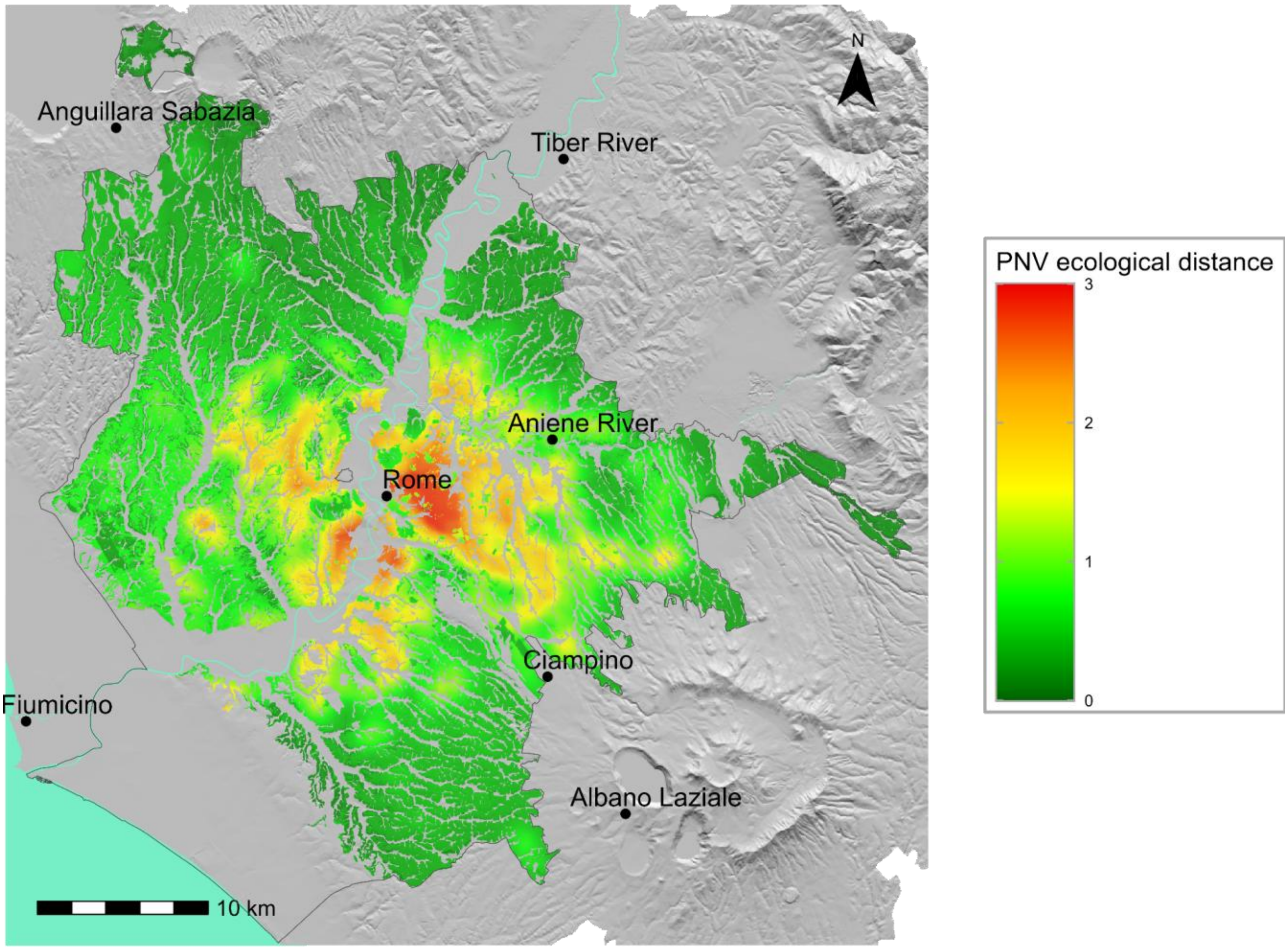

Figure 8: Map of ecological distances (adimensional) from reference PNV in the considered ELU of the Municipality of Rome Capital, based on the selected GAM. Independently from the actual land cover, highest values, in red, represent

areas where woodland successional stages are estimated to diverge most from the reference model, while lowest values, in dark green, represent areas where woodland successional stages are estimated to be most consistent with the reference model.

3.3 Soil characteristics

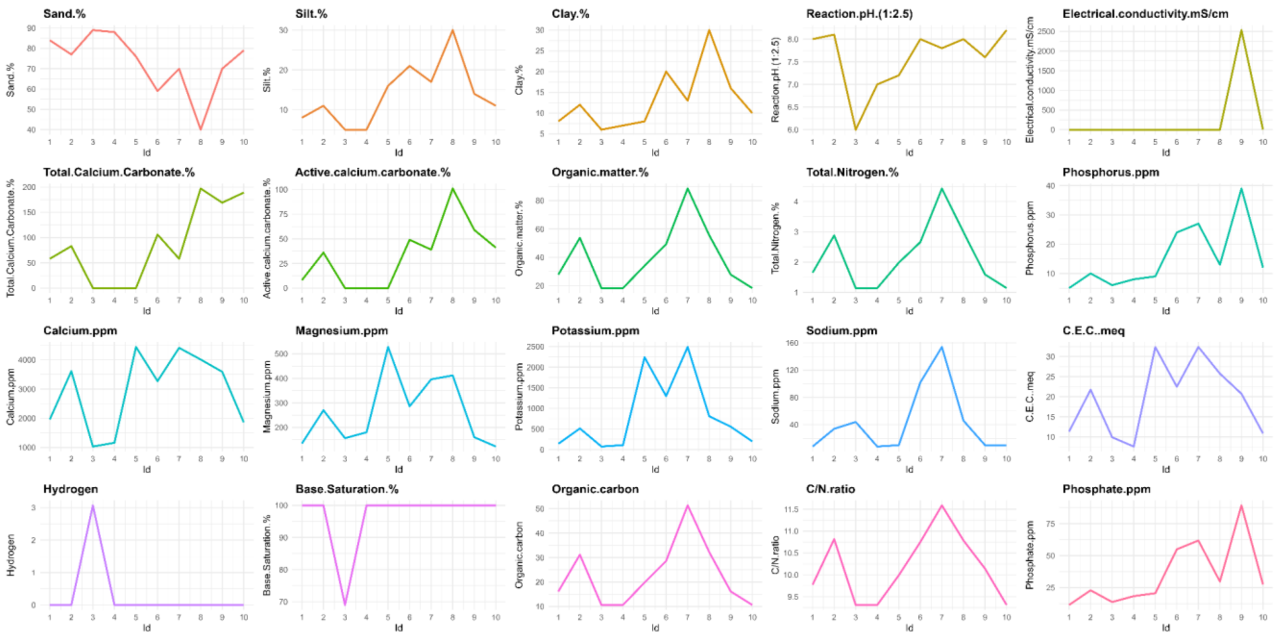

Figure 9: Chemical and physical properties of soils underpinning HNW, LNW and ALW in sandy ELU (Id = sample number).

The analysis on soils underpinning the three types of woodlands within the sandy ELU allowed some first evidence to be pointed out (Figure 9). Notwithstanding earthworks, almost all the soils in altered sites were found to be either sandy or sandy-loam as the less disturbed ones (sample nr 1 and 2), except for one sample (nr 8) with higher clay and silt proportion. Less disturbed soils presented similar physical characteristics, even if nitrogen and organic matter were higher in LNW (nr 2) than in HNW (nr 1). ALW soils showed a marked internal variability with some samples (nr 3, 4, and 10) more similar to those collected in HNW, especially for physical features (in line with previous findings about ecological distances between woodland types in the sandy ELU). The remaining five samples showed instead quite different physico-chemical characteristics, having higher percentages of clay and silt, as well as higher levels of organic matter, nitrogen, and phosphorus probably due to fertiliser residue. Furthermore, two samples (nr 5 and 7) stood out with significantly higher potassium levels, up to five times greater than the others, indicating a probable contamination with volcanic soils. Another sample (nr 9) exhibited high electrical conductivity induced by soluble anions mainly represented by sulfates (90%) due to high levels of gypsum, probably due to construction wastes used as filling materials.

4. Discussion

Urban ecology has gained significant momentum in recent years, with an increasing number of studies on urban biodiversity at different levels of complexity (Rega-Brodsky et al., 2022). Previous research has extensively documented the impact of land-use changes, habitat fragmentation, and biological invasions on species within and around cities (Celesti et al., 2006; Haddad et al., 2015; Boscutti et al., 2022). Many studies were also focused on vegetation composition and condition along rural-urban gradients and some on the definition of novel ecosystem conceptual models, but often lacking of quali-quantitative assessments of alteration degree (Salvati & Zitti, 2012; Malkinson et al., 2018; Knapp et al., 2022). Moreover, there are still few examples of methodological approaches designed and applied to measure the impact on the assembly, and especially on the dynamics, of plant communities by considering multiple aspects of urbanisation (Kumar et al., 2006; Williams et al., 2015; Andrade et al., 2021). The present study contributed, in particular, to filling the knowledge gap that still concerns the effects of anthropogenic landform alteration on both actual vegetation communities and present PNV (*sensu* Somodi et al., 2021).

Namely, the hypothesis that deep alteration of the substrate persistently affects environmental conditions and, in turn, determine a wider shift from potential natural state than other urban disturbances alone, was confirmed from the experimental evidence in the analysed case study. More interestingly however, these differences have been qualified in terms of species composition and diversity for woodlands on anthropogenic landforms, with respect to both controls under other kinds of urban pressures and controls in a more near-natural condition. For all these types of woodlands, the ecological distance from PNV was also quantified, always based on the observed species composition, and spatialised across the overall analysed environmental units, according to associated landscape variables.

Vegetation characteristics, in terms of indicator species, chorological types, life forms, non-native proportion and α- and β-diversity, revealed a clear pattern along the disturbance gradient of the three woodland types. As for these features, HNW, selected among the core areas of most ecologically connected nodes, demonstrated that woodlands under poorly disturbed conditions align well with PNV models. The indication value of the node *importance* metric for identifying well-conserved forest patches was therefore empirically proved, and the recognised role of ecological connectivity in maintaining ecosystem integrity confirmed (Pascual-Hortal & Saura, 2008), also in an urban context. LNW exhibited instead a more pronounced legacy of the previous traditional agricultural landscape, with higher rates of ruderal and light-demanding species typical of abandoned fields, as also stated in Prach et al. (2001). Being more isolated than HNW, these plant communities also showed a lower α-diversity probably due to local extinction (Ramalho et al., 2014). The differences between LNW and HNW can thus be related to a landscape effect, even though some species typical of mature PNV stages still occur, suggesting the capacity of these semi-natural areas to recover spontaneously over time depending on environmental and historical factors (Motta et al., 2009; Cuddington, 2011). ALW were found to be more impacted than LNW as for all the assessed features. Indicator species typical of PNV models were never found in these communities, likely due to the combined effects of local extinction and long-term removal of the seed bank resulting from landscape configuration and anthropogenic alteration of the sites. However, ALW had a higher α-diversity compared to LNW, due to a higher entry rate of exotic species in the former that counterbalances the local extinction of native species characterising both typologies. While the relationship between the high proportion of non-native species and high levels of urbanisation has been already well documented (Polce et al., 2011; Boscutti et al., 2022), the observed frequency in Mediterranean-Turanian species represents an original finding. Within the general bioclimatic and biogeographic framework of the study area, these species may be facilitated by a higher light availability, due to more open canopy cover and less complex structure of disturbed woodlands with respect to natural forests (as already observed for example by Williams et al., 2015). Still consistent with previous research on urban floras (Preston et al., 2000; Williams et al., 2005; Knapp et al., 2008), both disturbed woodland types showed a relative decrease in geophytes and an increase in therophytes with respect to HNW. The observed shift was however more pronounced in ALW, suggesting an additional effect of geomorphological disturbance, probably due to soil alteration prior to or during landform modification, subsequent removal of geophytes subterranean regenerative structures (e.g., rhizomes or bulbs), and indirect facilitation of therophytes, which persist through seeds under unfavourable conditions. Additionally, the absence of PNV typical species and the prevalence of non-native plants among the indicator ones suggest passive recovery may not represent the best solution for forest restoration in ALW. In fact, at these sites succession could have been permanently diverted toward novel stable communities (Prach et al., 2001; Kowarik & Von der Lippe, 2018), far from local reference models, and altered soils may sometimes significantly hurdle the development of root systems of native trees with respect to more competitive and ruderal exotics (Gillini et al., 2025).

The different degree of disturbance for the three woodland types was corroborated by the RDA, whose reliability is consistent with many multivariate ecological studies (Peres-Neto et al., 2006; Bisigato et al., 2016; Solé-Senan et al., 2018), where a substantial portion of variation in species data is often attributed to unconsidered environmental factors, stochasticity, and biotic interactions.

The distance of investigated woodlands from reference mature stages, modelled with PNV centroids, further confirmed the observed gradient, with higher observed alteration in ALW than in LNW types. Compositional shifts of LNW, mainly ascribable to landscape-level fragmentation and historical land use, are therefore discernibly added to those due to site-level geomorphological alteration in ALW. Local environmental preconditions, here modelled by ELU, seem also to play a significant role in the dimension of the shift, with original sandy substrata less affected than volcanic ones also for soil physical-chemical features. Such preliminary evidence should be however further tested by increasing the number of vegetation and soil samples per different extent/magnitude of modified substrata and per additional ELU types, such as alluvial valleys and coastal plains.

Combined landscape-level and site-level effects, resumed by the selected GAM, especially highlighted the positive importance of the age of vegetation cover, which in turn relates to long-lasting processes such as soil recovery after anthropogenic interventions (Bart & Devenport, 2015). The opposite influence on shifts from PNV of surrounding artificial cover confirmed instead the diffuse negative effect of the urban matrix, which has already been related with propagule pressure by exotic species (Heinrichs & Pauchard, 2015) and limitation in the dispersal of native ones (Lopez et al., 2018). As an additional result from herein research, the percentage of artificial cover also represents a reliable proxy for the anthropogenic alteration of landforms, and usefully contributed to the spatial modelling of these multiple-level effects.

The map of expected shifts from natural baselines (PNV types) not only represents a synthesis of the research results, but also provides a complementary approach to the analysis and representation of the urban-rural gradient and its determinism on ecosystem extent and condition (Medley et al., 1995; Kaminski et al., 2021). Practical implications, with respect to current challenges for ecosystem restoration in cities, include a better understanding of the deviation from natural reference models (Blasi et al., 2025) and, therefore, a more in-depth indicative capacity on where the local potential vegetation is expected to spontaneously recover and where instead is expected to be permanently replaced. This knowledge may effectively support a fine-scale prioritisation of sites to be alternatively conserved or restored and respective actions, spanning from the selection of the right species for urban forestry and tree planting to a better balanced choice between native species that are typical of natural forests and those that, even if not native, still embody important cultural values for historical cities (e.g *Pinus pinea* and *Platanus x hispanica* in Rome; Caneva et al., 2020).

Although satisfying results already emerged, the research could have been limited by the quite few number of vegetation surveys, with an increased sensitivity to outliers and a potentially underestimated effect of type (constructional, erosional, or mixed) and magnitude of morphological alterations. Supplementary geomorphological investigation could help identify additional suitable sites to sample ALW, along with a more in-depth understanding of urban geo-morphodiversity, due to alterations that alternatively articulate or flatten the relief (Burnelli et al., 2025), and an improved interpretation of the effects on spontaneous woodlands according to quantitative thresholds posed by transformation magnitude (e.g. in terms of slope shape changes, accumulation thickness, soil and drainage changes).

Concomitantly, the preliminary results derived from soil investigation should be strengthened with additional field samplings, in order to extend evidence to other ELU and establish more robust baselines for comparing altered and reference conditions, also with respect to the not yet investigated concentration of heavy metals (Gillini et al., 2025).

5. Conclusions

Urban areas represent complex ecosystems where many concurrent factors influence and determine not only the composition but also the spontaneous dynamics of vegetation communities. The present research was aimed at qualifying and quantifying the ecological distance of spontaneous woodlands from local natural reference models due to anthropogenic geomorphological alteration.

Results confirmed that a distinct disturbance gradient is reflected by species composition across woodland on anthropogenic landforms as compared to other disturbed and near-natural controls. Ecological distances from PNV were thus measured and implemented in a GAM to model this phenomenon against a set of explanatory variables. The best-performing model was subsequently spatialised, obtaining a map of predicted distances across the entire study area. Based on these estimates, it was possible to identify areas more suitable for spontaneous or passive ecological restoration against those where active measures are especially needed for helping the recovery of reference ecosystems. An original support to the definition of the urban-rural gradient was therefore provided that could be useful to inform ecological restoration in cities and, in some cases, also to identify sites where preference for species with just historical and cultural values may override strict ecological integrity needs without important trade-offs.

References

1. Abbate, G., Iberite, M., Bonacquisti, S., Giovi, E., Iamonico, D., & Scassellati, E., 2012. Taxonomical and chorological diversity of native woody flora of Italy at regional scale. Bocconea, 24, 169-175. https://herbmedit.org/papers/fe7466c5-edc5-4ef6-95f7-c53499ca7f18.
2. Adla, K., Dejan, K., Neira, D., & Dragana, Š., 2022. Degradation of ecosystems and loss of ecosystem services. In One health, pp. 281-327. Academic Press. https://doi.org/10.1016/B978-0-12-822794-7.00008-3.
3. Andrade, R., Franklin, J., Larson, K. L., Swan, C. M., Lerman, S. B., Bateman, H. L., Warren, P. S., & York, A., 2021. Predicting the assembly of novel communities in urban ecosystems. Landscape Ecol 36, 1–15. https://doi.org/10.1007/s10980-020-01142-1.
4. Bart, D., & Davenport, T., 2015. The influence of legacy impacted seed banks on vegetation recovery in a post-agricultural fen complex. Wetlands Ecology and Management, 23, 405-418. https://doi.org/10.1007/s11273-014-9391-1.
5. Bart, D., Davenport, T., & Carpenter, Q. (2014). Stress and land-use legacies alter the relationship between invasive- and native- plant richness. Journal of Vegetation Science, 26(1), 80-88. https://doi.org/10.1111/jvs.12220.
6. Bartolucci, F., Peruzzi, L., Galasso, G., Alessandrini, A., Ardenghi, N. M. G., Bacchetta, G., Banfi, E., Barberis, G., Bernardo, L., Bouvet, D., Bovio, M., Calvia, G., Castello, M., Cecchi, L., Del Guacchio, E., Domina, G., Fascetti, S., Gallo, L., Gottschlich, G., Guarino, R., Gubellini, L., Hofmann, N., Iberite, M., Jiménez-Mejías, P., Longo, D., Marchetti, D., Martini, F., Masin, R. R., Medagli, P., Peccenini, S., Prosser, F., Roma-Marzio, F., Rosati, L., Santangelo, A., Scoppola, A., Selvaggi, A., Selvi, F., Soldano, A., Stinca, A., Wagensommer, R. P., Wilhalm, T., & Conti, F., 2024. A second update to the checklist of the vascular flora native to Italy. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(2), 219–296. https://doi.org/10.1080/11263504.2024.2320126.
7. Bisigato, A. J., Hardtke, L. A., del Valle, H. F., Bouza, P. J., & Palacio, R. G., 2016. Regional-scale vegetation heterogeneity in northeastern Patagonia: Environmental and spatial components. Community Ecology, 17(1), 8-16. https://doi.org/10.1556/168.2016.17.1.2.
8. Blasi, C., Capotorti, G., & Frondoni, R., 2005. Defining and mapping typological models at the landscape scale. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 139(2), 155-163. https://doi.org/10.1080/11263500500163629.
9. Blasi, C., Capotorti, G., Copiz, R., Guida, D., Mollo, B., Smiraglia, D., & Zavattero, L., 2014. Classification and mapping of the ecoregions of Italy. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 148(6), 1255-1345. https://doi.org/10.1080/11263504.2014.985756.
10. Blasi C., Frondoni R., 2011. Modern perspectives for plant sociology: The case of eco-logical land classification and the ecoregions of Italy. Plant Biosyst145: 30–37. https://doi.org/10.1080/11263504.2011.602747.
11. Blasi, C., Capotorti, G., Del Vico, E., Bonacquisti, S., & Zavattero, L., 2025. Native Flora and Potential Natural Vegetation References for Effective Forest Restoration in Italian Urban Systems. Plants, 14(15), 2396. https://doi.org/10.3390/plants14152396.
12. Borcard, D., Gillet, F., & Legendre, P., 2011. Numerical ecology with R, Vol. 2, p. 688. New York: springer. Numerical Ecology with R | SpringerLink
13. Boscutti, F., Lami, F., Pellegrini, E., Buccheri, M., Busato, F., Martini, F., Sibella, R., Sigura, M., & Marini, L., 2022. Urban sprawl facilitates invasions of exotic plants across multiple spatial scales. Biological Invasions, 24(5), 1497-1510. https://doi.org/10.1007/s10530-022-02733-6.
14. Boscutti, F., Sigura, M., De Simone, S., Marini, L., 2018. Exotic plant invasion in agricultural landscapes: a matter of dispersal mode and disturbance intensity. Appl. Veg. Sci. 21, 250–257. https://doi.org/10.1111/avsc.12351.
15. Burnelli M., Pica A., Del Monte M., Delchiaro M., Melelli L., Reame F., Vergari F., Alvioli M., 2025. Does anthropogenic morphogenesis contribute to geomorphodiversity in urban environments? Geomorphology, Volume 471, 109582. https://doi.org/10.1016/j.geomorph.2024.109582.

16. Campobasso, C., Carton, A., Chelli, A. et al., 2018. Aggiornamento ed Integrazioni delle Linee Guida della Carta Geomorfologica d'Italia alla Scala 1:50.000; ISPRA: Rome, Italy. https://www.isprambiente.gov.it/files2023/pubblicazioni/periodici-tecnici/q13_fascicolo_ii-ver1-0-1.pdf (accessed 14 May 2024)
17. Caneva, G., Bartoli, F., Zappitelli, I., & Savo, V., 2020. Street trees in italian cities: story, biodiversity and integration within the urban environment. Rendiconti Lincei. Scienze Fisiche e Naturali, 31(2), 411-417. https://doi.org/10.1007/s12210-020-00907-9.
18. CIRBFEP (Centro di Ricerca Interuniversitario Biodiversità, Fitosociologia ed Ecologia del paesaggio), 2013. Carta delle serie di vegetazione e vegetazione naturale potenziale della Provincia di Roma. https://geoportale.cittametropolitanaroma.it/progetti/banche-dati-vegetazionali-2013-2014/serie-di-vegetazione. (Accessed 22 October 2025).
19. E., Bartoli, F., Bazzato, E., Beccaccioli, M., Braglia, R., Bretzel, F., Brighetti, M. A., Brundu, G., Burnelli, M., Calfapietra, C., Cambria, V. E., Caneva, G., Canini, A., Caronni, S., Castello, M., Catalano, C., Celesti-Grapow, L., Cicinelli, E., Cipriani, L., Citterio, S., Concu, G., Coppi, A., Corona, E., Del Duca, S., Del Vico, E., Di Gristina, E., Domina, G., Faino, L., Fano, E. A., Fares, S., Farris, E., Farris, S., Fornaciari, M., Gaglio, M., Galasso, G., Galletti, M., Gargano, M. L., Gentili, R., Giannotta, A. P., Guarino, C., Guarino, R., Iaquinta, G., Iiriti, A., Lallai, E., Lallai, E., Lattanzi, M., Manca, S., Manes, F., Marignani, M., Marinangeli, F., Mariotti, M., Mascia, F., Mazzola, P., Meloni, G., Michelozzi, P., Miraglia, A., Montagnani, C., Mundula, L., Muresan, A. N., Musanti, F., Nardini, A., Nicosia, E., Oddi, L., Orlandi, F., Pace, R., Palumbo, M. E., Palumbo, S., Parrotta, L., Pasta, S., Perini, K., Poldini, L., Postiglione, A., Prigioniero, A., Proietti, C., Raimondo, F. M., Ranfa, A., Redi, E. L., Reverberi, M., Roccotiello, E., Ruga, L., Savo, V., Scarano, P., Schirru, F., Sciarrillo, R., Scuderi, F., Sebastiani, A., Siniscalco, C., Sordo, A., Suanno, C., Tartaglia, M., Tilia, A., Toffolo, C., Toselli, E., Travaglini, A., Ventura, F., Venturella, G., Vincenzi, F., & Blasi, C., 2020. More nature in the city. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 154(6), 1003-1006. https://doi.org/10.1080/11263504.2020.1837285.
20. Capotorti, G., Ortí, M. M. A., Copiz, R., Fusaro, L., Mollo, B., Salvatori, E., & Zavattero, L., 2019. Biodiversity and ecosystem services in urban green infrastructure planning: A case study from the metropolitan area of Rome (Italy). Urban Forestry & Urban Greening, 37, 87-96. https://doi.org/10.1016/j.ufug.2017.12.014.
21. Celesti-Grapow, L., Capotorti, G., Del Vico, E., Lattanzi, E., Tilia, A., & Blasi, C., 2013. The vascular flora of Rome. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 147(4), 1059-1087. https://doi.org/10.1080/11263504.2013.862315.
22. Celesti-Grapow, L., Marzio, P. D., & Blasi, C., 2001. The importance of alien and native species in the urban flora of Rome (Italy). cabidigitallibrary.org/doi/full/10.5555/20023037492.
23. Celesti-Grapow, L., Pyšek, P., Jarošík, V., & Blasi, C., 2006. Determinants of native and alien species richness in the urban flora of Rome. Diversity and Distributions, 12(5), 490-501. https://doi.org/10.1111/j.1366-9516.2006.00282.x.
24. Convention on Biological Diversity., 2022. Kunming-Montreal Global Biodiversity Framework. CBD/COP/15/L25. Goal 11 | Department of Economic and Social Affairs.
25. Cuddington, K., 2011. Legacy effects: the persistent impact of ecological interactions. Biological Theory, 6(3), 203-210. https://doi.org/10.1007/s13752-012-0027-5.
26. Del Monte, M., D'Orefice, M., Luberti, G. M., Marini, R., Pica, A., & Vergari, F., 2016. Geomorphological classification of urban landscapes: the case study of Rome (Italy). Journal of Maps, 12(sup1), 178–189. https://doi.org/10.1080/17445647.2016.1187977.
27. Delchiaro, M., Vergari, F., Esposito, C., & Del Monte, M., 2025. The influence of anthropogenic topographic changes on geomorphological processes in the city of Rome (Italy): A case study of the Malagrotta area. Earth Surface Processes and Landforms, 50(3), e70033. https://doi.org/10.1002/esp.70033.
28. Dondina, O., Tirozzi, P., Viviano, A., Mori, E., Orioli, V., Tommasi, N., Tanzi, A., Bazzoli, L., Caprio, E., Patetta, C., Pastore, M. C., Bani, L., & Ancillotto, L., 2025. Spatial and habitat determinants of small-mammal biodiversity in urban green areas: Lessons for nature-based solutions. Urban Forestry & Urban Greening, 104, 128641. https://doi.org/10.1016/j.ufug.2024.128641.
29. European Commission, 2022. LUCAS Soil Component – Field Survey Guidelines. LUCAS - ESDAC - European Commission

30. European Commission., 2020. EU Biodiversity Strategy for 2030: Bringing nature back into our lives. COM, 380 final. EUR-Lex - 52020DC0380 - EN - EUR-Lexhttps://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52020DC0380.
31. European environmental agency (EEA), 2018. https://doi.org/10.2909/4605463b-7150-49c6-9b45-01bf542891a9
32. European environmental agency (EEA), 2021. https://doi.org/10.2909/08e2387f-0d1c-4747-a5f0-8b3c2f382d4a.
33. Frondoni, R., Mollo, B., Capotorti., G., 2011. A landscape analysis of land cover change in the Municipality of Rome (Italy): Spatio-temporal characteristics and ecological implications of land cover transitions from 1954 to 2001. Landscape and Urban planning, 100(1-2), 117-128. https://doi.org/10.1016/j.landurbplan.2010.12.002.
34. Funiciello, R., Giordano, G. (Eds.), 2008. Noteillustrative della carta geologica d'Italia alla scala 1:50.000, foglio 374 Roma. Università degli Studi Roma Tre, APAT.
35. Galasso, G., Conti, F., Peruzzi, L., Alessandrini, A., Ardenghi, N. M. G., Bacchetta, G., Banfi, E., Barberis, G., Bernardo, L., Bouvet, D., Bovio, M., Castello, M., Cecchi, L., Del Guacchio, E., Domina, G., Fascetti, S., Gallo, L., Guarino, R., Gubellini, L., Guiggi, A., Hofmann, N., Iberite, M., Jiménez-Mejías, P., Longo, D., Marchetti, D., Martini, F., Masin, R. R., Medagli, P., Musarella, C. M., Peccenini, S., Podda, L., Prosser, F., Roma-Marzio, F., Rosati, L., Santangelo, A., Scoppola, A., Selvaggi, A., Selvi, F., Soldano, A., Stinca, A., Wagensommer, R. P., Wilhalm, T., & Bartolucci, F., 2024. A second update to the checklist of the vascular flora alien to Italy. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(2), 297–340. https://doi.org/10.1080/11263504.2024.2320129.
36. Gillini, A., Bilyera, N., Sferra, G., Bucci, A., Monaco, P., Loginova, I., ... & Trupiano, D., 2025. Insights into plant-microbe interactions in urban rhizosphere: feedback from Quercus cerris L. root morphology, soil zymography, and microbial community composition in rhizobox system. Journal of Plant Interactions, 20(1), 2554974. https://doi.org/10.1080/17429145.2025.2554974.
37. Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M., 2008. Global change and the ecology of cities. science, 319(5864), 756-760. 10.1126/science.115019.
38. Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., Melbourne, B. A., Nicholls, A. O., Orrock, J. L., Song, D.-X., & Townshend, J. R., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Science advances, 1(2), e1500052. https://doi.org/10.1126/sciadv.1500052.
39. Heinrichs, S., & Pauchard, A., 2015. Struggling to maintain native plant diversity in a peri-urban reserve surrounded by a highly anthropogenic matrix. Biodiversity and Conservation, 24(11), 2769-2788. https://doi.org/10.1007/s10531-015-0964-y
40. Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R., & Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global ecology and biogeography, 15(1), 1-7. https://doi.org/10.1111%2Fj.1466-822X.2006.00212.x.
41. Hutchinson, M.F., Xu, T. & Stein, J.A., (2011). Recent progress in the ANUDEM elevation gridding procedure. Geomorphometry, 2011(2), 19–22. https://doi.org/10.1002/osp4.29.
42. ISPRA Sinanet, SINAnet Groupware, last access on 13/05/2025.
43. Kaminski, A., Bauer, D. M., Bell, K. P., Loftin, C. S., & Nelson, E. J., 2021. Using landscape metrics to characterize towns along an urban-rural gradient. Landscape Ecology, 36(10), 2937-2956. https://doi.org/10.1007/s10980-021-01287-7.
44. Knapp, S., Kuehn, I., Wittig, R., Ozinga, W. A., Poschlod, P., & Klotz, S., 2008. Urbanization causes shifts of species' trait state frequencies. https://hdl.handle.net/2066/72378.
45. Knapp, S., von der Lippe, M., & Kowarik, I., 2022. Interactions of functional traits with native status and ecosystem novelty explain the establishment of plant species within urban ecosystems: Evidence from Berlin, Germany. Frontiers in Ecology and Evolution, 10, 790340. https://doi.org/10.3389/fevo.2022.790340.
46. Kowarik I, von der Lippe M., 2018. Plant population success across urban ecosystems: A framework to inform biodiversity conservation in cities. J Appl Ecol.; 55: 2354–2361. https://doi.org/10.1111/1365-2664.13144.

47. Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. Environmental pollution, 159(8-9), 1974-1983. https://doi.org/10.1016/j.envpol.2011.02.022.
48. Kraft, N. J., Adler, P. B., Godoy, O., James, E. C., Fuller, S., & Levine, J. M., 2014. Community assembly, coexistence and the environmental filtering metaphor. Functional ecology, 29(5), 592-599. https://doi.org/10.1111/1365-2435.12345.
49. Kumar, S., Stohlgren, T.J., Chong, G.W., 2006. Spatial heterogeneity influences native and nonnative plant species richness. Ecology 87, 3186–3199. https://doi.org/10.1890/0012-9658(2006)87[3186:SHINAN]2.0.CO;2.
50. Kurek, P., Dobrowolska, D., & Wiatrowska, B., 2019. Dispersal distance and burial mode of acorns in Eurasian Jays Garrulus glandarius in European temperate forests. Acta Ornithologica, 53(2), 155-162.https://doi.org/10.3161/00016454AO2018.53.2.005.
51. La Vigna, F., Mazza, R., Amanti, M., Di Salvo, C., Petitta, M., Pizzino, L., Pietrosante, A., Martarelli, L., Bonfà, I., Capelli, G., Cinti, D., Ciotoli, F., Ciotoli, G., Conte, G., Del Bon, A., Dimasi, M., Falcetti, S., Gafà, R. M., Lacchini, A., Mancini, M., Martelli, S., Mastrorillo, L., Monti, G. M., Procesi, M., Roma, M., Sciarra, A., Silvi, A., Stigliano, F., & Succhiarelli, C., 2016. Groundwater of Rome. Journal of Maps, 12(sup1), 88–93. https://doi.org/10.1080/17445647.2016.1158669.
52. Lopez, B. E., Urban, D., & White, P. S. (2018). Nativity and seed dispersal mode influence species' responses to habitat connectivity and urban environments. Global Ecology and Biogeography, 27(9), 1017-1030. https://doi.org/10.1111/geb.12760.
53. Lundholm, J. T., & Richardson, P. J., 2010. MINI-REVIEW: Habitat analogues for reconciliation ecology in urban and industrial environments. Journal of Applied Ecology, 47(5), 966-975. https://doi.org/10.1111/j.1365-2664.2010.01857.x.
54. Malkinson, D., Kopel, D., & Wittenberg, L., 2018. From rural-urban gradients to patch–matrix frameworks: Plant diversity patterns in urban landscapes. Landscape and Urban Planning, 169, 260-268. http://dx.doi.org/10.1016/j.landurbplan.2017.09.021.
55. Mandarino, A., Faccini, F., Terrone, M., & Paliaga, G., 2021. Anthropogenic landforms and geo-hydrological hazards of the Bisagno Stream catchment (Liguria, Italy). Journal of Maps, 17(3), 122–135. https://doi.org/10.1080/17445647.2020.1866704.
56. Manes, F., Incerti, G., Salvatori, E., Vitale, M., Ricotta, C. and Costanza, R., 2012. Urban ecosystem services: tree diversity and stability of tropospheric ozone removal. Ecological Applications, 22: 349-360. https://doi.org/10.1890/11-0561.1.
57. McCune, B., and D. Keon, 2002. Equations for potential annual direct incident radiation and heat load index. Journal of Vegetation Science. 13:603-606. https://doi.org/10.1111/j.1654-1103.2002.tb02087.x.
58. McDonald, R. I., Mansur, A. V., Ascensão, F., Colbert, M., Crossman, K., Elmqvist, T., Gonzalez, A., Güneralp, B., Haase, D., Hamann, M., Hillel, O., Huang, K., Kahnt, B., Maddox, D., Pacheco, A., Pereira, H. M., Seto, K. C., Simkin, R., Walsh, B., Werner, A. S., & Ziter, C., 2020. Research gaps in knowledge of the impact of urban growth on biodiversity. Nature Sustainability, 3(1), 16-24. https://doi.org/10.1038/s41893-019-0436-6.
59. McGarigal, K., 2015. FRAGSTATS Help. University of Massachusetts, Amherst, MA, USA, p. 182. https://doi.org/10.2737/PNW-GTR-351.
60. McLauchlan, K., 2006. The nature and longevity of agricultural impacts on soil carbon and nutrients: a review. Ecosystems, 9, 1364-1382. https://doi.org/10.1007/s10021-005-0135-1.
61. Medley, K. E., McDonnell, M. J., & Pickett, S. T., 1995. Forest-landscape structure along an urban-to-rural gradient. The Professional Geographer, 47(2), 159-168. https://doi.org/10.1111/j.0033-0124.1995.159_c.x.
62. Midolo, G., Axmanová, I., Divíšek, J., Dřevojan, P., Lososová, Z., Večeřa, M., Karger, D. N., Thuiller, W., Bruelheide, H., Aćić, S., Attorre, F., Biurrun, I., Boch, S., Bonari, G., Čarni, A., Chiarucci, A., Ćušterevska, R., Dengler, J., Dziuba, T., Garbolino, E., Jandt, U., Lenoir, J., Marcenò, C., Rūsiņa, S., Šibík, J., Škvorc, Ž., Stančić, Z., Stanišić-Vujačić, M., Svenning, J.-C., Swacha, G., Vassilev, K., & Chytrý, M., 2024. Diversity and distribution of Raunkiær's life forms in European vegetation. Journal of Vegetation Science, 35(1), e13229. https://doi.org/10.1111/jvs.13229.
63. Motta, R., Nola, P., & Berretti, R., 2009. The rise and fall of the black locust (Robinia pseudoacacia L.) in the "Siro Negri" Forest Reserve (Lombardy, Italy): lessons learned and future uncertainties. Annals of Forest Science, 66(4), 1-10. http://dx.doi.org/10.1051/forest/2009012.

64. Nelson, C.R., Hallett, J.G., Romero Montoya, A.E., Andrade, A., Besacier, C., Boerger, V., Bouazza, K., Chazdon, R., Cohen-Shacham, E., Danano, D., Diederichsen, A., Fernandez, Y., Gann, G.D., Gonzales, E.K., Gruca, M., Guariguata, M.R., Gutierrez, V., Hancock, B., Innecken, P., Katz, S.M., McCormick, R., Moraes, L.F.D., Murcia, C., Nagabhatla, N., Pouaty Nzembialela, D., Rosado-May, F.J., Shaw, K., Swiderska, K., Vasseur, L., Venkataraman, R., Walder, B., Wang, Z., & Weidlich, E.W.A., 2024. Standards of practice to guide ecosystem restoration – A contribution to the United Nations Decade on Ecosystem Restoration 2021-2030. Rome, FAO, Washington, DC, SER & Gland, Switzerland, IUCN CEM. https://doi.org/10.4060/cc9106e.
65. Pascual-Hortal, L., & Saura, S., 2008. Integrating landscape connectivity in broad-scale forest planning through a new graph-based habitat availability methodology: application to capercaillie (Tetrao urogallus) in Catalonia (NE Spain). European Journal of Forest Research, 127(1), 23-31. https://doi.org/10.1007/s10342-006-0165-z.
66. Peres-Neto, P. R., Legendre, P., Dray, S., & Borcard, D., 2006. Variation partitioning of species data matrices: estimation and comparison of fractions. Ecology, 87(10), 2614-2625. https://doi.org/10.1890/0012-9658(2006)87[2614:VPOSDM]2.0.CO;2.
67. Perring, M. P., Standish, R. J., Price, J. N., Craig, M. D., Erickson, T. E., Ruthrof, K. X., Whiteley, A. S., Valentine, L. E., & Hobbs, R. J., 2015. Advances in restoration ecology: rising to the challenges of the coming decades. Ecosphere, 6(8), 1-25. https://doi.org/10.1890/ES15-00121.1.
68. Pica, A., Lämmle, L., Burnelli, M., Del Monte, M., Donadio, C., Faccini, F., Lazzari, M., Mandarino, A., Melelli, L., Perez Filho, A., Russo, F., Stamatopoulos, L., Stanislao, C., & Brandolini, P., 2024. Urban Geomorphology Methods and Applications as a Guideline for Understanding the City Environment. Land. 13(7):907. https://doi.org/10.3390/land13070907.
69. Pignatti, S., Guarino, R., & La Rosa, M., 2017. Flora d'italia, Vol. 1.
70. Polce, C., Kunin, W. E., Biesmeijer, J. C., Dauber, J., Phillips, O. L., & ALARM Field Site Network., 2011. Alien and native plants show contrasting responses to climate and land use in Europe. Global Ecology and Biogeography, 20(3), 367-379. https://doi.org/10.1111/j.1466-8238.2010.00619.x.
71. Prach, K., Pyšek, P., & Bastl, M., 2001. Spontaneous vegetation succession in human-disturbed habitats: A pattern across seres. Applied Vegetation Science, 4(1), 83-88. https://doi.org/10.1111/j.1654-109X.2001.tb00237.x.
72. Preston, C.D., 2000. Engulfed by suburbia or destroyed by the plough: The ecology of extinction in Middlesex and Cambridgeshire. Watsonia 23, 59–81.
73. R Core Team R, 2024. A Language and Environment for Statistical Computing. R Foundation for Statistical Computing. https://www.R-project.org/.
74. Ramalho, C. E., Laliberté, E., Poot, P., & Hobbs, R. J., 2014. Complex effects of fragmentation on remnant woodland plant communities of a rapidly urbanizing biodiversity hotspot. Ecology, 95(9), 2466-2478. https://doi.org/10.1890/13-1239.1.
75. Rega-Brodsky, C. C., Aronson, M. F. J., Piana, M. R., Carpenter, E.-S., Hahs, A. K., Herrera-Montes, A., Knapp, S., Kotze, D. J., Lepczyk, C. A., Moretti, M., Salisbury, A. B., Williams, N. S. G., Jung, K., Katti, M., MacGregor-Fors, I., MacIvor, J. S., La Sorte, F. A., Sheel, V., Threfall, C. G., & Nilon, C. H., 2022. Urban biodiversity: State of the science and future directions. Urban Ecosystems, 25(4), 1083-1096. https://doi.org/10.1007/s11252-022-01207-w.
76. Regione Lazio, 2002. Regional Technical Map http://dati.lazio.it/catalog/it/dataset/carta-tecnica-regionale-2002-2003-5k-roma.
77. Regione Lazio., 2012. Computerized Geological Map of the Lazio Region (1:25,000).
78. Roma Capitale, 2022. Roma Capitale | Sito Istituzionale | I numeri di Roma Capitale
79. Rosenzweig, M. L., 2003. Reconciliation ecology and the future of species diversity. Oryx, 37(2), 194-205. doi:10.1017/S0030605303000371.
80. Sabatini, F. M., Zanini, M., Dowgiallo, G., & Burrascano, S., 2015. Multiscale heterogeneity of topsoil properties in southern European old-growth forests. *European Journal of Forest Research*, *134*(5), 911-925. https://doi.org/10.1007/s10342-015-0899-6.
81. Salvati, L., & Zitti, M., 2012. Monitoring vegetation and land use quality along the rural–urban gradient in a Mediterranean region. Applied Geography, 32(2), 896-903. https://doi.org/10.1016/j.apgeog.2011.09.002.
82. Sanderson, E. W., Jaiteh, M., Levy, M. A., Redford, K. H., Wannebo, A. V., & Woolmer, G., 2002. The human footprint and the last of the wild: the human footprint is a global map of human influence on the land surface,

which suggests that human beings are stewards of nature, whether we like it or not. BioScience, 52(10), 891-904.https://doi.org/10.1641/0006-3568(2002)052[0891:THFATL]2.0.CO;2.
83. Saura, S., Vogt, P., Velázquez, J., Hernando, A., & Tejera, R., 2011. Key structural forest connectors can be identified by combining landscape spatial pattern and network analyses. Forest Ecology and Management, 262(2), 150-160. https://doi.org/10.1016/j.foreco.2011.03.017.
84. Saura, S., & Pascual-Hortal, L., 2007. A new habitat availability index to integrate connectivity in landscape conservation planning: comparison with existing indices and application to a case study. Landscape and urban planning, 83(2-3), 91-103. https://doi.org/10.1016/j.landurbplan.2007.03.005.
85. Smiraglia, D., Capotorti, G., Guida, D., Mollo, B., Siervo, V., & Blasi, C., 2013. Land units map of Italy. Journal of Maps, 9(2), 239–244. https://doi.org/10.1080/17445647.2013.771290.
86. Solé-Senan, X. O., Juárez-Escario, A., Conesa, J. A., & Recasens, J., 2018. Plant species, functional assemblages and partitioning of diversity in a Mediterranean agricultural mosaic landscape. Agriculture, Ecosystems & Environment, 256, 163-172. https://doi.org/10.1016/j.agee.2018.01.014.
87. Somodi, I., Ewald, J., Bede-Fazekas, Á., & Molnár, Z., 2021. The relevance of the concept of potential natural vegetation in the Anthropocene. Plant Ecology & Diversity, 14(1–2), 13–22. https://doi.org/10.1080/17550874.2021.1984600.
88. Stephens, C. M., Lall, U., Johnson, F. M., & Marshall, L. A., 2021. Landscape changes and their hydrologic effects: Interactions and feedbacks across scales. Earth-Science Reviews, 212, 103466
89. Szabó, J., Dávid, L., & Lóczy, D., 2010. Anthropogenic Geomorphology: A Guide to Man-Made Landforms. Springer. https://doi.org/10.1007/978-90-481-3058-0.
90. Thornbush, M. J., & Allen, C. D. (Eds.), 2018. Urban Geomorphology: Landforms and Processes in Cities. Elsevier.
91. TUXEN, R., 1956. Die heutige potentille natuerliche vegetation als Gegenst vegetation skartierung. Angew Pflanzensoziologie, 13, 5-42.
92. United Nations., 2015. Transforming our world: The 2030 Agenda for Sustainable Development. Goal 11 | Department of Economic and Social Affairs.
93. Valeri, S., Zavattero, L., & Capotorti, G., 2021. Ecological Connectivity in Agricultural Green Infrastructure: Suggested Criteria for Fine Scale Assessment and Planning. Land, 10(8), 807. https://doi.org/10.3390/land10080807.
94. Vergari, F., Marco Luberti, G., Pica, A., & Del Monte, M. (2021). Geomorphology of the historic centre of the Urbs (Rome, Italy). Journal of Maps, 17(4), 6-17.https://doi.org/10.1080/17445647.2020.1761465.
95. Vogt, P., Riitters, K., Rambaud, P., d'Annunzio, R., Lindquist, E., & Pekkarinen, A., 2022. GuidosToolbox Workbench: spatial analysis of raster maps for ecological applications. Ecography, 2022(3), e05864. https://doi.org/10.1111/ecog.05864.
96. Williams, N. S., Hahs, A. K., & Vesk, P. A., 2015. Urbanisation, plant traits and the composition of urban floras. Perspectives in Plant Ecology, Evolution and Systematics, 17(1), 78-86. http://dx.doi.org/10.1016/j.ppees.2014.10.002.
97. Williams, N. S., Morgan, J. W., McDonnell, M. J., & Mccarthy, M. A., 2005. Plant traits and local extinctions in natural grasslands along an urban–rural gradient. Journal of Ecology, 93(6), 1203-1213.https://doi.org/10.1111/j.1365-2745.2005.01039.x.
98. Zapolska, A., Serge, M. A., Mazier, F., Quiquet, A., Renssen, H., Vrac, M., ... & Roche, D. M., 2023. More than agriculture: Analysing time-cumulative human impact on European land-cover of second half of the Holocene. Quaternary Science Reviews, 314, 108227.https://doi.org/10.1016/j.quascirev.2023.108227.
99. Zerbe, S., 1998. Potential natural vegetation: validity and applicability in landscape planning and nature conservation. Applied Vegetation Science, 1(2), 165-172. https://doi.org/10.2307/1478945.
100. Zipperer, W. C., & Guntenspergen, G. R., 2009. Vegetation composition and structure of forest patches along urban-rural gradients. In: McDonnell, MJ; Hahns, AK; Breuste, JH, eds. Ecology of Cities and Towns. Cambridge, UK: Cambridge University Press. 274-286. ja_2009_zipperer_001.pdf

Part III: Green Infrastructure design for the containment of biological invasions. Insights from a peri-urban case study in Rome, Italy

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The published version of the article is provided in Appendix and online resources III.

Abstract

Secondary shrublands and transitional woodland/shrub formations are recognised to be particularly susceptible to plant invasions, one of the main global threats to biodiversity, especially in dynamic peri-urban landscapes. Urban fringes are in fact often the place for the sprawl of artificial surfaces, fragmentation of habitats, and complex land transitions (including both agriculture intensification and abandonment), which in turn increase propagule pressure of exotic species over residual semi-natural ecosystems. Within this framework, the present study was aimed at analysing i) how landscape composition and configuration affect the richness of woody exotic species in shrubland and transitional woodland/shrub patches, and ii) how this threat can be addressed by means of green infrastructure design in a peri-urban case study (Metropolitan City of Rome, Italy). Accordingly, the occurrence of exotic plants was recorded with field surveys and then integrated with landscape analyses, both at patch level and over a 250 m buffer area around each patch. Thus, the effect of landscape features on exotic plant richness was investigated with Generalised Linear Models, and the best model identified (pseudo R-square = 0.62) for inferring invasibility of shrublands throughout the study area. Finally, a Green Infrastructure (GI) to contain biological invasion was planned, based on inferred priority sites for intervention and respective, site-tailored, actions. The latter included not only the removal of invasive woody alien plants, but also reforestation and planting of native trees for containment of dispersal and subsequent establishment. Even though specifically developed for the study site, and consistent with local government needs, the proposed approach represents a pilot planning process that might be applied to other peri-urban regions for the combined containment of biological invasions and sustainable development of peripheral complex landscapes.

Keywords: Invasive plants, Urban sprawl, Propagule pressure, Peri-urban landscapes, Environmental management

Highlights

- Containment of plant invasion is needed in peri-urban areas.
- A landscape ecology approach helps detect the habitats more susceptible to invasion.
- Passive and active containment measures should be enacted.
- An inference-based model for Green Infrastructure planning is proposed.

1. Introduction

Plant invasion is a global phenomenon that is endangering biodiversity and natural resources, simultaneously causing economic losses and health diseases due, for example, to allergenic pollen and toxic compounds (Hattendorf et al., 2007; Prank et al., 2013).
The spread of exotic species is intentionally or unintentionally aided by humans, and cities represent preferential sites for importation and cultivation (Kowarik, 2011). Richness of allochthonous species and abundance of respective individuals are among the main factors enhancing plant invasions and have been defined as “propagule pressure” (Lockwood et al., 2013; Theoharides and Dukes 2007). Once naturalised, exotic plants may spread via seed dispersal or vegetative reproduction from urban centres, to progressively reach and eventually colonise more peripheral and remote areas (Celesti-Grapow et al., 2001; Campagnaro et al., 2022). Since the spread may be affected by landscape filters, several studies have explored the role of composition, configuration and condition of the land cover mosaic in facilitating the invasion process (Kumar et al., 2006; Vilà and Ibáñez 2011). As a matter of fact, fragmentation of natural habitats and anthropogenic land uses (urban and/or agricultural) directly enhance the invasibility of natural areas (González-Moreno et al., 2013; Boscutti et al., 2018). Particularly in cities, a broad array of disturbed habitats occurs that may be easily invaded (roadsides, gardens) and the urban sprawl consistently facilitates this process, representing a proxy for the propagule pressure at the local level (Botham et al., 2009; Polce et al., 2011; Boscutti et al., 2022).
In peri-urban contexts, close to main urban input sources, exotic invasion is expected to be especially harmful for residual native communities and may easily affect spontaneous ecosystem dynamics and ecosystem service capacity, sometimes also causing ecosystem disservices (Gaertner et al., 2014; Shackleton et al., 2016; Simberloff et al., 2013). Concurrently, at least in European countries and over the last few decades, various social and economic changes have driven the abandonment of wide arable lands in these sectors (Stoate et al., 2009). Substitution stages of the vegetation series, such as secondary grasslands and shrublands, evolve over abandoned fields with potential positive effects on natural biodiversity and ecosystem service capacity. However, abandoned and recolonized fields may be particularly susceptible to plant invasion because of the scarcity of autochthonous seed sources, lack of competition by native trees, and high soil nutrient availability due to previous fertilisation practices (Fenesi et al., 2015). Once established, invasive plants can alter the natural composition of vegetation communities, with long-lived species that may persist until later successional stages are reached (Meiners et al., 2002).
Shrublands, along with transitional woodland-shrubs and abandoned pastures and grasslands, typically offer a significant opportunity for invasive phanerophytes to succeed because of few structural barriers, such asclosed tree canopies, high levels of light availability facilitating pioneer species, and altered soil conditions (Fernandes et al., 2018). Conversely, natural forest ecosystems are usually more resistant to invasions, because native trees already occupy most of the ecological niches and leave little space and resources for the establishment of exotic phanerophytes (Funk et al., 2008; Santala et al., 2022). Thus, shrublands and other dynamic vegetation stages represent key elements to be restored, while woodlands dominated by native species represent primary components to be conserved in peri-urban areas.
Green Infrastructure (GI), defined by the European Commission as a “strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services” (EC, 2013), represents a useful tool for addressing these needs.

GIs explicitly devoted to the containment of biological invasions have already been proposed at broad scales (Vallecillo et al., 2018), but no evidence has been yet provided as regards planning criteria at the local scale.
Accordingly, this work is aimed at i) investigating whether and how landscape mosaic characteristics affect the spread of woody non-native plants over dynamic peri-urban ecosystems and ii) proposing an inference-based GI planning process devoted to the containment of plant invasion at the local scale.

2. Materials and methods

2.1. Study area

The research was conducted in the "Wide Area of Valle Galeria", western Metropolitan City of Rome, that is an area of particular concern to the metropolitan government for the deployment of environmental monitoring and mitigation actions against land pollution and degradation. Besides being affected by scattered urban expansion and land consumption (Salvati, 2015), it harbours the former Malagrotta landfill (the largest in Europe) along with a petrochemical centre, a gasifier, an incinerator for special hospital waste, two waste treatment plants to produce fuel, and liquefied gas and mineral oil deposits.
The exact study site (Figure 1) embraces the median sector of the river basins of Arrone River and Rio Galeria and covers 22900 ha. It belongs to the Roman Area ecoregional subsection, characterised by Mediterranean and transitional bioclimates, average annual precipitation between 660 mm and 1086 mm, yearly temperatures between 14 and 17 C°, composite sedimentary and volcanic lithological substrata, and morphological plateaus and slopes interspersed with alluvial valleys, with an average altitude of 70 m a.s.l. (Blasi et al., 2014).
Prevailing potential natural vegetation (PNV), determined by varying combination of abiotic and biotic environmental features, is for oak forests and hygrophilous woods. More in detail, dominant PNV types embrace meso-hygrophilous *Quercus robur* forests of the alluvial plains, riparian forests with *Alnus glutinosa*, *Populus* sp. pl. and *Salix* sp. pl., and deciduous and mixed forests with *Quercus cerris, Q. frainetto Q. pubescens, Q.ilex* and *Q. suber* on volcanic plateaux and sedimentary hills (CIRBFEP, 2013; Fig. 1).

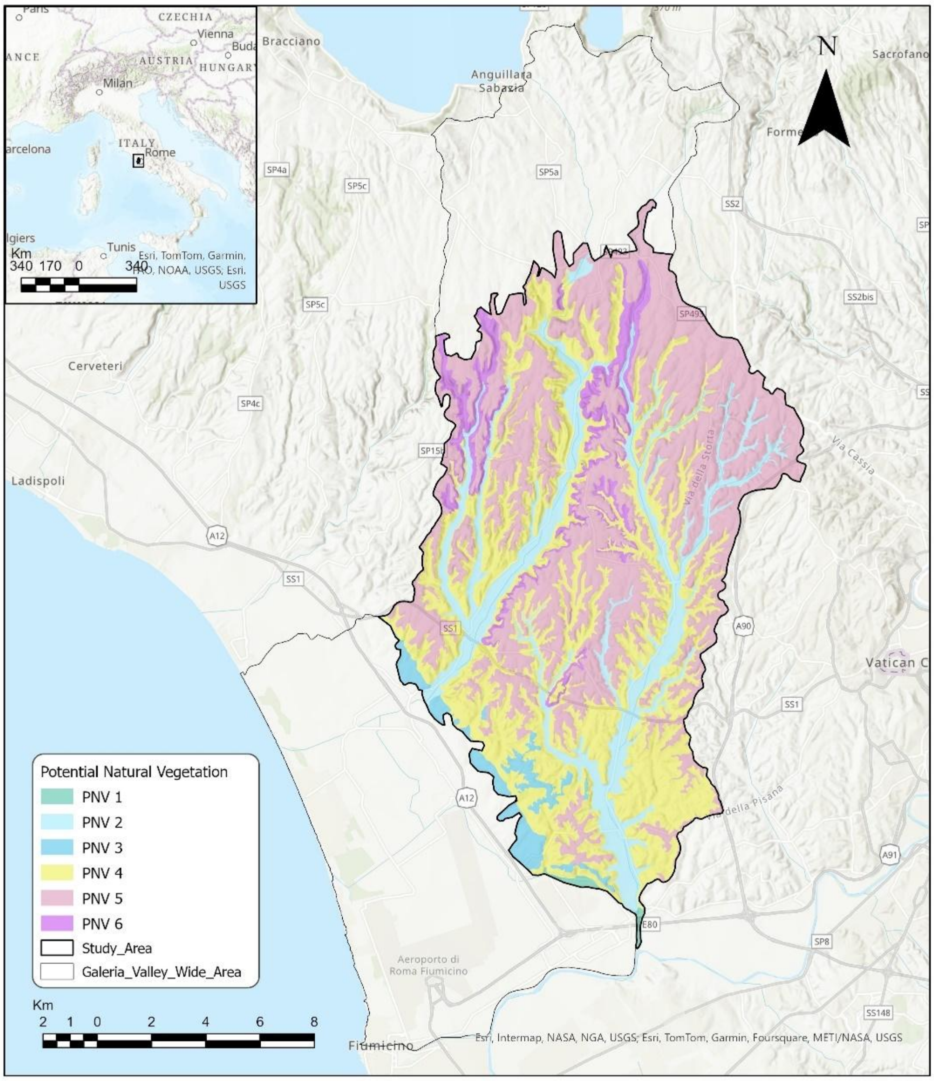


Figure 1. Geographic location (upper left corner) and internal arrangement into Potential Natural Vegetation types of the study area (main picture): PNV 1 - *Quercus robur, Ulmus minor and Fraxinus angustifolia* subsp. *oxycarpa* forests of the Tiber River alluvial plain; PNV 2 - *Quercus robur, Q. cerris, Q. frainetto, Ulmus minor, Alnus glutinosa, Populus* sp.pl. and *Salix* sp.pl. forests vegetation complex of alluvial valleys; PNV 3- Acidophilous forests with *Quercus cerris, Q. frainetto, Q. pubescens* and *Q.suber;* PNV 4 - Thermophilous forests with *Quercus pubescens* and *Quercus cerris;* PNV 5 - *Quercus cerris, Q. ilex, Fraxinus ornus* and *Carpinus orientalis* forests on volcanic substrata; PNV 6 - *Quercus cerris, Q. robur* and *Ulmus minor* forests of colluvial valleys. Base map: World Topographic Map (WGS 84).

With respect to this potential arrangement, the actual land cover mosaic is dominated by agricultural surfaces (53.4 % of the territory), mainly embracing extensive and traditional arable lands (Biasi et al., 2015), with interspersed natural and semi-natural vegetation communities (26.4 %) and artificial areas (20.2%).

2.2. Methodological framework

In order to provide an inference-based model for the design of a GI devoted to the containment of biological invasions, the occurrence of exotic woody species in semi-natural shrubland patches was first investigated with field surveys. Second, landscape metrics to measure the pattern and quality of land use/land cover mosaic around these patches were calculated. Thus, the correlation between landscape metrics and exotic plant richness was investigated through Generalised Linear Models (GLM) and, finally, GI components and respective actions were prioritised according to statistical inferences provided by the best resulting GLM (Fig. 2).

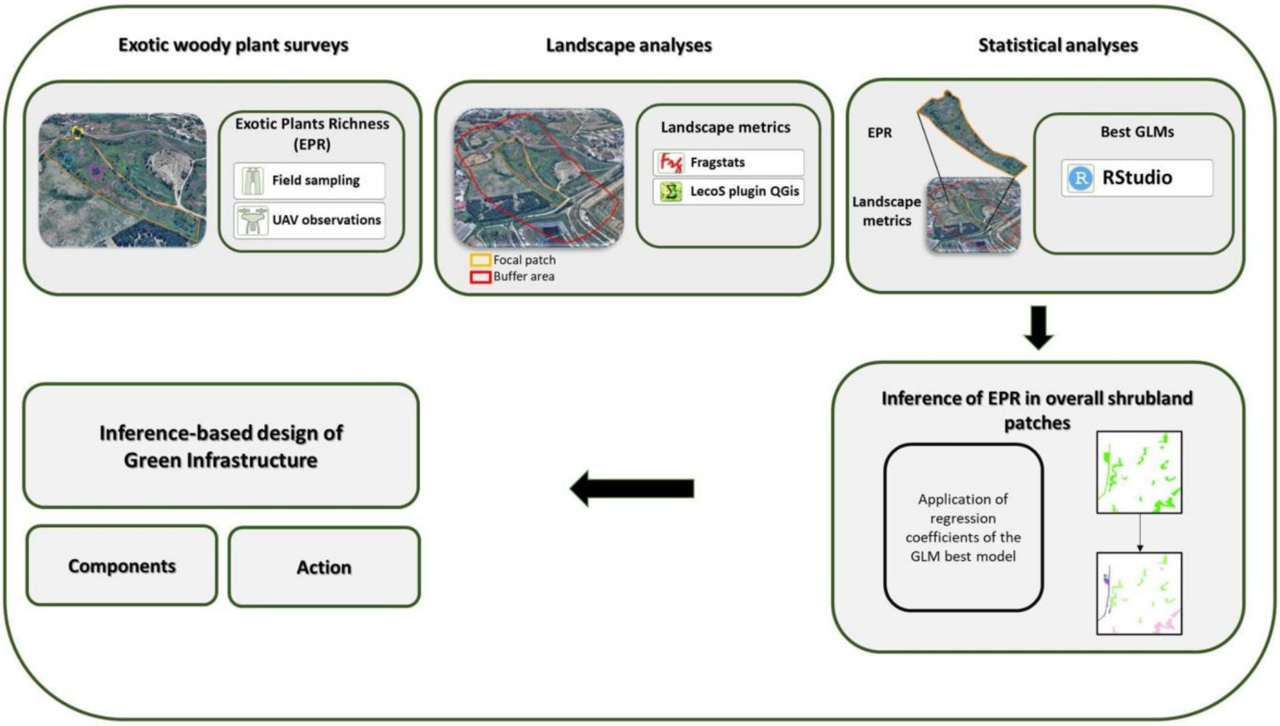


Figure 2. Workflow for the design of GI proposal.

2.2.1. Basic data

The Land use and land cover map of Valle Galeria, realised for the Environmental Monitoring Project of the site at 1:10000 scale with a minimum mapping unit of 0.01 ha (Interuniversity Research Center Biodiversity Ecosystem Services and Sustainability, unpublished results), was adopted for the stratification of field samplings, landscape analyses and definition of GI components. The map, which returns a detailed representation of natural and semi-natural areas (Table 1), was further improved as regards the road network with the assignment of different levels of disturbance to each road segment according to OpenStreetMap (https://www.openstreetmap.org).

The PNV map of the Province of Rome at 1:25000 scale (CIRBFEP, 2013) was instead adopted to recognise homogeneous environmental land units and to define local tailored reference models for restoration actions (Palmer et al., 2016). The map actually reports the mature vegetation types that would develop under undisturbed successional dynamics, consistently with climatic, lithological and morphological features, and without human disturbance (Zerbe, 1998).

Table 1: Natural and semi-natural cover types in the Land use and land cover map of Valle Galeria.

Map Code	Description
3.1.1.1.1	Broad-leaved forests with *Quercus cerris*, *Q. virgiliana*, *Carpinus orientalis*
3.1.1.1.2	Broad-leaved forests with *Quercus cerris*, *Q. frainetto*, *Q. suber*
3.1.1.1.3	Broad-leaved forests with *Quercus virgiliana, Fraxinus ornus*, *Q. ilex*, *Acer campestre*
3.1.1.1.4	Broad-leaved forests with *Quercus suber, Q. virgiliana*, *Ulmus minor*, *Q. frainetto*, *Q. cerris*
3.1.1.1.5	Mesophilous broad-leaved forests with *Quercus robur*, *Ulmus minor*, *Q. cerris*, *Carpinus betulus*
3.1.1.1.6	Hygrophilous and meso-hygrophilous broad-leaved forests with *Populus* sp. pl., *Salix* sp. pl., *Alnus glutinosa*, *Fraxinus angustifolia* subsp. *oxycarpa*, *Quercus robur*
3.1.1.2.1	Broad-leaved forests with *Quercus ilex*, *Fraxinus ornus*, *Quercus virgiliana*, *Q. cerris*, *Carpinus orientalis*
3.1.1.2.2	High Mediterranean maquis and coastal forests with *Quercus ilex*, *Phillyrea* sp. pl., *Pistacia lentiscus*, *Myrtus communis*
3.1.3	Mixed coniferous and broadleaved forest plantations
3.1.4	Exotic pine plantations, mainly with *Pinus pinea*
3.1.5	Allochthonous forests dominated by *Eucalyptus* sp. pl. and *Robinia pseudoacacia*
3.2.1	Semi-natural pastures, meadows and fallows
3.2.2.1	Deciduous shrublands with *Rubus ulmifolius*, *Prunus spinosa*, *Ulmus minor*, *Spartium junceum*, *Pteridium aquilinum*
3.2.2.2	Hygrophilous and meso-hygrophilous shrublands with *Salix* sp. pl., *Rubus* sp. pl., *Sambucus* sp. pl., *Cornus sanguinea*, *Arundo donax*, *Phragmites australis*
3.2.3.1	Evergreen shrublands with *Pistacia lentiscus*, *Phillyrea latifolia*, *Myrtus communis*, *Rhamnus alaternus*
3.2.3.2	Coastal shrublands and garrigues with *Pistacia lentiscus*, *Phillyrea* sp. pl., *Myrtus communis*, *Juniperus macrocarpa*
3.2.4	Transitional woodland-shrub vegetation communities
3.4	Hedgerows

2.2.2. Field sampling design

The occurrence of exotic woody species have been especially investigated in secondary shrub communities, because of their high susceptibility to plant invasion and considerable extent in the study area, by focusing field samplings in the following land use /land cover types (hereafter generically named "shrublands"):

• Deciduous shrublands with *Rubus ulmifolius, Prunus spinosa, Ulmus minor, Spartium junceum, Pteridium aquilinum*

• Hygrophilous and meso-hygrophilous shrublands with *Salix* sp. pl., *Rubus* sp. pl., *Sambucus* sp. pl., *Cornus sanguinea, Arundo donax*, *Phragmites australis*

• Evergreen shrublands with *Pistacia lentiscus, Phillyrea latifolia, Myrtus communis, Rhamnus alaternus*
• Transitional woodland-shrub vegetation communities.

Due to widespread private properties in the study area, the field campaign was opportunistic, and all the accessible patches were sampled until a considerable number of samplings was reached (i.e. 80 according to Kumar et al., 2006). When possible, some of the inaccessible sites were sampled by means of UAV (Unmanaged Air Vehicle, Model DJI MINI SE).

The number of exotic plants for each explored shrubland patch was recorded to estimate the Exotic Plant Richness (EPR). In particular, exotic phanerophytes were recorded, because of their capability to alter the mature stages of vegetation series (especially oak woods) by replacing native trees and persist in the long term (Badalamenti et al., 2018; Kowarik, 2011), along with the giant geophyte reed *Arundo donax,* because of its ability to form dense stands that impair spontaneous recovery of natural forests (Lambert et al., 2010).

Sampled species were thus identified and characterised in terms of biological form (Pignatti et al., 2017), non-native status (archaeophytes *vs* neophytes, respectively introduced before and after the discovery of America)(Celesti-Grapow et al., 2013), invasive status in Italy (Portal to the Flora of Italy, https://dryades.units.it/floritaly/index.ph), and acknowledged level of impact (European Alien Species Information network, https://easin.jrc.ec.europa.eu/easin). The nomenclature mainly followed Bartolucci et al. (2018) for native plant taxa and Galasso et al. (2018) for non-native ones.

2.2.3. Landscape and statistical analyses

A set of landscape metrics was selected according to recognised relationships between EPR and structural properties of the landscape mosaic (Malavasi et al., 2014). Namely, thirteen metrics have been measured at the patch level and within a distance of 250 m from the border of each shrubland patch (Table 2), *i.e.* in a buffer area compliant with spread capacity of exotic plants in similar environments (Boscutti et al., 2022). Along with structural landscape features, especially concerning artificial and agricultural land cover/uses, the overall status of the mosaic around the patches was also assessed by means of the Index of Landscape Conservation (ILC; Pizzolotto and Brandmayr, 1996), which ranges from 0 (completely artificialised landscapes) to 1 (completely natural landscapes).

The analyses were run in Fragstats 4.0 and LecoS Plugin for Quantum GIS, on a raster version of the Land cover and land use map of the Galeria Valley simplified into 5 main classes (1 - Artificial surfaces, 2 - Agricultural areas, 3 - Pastures, meadows and natural grasslands, 4 - Shrublands and transitional woody-shrub communities, 5 - Forests) (Supplementary Fig. 1).

Table 2: Landscape metrics calculated at the patch level and within a buffer area of 250 m from the border of each patch. Definitions and formulas for these metrics are available in McGarigal (2015) and in Pizzolotto and Brandmayr (1996).

Patch level metrics	
SHP	Shape index of the focal patch (adimensional)
AC	Proportion of contacts of the focal patch with artificial surfaces (%)
Class level metrics in the buffer area	
PLAND1	Proportional extent of artificial surfaces (%)
TE1	Total edge of artificial surfaces (m)
ED1	Edges density of artificial surfaces (m/m^2)
SHAPE_MN1	Mean shape ratio of artificial surfaces (adimensional)
FRAC_ID1	Fractal dimension index of artificial surfaces (adimensional)
PLAND2	Proportional extent of agricultural areas (%)
TE2	Total edge of agricultural areas (m)
ED2	Edge density of agricultural areas (m/m^2)
SHAPE_MN2	Mean shape ratio of agricultural areas (adimensional)
FRAC_ID2	Fractal dimension index of agricultural areas (adimensional)
Landscape level metric in the buffer area	
ILC	Index of Landscape Conservation (adimensional)

The effects of landscape metrics on EPR were thus statistically analysed and the most explicative variables identified. A generalised linear model (GLM) with Poisson distribution was fit for EPR as a response variable, using landscape metrics as independent variables. Thus, a Stepwise model selection was performed to simplify the model, by selecting the best variables according to Akaike's Information Criterion (AIC), and the Variance Inflation Factor (VIF) analysis was used to evaluate collinearity among all the explanatory variables. Moreover, using Spearman's rank method, correlograms were used to assess the relationship between variables, and Leave-One-Out Cross-Validation (LOOCV) was performed to assess predictive capability of the chosen model.
The analyses were performed using the R Studio software (version 4.2.2) (R Core team, 2023).

2.2.4. Inference-based Green Infrastructure planning

To effectively manage non-native and invasive woody species, the previous outcomes have been framed within the "Global guidelines for the sustainable use and management of non-native trees", with special reference to "Preventing and mitigating the escape of Non-Native Trees" and "Mitigating the negative effects of Invasive Non-Native Trees (INNT)" goals (Brundu and Richardson, 2017).
Accordingly, a set of evidence-based practices for INNTs predictive detection and control were designed that strategically combine i) residual native forest conservation, under a landscape-level

restoration perspective (D'Antonio et al., 2016), ii) nature-based reforestation, under the European guidelines for biodiversity-friendly afforestation, reforestation and tree planting (EC, 2023), and iii) invasive alien plant removal (Maron and Marler, 2007). GI components to be alternatively conserved or restored have been thus selected according to resistance capacity against woody plant invasion, potential role as native seed sources (Rey-Benayas et al., 2008), structural and dynamic attitude towards invasion from woody plants (Santala et al., 2022), and potential exotic plant keeping and dispersal due to high levels of disturbance along roads (Parendes and Jones, 2000). For shrublands, chosen as the backbone for the GI deployment, a further prioritisation based on the above described inferential statistical outcomes was applied.

3. Results

3.1. Exotic woody species sampled in Galeria Valley shrublands

An overall number of 88 patches were explored during the field campaign, counting 22 exotic species (18 scapose phanerophytes , 2 caespitose phanerophytes , 1 succulent phanerophyte , and 1 rhizomatous geophyte) (Table 3; Fig. 3). Out of these 22 species, 17 are neophytes and 5 archaeophytes (including *Juglans regia,* which is considered cryptogenic but just in northern Italian regions; Mercuri et al., 2013).

Ten species, namely *Acacia dealbata*, *Acer negundo*, *Ailanthus altissima, Arundo donax, Eucalyptus camaldulensis, Ligustrum lucidum, Opuntia ficus-indica, Populus x canadensis, Robinia pseudoacacia,* and *Yucca gloriosa* are reported to be invasive in Italy, and seven of them have a high impact in Europe, with *A. altissima* also recognised of Union concern (Table 3). The number of exotic species detected in each patch ranged between 0 and 7, with an average value of 2.72 (2.04 St.dev).

Table 3: Exotic species sampled in the shrublands of Valle Galeria, comprehensive of the non-native status (N=Neophytes, A=Archeophytes), biological form, level of acknowledged impact, invasiveness in Italy, and Union concern (marked with an asterisk).

Species	Non-native status	Biological form	High level of impact	Invasive species
Acacia dealbata Link subsp. *dealbata*	N	P scap	Yes	Yes
Acer negundo L.	N	P scap	Yes	Yes
Ailanthus altissima (Mill.) Swingle*	N	P scap	Yes	Yes
Albizia julibrissin Durazz.	N	P scap	Yes	No
Arundo donax L.	A	G rhiz	Yes	Yes
Cedrus cfr. *atlantica* (Endl.) G.Manetti ex Carrière	N	P scap	No	No
Cupressus sempervirens L.	A	P scap	Yes	No
Eucalyptus camaldulensis Dehnh. subsp. *camaldulensis*	N	P scap	Yes	Yes

Gleditsia triacanthos L.	N	P scap	Yes	No
Hesperocyparis arizonica (Greene) Bartel	N	P scap	No	No
Juglans regia L.	A	P scap	No	No
Ligustrum lucidum W.T. Aiton	N	P scap	No	Yes
Morus alba L.	A	P scap	Yes	No
Opuntia ficus-indica (L.) Mill.	N	P succ	Yes	Yes
Phoenix canariensis H.Wildpret	N	P scap	Yes	No
Phyllostachys aurea Carrière ex Rivière & C.Rivière	N	P caesp	No	No
Pinus pinea L.	A	P scap	No	No
Populus x *canadensis* Moench	N	P scap	No	Yes
Eriobotrya japonica (Thunb.) Lindl.	N	P scap	Yes	No
Robinia pseudoacacia L.	N	P scap	Yes	Yes
Salix babylonica L.	N	P scap	No	No
Yucca gloriosa L.	N	P caesp	No	Yes

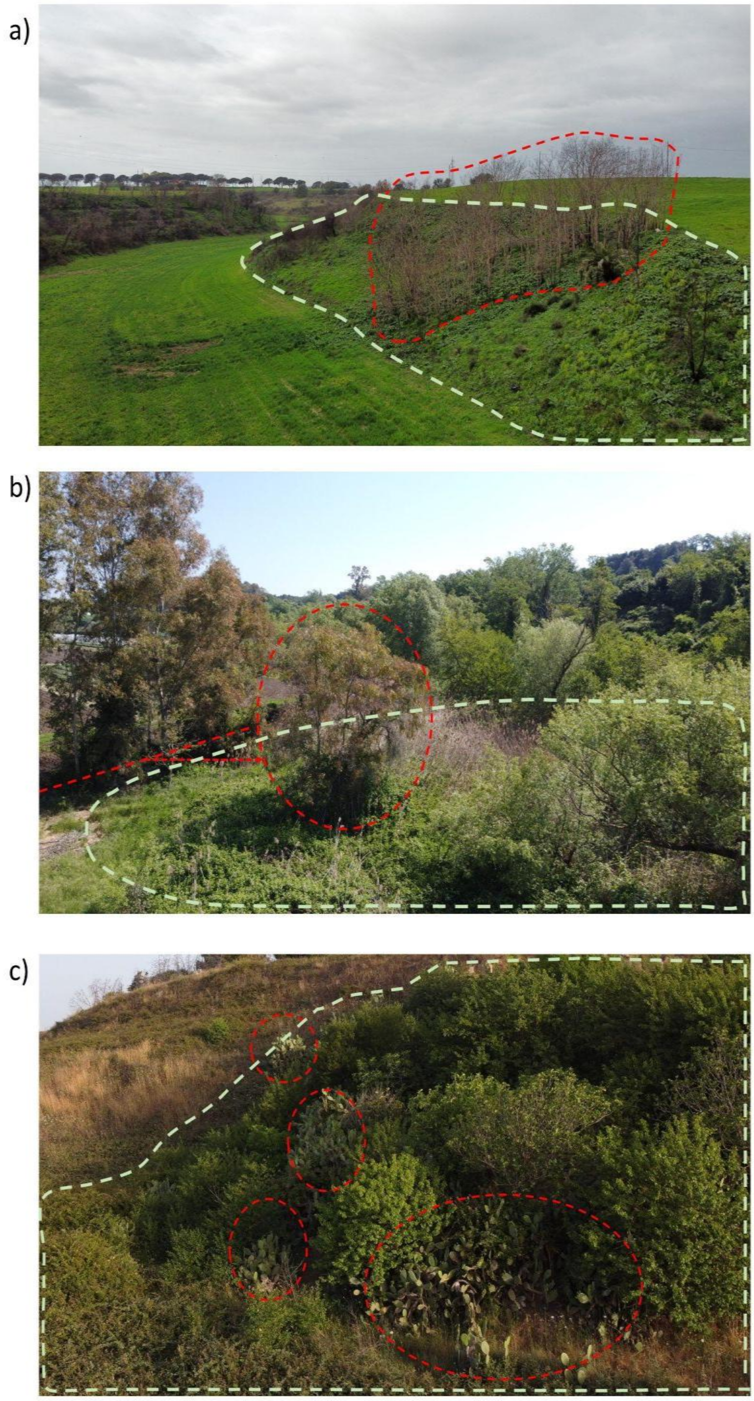


Figure 3. Examples of invaded secondary shrublands: (a) on a sandy slope (Quercus pubescens, Q. suber and Q. cerris PNV), with a nucleus of Ailanthus altissima and a young individual of Phoenix canariensis; (b) on an alluvial plain (Quercus robur and Ulmus minor PNV), colonised by Eucalyptus camaldulensis escapees from an adjacent tree line; (c) on a volcanic slope (Quercus ilex, Q. suber and Fraxinus ornus PNV), colonised by Opuntia ficus-indica.

3.2. Landscape metrics and statistical relationships with exotic plant richness

Landscape metrics concerning individual shrubland patches and pertaining buffer areas are reported in supplementary materials (Supplementary Table 1).

Based on these results, the best model for estimating exotic plant richness in shrubland patches, first selected by the Stepwise procedure, comprised seven variables. After removing non-significant ones, until the best LOOCV, AIC and pseudo R-square were obtained, a final model with five variables was retained that explains almost 62% of the deviance ($pseudo\ R^2 = 0.618$) (Table 4).

On the linear scale, the formula of the best model is:

$$log(EPR) = \beta_0 + \beta_1 ED1 + \beta_2 SHAPE\_MN1 + \beta_3 TE2 + \beta_4 ED2 + \beta_5 AC$$

Table 4. Best GLM obtained by Stepwise procedure and supervised selection. For each spatial parameter, estimated coefficients, z-value, variance inflation factor (VIF) and p-value are reported. AIC (Akaike's Information Criterion), R-square, and Leave-One-Out Cross-Validation (LOOCV)-error are reported at the bottom of the table.

Independent variables	Estimated coefficients β	z value	VIF	p-value
Intercept	-1.43 E-01	-0.366		0.71416
ED1	4.21 E+01	2.638	1.30	8.34 E-03
SHAPE_MN1	-2.67 E-01	-3.154	1.27	1.61 E-03
TE2	6.80 E-05	2.864	1.66	4.19 E-03
ED2	-2.47 E+01	-2.328	1.01	1.99 E-02
AC	2.06 E-02	7.416	1.38	1.21 E-13
AIC	268.54			
pseudo R-square	0.62			
LOOCV-error	1.93			

A similar model was also obtained by replacing ED2 with PLAND2, but with a worse predictive ability (larger LOOCV value, equal to 2.25).

To assess the relative importance of individual variables in the model, we initially standardised all variables within the optimal model. Subsequently, we re-estimated the model and arranged the coefficients in descending order, thereby ranking their relevance from the largest to the smallest.

Based on the standardisation of variables and their coefficients, the proportion of contacts of the focal patch with artificial surfaces (AC) resulted to be the most significant variable. As expected, apart from the artificial contacts (AC), EPR was found to also be positively affected by artificial edge density (ED1), as consequence of the recognised urban sprawl in the study area, and by the total edge of agricultural surfaces (TE2), often characterised by field margins and hedgerows with exotic species (such as *Arundo donax, Populus* x *canadensis* and *Eucalyptus camaldulensis*).

The opposite effect of the edge density of agricultural surfaces (ED2), akin to that of the prevailing agricultural matrix in the surrounding landscape (PLAND2), is instead ascribable to a protection effect from invasion provided by dominant traditional agricultural practices in the study area (mainly extensive and non-irrigated arable land). A reduction in EPR was also observed with respect to the complex shape of artificial surfaces (SHAPE_MN1), a parameter that increases with the prevalence of roads over the urban fabric due to the high thematic resolution of the basic land cover map. Indeed, although roads represent important fragmenting elements, in the study area they drive a huge amount of individuals but a limited number of species, mainly represented by *Robinia pseudoacacia* and *Ailanthus altissima*.

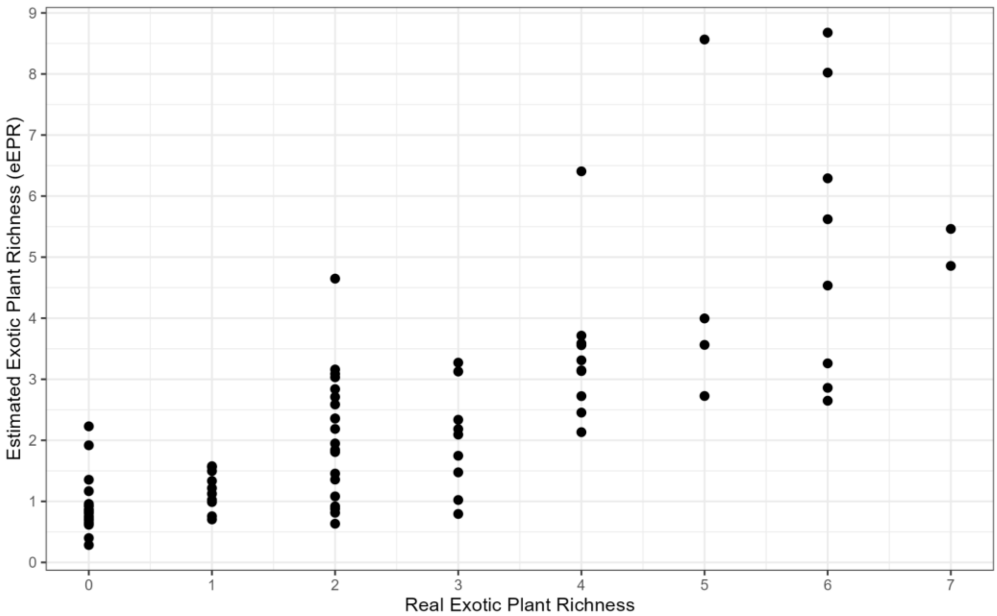


Figure 4. Dispersion plot between estimated EPR (GLM coefficients) and real EPR in explored patches. Spearman's correlation = 0.86.

The dispersion plot referred to the 88 sampled patches, obtained with GLM coefficients, showed a strong correlation between estimated and real EPR (Spearman's rank of 0.86; Fig. 4). The calculation was thus repeated to infer the EPR for the overall 1180 shrubland patches in the study area, suddenly arranged into four different categories of invasibility, useful for the subsequent prioritisation of GI actions (Fig. 5):

- low threatened shrublands, with an estimated EPR (eEPR) between 0.0 and 2.0 and covering a total of 1457 ha;
- medium-low threatened shrublands, with an eEPR between 2.1 and 4.0 and covering a total of 252 ha;
- medium-high threatened shrublands, with an eEPR between 4.1 and 6.0 and covering a total of 120 ha;
- highly threatened shrublands, with an eEPR higher than 6.0 and covering a total of 94 ha.

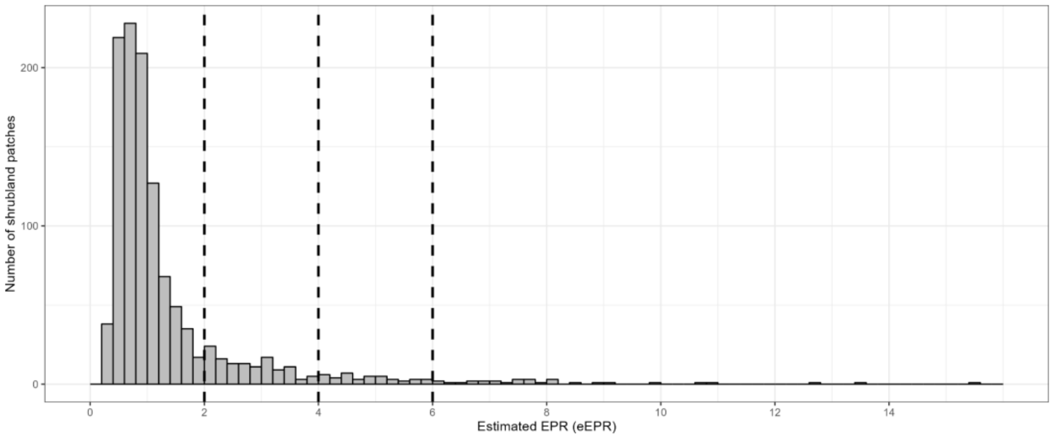


Figure 5. Inferred invasibility for the shrublands in Valle Galeria.

3.3. Green Infrastructure proposal

The proposed GI for the containment of plant invasion (Fig. 6), is composed of areal (4744 ha) and linear components (95 km of primary road verges), to be alternatively conserved or restored according to their current status and inferred vulnerability to invasion (Table 5).

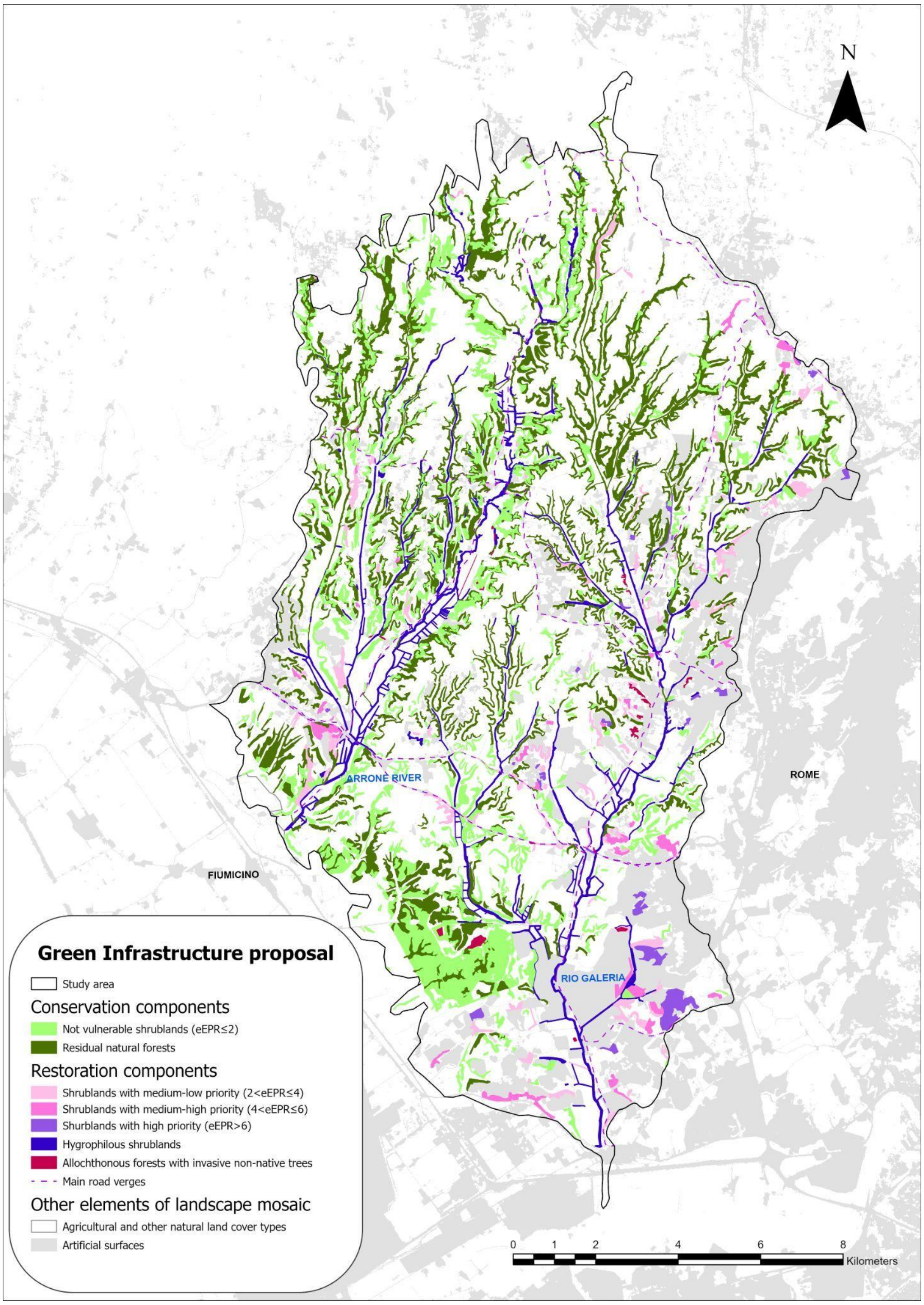


Figure 6. Green Infrastructure proposal for the containment of plant invasion in Valle Galeria.

Table 5. Absolute and proportional extent of GI components.

GI component types	Area/Length	% of the GI	% of the study area
Conservation components			
Residual natural forests	2500 ha	52.4	10.9
Not vulnerable shrublands (eEPR≤2)	1457 ha	30.5	6.3
Restoration components			
Shrublands with medium-low priority (2<eEPR≤4)	252 ha	5.3	1.1
Shrublands with medium-high priority (4<eEPR≤6)	120 ha	2.5	0.5
Shrublands with high priority (eEPR>6)	94 ha	1.9	0.4
Hygrophilous shrublands	292 ha	6.1	1.3
Allochthonous forests with invasive non-native trees	59 ha	1.2	0.3
Main road verges	95 km		(22% of the overall road network = 421 km)
Total	4774 ha	100	20.8

Conservation components embrace residual natural forests, along with those shrublands that emerged as not vulnerable to invasion from the inference process.

The nuclei of natural forests, intrinsically resistant to plant invasion, were selected to be preserved both as important barriers against the dispersal of non-native trees and as seed sources for the eventual recolonisation by native woody species of abandoned fields, residual spaces in the agricultural matrix, and earlier successional vegetation stages (Motta et al., 2009). In the case of coppices, not discerned from less disturbed forest patches in the adopted land use/land cover map, the eventual spread of exotic trees is meant to be prevented by proactive involvement of landowners and promotion of sustainable forestry (Radtke et al., 2013; Brundu et al., 2020).

Similarly, low threatened shrublands are subject to minimal interventions, mainly devoted at preventing disturbances other than biological invasion (e.g. fires and soil degradation) and eventually allowing the spontaneous recovery of mature forest ecosystems (i.e. "passive restoration"; Chazdon et al., 2021).

Restoration components include threatened shrublands, with an increasing priority assigned along with increasing vulnerability to woody plant invasion (inferred EPR value), and hygrophilous shrublands (not inferred, but considered very vulnerable by default), allochthonous forests, and verges of main roads.

For all these types of components, either alternative or combined removal of invasive exotic species, with a special reference to available guidelines for effective eradication (EPPO, 2019), and reforestation, with a special reference to biogeographic and ecological coherence of planted trees (Capotorti et al., 2016; EC, 2023), have to be pursued. Such actions involve a total of 817 ha, plus 95 km of road verges, that is 3.6% of the entire territory for the areal components and 22% of the entire road network for the linear ones.

Threatened shrublands, identified by means of statistical inference, represent preferential reforestation sites, as they fall for sure in no more cultivated lands and would not cause conflicts for land use allocation between ecological restoration and agricultural production. The respective patches that exceed 1 ha (95 patches, for a total surface of 409 ha) also meet the minimum eligibility criteria posed by the NRRP investment for urban and peri-urban forestation in Italy (MASE, 2021).

Hygrophilous shrublands contribute to the same target, especially with 8 patches larger than 1 ha. This type of secondary ecosystem was directly integrated into restoration components, without passing through inference, due to its belonging to the most altered and exploited environmental unit in the study area and subsequent susceptibility to invasions (Moss & Monstadt, 2008).

Allochthonous forests, dominated by *Eucalyptus camaldulensis* and *Robinia pseudoacacia,* were directly included among components to be converted into native forests as well. Notwithstanding some associated regulating values, they actually represent additional and widespread sources of propagules and seeds of non-native invasive trees, able to colonise semi-natural ecosystems even far away from residential areas and grey infrastructures.

Verges of main roads with high vehicular traffic may serve as dispersal pathways for both native and non-native species (Arevalo et al., 2010; Von der Lippe and Kowarik 2007), but the latter are often better adapted to altered substrata and more resistant against disturbance from road maintenance (Pollnac et al., 2012; Szilassi et al., 2021). Thus, in these habitats, both removal of allochthonous trees and plantation of native woody species can help contain the invasion process while facilitating the recovery of autochthonous forests.

4. Discussion

In keeping with the strategic targets of the Convention on Biological Diversity for the control of biological invasions (https://www.cbd.int/gbf/targets/), a GI planning procedure is proposed and applied, aimed at controlling the spread of woody non-native plants in a dynamic peri-urban case context. Potentially invasive species were identified with field surveys, while fine scale investigation of the relationship between EPR and landscape pattern provided new insights about the characteristics of land cover mosaic that mainly facilitate plant invasion. This way, it was possible to construct an original, inference-based procedure for setting ecosystem conservation and restoration priorities, useful to the protection of local biodiversity and enhancement of territorial resilience.

4.1. Threatening non-native plants in the western peri-urban sector of Rome

A number of invasive plants have been sampled in the investigated peri-urban sector, which can synergically threaten residual natural and semi-natural ecosystems, respective biodiversity, and capacity to provide services, therefore deserving targeted GI actions to be managed and contained (Potgieter et al., 2022).

Among these, *Ailanthus altissima* is recognised to directly threaten various Mediterranean vegetation types, including grasslands, shrublands, degraded forests, and riparian forests. As a light-demanding species, it also easily colonises and thrives in urban environments, along roadsides/railways, and in abandoned agricultural fields. Once established, the species forms almost pure stands that limit species diversity and alter natural soil conditions (Sladonja et al., 2015; Constán-Nava et al., 2010), thus preventing and/or altering spontaneous natural succession in the invaded sites.

Robinia pseudoacacia is considered among the top 40 most invasive woody angiosperms worldwide. It thrives in high light environments and can quickly spread over different types of habitats, from dry to moderately moist. Along with *Ailanthus altissima*, *R. pseudoacacia* can quickly take over abandoned fields, pastures, grasslands, shrublands, coppice forests, urban areas, roadsides, and alluvial habitats, especially in modern landscapes subject to widespread disturbances (Vítková et al., 2017; Gentili et al., 2019).

Acer negundo is able to outcompete native tree species, reducing undergrowth density and biodiversity while modifying forest structure, especially in resource-rich environments like riparian forests with canopy gaps (Sikorska et al., 2019).

Acacia dealbata is one of the most invasive species in Mediterranean Europe, usually introduced for ornamental purposes and forestry. It exerts marked effects on soil (increase in total Nitrogen and decrease in pH) and on plant community structure and composition. *A. dealbata* especially invades sclerophyllous native vegetation, representing a serious threat for Mediterranean maquis (Lazzaro et al., 2014). In Spain, it has also been reported that invasion of *A. dealbata* in the understory of *Quercus robur* forests reduces species richness and plant cover but also modifies the soil seed bank, by facilitating the abundance of exotic species seeds (e.g. *Conyza* sp.) (González-Muñoz et al., 2012).

Eucalyptus camaldulensis is the only species of the genus *Eucalyptus* reported to be invasive in Italy, especially in sectors with a Mediterranean climate. Although not as widespread as *A. altissima* and *R. pseudoacacia*, it shows a great ability to exploit disturbed areas and quickly spread over riparian habitats, maquis, roadsides, and urban environments. Riparian ecosystems seem to be the most endangered, except for poplar woodlands, with marked native species replacement and alteration of ecosystem functioning (Badalamenti et al., 2018). The species is actually able to diminish water availability in soils, which can lead to reduced streamflow and altered water regimes and may especially impact hygrophilous ecosystems in dry climates (Dzikiti et al., 2016).

Arundo donax is very invasive worldwide and especially threatens riparian and floodplain areas, where it forms dense pure stands that reduce plant richness, almost completely alter natural vegetation structure, and change fire regimes. Out of its native range, however, *A. donax* propagates only by vegetative propagation, making its spread predictable. For the containment, *Salix* species have been proven to actively compete with the giant reed for space and light, whilst other plants, such as *Populus* sp., *Tamarix* sp., and *Sambucus* sp., are less effective (Jiménez-Ruiz et al., 2021).

4.2. Local-scale landscape context effect and Green infrastructure planning aimed at plant invasion control in peri-urban contexts

Even though the present research focused specifically on secondary shrublands and woody species, it confirmed that richness of exotic plants in semi-natural ecosystems is highly dependent on the surrounding landscape pattern (Lázaro-Lobo and Ervin, 2021), and especially on artificial surfaces (González-Moreno et al., 2013). In the present case study, this effect was found to be mainly related to the direct contiguity with urban fabric, but not to the road network in the neighbouring buffer zone. Notwithstanding roadsides have been recognised to facilitate invasive plants dispersal (Christen and Matlack, 2006), the little effect that has been observed here may be due to a different behaviour of woody species compared to herbaceous ones and to specific features of the analysed landscape context (prone to convey the dispersal of many individuals of invasive woods, but just of the few species commonly planted as roadside trees).

Peculiarity of the landscape matrix, with residual traditional features (Biasi et al., 2015), may also explain the observed buffering effect of agricultural areas against EPR in shrublands, validating the hypothesis that intensity, rather than just occurrence, of agricultural practices affects the degree of plant invasion (Pellegrini et al., 2021).

In keeping with these results, the landscape-level actions that emerged as important priorities for the containment of plant invasion in such a peri-urban context concomitantly include: i) the preservation of natural and semi-natural ecosystems from diffuse direct contacts with artificial surfaces, that is the prevention and mitigation of urban sprawl over abandoned agricultural lands (Frondoni et al., 2011, Smiraglia et al., 2021; and ii) the maintenance and promotion of traditional agricultural practices (Zavattero et al., 2021), able to provide buffer zones for the preservation of biogeographically coherent species in present day semi-natural remnant ecosystems and along their progressive dynamics. These actions should complement targeted interventions at the patch level, aimed at alternatively conserving natural woodland remnants, converting woodlands dominated by alien plants, and assisting forest recovery in unexploited areas, for a general improvement of the landscape mosaic condition and resistance against pressures posed by alien species.

As regards the overall GI planning process, which goes beyond defining conservation and restoration actions at multiple levels, several principles have been already outlined to deal with different aspects of sustainability and resilience in urban systems (Monteiro et al., 2020). In the same Metropolitan City of Rome, a number of GI plans have been proposed focusing on the combination between biodiversity support, ecological connectivity and ecosystem service provision, both in the inner city and at the rural-urban interface (Capotorti et al., 2019, 2023; Valeri et al., 2021).

Moving from this quite consolidated knowledge, the present work suggests an advancement also in the prioritisation of GI components, which was based upon a statistical inference of ecosystem vulnerability to plant invasion. Even though the strong representativeness of the adopted inference model (pseudo R-square = 0.62) is limited to few life-forms of invaders (mainly phanerophytes), the special focus on shrublands, as potentially invaded ecosystems, allowed ecological restoration efforts to be concentrated on a limited portion of the territory. The occurrence of secondary communities in such a peri-urban landscape should actually be interpreted as a symptom of loosening of cultivation practices (Fayet et al., 2022) and, therefore, their restoration can avoid, or at least minimise, potential land use conflicts with persistent production activities (Cortina-Segarra et al., 2021). Moreover, this type of ecosystem easily accommodates the establishment of shade-tolerant species typical of late-successional stages, making reforestation efforts more rapid and potentially effective with respect to open field and grassland restoration (Chazdon et al., 2021). Active guidance of the forest recovery process, by facilitating PNV coherent species, is anyway essential to avoid spontaneous trajectories towards novel ecosystems dominated by invasive aliens and with low natural values (Kowarik and von der Lippe, 2018).

Along with dynamic shrublands, overall forest remnants have been included among the GI priority components. Conservation of more natural types is expected to provide resilience to the ecological network by means of widespread nuclei of native species seeds (Martinez-Baroja et al., 2022; Holmes et al., 2020), whilst conversion of non-native woodlands would support an effective containment of invasive alien spread over treeless areas (Rundel et al., 2014).

Such combined GI actions and components are consistent with the post-2020 global biodiversity framework (Nicholson et al., 2021), especially as regards goal 3, for the retainment of remnant valuable ecosystems (mature forests and low vulnerable shrublands, in the case study), goal 2, for restoration of degraded ones (high vulnerable shrublands, allochthonous forests, lowland riparian shrublands and road verges), and goal 6, to reduce the spread of alien species. Moreover, they contribute addressing two sustainability challenges typical of the urban-rural interface (Geneletti et al., 2017), i.e. the re-activation and valorisation of declining traditional agricultural landscapes and the enhancement of peri-urban forests as ecosystem service providers close to resident population (Vallecillo et al., 2018).

Optimisation of benefits, with respect to costs, should hence take into account additional deployment and maintenance options, not specifically addressed by the present research, especially for the active management of alien species. Anyway, the proposed GI is not intended to be a measure for their complete eradication, but rather a tool for supporting widespread self-sustaining natural ecosystems.

Finally, the GI proposal could be further improved with i) the inclusion of semi-natural grasslands, as GI components susceptible to invasions (Fernandes et al., 2018), and ii) a multi-temporal analysis of land cover dynamics, as an additional variable to be considered in the inference model (Malavasi et al., 2014).

5. Conclusions

Understanding the mechanism of biological invasions across landscapes is fundamental to promote environmental management practices that are able to effectively preserve local biodiversity. The integration between landscape measures and floristic samplings enabled the research to shed light on which landscape features are most involved in the successful spread of woody exotic plants over semi-natural ecosystem remnants in a peri-urban context. In particular, direct contacts with artificial surfaces, due to the documented urban sprawl in the study area, resulted to be the major driver of plant invasions. On the contrary, the agricultural matrix, mainly retaining a traditional character, was proven to be responsible for a buffer effect that decreases the probability of exotic plants spreading.

To the best of our knowledge, the inferred susceptibility to plant invasion, derived from a predictive GLM, has been thus adopted in a local-scale GI planning process for the first time. Notwithstanding some margins for improvement, the proposed GI could support local authorities in achieving key targets for biodiversity conservation and restoration, and testing win-win solutions for both nature and people aimed at the sustainable development of a metropolitan peripheral sector.

References

1. Arévalo, J. R., Otto, R., Escudero, C., Fernández-Lugo, S., Arteaga, M., Delgado, J. D., & Fernández-Palacios, J. M., 2010. Do anthropogenic corridors homogenize plant communities at a local scale? A case studied in Tenerife (Canary Islands). Plant Ecology, 209, 23-35. https://doi.org/10.1007/s11258-009-9716-y.
2. Badalamenti, E., Cusimano, D., La Mantia, T., Pasta, S., Romano, S., Troia, A., & Ilardi, V., 2018. The ongoing naturalisation of Eucalyptus spp. in the Mediterranean Basin: new threats to native species and habitats. Australian Forestry, 81(4), 239-249. https://doi.org/10.1080/00049158.2018.1533512.
3. Bartolucci, F., Peruzzi, L., Galasso, G., Albano, A., Alessandrini, A. N. M. G., Ardenghi, N. M. G., ... & Conti, F., 2018. An updated checklist of the vascular flora native to Italy. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 152(2), 179-303. https://doi.org/10.1080/11263504.2017.1419996.
4. Biasi, R., Colantoni, A., Ferrara, C., Ranalli, F., & Salvati, L., 2015. In-between sprawl and fires: Long-term forest expansion and settlement dynamics at the wildland–urban interface in Rome, Italy. International Journal of Sustainable Development & World Ecology, 22(6), 467-475. https://doi.org/10.1080/13504509.2015.1064488.
5. Blasi, C., Capotorti, G., Copiz, R., Guida, D., Mollo, B., Smiraglia, D., & Zavattero, L., 2014. Classification and mapping of the ecoregions of Italy. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 148(6), 1255-1345. https://doi.org/10.1080/11263504.2014.985756.
6. Boscutti, F., Lami, F., Pellegrini, E., Buccheri, M., Busato, F., Martini, F., Sibella, R., Sigura, M., Marini, L., 2022. Urban sprawl facilitates invasions of exotic plants across multiple spatial scales. Biological Invasions, 24(5), 1497-1510. https://doi.org/10.1007/s10530-022-02733-6.
7. Boscutti, F., Sigura, M., De Simone, S., Marini, L., 2018. Exotic plant invasion in agricultural landscapes: A matter of dispersal mode and disturbance intensity. Applied Vegetation Science, 21(2), 250-257. https://doi.org/10.1111/avsc.12351.
8. Botham, M. S., Rothery, P., Hulme, P. E., Hill, M. O., Preston, C. D., & Roy, D. B., 2009. Do urban areas act as foci for the spread of alien plant species? An assessment of temporal trends in the UK. Diversity and Distributions, 15(2), 338-345.https://doi.org/10.1111/j.1472-4642.2008.00539.x.
9. Brundu, G., & Richardson, D. M., 2017. Code of conduct for invasive alien trees. Council of Europe Publishing, Strasbourg. easin.jrc.ec.europa.eu.
10. Brundu, G., Pauchard, A., Pyšek, P., Pergl, J., Bindewald, A. M., Brunori, A., Canavan, S., Campagnaro, T., Celesti-Grapow, L., Dechoum, M. S. D., Dufour-Dror, J. M., Essl, F., Flory, S. L., Genovesi, P., Guarino, F., Guangzhe, L., Hulme, P. E., Jäger, H., Kettle, C. J., Krumm, F., Langdon, B., Lapin, K., Lozano, V., Le Roux, J. J., Novoa, A., Nuñez, M. A., Porté, A. J., Silva, J. S., Schaffner, U., Sitzia, T., Tanner, R., Tshidada, N., Vítková, M., Westergren, M., Wilson, J. R. U., & Richardson, D. M. (2020). Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. NeoBiota 61: 65–116. 10.3897/neobiota.65.58380 http://neobiota.pensoft.net.
11. Campagnaro, T., Brundu, G., Burrascano, S., Celesti-Grapow, L., La Mantia, T., Sitzia, T., & Badalamenti, E., 2022. Tree invasions in Italian forests. Forest Ecology and Management, 521, 120382. https://doi.org/10.1016/j.foreco.2022.120382.
12. Capotorti, G., Del Vico, E., Anzellotti, I., Celesti-Grapow, L., 2016. Combining the conservation of biodiversity with the provision of ecosystem services in urban green infrastructure planning: Critical features arising from a case study in the metropolitan area of Rome. Sustainability, 9(1), 10. https://doi.org/10.3390/su9010010.
13. Capotorti, G., Ortí, M. M. A., Copiz, R., Fusaro, L., Mollo, B., Salvatori, E., Zavattero, L., 2019. Biodiversity and ecosystem services in urban green infrastructure planning: A case study from the metropolitan area of Rome (Italy). Urban Forestry & Urban Greening, 37, 87-96. https://doi.org/10.1016/j.ufug.2017.12.014.
14. Capotorti, G., Valeri, S., Giannini, A., Minorenti, V., Piarulli, M., & Audisio, P., 2023. On the Role of Natural and Induced Landscape Heterogeneity for the Support of Pollinators: A Green Infrastructure Perspective Applied in a Peri-Urban System. Land, 12(2), 387. https://doi.org/10.3390/land12020387.

15. Celesti-Grapow, L., Capotorti, G., Del Vico, E., Lattanzi, E., Tilia, A., Blasi, C., 2013. The vascular flora of Rome. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 147(4), 1059-1087. https://doi.org/10.1080/11263504.2013.862315.
16. Celesti-Grapow, L., Marzio, P. D., Blasi, C., 2001. The importance of alien and native species in the urban flora of Rome (Italy). Plant invasions: species ecology and ecosystem management, 209-220. https://doi.org/10.1016/j.biocon.2011.12.028.
17. Chazdon, R. L., Falk, D. A., Banin, L. F., Wagner, M., J Wilson, S., Grabowski, R. C., Suding, K. N., 2021. The intervention continuum in restoration ecology: rethinking the active–passive dichotomy. Restoration Ecology, e13535. https://doi.org/10.1111/rec.13535.
18. Christen, D., Matlack, G., 2006. The role of roadsides in plant invasions: a demographic approach. Conservation Biology, 20(2), 385-391. https://doi.org/10.1111/j.1523-1739.2006.00315.x.
19. CIRBFEP (Centro di Ricerca Interuniversitario Biodiversità, Fitosociologia ed Ecologia del paesaggio), 2013. Carta delle serie di vegetazione e vegetazione naturale potenziale della Provincia di Roma. https://geoportale.cittametropolitanaroma.it/progetti/banche-dati-vegetazionali-2013-2014/serie-di-vegetazione Accessed on: 21/03/2024
20. Constán-Nava, S., Bonet, A., Pastor, E., Lledó, M. J., 2010. Long-term control of the invasive tree Ailanthus altissima: Insights from Mediterranean protected forests. Forest Ecology and Management, 260(6), 1058-1064. https://doi.org/10.1016/j.foreco.2010.06.030.
21. Cortina-Segarra, J., García-Sánchez, I., Grace, M., Andrés, P., Baker, S., Bullock, C., Decleer, K., Dicks, L. V., Fisher, J. L., Frouz, J., Klimkowska, A., Kyriazopoulos, A. P., Moreno-Mateos, D., Rodríguez-González, P. M., Sarkki, S., Ventocilla, J. L., 2021. Barriers to ecological restoration in Europe: expert perspectives. Restoration Ecology, 29(4), e13346. https://doi.org/10.1111/rec.13346.
22. D'Antonio, C.M., August-Schmidt, E., Fernandez-Going, B., 2016. Invasive Species and Restoration Challenges, in: Palmer, M.A., Zedler, J.B., Falk, D.A. (Eds) Foundations of Restoration Ecology. Island Press, Washington, DC. https://doi.org/10.5822/978-1-61091-698-1_8.
23. Dzikiti, S., Gush, M. B., Le Maitre, D. C., Maherry, A., Jovanovic, N. Z., Ramoelo, A., Cho, M. A., 2016. Quantifying potential water savings from clearing invasive alien Eucalyptus camaldulensis using in situ and high resolution remote sensing data in the Berg River Catchment, Western Cape, South Africa. Forest Ecology and Management, 361, 69-80. https://doi.org/10.1016/j.foreco.2015.11.009.
24. EC (European Commission), 2013. Green Infrastructure (GI) – Enhancing Europe's Natural Capital. COM (2013) 249 Final. Available online: http://ec.europa.eu/environment/nature/ecosystems/docs/green_infrastructures/1_EN_ ACT_part1_v5.pdf (Accessed on 21 March 2024).
25. EC (European Commission), 2023. Guidelines on Biodiversity-Friendly Afforestation, Reforestation and Tree Planting. Guidelines on Biodiversity-Friendly Afforestation, Reforestation and Tree Planting https://environment.ec.europa.eu/publications/guidelines-biodiversity-friendly-afforestation-reforestation-and-tree-planting_en.
26. EPPO (European and Mediterranean Plant Protection Organization), 2019. PM 9/29 (1) Ailanthus altissima. EPPO Bulletin, 50, 148-155. https://doi.org/10.1111/epp.12621.
27. European Alien Species Information Network. EASIN - European Alien Species Information Network. Accessed 21 March 2024.
28. Fayet, C. M., Reilly, K. H., Van Ham, C., Verburg, P. H., 2022. What is the future of abandoned agricultural lands? A systematic review of alternative trajectories in Europe. Land Use Policy, 112, 105833. https://doi.org/10.1016/j.landusepol.2021.105833
29. Fenesi, A., Vágási, C. I., Beldean, M., Földesi, R., Kolcsár, L. P., Shapiro, J. T., Török, E., Kovács-Hostyánszki, A., 2015. Solidago canadensis impacts on native plant and pollinator communities in different-aged old fields. Basic and Applied Ecology, 16(4), 335-346. https://doi.org/10.1016/j.baae.2015.03.003.
30. Fernandes, P., Máguas, C., Correia, O., González-Moreno, P., 2018. What drives Eucalyptus globulus natural establishment outside plantations? The relative importance of climate, plantation and site characteristics. Biological Invasions, 20(5), 1129-1146. https://doi.org/10.1007/s10530-017-1614-y.
31. Frondoni, R., Mollo, B., Capotorti, G., 2011. A landscape analysis of land cover change in the Municipality of Rome (Italy): Spatio-temporal characteristics and ecological implications of land cover transitions from 1954 to 2001. Landscape and Urban planning, 100(1-2), 117-128. https://doi.org/10.1016/j.landurbplan.2010.12.002.

32. Funk, J.L., Cleland, E.E., Suding, K.N., Zavaleta, E.S., 2008. Restoration through reassembly: plant traits and invasion resistance. Trends Ecol. Evol. 23, 695–703. https://doi.org/10.1016/j.tree.2008.07.013.
33. Gaertner, M., Biggs, R., Te Beest, M., Hui, C., Molofsky, J., Richardson, D. M., 2014. Invasive plants as drivers of regime shifts: identifying high-priority invaders that alter feedback relationships. Diversity and Distributions, 20(7), 733-744. https://doi.org/10.1111/ddi.12182.
34. Galasso, G., Conti, F., Peruzzi, L., Ardenghi, N. M. G., Banfi, E., Celesti-Grapow, L., Albano, A., Alessandrini, A., Bacchetta, G., Ballelli, S., Bandini Mazzanti, M., Barberis, G., Bernardi, L., Blasi, C., Bouvet, D., Bovio, M., Cecchi, L., Del Guacchio, E., Domina, G., Fascetti, S., Gallo, L., Gubellini, L., Guiggi, A., Iamonico, D., Iberite, M., Jiménez-Mejías, P., Lattanzi, E., Marchetti, D., Martinetto, E., Masin, R. R., Medagli, P., Passalacqua, N. G., Peccenini, S., Pennesi, R., Pierini, B., Podda, L., Poldini, L., Prosser, F., Raimondo, F. M., Roma-Marzio, F., Rosati, L., Santangeli, A., Scoppola, A., Scortegagna, S., Selvaggi, A., Selvaggio, F., Soldano, A., Stinca, A., Wagensommer, R. P., Wilhalm, T., Bartolucci, F., 2018. An updated checklist of the vascular flora alien to Italy. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 152(3), 556-592. https://doi.org/10.1080/11263504.2018.1441197.
35. Geneletti, D., La Rosa, D., Spyra, M., Cortinovis, C., 2017. A review of approaches and challenges for sustainable planning in urban peripheries. Landscape and Urban Planning, 165, 231-243. https://doi.org/10.1016/j.landurbplan.2017.01.013.
36. Gentili, R., Ferrè, C., Cardarelli, E., Montagnani, C., Bogliani, G., Citterio, S., Comolli, R., 2019. Comparing negative impacts of Prunus serotina, Quercus rubra and Robinia pseudoacacia on native forest ecosystems. Forests, 10(10), 842. https://doi.org/10.3390/f10100842.
37. González-Moreno, P., Pino, J., Carreras, D., Basnou, C., Fernández-Rebollar, I., Vila, M., 2013. Quantifying the landscape influence on plant invasions in Mediterranean coastal habitats. Landscape ecology, 28, 891-903. https://doi.org/10.1007/s10980-013-9857-1.
38. González-Muñoz, N., Costa-Tenorio, M., Espigares, T., 2012. Invasion of alien Acacia dealbata on Spanish Quercus robur forests: impact on soils and vegetation. Forest Ecology and Management, 269, 214-221. https://doi.org/10.1016/j.foreco.2011.12.026.
39. Hattendorf, J., Hansen, S. O., Nentwig, W., 2007. Defence systems of Heracleum mantegazzianum. In Ecology and management of giant hogweed (Heracleum mantegazzianum). Wallingford UK: CABI pp. 209-225. https://doi.org/10.1079/9781845932060.020.
40. Holmes, P. M., Esler, K. J., Van Wilgen, B. W., Richardson, D. M., 2020. Ecological restoration of ecosystems degraded by invasive alien plants in South African Fynbos: Is spontaneous succession a viable strategy?. Transactions of the Royal Society of South Africa, 75(2), 111-139. https://hdl.handle.net/10520/ejc-royalsa_v75_i2_a111.
41. Jiménez-Ruiz, J., Hardion, L., Del Monte, J. P., Vila, B., Santín-Montanyá, M. I., 2021. Monographs on invasive plants in Europe N° 4: Arundo donax L. Botany Letters, 168(1), 131-151. https://doi.org/10.1080/23818107.2020.1864470.
42. Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. Environmental pollution, 159(8-9), 1974-1983. https://doi.org/10.1016/j.envpol.2011.02.022.
43. Kowarik, I., von der Lippe, M., 2018. Plant population success across urban ecosystems: A framework to inform biodiversity conservation in cities. Journal of Applied Ecology, 55(5), 2354-2361. https://doi.org/10.1111/1365-2664.13144.
44. Kumar, S., Stohlgren, T. J., Chong, G. W., 2006. Spatial heterogeneity influences native and nonnative plant species richness. Ecology, 87(12), 3186-3199. https://doi.org/10.1890/0012-9658(2006)87[3186:SHINAN]2.0.CO;2.
45. Lambert, A., Dudley, T., Saltonstall, K., 2010. Ecology and Impacts of the Large-Statured Invasive Grasses Arundo donax and Phragmites australis in North America. Invasive Plant Science and Management, 3(4). 489-494. https://doi.org/10.1614/IPSM-D-10-00031.1.
46. Lázaro-Lobo, A., Ervin, G. N., 2021. Native and exotic plant species respond differently to ecosystem characteristics at both local and landscape scales. Biological Invasions, 23, 143-156. https://doi.org/10.1007/s10530-020-02361-y.
47. Lazzaro, L., Giuliani, C., Fabiani, A., Agnelli, A. E., Pastorelli, R., Lagomarsino, A., Benesperi, R., Calamassi, R., Foggi, B., 2014. Soil and plant changing after invasion: the case of Acacia dealbata in a Mediterranean ecosystem. Science of the total environment, 497, 491-498. https://doi.org/10.1016/j.scitotenv.2014.08.014.

48. Lockwood, J. L., Hoopes, M. F., Marchetti, M. P., 2013. Invasion ecology. John Wiley & Sons. https://doi.org/10.1111/aec.12295.
49. Malavasi, M., Carboni, M., Cutini, M., Carranza, M. L., & Acosta, A. T., 2014. Landscape fragmentation, land-use legacy and propagule pressure promote plant invasion on coastal dunes: a patch-based approach. Landscape Ecology, 29, 1541-1550. https://doi.org/10.1007/s10980-014-0074-3.
50. Maron, J., Marler, M., 2007. Native plant diversity resists invasion at both low and high resource levels. Ecology, 88(10), 2651-2661. https://doi.org/10.1890/06-1993.1.
51. Martinez-Baroja, L., Rey-Benayas, J. M., Perez-Camacho, L., Villar-Salvador, P., 2022. Drivers of oak establishment in Mediterranean old fields from 25-year-old woodland islets planted to assist natural regeneration. European Journal of Forest Research, 141(1), 17-30. https://doi.org/10.1007/s10342-021-01423-7.
52. MASE (Italian Ministry of Environment and Energy Security), 2021. Piano di riforestazione urbana ed extraurbana, PNRR Misura 2 – Componente 4. PNRR_INVESTIMENTO_3.1_Avviso_pubblico.pdf https://www.mase.gov.it/sites/default/files/archivio/bandi/PNRR_INVESTIMENTO_3.1_Avviso_pubblico.pdf .
53. McGarigal, K., 2015. FRAGSTATS help. University of Massachusetts: Amherst, MA, USA, 182. https://doi.org/10.2737/PNW-GTR-351.
54. Meiners, S. J., Pickett, S. T., Cadenasso, M. L., 2002. Exotic plant invasions over 40 years of old field successions: community patterns and associations. Ecography, 25(2), 215-223. https://doi.org/10.1034/j.1600-0587.2002.250209.x.
55. Mercuri, A. M., Mazzanti, M. B., Florenzano, A., Montecchi, M. C., & Rattighieri, E., 2013. Olea, Juglans and Castanea: the OJC group as pollen evidence of the development of human-induced environments in the Italian peninsula. Quaternary international, 303, 24-42. https://doi.org/10.1016/j.quaint.2013.01.005.
56. Monteiro, R., Ferreira, J. C., Antunes, P., 2020. Green infrastructure planning principles: An integrated literature review. Land, 9(12), 525. https://doi.org/10.3390/land9120525.
57. Moss, T., Monstadt, J., 2008. Restoring floodplains in Europe: Policy contexts and project experiences, IWA Publishing, London.
58. Motta, R., Nola, P., Berretti, R., 2009. The rise and fall of the black locust (Robinia pseudoacacia L.) in the "Siro Negri" Forest Reserve (Lombardy, Italy): lessons learned and future uncertainties. Annals of Forest Science, 66(4), 1-10. https://doi.org/10.1051/forest/2009012.
59. Mtati, N., Sachikonye, M., & Thondhlana, G., 2016. Unpacking Pandora's box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems, 19, 587-600. https://doi.org/10.1007/s10021-015-9952-z.
60. Nicholson, E., Watermeyer, K. E., Rowland, J. A., Sato, C. F., Stevenson, S. L., Andrade, A., Brooks, T. M., Burgess, N. D., Cheng, S.-T., Grantham, H. S., Hill, S. L., Keith, D. A., Maron, M., Metzke, D., Murray, N. J., Nelson, C. R., Obura, D., Plumptre, A., Skowno, A. L., Watson, J. E. M., 2021. Scientific foundations for an ecosystem goal, milestones and indicators for the post-2020 global biodiversity framework. Nature Ecology & Evolution, 5(10), 1338-1349. https://doi.org/10.1038/s41559-021-01538-5.
61. Palmer, M.A., Zedler, J.B., Falk, D.A., 2016. Ecological Theory and Restoration Ecology. In: Palmer, M.A., Zedler, J.B., Falk, D.A. (Eds) Foundations of Restoration Ecology. Island Press, Washington, DC. https://doi.org/10.5822/978-1-61091-698-1_1.
62. Parendes, L. A., Jones, J. A., 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the HJ Andrews Experimental Forest, Oregon. Conservation Biology, 14(1), 64-75. https://doi.org/10.1046/j.1523-1739.2000.99089.x.
63. Pellegrini, E., Buccheri, M., Martini, F., Boscutti, F., 2021. Agricultural land use curbs exotic invasion but sustains native plant diversity at intermediate levels. Scientific reports, 11(1), 8385. https://doi.org/10.1038/s41598-021-87806-7.
64. Pignatti, S., Guarino, R., La Rosa, M., 2017. Flora d'Italia, Edagricole, Milano.
65. Pizzolotto, R., Brandmayr, P., 1996. An index to evaluate landscape conservation state based on land-use pattern analysis and Geographic Information System techniques. Coenoses, 37-44. https://www.jstor.org/stable/43461163.
66. Polce, C., Kunin, W. E., Biesmeijer, J. C., Dauber, J., Phillips, O. L., ALARM Field Site Network., 2011 . Alien and native plants show contrasting responses to climate and land use in Europe. Global Ecology and Biogeography, 20(3), 367-379. https://doi.org/10.1111/j.1466-8238.2010.00619.x.

67. Pollnac, F., Seipel, T., Repath, C., Rew, L. J., 2012. Plant invasion at landscape and local scales along roadways in the mountainous region of the Greater Yellowstone Ecosystem. Biological Invasions, 14(8), 1753-1763. https://doi.org/10.1007/s10530-012-0188-y.
68. Portal of the Flora of Italy. Available at Portale della Flora d'Italia - Portal to the Flora of Italy (units.it). Accessed 21 March 2024.
69. Potgieter, L. J., Shrestha, N., Cadotte, M. W., 2022. Prioritizing sites for terrestrial invasive alien plant management in urban ecosystems. Ecological Solutions and Evidence, 3(3), e12160. https://doi.org/10.1002/2688-8319.12160.
70. Prank, M., Chapman, D. S., Bullock, J. M., Belmonte, J., Berger, U., Dahl, A., Jäger, S., Kovtunenko, I., Magyar, D., Niemelä, S., Rantio-Lehtimäki, A., Rodinkova, V., Sauliene, I., Severova, E., Sikoparija, B., Sofiev, M., 2013. An operational model for forecasting ragweed pollen release and dispersion in Europe. Agricultural and forest meteorology, 182, 43-53. https://doi.org/10.1016/j.agrformet.2013.08.003.
71. R Core Team, 2023. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.
72. Radtke, A., Ambraß, S., Zerbe, S., Tonon, G., Fontana, V., Ammer, C., 2013. Traditional coppice forest management drives the invasion of Ailanthus altissima and Robinia pseudoacacia into deciduous forests. Forest Ecology and Management, 291, 308-317. https://doi.org/10.1016/j.foreco.2012.11.022.
73. Rey-Benayas, J. M. R., Bullock, J. M., & Newton, A. C., 2008. Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. Frontiers in Ecology and the Environment, 6(6), 329-336. https://doi.org/10.1890/070057.
74. Rundel, P. W., Dickie, I. A., Richardson, D. M., 2014. Tree invasions into treeless areas: mechanisms and ecosystem processes. Biological invasions, 16, 663-675. https://doi.org/10.1007/s10530-013-0614-9 .
75. Salvati, L., 2015. Lost in complexity, found in dispersion:'Peripheral'development and deregulated urban growth in Rome. Cities, 47, 73-80. https://doi.org/10.1016/j.cities.2015.04.001.
76. Santala, K., Cardou, F., Yemshanov, D., Campioni, F., Simpson, M., Handa, I. T., Ryser, P., Aubin, I., 2022. Finding the perfect mix: An applied model that integrates multiple ecosystem functions when designing restoration programs. Ecological Engineering, 180, 106646. https://doi.org/10.1016/j.ecoleng.2022.106646.
77. Shackleton, C. M., Ruwanza, S., Sinasson Sanni, G. K., Bennett, S., De Lacy, P., Modipa, R.,
78. Sikorska, D., Sikorski, P., Archiciński, P., Chormański, J., Hopkins, R. J., 2019. You can't see the woods for the trees: invasive Acer negundo L. in urban riparian forests harms biodiversity and limits recreation activity. Sustainability, 11(20), 5838. https://doi.org/10.3390/su11205838.
79. Simberloff, D., Martin, J. L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vila, M., 2013. Impacts of biological invasions: what's what and the way forward. Trends in ecology & evolution, 28(1), 58-66. https://doi.org/10.1016/j.tree.2012.07.013.
80. Sladonja, B., Sušek, M., Guillermic, J., 2015. Review on invasive tree of heaven (Ailanthus altissima (Mill.) Swingle) conflicting values: assessment of its ecosystem services and potential biological threat. Environmental management, 56, 1009-1034. https://doi.org/10.1007/s00267-015-0546-5.
81. Smiraglia, D., Salvati, L., Egidi, G., Salvia, R., Giménez-Morera, A., Halbac-Cotoara-Zamfir, R., 2021. Toward a new urban cycle? A closer look to sprawl, demographic transitions and the environment in Europe. Land, 10(2), 127. https://doi.org/10.3390/land10020127.
82. Stoate, C., Baldi, A., Beja, P., Boatman, N. D., Herzon, I., Van Doorn, A., de Snoo, G. R., Rakosy, L., Ramwell, C., 2009. Ecological impacts of early 21st century agricultural change in Europe–a review. Journal of environmental management, 91(1), 22-46. https://doi.org/10.1016/j.jenvman.2009.07.005.
83. Szilassi, P., Soóky, A., Bátori, Z., Hábenczyus, A. A., Tölgyesi, C., Van Leeuwen, B., Tobak., Z., Csikós, N., 2021. Natura 2000 areas, road, railway, water, and ecological networks may provide pathways for biological invasion: A country scale analysis. Plants, 10(12), 2670. https://doi.org/10.3390/plants10122670.
84. Theoharides, K. A., Dukes, J. S., 2007. Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. New phytologist, 176(2), 256-273. https://doi.org/10.1111/j.1469-8137.2007.02207.x.
85. Valeri, S., Zavattero, L., Capotorti, G., 2021. Ecological connectivity in agricultural green infrastructure: Suggested criteria for fine scale assessment and planning. Land, 10(8), 807. https://doi.org/10.3390/land10080807.

86. Vallecillo, S., Polce, C., Barbosa, A., Castillo, C. P., Vandecasteele, I., Rusch, G. M., Maes, J., 2018. Spatial alternatives for Green Infrastructure planning across the EU: An ecosystem service perspective. Landscape and Urban Planning, 174, 41-54. https://doi.org/10.1016/j.landurbplan.2018.03.001.
87. Vila, M., Ibáñez, I., 2011. Plant invasions in the landscape. Landscape ecology, 26, 461-472. https://doi.org/10.1007/s10980-011-9585-3.
88. Vítková, M., Müllerová, J., Sádlo, J., Pergl, J., Pyšek, P., 2017. Black locust (Robinia pseudoacacia) beloved and despised: A story of an invasive tree in Central Europe. Forest ecology and management, 384, 287-302. https://doi.org/10.1016/j.foreco.2016.10.057.
89. Von der Lippe, M., Kowarik, I., 2007. Long-distance dispersal of plants by vehicles as a driver of plant invasions. Conserv Biol 21:986–996. https://doi.org/10.1111/j.1523-1739.2007.00722.x.
90. Zavattero, L., Frondoni, R., Capotorti, G., Copiz, R., Blasi, C., 2021. Towards the identification and mapping of traditional agricultural landscapes at the national scale: an inventory approach from Italy, Landscape Research, 46:7, 945-958. https://doi.org/10.1080/01426397.2021.1923678.
91. Zerbe, S., 1998. Potential natural vegetation: validity and applicability in landscape planning and nature conservation. Applied Vegetation Science, 1(2), 165-172. https://doi.org/10.2307/1478945.

Discussion

The research project has focused on the impact of urbanisation on vegetation and biodiversity, with the objective of advancing knowledge on key aspects, such as reference ecosystem typification and quantification of alteration processes. The ecological study of urban environments has gained increasing importance in recent years, driven in particular by the "One Health" paradigm, which recognises the close interconnection between the quality of life of citizens and that of their surrounding environment (WHO, 2023). Nevertheless, the scientific debate remains open on multiple issues, necessitating further investigation. Current environmental policies, increasingly supported by scientific evidence, call for large-scale restoration actions in urban ecosystems (UN, 2019; EC, 2020). However, the effectiveness of such actions depends on the availability of a robust body of ecological knowledge, which is often fragmented or absent (Maes et al., 2020). This study has sought to fill some of these knowledge gaps and provide applicative tools for the implementation of recently approved and binding European regulations. Indeed, a recurring limitation in restoration projects is the lack of an integrated approach that provides a general ecological framework and defines baselines for the actions to be undertaken (Suding, 2011). It would be methodologically incorrect and operationally unsuccessful, for example, to apply the same restoration actions across diverse urban environments regardless of different natural potentials. Different ecosystems require tailored management measures, based on sound environmental knowledge (Dickson et al., 2014). Such knowledge should be acquired through robust, formalised protocols that ensure the comparability of results while maintaining the necessary flexibility to adapt to different contexts (Konrad et al., 2022). The definition of PNV, and thus of reference ecosystems, is entrusted to heterogeneous methodologies that vary across scales, study areas and schools of thought (Fischer et al., 2019; Nelson et al., 2024). These works, while in many cases representing milestones in the research on potential vegetation, collectively constitute a disjointed framework in which the thematic and geometric resolution of the produced maps is not uniform, and the result is often influenced by the subjectivity of expert judgment and by the different environmental characteristics of investigated areas (Bohn et al., 2004; Vuerich et al., 2001; Blasi et al., 2005). Moreover, in urban contexts, a second level of complexity is superimposed to this environmental heterogeneity, which relates to varying degrees of artificialisation and different configuration and composition of the landscape mosaic. Although the effects of such characteristics on species have been widely documented in the literature, studies that have assessed and quantified their impact on overall plant communities and ecosystems are less common (Malkinson et al., 2018; Celesti et al., 2006; Andrade et al., 2021). In particular, as far as our knowledge, the specific effect of anthropogenic landforms on PNV was poorly investigated, so that it represents one of the main novel contributions of the present research. Intending to fill the aforementioned gaps, this research project has thus contributed to advancing the state of the art in the study of PNV in urban environments.

With the first presented paper, a protocol for classifying PNV in the Functional Urban Area (FUA) of Campobasso was developed and implemented that updates the proposal drawn for the Italian landscapes and its subsequent reviews (Blasi et al., 2000; Blasi et al., 2005; Smiraglia et al 2013) . The FUA of Campobasso is characterised by a substantial persistence of forest cover, which offers interesting opportunities for nature conservation in an urban system, and by an extended agricultural cover and a compact (historical) urban centre, which leave space for restoration actions aimed at improving ecosystem services and increasing ecological connectivity (Varricchione et al., 2024;

Valeri et al., 2021). The study area is therefore representative of many medium-sized urban areas, which can benefit from ecological knowledge for environmental management and restoration. For these purposes, an environmental classification was carried out, hierarchically dividing the territory into land subsystems and identifying mature vegetation types (forests) related to them (Blasi et al., 2025). Vegetation surveys and analyses allowed the main forest types of the area to be identified, which were mainly ascribable to the mesophilous forests with *Quercus cerris* and *Quercus frainetto,* to the thermophilous forests with *Quercus pubescens* and to the mese-hygrophilous and higrophilous forests with *Populs nigra, Ulmus minor, Populs alba* and *Salix alba*, as also recognised by previous research (Paura & Blasi, 1995; Pirone et al., 2004; Taffetani et al., 2012). These communities, representing present-day PNV models (Somodi et al., 2021), were automatically spatialised with respect to environmental variables alternatively using a Random Forest Classification Tree (RFCT) and a Multinomial Logistic Regression (MLR). Other studies adopted a automatic methods to spatialise PNV, but predominantly focused on large-scale analyses, utilising existing vegetation databases, remote-sensing data, and machine learning techniques with large-grain environmental layers (Hengl et al., 2018; Konrad et al., 2022; Bourdouxhe et al., 2023). Even though the present work used a smaller internal dataset to model high-resolution PNV maps at the local level, the good performance of the two statistical models implemented indicates that the geographical and thematic stratification of the surveys has been able to capture a significant portion of the territory's ecological information. In addition to the sampling design and consequent PNV map modelling, the research was innovative in its selection of environmental variables, highlighting the flexibility of the ecological classification by evaluating the most proper environmental features to be considered case-by-case.

Especially, a series of morphometric variables was used to distinguish ecological boundaries among riparian forests, meso-hygrophilous lowland forests, and slope forests. Topographic wetness index, vertical distance from channel network, horizontal distance to channel network and slope-area domains demonstrated their value as useful predictors for PNV types. In particular, the application of the slope-area domains, which is the functional relationship between drainage area and slope, represents an innovative approach for the classification of geomorphology underpinning a vegetation study. It allows for an objective delineation of geomorphic domains (Delchiaro et al., 2023, 2025) and, in the present study, was explored as a proxy for vegetation distribution, based on the hypothesis that water availability and erosion regimes significantly influence vegetation composition. This method is potentially more effective than traditional terrain descriptors (such as slope or elevation alone), as it integrates dynamic flow accumulation and dissipation processes at the basin scale. Finally, a ELU/PNV map was synthesised by combining the required local scale accuracy in PNV spatialisation, mainly provided by MLR output, with the more easily employable geographical scope of the RFCT output. The final map gathered precise ecological information and may be helpful in addressing the growing need for practical restoration actions, serving as a valuable tool for ecosystem monitoring, management, and restoration. The application of ecological classification, in combination with robust statistical methods, at the local level produced a robust map of PNV gathering environmental information. It represents the ecological reference model and the starting point for many restoration interventions and reforestation programs, including local-scale restoration of industrial sites (Török et al., 2018; Fischer et al., 2019), but also nationwide programmes for urban and periurban reforestation in metropolitan cities (MASE, 2021).

However, in complex and profoundly altered ecosystems such as urban ones, reference models need to be calibrated based on the degree of artificialisation. Consequently, the second study focused on quantifying the impacts that urbanisation exerts on PNV, with specific attention to the effect of anthropogenic landforms, which in Rome represent a widespread alteration factor. The urban landscape of Rome has been shaped in centuries of human activities, which also changed natural morphology and, consequently, ecosystems functioning and dynamics. Thus, a thorough understanding of such divergences from local PNV models could be useful for planning the effective restoration of self-sustaining ecosystems in the area, as well as complying with current strategic and policy requirements (EC, 1991/2024), which also apply to other cities. Starting from pre-existing ELU/PNV maps, a stratified sampling design was employed to investigate three types of woodland across a disturbance gradient within two selected ELU. Vegetation characteristics, in terms of indicator species, chorological types, life forms, non-native proportion and α- and β-diversity, revealed a clear pattern along the disturbance gradient of the three woodland types. High-naturalness woodlands (HNW) selected among the core areas of most ecologically connected nodes, were the best preserved forests, aligning well with PNV models. The indication value of the node importance metric for identifying well-conserved forest patches was therefore empirically demonstrated, and the recognised role of ecological connectivity in maintaining ecosystem integrity was also confirmed in the urban context (Pascual-Hortal & Saura, 2008). Low-naturalness woodlands (LNW) exhibited instead a more pronounced legacy of the previous traditional agricultural landscape, with higher rates of ruderal and light-demanding species (Prach et al., 2001), and lower α-diversity, probably due to local extinction being more pronounced in LNW than in HNW (Ramalho et al., 2014). The differences between LNW and HNW can thus be related to a landscape effect, even though some species typical of mature PNV stages still occur, suggesting the capacity of these semi-natural areas to recover spontaneously over time depending on environmental and historical factors (Motta et al., 2009; Cuddington, 2011). Anthropogenic landform woodlands (ALW) were found to be more impacted than LNW despite occurring in comparable landscape contexts. Indicator species typical of PNV models were never found in these communities, likely due to the combined effects of local extinction and long-term removal of the seed bank resulting from landscape configuration and anthropogenic alteration of the sites. Instead, these communities were characterised by nitrophilous, ruderal and invasive species. Both LNW and ALW showed higher rates of therophytes and lower levels of geophytes with respect to HNW, as also reported by previous research studying urban floras (Preston et al., 2000; Williams et al., 2005; Knapp et al., 2008), despite the shift being even more pronounced in ALW, suggesting an additional effect of geomorphological disturbance. PNV centroids, obtained by averaging the Principal Components scores of HNW, were used to measure the ecological distance of the investigated woodlands from reference mature stages. Ecological distances confirmed previous results, with higher alteration observed in ALW than in LNW types. Compositional shifts of LNW, mainly ascribable to landscape-level fragmentation and historical land use, are therefore discernibly added to those due to site-level geomorphological alteration in ALW. The ecological distance from PNV was modelled with respect to urbanisation gradient (percentage of artificial cover) and legacy (age of vegetation cover), and spatialised to the Municipality of Rome. The predicted distance from PNV was therefore modelled across the city, independently of actual land cover at each site, obtaining a map of expected shifts from natural baselines. The map provides a complementary approach to the analysis and representation of the urban-rural gradient and its determinism on ecosystem extent and condition (Medley et al., 1995; Kaminski et al., 2021). In terms of current challenges for ecosystem restoration in cities, practical implications include a better understanding of deviation from natural reference models (Blasi et al., 2025), providing a more in-depth indication of where local vegetation is expected to recover spontaneously and where it is expected to be permanently replaced. This knowledge could effectively support the fine-scale prioritisation of sites for conservation or restoration, as well as the actions to be taken. These could range from selecting the right species for urban forestry and tree planting, to making a more balanced choice between native species typical of natural forests, and non-native species that embody important cultural values for historical cities (e.g. Pinus pinea and Platanus x hispanica in Rome; Caneva et al., 2020).

The final step (Paper III) of the research concerned the application of acquired knowledge to the design of an environmental restoration intervention in a peri-urban area of Rome, subject to multiple sources of disturbance. If the first and second studies indicated what to restore based on the reference model and potential alteration factors, the third study defined how and where to act to address a specific aim, specifically, biological invasions, one of the main threats to biodiversity (Pyšek et al., 2020). The landscape analysis indicated that the landscape structure, characterised by fragmentation and a high edge density, creates ideal conditions for the spread of exotic woody species, often associated with artificial covers (Boscutti et al., 2022). The progressive agricultural abandonment of less easily cultivable lands has favoured vegetation dynamics, and today the territory is characterised by many shrublands and evolving natural areas that have emerged as particularly prone to invasions when the contact with artificial surfaces was particularly extensive, confirming and deepening what has been observed in previous research (Fenesi et al., 2015). The abandonment trend is common across many agricultural areas, therefore, the problem addressed in this paper can be extended elsewhere in Italy and Europe (Quintas-Soriano et al., 2022). A number of invasive plants have been sampled in the investigated peri-urban sector, which can threaten residual natural and semi-natural ecosystems, their respective biodiversity, and their capacity to provide services deserving targeted GI actions to be managed and contained (Potgieter et al., 2022). The most threatening species detected were *Ailanthus altissima, Robinia pseudoacacia, Acer negundo, Acacia dealbata, Eucalyptus camaldulensis*, and the archeophyte *Arundo donax.* These species are capable of rapidly colonising abandoned fields, forming almost pure stands that limit species diversity and hinder native species (Sladonja et al., 2015; Gentili etal., 2019; Sikorska et al., 2019; Lazzaro et al., 2014; Badalamenti et al., 2018). It was confirmed that exotic plant richness (EPR) in semi-natural ecosystems depends on the surrounding landscape pattern (Làzaro-Lobo and Ervin, 2021) and is associated with artificial surfaces (Gonzàlez-Moreno et al., 2013), particularly with direct contiguity to the urban fabric. Instead, agricultural areas representing the landscape matrix, with residual traditional features (Biasi et al., 2015), acted as a buffer against EPR in shrublands, validating the hypothesis that the intensity, rather than just the occurrence, of agricultural practices affects the degree of plant invasion (Pellegrini et al., 2021). Based on this evidence, a Green Infrastructure (GI) was designed to curb biological invasions across the study area, following a proactive, robust, science-based approach. The GI has been based on information from the Generalised Linear Model (GLM), which estimated the EPR across brushlands and evolving green areas using a set of landscape variables. Following a prioritisation process based on the susceptibility to invasion of the different natural areas, inferred through the inferential model, the GI indicates components to be requalified and those to be conserved, thereby optimising resources and increasing the probability of success of the interventions.

In keeping with these results, the landscape-level actions that emerged as essential priorities for the containment of plant invasion in such a peri-urban context concomitantly included: i) preservation of natural and semi-natural ecosystems from diffuse direct contacts with artificial surfaces, that is, the prevention and mitigation of urban sprawl over abandoned agricultural lands (Frondoni et al., 2011; Smiraglia et al., 2021); ii) maintenance and promotion of traditional agricultural practices (Zavattero et al., 2021), able to provide buffer zones for the preservation of biogeographically coherent species in present-day semi-natural remnant ecosystems and along their progressive dynamics; iii) conservation of less vulnerable components, through proactive involvement of landowners and promotion of sustainable forestry (Radtke et al., 2013; Sitzia et al., 2016; Brundu et al., 2020) and minimal interventions, mainly devoted at preventing disturbances other than biological invasion (e.g. fires and soil degradation) and eventually allowing the spontaneous recovery of mature forest ecosystems (i.e. "passive restoration"; Chazdon et al., 2021); and iv) active restoration of highly susceptible components, through either alternative or combined removal of invasive exotic species, with a special reference to available guidelines for effective eradication (EPPO, 2019), and reforestation, with a special reference to biogeographic and ecological coherence of planted trees, thus belonging to PNV (Capotorti et al., 2016; EC, 2023). Regarding the costs, the present work did not address a cost-benefit analysis, despite the fact that the eradication of invasive alien plant species (IAPS) would require impressive financial support, considering also monitoring actions in post-intervention stands (Stafford et al., 2017; Lazzaro et al., 2023). Moreover, these interventions may be unsuccessful due to the persistent nature of IAPS and considering the continuous propagule pressure due to contact with adjacent artificial areas. It is therefore important to stress once again that the eradication of IAPS from these priority areas should always follow a logic whereby packages of actions are implemented that include both the removal and/or management of prioritised IAPS (Lazzaro et al., 2019), and restoration through the planting of native species that are ecologically suited to the environment, favouring the future control of invasive species thanks to competition mechanisms (Kattenrig et al., 2011; Sitzia et al., 2016; Fernandes et al., 2018).This multifunctional approach transforms the GI into an active land management tool capable of simultaneously providing ecosystem services, increasing connectivity for native species, and addressing a specific ecological threat, offering a concrete model for implementing current European environmental policies (EC, 2020).

Conclusions

The doctoral thesis has developed and validated an integrated methodological framework able to connect fundamental ecological analysis with applied spatial planning, providing a logical pathway to support nature restoration in urban ecosystems. The work has demonstrated how, starting from a solid scientific basis, it is possible to address complex management challenges of human-dominated landscapes.

The first critical contribution of the research was a robust and repeatable methodology for identifying and mapping PNV using a semi-automatic approach. The protocol, which leverages stratified field observations to optimise an Ecological Land Classification (ELC) down to the land unit level, provides the essential foundation for any subsequent analysis or intervention. Given its flexibility, the process can be adapted to various contexts beyond a medium-sized Functional Urban Area, with site-specific adjustments, especially in highly impacted urban systems. The resulting PNV maps, along with connected tools such as local species databases, represent fundamental information for carrying out restoration actions and implementing Nature-Based Solutions that are coherent with local ecological potential and biogeographic settings. This aligns with recent applications, from local-scale restoration of industrial sites to nationwide programmes for urban and peri-urban reforestation. The utility and potential of these products underscore the need to incorporate high-resolution maps and precise ecological information to address the growing need for practical restoration actions, as mandated by recent environmental policies (EC, 1991/2024). Building on this ecological baseline, the research examined the implications of anthropogenic drivers for vegetation dynamics. The second part of the thesis assessed how PNV is significantly affected by anthropogenic landforms, which alter successional pathways and can divert them from the mature stage of the local PNV towards a replacement community. The investigation revealed a distinct naturalness gradient reflected in the species composition across different woodland types. To quantify the phenomenon, a synthetic index of deviation from PNV was developed and modelled against a set of explanatory variables using a GAM. The spatialisation of the best-performing model produced a map of ecological distance, a critical tool for strategic planning. Based on these distance values, it was possible to identify areas suitable for spontaneous or passive ecological restoration, as well as those where active, targeted measures are necessary. These results have direct implications for management within urban environments, where disturbed sites are widespread. Areas with high estimated ecological distances suggest the presence of novel ecosystems that may no longer support PNV development. This information is crucial for prioritising conservation resources, as it helps to identify where restoring PNV forests is feasible and where it is more suitable to invest in designing effective Green Infrastructure or introducing well-adapted species tailored to the specific urban context. Finally, the framework's practical application was demonstrated by exploring the mechanisms of biological invasions in peri-urban landscapes. By integrating landscape metrics with floristic sampling, the research shed light on which landscape features are most involved in the successful spread of exotic woody plants. Direct contact with artificial surfaces, driven by urban sprawl, was identified as the major driver of plant invasions, while the traditional agricultural matrix was shown to provide a buffering effect. In what is believed to be an innovative application, the inferred susceptibility to invasion, derived from a predictive GLM, was used to inform a local-scale Green Infrastructure planning process. Notwithstanding margins for improvement, the proposed GI could support local authorities in achieving key targets for biodiversity conservation and restoration, testing win-win solutions aimed at the sustainable development of metropolitan peripheral sectors.

In conclusion, this work provides a comprehensive scientific and operational toolkit for planners and land managers who must translate the ambitious goals of environmental policies into practice. The path outlined here, from defining ecological potential to understanding disturbance factors to designing targeted solutions, offers a replicable model for promoting biodiversity and resilience, thereby contributing to the crucial challenge of making our cities more sustainable socio-ecological systems.

References

1. Andrade, R., Franklin, J., Larson, K. L., Swan, C. M., Lerman, S. B., Bateman, H. L., Warren, P. S., & York, A., 2021. Predicting the assembly of novel communities in urban ecosystems. Landscape Ecol 36, 1–15. https://doi.org/10.1007/s10980-020-01142-1.
2. Babi Almenar, J., Marando, F., Vallecillo, S., Zulian, G., Cortinovis, C., Zurbaran-Nucci, M., Chrysoulakis, N., Parastatidis, D., Heris, M. and Grammatikopoulou, I., Urban Ecosystem accounts following the SEEA EA standard: A pilot application in Europe, Publications Office of the European Union, Luxembourg, 2023, doi:10.2760/741116, JRC133240.
3. Badalamenti, E., Cusimano, D., La Mantia, T., Pasta, S., Romano, S., Troia, A., & Ilardi, V., 2018. The ongoing naturalisation of Eucalyptus spp. in the Mediterranean Basin: new threats to native species and habitats. Australian Forestry, 81(4), 239-249. https://doi.org/10.1080/00049158.2018.1533512.
4. Bailey, R. G., 2004. Identifying ecoregion boundaries. Environmental management, 34(Suppl 1), S14-S26. https://doi.org/10.1007/s00267-003-0163-6.
5. Biasi, R., Colantoni, A., Ferrara, C., Ranalli, F., & Salvati, L., 2015. In-between sprawl and fires: Long-term forest expansion and settlement dynamics at the wildland–urban interface in Rome, Italy. International Journal of Sustainable Development & World Ecology, 22(6), 467-475. https://doi.org/10.1080/13504509.2015.1064488.
6. Blasi, C., Capotorti, G., & Frondoni, R., 2005a: Defining and mapping typological models at the landscape scale, Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology: Official Journal of the Societa Botanica Italiana, 139:2, 155-163. https://doi.org/10.1080/11263500500163629.
7. Blasi, C., Capotorti, G., Del Vico, E., Bonacquisti, S., & Zavattero, L., 2025. Native Flora and Potential Natural Vegetation References for Effective Forest Restoration in Italian Urban Systems. Plants, 14(15), 2396. https://doi.org/10.3390/plants14152396.
8. Blasi, C., Carranza, M. L., Frondoni, R., & Rosati, L., 2000. Ecosystem classification and mapping: a proposal for Italian landscapes. Applied Vegetation Science, 3(2), 233-242. https://doi.org/10.2307/1479002.
9. Boscutti, F., Lami, F., Pellegrini, E., Buccheri, M., Busato, F., Martini, F., Sibella, R., Sigura, M., & Marini, L., 2022. Urban sprawl facilitates invasions of exotic plants across multiple spatial scales. Biological Invasions, 24(5), 1497-1510. https://doi.org/10.1007/s10530-022-02733-6.
10. Boscutti, F., Sigura, M., De Simone, S., Marini, L., 2018. Exotic plant invasion in agricultural landscapes: a matter of dispersal mode and disturbance intensity. Appl. Veg. Sci. 21, 250–257. https://doi.org/10.1111/avsc.12351.
11. Bourdouxhe, A., Wibail, L., Claessens, H., & Dufrêne, M., 2023. Modeling potential natural vegetation: A new light on an old concept to guide nature conservation in fragmented and degraded landscapes. Ecological Modelling, 481, 110382. https://doi.org/10.1016/j.ecolmodel.2023.110382.
12. Brundu, G., Pauchard, A., Pyšek, P., Pergl, J., Bindewald, A. M., Brunori, A., Canavan, S., Campagnaro, T., Celesti-Grapow, L., Dechoum, M. S. D., Dufour-Dror, J. M., Essl, F., Flory, S. L., Genovesi, P., Guarino, F., Guangzhe, L., Hulme, P. E., Jäger, H., Kettle,

C. J., Krumm, F., Langdon, B., Lapin, K., Lozano, V., Le Roux, J. J., Novoa, A., Nuñez, M. A., Porté, A. J., Silva, J. S., Schaffner, U., Sitzia, T., Tanner, R., Tshidada, N., Vítková, M., Westergren, M., Wilson, J. R. U., & Richardson, D. M. (2020). Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. NeoBiota 61: 65–116. 10.3897/neobiota.65.58380 http://neobiota.pensoft.net.
13. Caneva, G., Bartoli, F., Zappitelli, I., & Savo, V., 2020. Street trees in italian cities: story, biodiversity and integration within the urban environment. Rendiconti Lincei. Scienze Fisiche e Naturali, 31(2), 411-417. https://doi.org/10.1007/s12210-020-00907-9.
14. Capotorti, G., De Lazzari, V., & Alós Ortí, M., 2019. Local scale prioritisation of green infrastructure for enhancing biodiversity in Peri-Urban agroecosystems: A multi-step process applied in the Metropolitan City of Rome (Italy). Sustainability, 11(12), 3322. https://doi.org/10.3390/su11123322. (a)
15. Capotorti, G., Del Vico, E., Anzellotti, I., Celesti-Grapow, L., 2016. Combining the conservation of biodiversity with the provision of ecosystem services in urban green infrastructure planning: Critical features arising from a case study in the metropolitan area of Rome. Sustainability, 9(1), 10. https://doi.org/10.3390/su9010010.
16. Capotorti, G., Del Vico, E., Lattanzi, E., Tilia, A., & Celesti-Grapow, L., 2013. Exploring biodiversity in a metropolitan area in the Mediterranean region: The urban and suburban flora of Rome (Italy). Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 147(1), 174–185. https://doi.org/10.1080/11263504.2013.771715.
17. Capotorti, G., Guida, D., Siervo, V., Smiraglia, D., & Blasi, C., 2012. Ecological classification of land and conservation of biodiversity at the national level: The case of Italy. Biological Conservation, 147(1), 174-183. https://doi.org/10.1016/j.biocon.2011.12.028.
18. Celesti-Grapow, L., Pyšek, P., Jarošík, V., & Blasi, C., 2006. Determinants of native and alien species richness in the urban flora of Rome. Diversity and Distributions, 12(5), 490-501. https://doi.org/10.1111/j.1366-9516.2006.00282.x.
19. Chazdon, R. L., Falk, D. A., Banin, L. F., Wagner, M., J Wilson, S., Grabowski, R. C., Suding, K. N., 2021. The intervention continuum in restoration ecology: rethinking the active–passive dichotomy. Restoration Ecology, e13535. https://doi.org/10.1111/rec.13535.
20. Convention on Biological Diversity., 2022. Kunming-Montreal Global Biodiversity Framework. CBD/COP/15/L25. Goal 11 | Department of Economic and Social Affairs.
21. Cuddington, K., 2011. Legacy effects: the persistent impact of ecological interactions. Biological Theory, 6(3), 203-210. https://doi.org/10.1007/s13752-012-0027-5.
22. Delchiaro, M., Della Seta, M., Martino, S., Nozaem, R., & Moumeni, M., 2023. Tectonic deformation and landscape evolution inducing mass rock creep driven landslides: the Loumar case-study (Zagros Fold and Thrust Belt, Iran). Tectonophysics, 846, 229655. https://doi.org/10.1016/j.tecto.2022.229655.
23. Delchiaro, M., Vergari, F., Esposito, C., & Del Monte, M., 2025. The influence of anthropogenic topographic changes on geomorphological processes in the city of Rome (Italy): A case study of the Malagrotta area. Earth Surface Processes and Landforms, 50(3), e70033. https://doi.org/10.1002/esp.70033.

24. Dickson, B., Blaney, R. Miles, L., Regan, E., van Soesbergen, A.,Väänänen, E., Blyth, S., Harfoot, M. Martin, C.S., McOwen, C., Newbold, T., van Bochove, J., 2014. Towards a global map of natural capital: key ecosystem assets. UNEP, Nairobi, Kenya.
25. EC (European Commission), 2020. https://environment.ec.europa.eu/topics/nature-and-biodiversity_mt
26. EC (European Commission), 2023. Guidelines on Biodiversity-Friendly Afforestation, Reforestation and Tree Planting. Guidelines on Biodiversity-Friendly Afforestation, Reforestation and Tree Planting https://environment.ec.europa.eu/publications/guidelines-biodiversity-friendly-afforestation-reforestation-and-tree-planting_en.
27. EPPO (European and Mediterranean Plant Protection Organization), 2019. PM 9/29 (1) Ailanthus altissima. EPPO Bulletin, 50, 148-155. https://doi.org/10.1111/epp.12621.
28. EUROSTAT. 2018. Methodological manual on territorial typologies. 2018 Edition. General and Regional Statistics, and guidelines, 132. https://ec.europa.eu/eurostat/web/products-manuals-and- guidelines/-/ks-gq-18-008.
29. Fenesi, A., Vágási, C. I., Beldean, M., Földesi, R., Kolcsár, L. P., Shapiro, J. T., Török, E., Kovács-Hostyánszki, A., 2015. Solidago canadensis impacts on native plant and pollinator communities in different-aged old fields. Basic and Applied Ecology, 16(4), 335-346. https://doi.org/10.1016/j.baae.2015.03.003.
30. Fernandes, P., Máguas, C., Correia, O., González-Moreno, P., 2018. What drives Eucalyptus globulus natural establishment outside plantations? The relative importance of climate, plantation and site characteristics. Biological Invasions, 20(5), 1129-1146. https://doi.org/10.1007/s10530-017-1614-y.
31. Fischer, S., Michler, B., & Fischer, A., 2019. High resolution predictive modelling of potential natural vegetation under recent site conditions and future climate scenarios: Case study Bavaria. Tuexenia, 39. 10.14471/2018.39.001.
32. Frondoni, R., Mollo, B., Capotorti., G., 2011. A landscape analysis of land cover change in the Municipality of Rome (Italy): Spatio-temporal characteristics and ecological implications of land cover transitions from 1954 to 2001. Landscape and Urban planning, 100(1-2), 117-128. https://doi.org/10.1016/j.landurbplan.2010.12.002.
33. Gallizia Vuerich, L., Poldini, L., & Feoli, E., 2001. Model for the potential natural vegetation mapping of Friuli Venezia-Giulia (NE Italy) and its application for a biogeographic classification of the region.
34. Gentili, R., Ferrè, C., Cardarelli, E., Montagnani, C., Bogliani, G., Citterio, S., Comolli, R., 2019. Comparing negative impacts of Prunus serotina, Quercus rubra and Robinia pseudoacacia on native forest ecosystems. Forests, 10(10), 842. https://doi.org/10.3390/f10100842.
35. González-Moreno, P., Pino, J., Carreras, D., Basnou, C., Fernández-Rebollar, I., Vila, M., 2013. Quantifying the landscape influence on plant invasions in Mediterranean coastal habitats. Landscape ecology, 28, 891-903. https://doi.org/10.1007/s10980-013-9857-1
36. Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M., 2008. Global change and the ecology of cities. science, 319(5864), 756-760. 10.1126/science.115019.

37. Hengl, T., Walsh, MG., Sanderman, J., Wheeler, I., Harrison, SP., Prentice, IC., 2018. Global mapping of potential natural vegetation: an assessment of machine learning algorithms for estimating land potential. PeerJ 6:e5457 https://doi.org/10.7717/peerj.5457.
38. Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R., & Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global ecology and biogeography, 15(1), 1-7. https://doi.org/10.1111%2Fj.1466-822X.2006.00212.x.
39. Hulme, P. E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., ... & Vilà, M., 2008. Grasping at the routes of biological invasions: a framework for integrating pathways into policy. Journal of Applied Ecology, 45(2), 403-414.
40. Kaminski, A., Bauer, D. M., Bell, K. P., Loftin, C. S., & Nelson, E. J., 2021. Using landscape metrics to characterize towns along an urban-rural gradient. Landscape Ecology, 36(10), 2937-2956. https://doi.org/10.1007/s10980-021-01287-7.
41. Kettenring, K. M., & Adams, C. R., 2011. Lessons learned from invasive plant control experiments: a systematic review and meta-analysis. *Journal of applied ecology*, *48*(4), 970-979. https://doi.org/10.1111/j.1365-2664.2011.01979.x
42. Knapp, S., Kuehn, I., Wittig, R., Ozinga, W. A., Poschlod, P., & Klotz, S., 2008. Urbanization causes shifts of species' trait state frequencies. https://hdl.handle.net/2066/72378.
43. Konrád, K. D., Bede-Fazekas, Á., Molnár, Z., & Somodi, I., 2022. Multilayer landscape classification based on potential vegetation. 10.23855/preslia.2022.631.
44. Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. Environmental pollution, 159(8-9), 1974-1983. https://doi.org/10.1016/j.envpol.2011.02.022.
45. Lázaro-Lobo, A., Ervin, G. N., 2021. Native and exotic plant species respond differently to ecosystem characteristics at both local and landscape scales. Biological Invasions, 23, 143-156. https://doi.org/10.1007/s10530-020-02361-y.
46. Lazzaro, L., Bolpagni, R., Barni, E., Brundu, G., Blasi, C., Siniscalco, C., & Celesti-Grapow, L., 2019. Towards alien plant prioritization in Italy: methodological issues and first results. *Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology*, *153*(5), 740-746. https://doi.org/10.1080/11263504.2019.1640310
47. Lazzaro, L., Giuliani, C., Fabiani, A., Agnelli, A. E., Pastorelli, R., Lagomarsino, A., Benesperi, R., Calamassi, R., Foggi, B., 2014. Soil and plant changing after invasion: the case of Acacia dealbata in a Mediterranean ecosystem. Science of the total environment, 497, 491-498. https://doi.org/10.1016/j.scitotenv.2014.08.014.
48. Lazzaro, L., Mugnai, M., Ferretti, G., Giannini, F., Giunti, M., & Benesperi, R., 2023. (Not) sweeping invasive alien plants under the carpet: results from the use of mulching sheets for the control of invasive Carpobrotus spp. *Biological Invasions*, *25*(8), 2583-2597. https://doi.org/10.1007/s10530-023-03059-7.
49. Maes, J., Teller, A., Erhard, M., Conde, S., Vallecillo Rodriguez, S., Barredo Cano, J. I., Paracchini, M.-L., Abdul Malak, D., Trombetti, M., Vigiak, O., Zulian, G., Addamo, A.,

Grizzetti, B., Somma, F., Hagyo, A., Vogt, P., Polce, C., Jones, A., Marin, A., Ivits, E., Mauri, A., Rega, C., Czucz, B., Ceccherini, G., Pisoni, E., Ceglar, A., De Palma, P., Cerrani, I., Meroni, M., Caudullo, G., Lugato, E., Vogt, J., Spinoni, J., Cammalleri, C., Bastrup-Birk, A., San-Miguel-Ayanz, J., San Román, S., Kristensen, P., Christiansen, T., Zal, N., De Roo, A., De Jesus Cardoso, A., Pistocchi, A., Del Barrio Alvarellos, I., Tsiamis, K., Gervasini, E., Deriu, I., La Notte, A., Abad Viñas, R., Vizzarri, M., Camia, A., Robert, N., Kakoulaki, G., Garcia Bendito, E., Panagos, P., Ballabio, C., Scarpa, S., Montanarella, L., Orgiazzi, A., Fernandez Ugalde, O., & Santos-Martín, F., 2020. Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment, Publications Office of the European Union, Luxembourg. JRC120383. https://data.europa.eu/doi/10.2760/757183.

50. Maes, J., Zulian, G., Thijssen, M., Castell, C., Baró, F., Ferreira, A. M., ... & Teller, A., 2016. Mapping and assessment of ecosystems and their services. Urban ecosystems. In Mapping and assessment of ecosystems and their services. Urban ecosystems. (pp. 1-100). Publications office of the European Union.
51. Malkinson, D., Kopel, D., & Wittenberg, L., 2018. From rural-urban gradients to patch–matrix frameworks: Plant diversity patterns in urban landscapes. Landscape and Urban Planning, 169, 260-268. http://dx.doi.org/10.1016/j.landurbplan.2017.09.021.
52. MASE (Italian Ministry of Environment and Energy Security), 2021. Piano di riforestazione urbana ed extraurbana, PNRR Misura 2 – Componente 4. PNRR_INVESTIMENTO_3.1_Avviso_pubblico.pdf https://www.mase.gov.it/sites/default/files/archivio/bandi/PNRR_INVESTIMENTO_3.1_Avviso_pubblico.pdf
53. Medley, K. E., McDonnell, M. J., & Pickett, S. T., 1995. Forest-landscape structure along an urban-to-rural gradient. The Professional Geographer, 47(2), 159-168. https://doi.org/10.1111/j.0033-0124.1995.159_c.x.
54. Motta, R., Nola, P., & Berretti, R., 2009. The rise and fall of the black locust (Robinia pseudoacacia L.) in the "Siro Negri" Forest Reserve (Lombardy, Italy): lessons learned and future uncertainties. Annals of Forest Science, 66(4), 1-10. http://dx.doi.org/10.1051/forest/2009012.
55. Nelson, C.R., Hallett, J.G., Romero Montoya, A.E., Andrade, A., Besacier, C., Boerger, V., Bouazza, K., Chazdon, R., Cohen-Shacham, E., Danano, D., Diederichsen, A., Fernandez, Y., Gann, G.D., Gonzales, E.K., Gruca, M., Guariguata, M.R., Gutierrez, V., Hancock, B., Innecken, P., Katz, S.M., McCormick, R., Moraes, L.F.D., Murcia, C., Nagabhatla, N., Pouaty Nzembialela, D., Rosado-May, F.J., Shaw, K., Swiderska, K., Vasseur, L., Venkataraman, R., Walder, B., Wang, Z., & Weidlich, E.W.A. 2024. Standards of practice to guide ecosystem restoration – A contribution to the United Nations Decade on Ecosystem Restoration 2021-2030. Rome, FAO, Washington, DC, SER & Gland, Switzerland, IUCN CEM. https://doi.org/10.4060/cc9106e.
56. Palmer, M.A., Zedler, J.B., Falk, D.A., 2016. Ecological Theory and Restoration Ecology. In: Palmer, M.A., Zedler, J.B., Falk, D.A. (Eds) Foundations of Restoration Ecology. Island Press, Washington, DC. https://doi.org/10.5822/978-1-61091-698-1_1.

57. Paura B., & Blasi C., 1995. Su alcune stazioni a Quercus frainetto Ten. in Campania e in Molise; analisi fitosociologica e fitogeografica. Ann. Bot. (Roma), Studi sul territorio, 51 (10) (1993): 353-366. https://hdl.handle.net/11695/7346.
58. Pellegrini, E., Buccheri, M., Martini, F., Boscutti, F., 2021. Agricultural land use curbs exotic invasion but sustains native plant diversity at intermediate levels. Scientific reports, 11(1), 8385. https://doi.org/10.1038/s41598-021-87806-7.
59. Pirone, G., Ciaschetti, G., & Frattaroli, A. R., 2004. Appunti sulla vegetazione della Valle del Trigno (Abruzzo meridionale). Inform. Bot. Ital, 36(1), 13-27.
60. Potgieter, L. J., Shrestha, N., Cadotte, M. W., 2022. Prioritizing sites for terrestrial invasive alien plant management in urban ecosystems. Ecological Solutions and Evidence, 3(3), e12160. https://doi.org/10.1002/2688-8319.12160.
61. Prach, K., Pyšek, P., & Bastl, M., 2001. Spontaneous vegetation succession in human-disturbed habitats: A pattern across seres. Applied Vegetation Science, 4(1), 83-88. https://doi.org/10.1111/j.1654-109X.2001.tb00237.x.
62. Preston, C.D., 2000. Engulfed by suburbia or destroyed by the plough: The ecology of extinction in Middlesex and Cambridgeshire. Watsonia 23, 59–81.
63. Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., ... & Richardson, D. M., 2020. Scientists' warning on invasive alien species. Biological Reviews, 95(6), 1511-1534.
64. Quintas-Soriano, C., Buerkert, A., & Plieninger, T., 2022. Effects of land abandonment on nature contributions to people and good quality of life components in the Mediterranean region: A review. Land use policy, 116, 106053.https://doi.org/10.1016/j.landusepol.2022.106053.
65. Radtke, A., Ambraß, S., Zerbe, S., Tonon, G., Fontana, V., Ammer, C., 2013. Traditional coppice forest management drives the invasion of Ailanthus altissima and Robinia pseudoacacia into deciduous forests. Forest Ecology and Management, 291, 308-317. https://doi.org/10.1016/j.foreco.2012.11.022.
66. Ramalho, C. E., Laliberté, E., Poot, P., & Hobbs, R. J., 2014. Complex effects of fragmentation on remnant woodland plant communities of a rapidly urbanizing biodiversity hotspot. Ecology, 95(9), 2466-2478. https://doi.org/10.1890/13-1239.1.
67. Rivas-Martinez, S., 2005. Notions on dynamic-catenal phytosociology as a basis of landscape science, Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology, 139:2, 135-144, DOI: 10.1080/11263500500193790.
68. Rosenzweig, M. L., 2003. Reconciliation ecology and the future of species diversity. Oryx, 37(2), 194-205. doi:10.1017/S0030605303000371.
69. Sanderson, E. W., Jaiteh, M., Levy, M. A., Redford, K. H., Wannebo, A. V., & Woolmer, G., 2002. The human footprint and the last of the wild: the human footprint is a global map of human influence on the land surface, which suggests that human beings are stewards of nature, whether we like it or not. BioScience, 52(10), 891-904.https://doi.org/10.1641/0006-3568(2002)052[0891:THFATL]2.0.CO;2.
70. Saura, S., & Pascual-Hortal, L., 2007. A new habitat availability index to integrate connectivity in landscape conservation planning: comparison with existing indices and application to a case study. Landscape and urban planning, 83(2-3), 91-103. https://doi.org/10.1016/j.landurbplan.2007.03.005.

71. Sikorska, D., Sikorski, P., Archiciński, P., Chormański, J., Hopkins, R. J., 2019. You can't see the woods for the trees: invasive Acer negundo L. in urban riparian forests harms biodiversity and limits recreation activity. Sustainability, 11(20), 5838. https://doi.org/10.3390/su11205838.
72. Sitzia, T., Campagnaro, T., Kowarik, I., & Trentanovi, G., 2016. Using forest management to control invasive alien species: helping implement the new European regulation on invasive alien species. *Biological invasions*, *18*(1), 1-7. https://doi.org/10.1007/s10530-015-0999-8.
73. Sladonja, B., Sušek, M., Guillermic, J., 2015. Review on invasive tree of heaven (Ailanthus altissima (Mill.) Swingle) conflicting values: assessment of its ecosystem services and potential biological threat. Environmental management, 56, 1009-1034. https://doi.org/10.1007/s00267-015-0546-5.
74. Smiraglia, D., Capotorti, G., Guida, D., Mollo, B., Siervo, V., C Blasi, C., 2013. Land units map of Italy. Journal of Maps, 9(2), 239-244. https://doi.org/10.1080/17445647.2013.771290.
75. Smiraglia, D., Salvati, L., Egidi, G., Salvia, R., Giménez-Morera, A., Halbac-Cotoara-Zamfir, R., 2021. Toward a new urban cycle? A closer look to sprawl, demographic transitions and the environment in Europe. Land, 10(2), 127. https://doi.org/10.3390/land10020127.
76. Somodi, I., Ewald, J., Bede-Fazekas, Á., & Molnár, Z., 2021. The relevance of the concept of potential natural vegetation in the Anthropocene. Plant Ecology & Diversity, 14(1-2), 13-22.https://doi.org/10.1080/17550874.2021.1984600.
77. Stafford, W. H., Von Maltitz, G. P., & Watson, H. K., 2018. Reducing the costs of landscape restoration by using invasive alien plant biomass for bioenergy. *Wiley Interdisciplinary Reviews: Energy and Environment*, *7*(1), e272. https://doi.org/10.1002%2Fwene.272
78. Suding, K. N., 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics, 42, 465-487.
79. Taffetani, F., Catorci, A., Ciaschetti, G., Cutini, M., Di Martino, L., Frattaroli, A. R., Paura B., Pirone G., Rismondo M. & Zitti, S., 2012. The Quercus cerris woods of the alliance Carpinion orientalis Horvat 1958 in Italy. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology, 146(4), 918-953. https://doi.org/10.1080/11263504.2012.682613.
80. Theodorou, P., 2022. The effects of urbanisation on ecological interactions. Current Opinion in Insect Science, 52, 100922. https://doi.org/10.1016/j.cois.2022.100922.
81. Török, K., Csecserits, A., Somodi, I., Kövendi-Jakó, A., Halász, K., Rédei, T., & Halassy, M., 2018. Restoration prioritization for industrial area applying multiple potential natural vegetation modeling. Restoration Ecology, 26(3), 476-488. https://doi.org/10.1111/rec.12584.
82. Tuxen, R., 1956. Die heutige potentille natuerliche vegetation als Gegenst vegetation skartierung. Angew Pflanzensoziologie, 13, 5-42.
83. U. Bohn, G. Gollub, C. Hettwer, Z. Neuhäuslová, Th. Raus, H. Schlüter, H. Weber. [Bearb./Eds.] Karte der natürlichenVegetation Europas/Map of the Natural Vegetation of

Europe, Maßstab/Scale 1:2.500.000, Interaktive/Interactive CD-ROM- Erläuterungstext, Legende, Karten / Explanatory Text, Legend, Maps. Landwirtschaftsverlag, Münster, 2004.
84. United Nations, 2019. Resolution adopted by the General Assembly: United Nations Decade on Ecosystem Restoration (2021–2030). Document Viewer
85. United Nations., 2015. Transforming our world: The 2030 Agenda for Sustainable Development. Goal 11 | Department of Economic and Social Affairs.
86. Valeri, S., Zavattero, L., & Capotorti, G., 2021. Ecological Connectivity in Agricultural Green Infrastructure: Suggested Criteria for Fine Scale Assessment and Planning. Land, 10(8), 807. https://doi.org/10.3390/land10080807.
87. Varricchione, M., Laura Carranza, M., D'Angeli, C., Carla de Francesco, M., Innangi, M., Santoianni, L. A., & Stanisci, A., 2024. Exploring the distribution pattern of native and alien forests and their woody species diversity in a small Mediterranean city. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 158(6), 1335–1346. https://doi.org/10.1080/11263504.2024.2415613.
88. Williams, N. S., Morgan, J. W., McDonnell, M. J., & Mccarthy, M. A., 2005. Plant traits and local extinctions in natural grasslands along an urban–rural gradient. Journal of Ecology, 93(6), 1203-1213.https://doi.org/10.1111/j.1365-2745.2005.01039.x.
89. World Health Organization, 2023. One Health, last access 07/07/2025.
90. Zavattero, L., Frondoni, R., Capotorti, G., Copiz, R., Blasi, C., 2021. Towards the identification and mapping of traditional agricultural landscapes at the national scale: an inventory approach from Italy, Landscape Research, 46:7, 945-958. https://doi.org/10.1080/01426397.2021.1923678.
91. Zerbe, S., 1998. Potential natural vegetation: validity and applicability in landscape planning and nature conservation. Applied Vegetation Science, 1(2), 165-172. https://doi.org/10.2307/1478945.
92. Zulian G., Marando F., Vogt P., Barbero Vignola G., Babí Almenar J., Zurbarán-Nucci M., Prince' K. BiodiverCities: A roadmap to enhance the biodiversity and green infrastructure of European cities by 2030, Publications Office of the European Union, Luxembourg, 2022, Editors, Publisher, Publisher City, Year of Publication, doi:10.2760/21172, JRC129888.

Related works

Ecological classification of land based on Potential Natural Vegetation: a new quantitative & multivariate approach to map ecosystems. The case study of Campobasso Functional Urban Area / Montaldi, Alessandro; Del Vico, Eva; Stanisci, Angela; Carla De Francesco, Maria; Varicchione, Marco; Antonietta Santoianni, Lucia; Delchiaro, Michele; Pica, Alessia; Jona Lasinio, Giovanna; Paura, Bruno; Capotorti, Giulia. - (2025). (Intervento presentato al convegno 33rd European Vegetation Survey tenutosi a Perugia). Abstract in atti di convegno

Effects of anthropogenic landforms on Potential Natural Vegetation in a wide urban ecosystem (Capital city of Rome, Italy) / Montaldi, Alessandro; Pica, Alessia; Del Vico, Eva; Iamonico, Duilio; Valeri, Simone; Iacopino, Asja; Camilla, Frank; Capotorti, Giulia. - (2025). (Intervento presentato al convegno 2nd Conference of Conservation Biology for Early Career Researchers tenutosi a L'Aquila). Abstract in atti di convegno

On biodiversity friendly Green Infrastructure planning. Contribution of plant ecology to the monitoring and valorisation of peri-urban environments in the Metropolitan City of Rome (Mediterranean Italy) / Capotorti, Giulia; Zavattero, Laura; Bonacquisti, Sandro; Del Vico, Eva; Facioni, Laura; Iamonico, Duilio; Montaldi, Alessandro; Valeri, Simone. - (2022). (Intervento presentato al convegno 117. Congresso della Società Botanica Italiana. 8. International plant science conference (IPSC) tenutosi a Bologna; Italy). Abstract in atti di convegno

De Francesco, M. C., Carranza, M. L., Capotorti, G., Del Vico, E., D'Angeli, C., Montaldi, A., ... & Stanisci, A. (2025). DALIA: a relational DAtabase of tree, shrub and LIAna taxa recorded in the Functional Urban Area of Campobasso (Italy). Vegetation Ecology and Diversity, 62, e155222.

Conferences

Oral presentation at the 33rd European Vegetation Society (EVS) congress held in Perugia on the work: "Ecological classification of land based on Potential Natural Vegetation: a new quantitative & multivariate approach to map ecosystems. The case study of Campobasso Functional Urban Area".

Oral presentation at the PhD Day held in Perugia on the work "Ecological classification of land based on Potential Natural Vegetation: a new quantitative & multivariate approach to define reference ecosystems. The case study of Campobasso Functional Urban Area"

Oral presentation at the 2nd Early Career Research at the Society of conservation Biology, held in L'Aquila on the work" Effects of anthropogenic landforms on Potential Natural Vegetation in a wide urban ecosystem (Capital city of Rome, Italy)"

Appendices

Appendices I: Mapping Potential Natural Vegetation through a quantitative and multivariate approach. The case study of Campobasso Functional Urban Area (Italy)

Supplementary material, section 1: climatic, morphological and lithological layers used forthe ecological land classification and the ecological modelling.

Figure S.1: Temperature.

Raster of mean monthly temperature in the functional urban areas (FUA) of Campobasso resolution 5mx5m. Derived from the elaboration of Mean monthly Temperature in the period 2011-2020, in QGIS 3.34.4 (ISPRA).

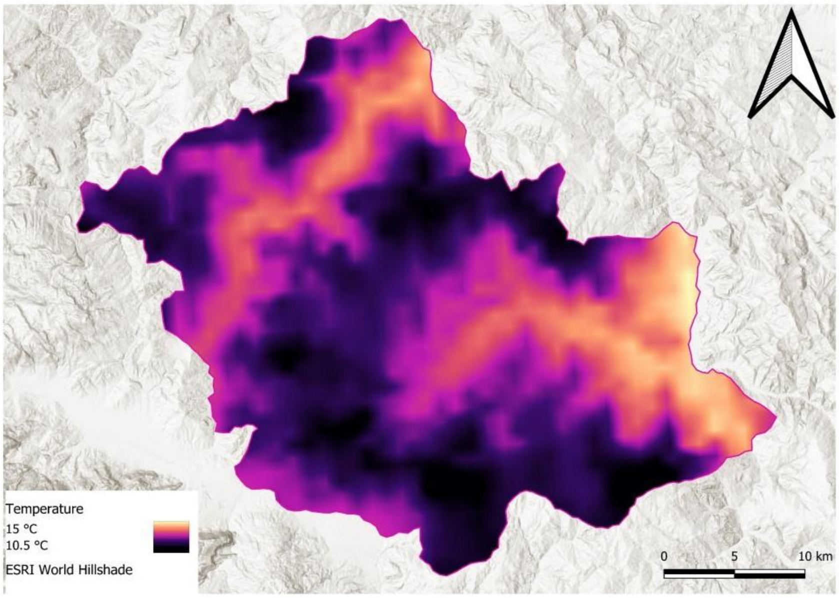


Figure S.2: Precipitation.

Raster of mean monthly precipitation in the FUA of Campobasso resolution 5mx5m. Derived from the elaboration of Mean monthly precipitation in the period 2011-2020, in QGIS 3.34.4 (ISPRA).

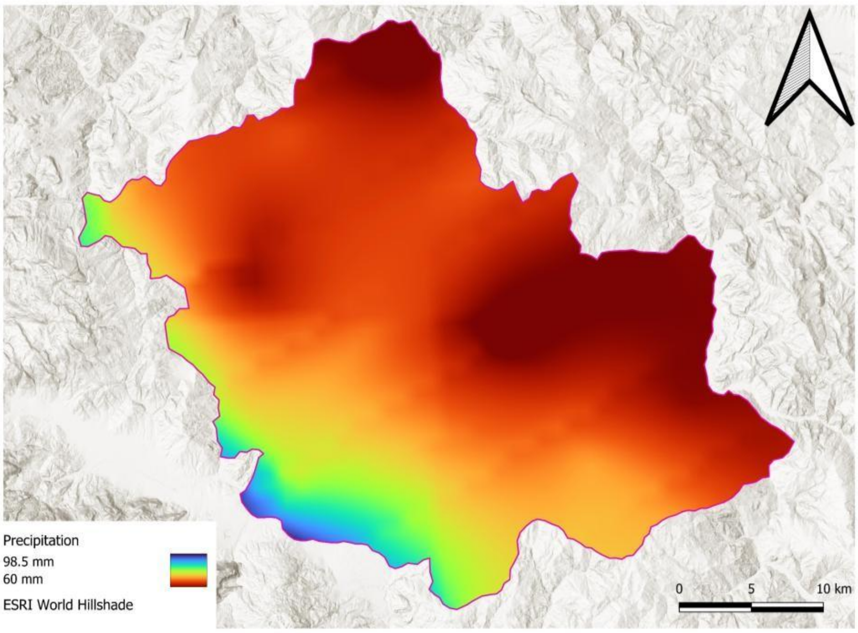


Figure S.3: Phytoclimatic map of Italy.

Phytoclimatic map of Italy expressed in macroclimate, 1:250 000 (Smiraglia et al., 2013).

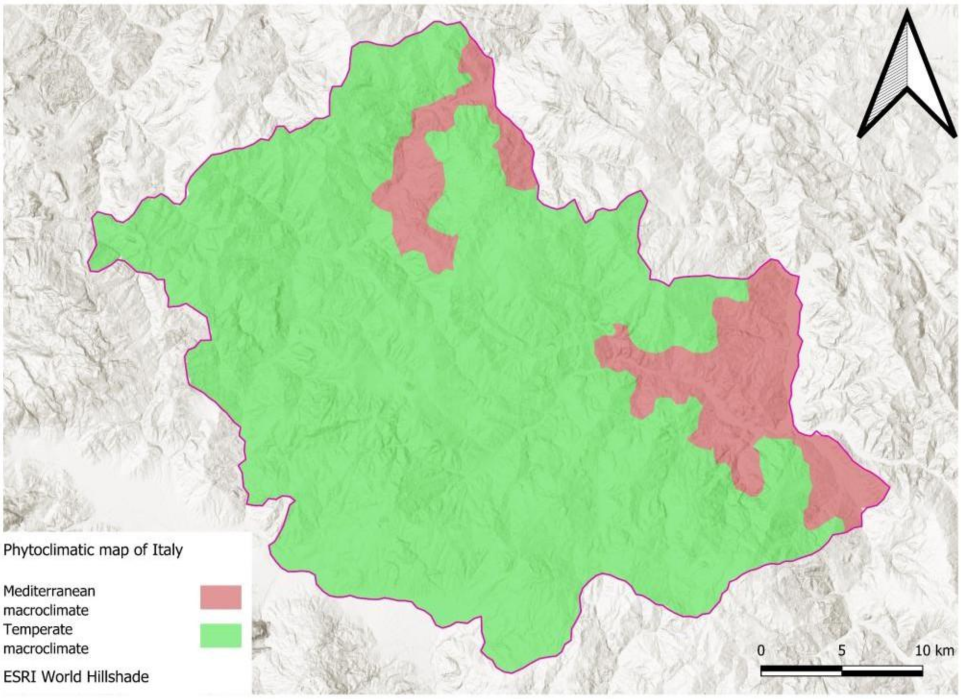


Figure S.4: Lithological map of Italy.

Lithological map of Italy reclassified into five classes, 1:100 000. (Bucci et al., 2022).

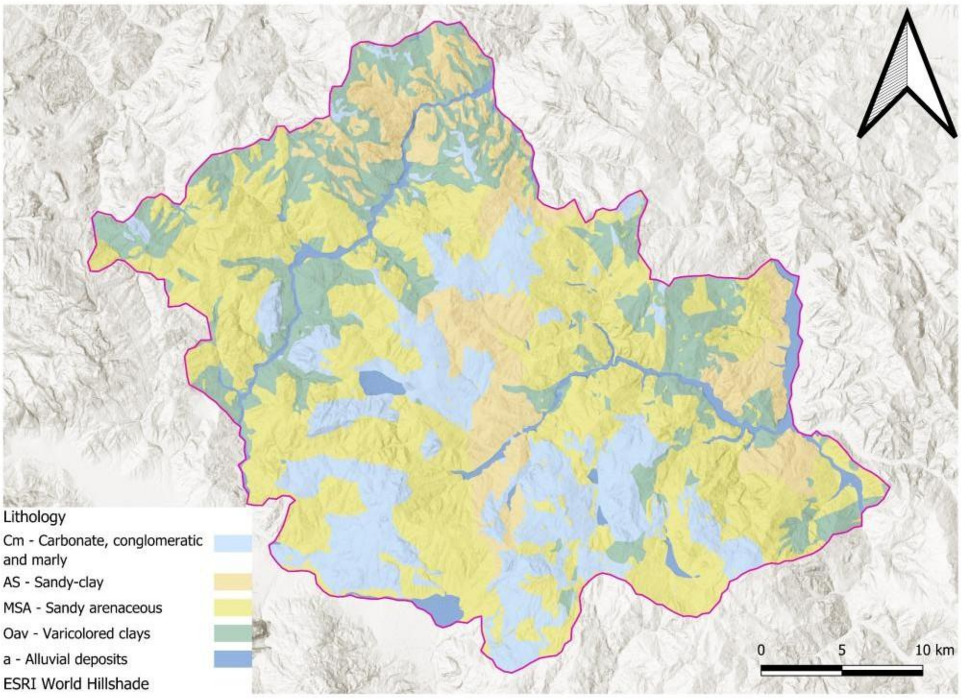


Figure S.5: digital elevation model (DEM).

A 5 m/pixel resolution derived from the 1:5000 scale topographic map (RTM 1:5000)

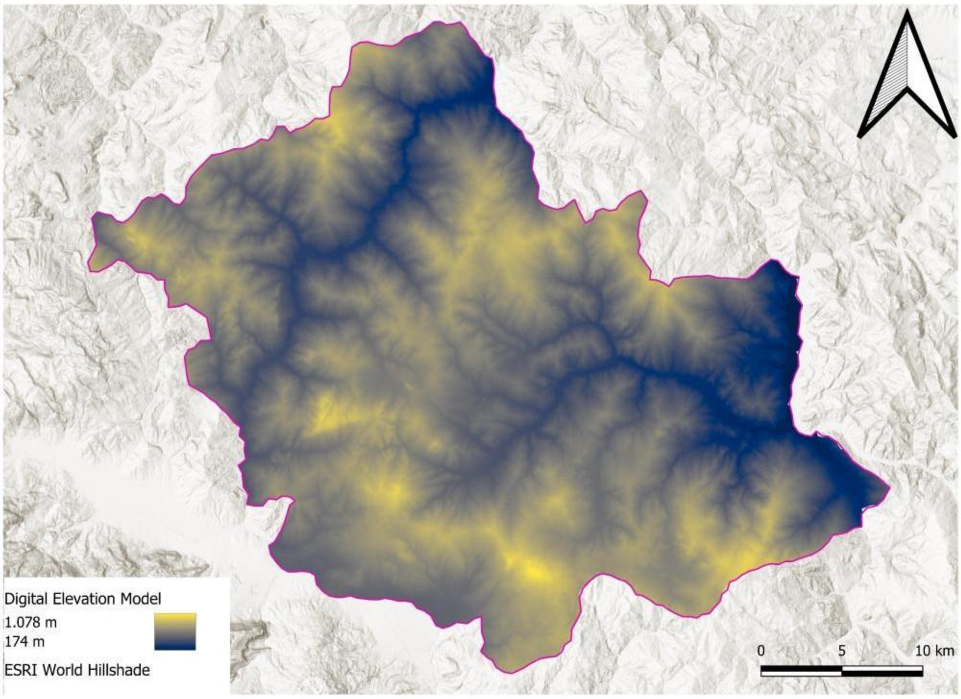


Figure S.6: Slope.

Slope influences both runoff velocity and the geometry of potential failure surfaces. It was computed using QGIS 3.34.4.

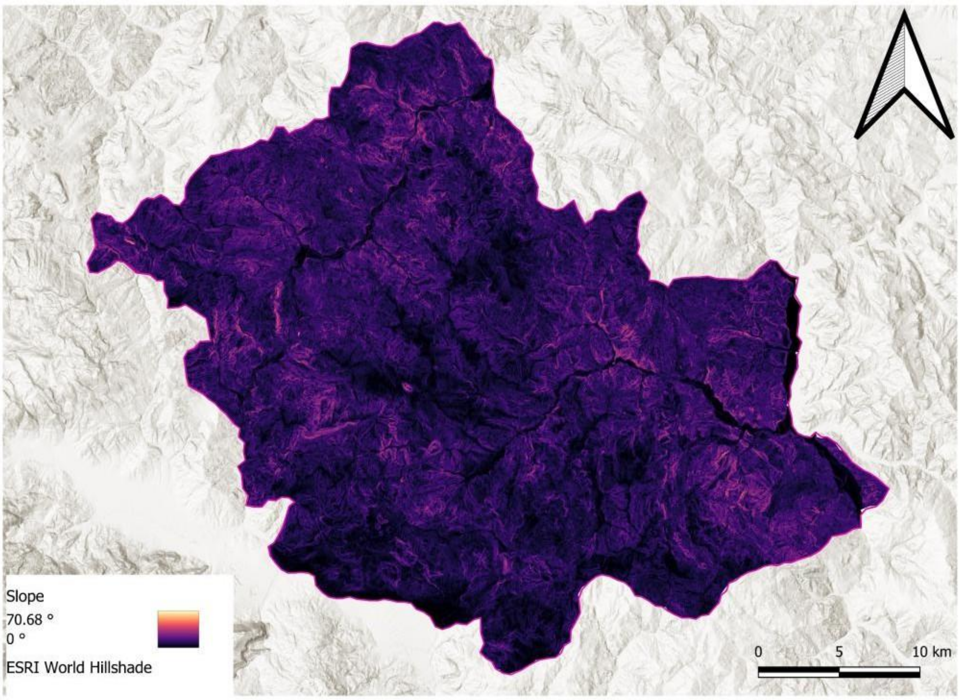


Figure S.7: Heatload.

Heatload measures the insolation based on aspect and slope (McCune C Keon, 2002). Aspect serves as a proxy for seasonal wetting and drying cycles of soils (Auslander et al., 2003). It was computed using QGIS 3.34.4.

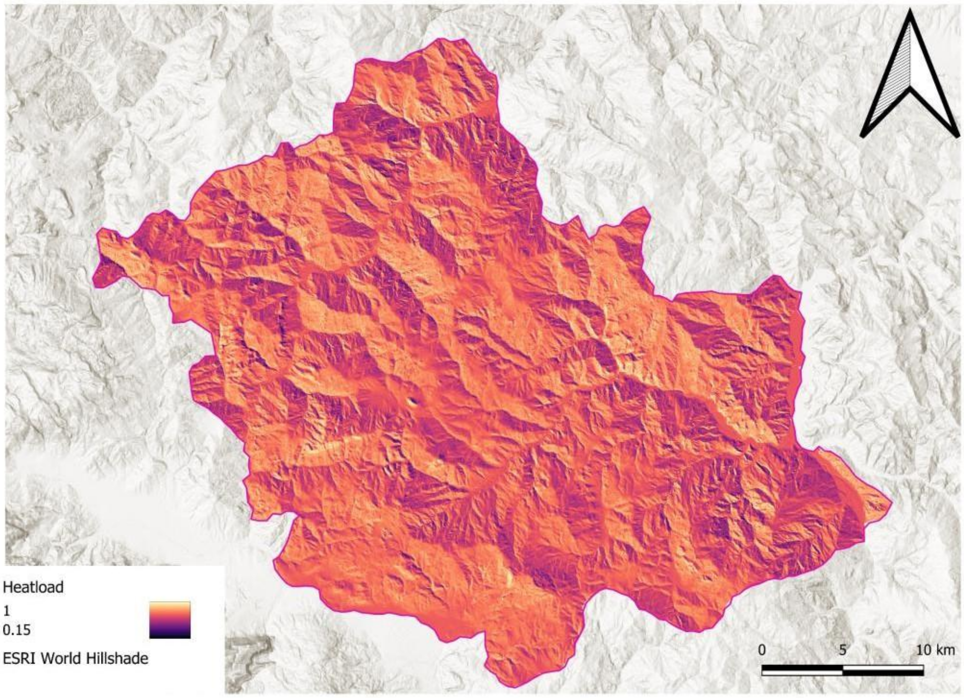


Figure S.8: Topographic Wetness Index (TWI).

TWI represents potential infiltration capacity and is used as a proxy for estimating the presence and thickness of saturated soil horizons (e.g., Martinello et al., 2024). It was computed using SAGA GIS 9.3.1.

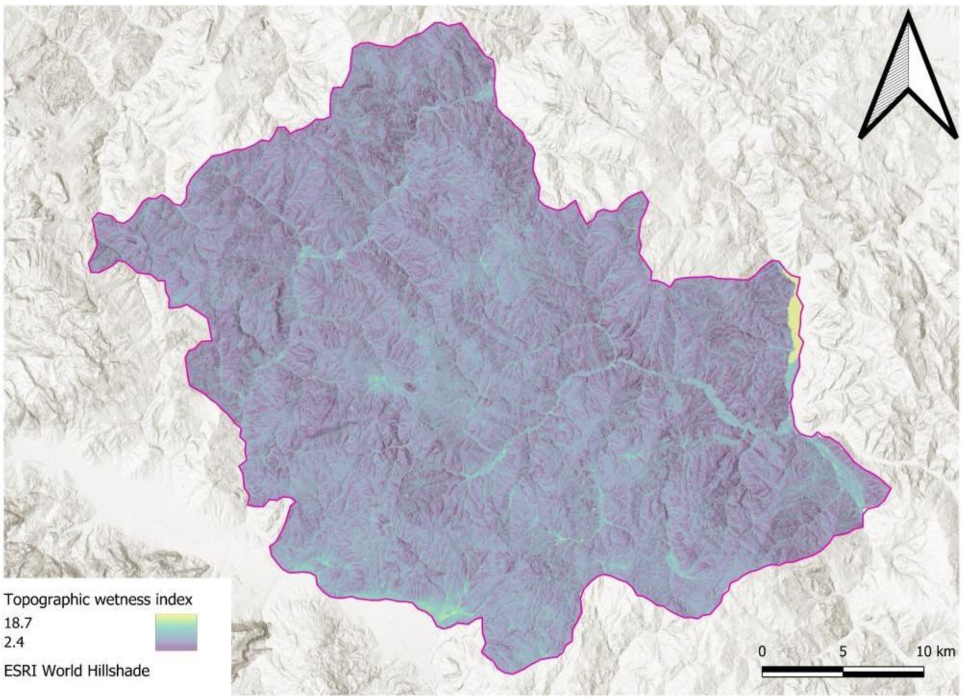


Figure S.9: Height Above Nearest Drainage (HAND) or Vertical distance from channel network.

HAND calculates the vertical distance of each DEM cell above the nearest stream cell along flow paths. It was computed using TopoToolbox.

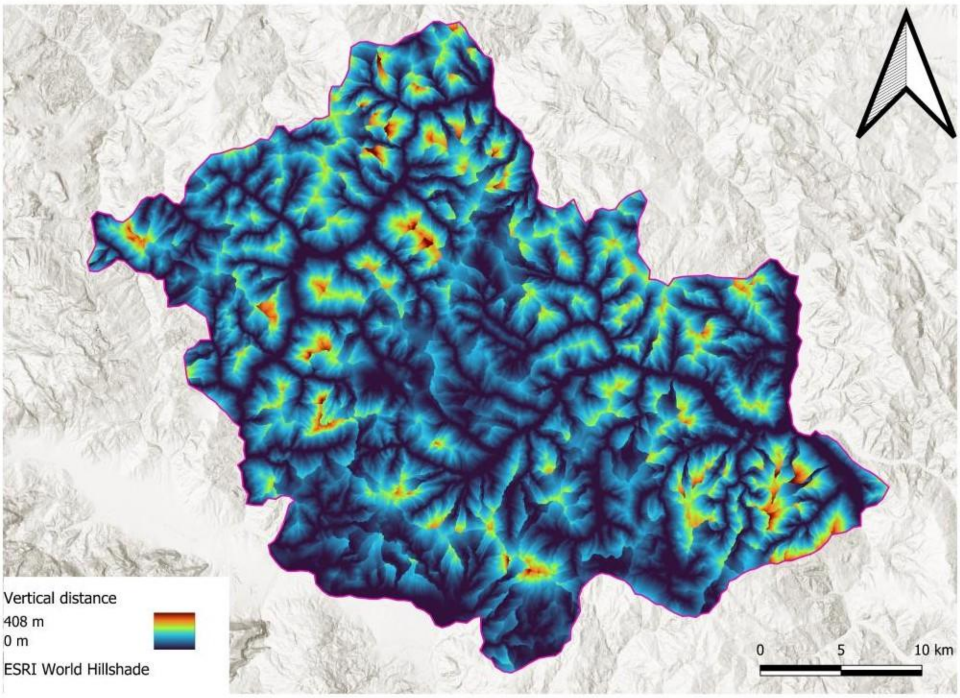


Figure S.10: Horizontal Distance to Channel Network.

This parameter measures the horizontal distance of each DEM cell to the stream network along flow paths. It was also derived using TopoToolbox

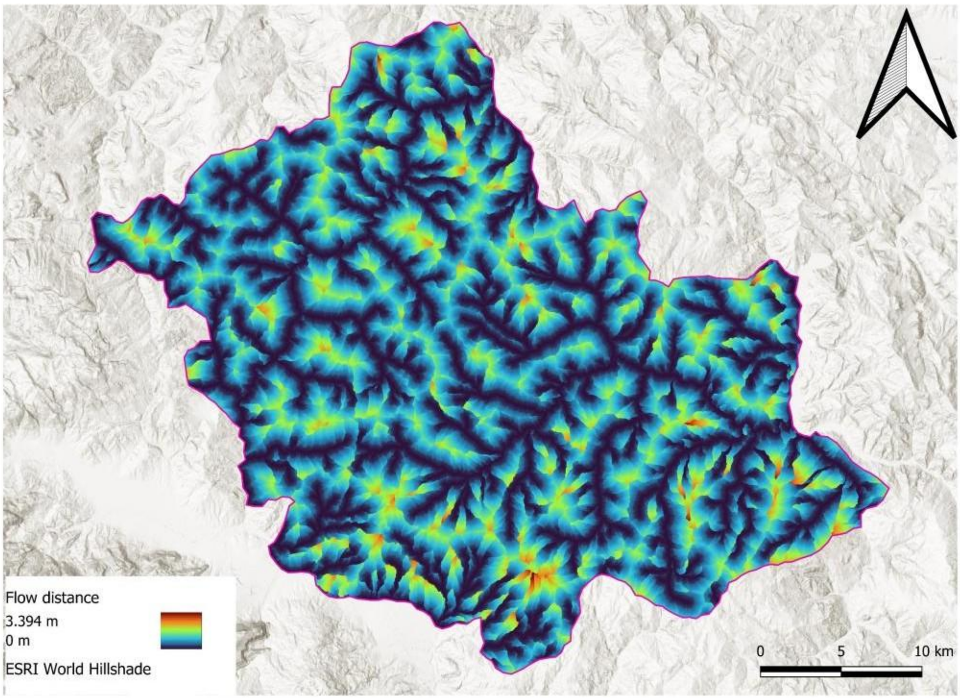


Figure S.11: Geomorphons.

Geomorphons classify landforms using a machine vision approach (Jasiewicz C Stepinski, 2013). The method identifies terrain features such as pits, flats, footslopes, hollows, ridges, shoulders, slopes, spurs, summits, and valleys. Calculations were performed using SAGA GIS 9.3.1.

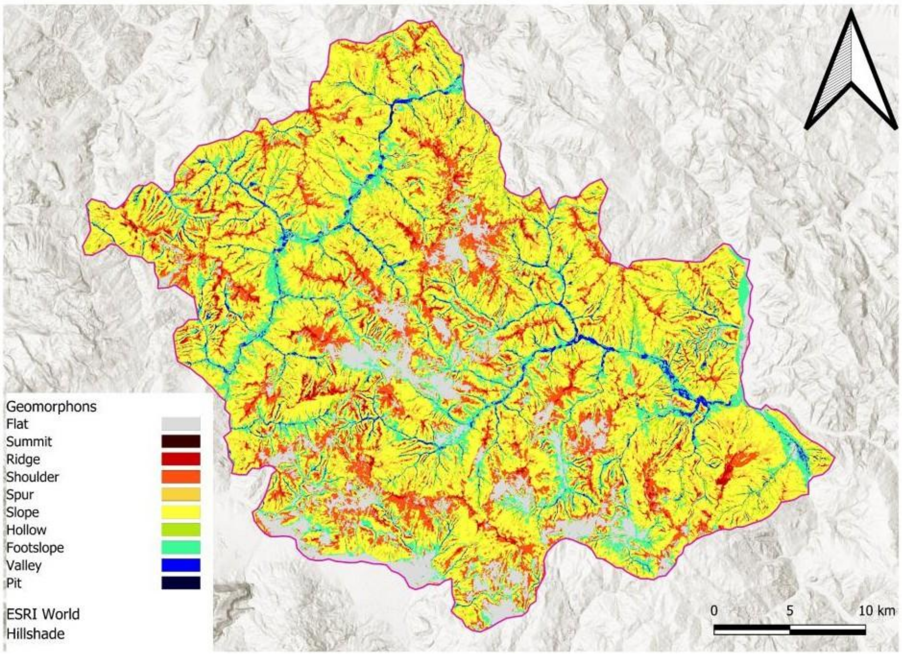
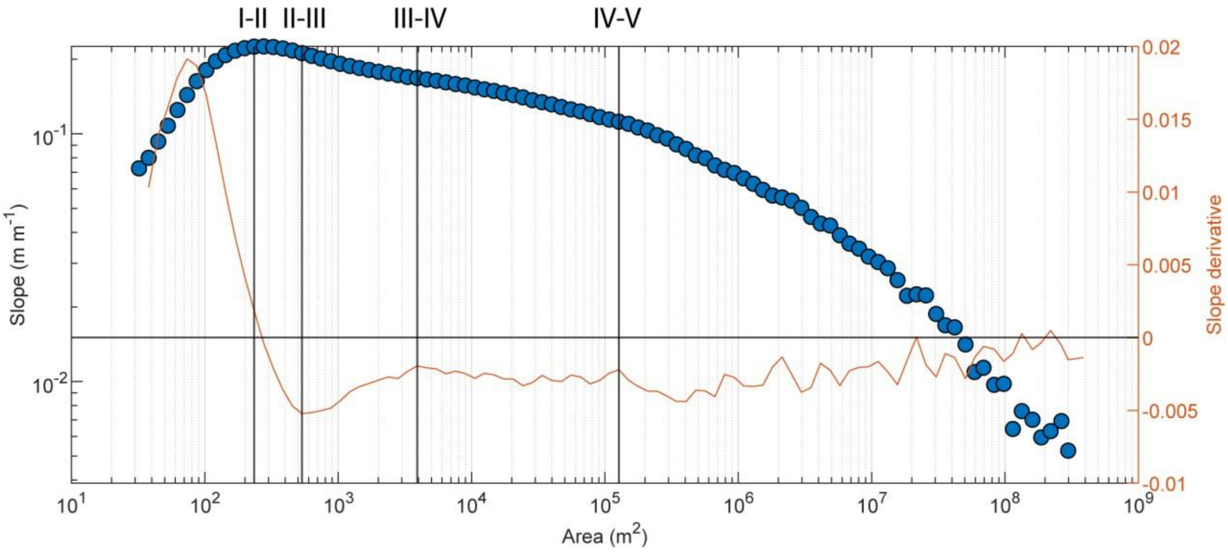
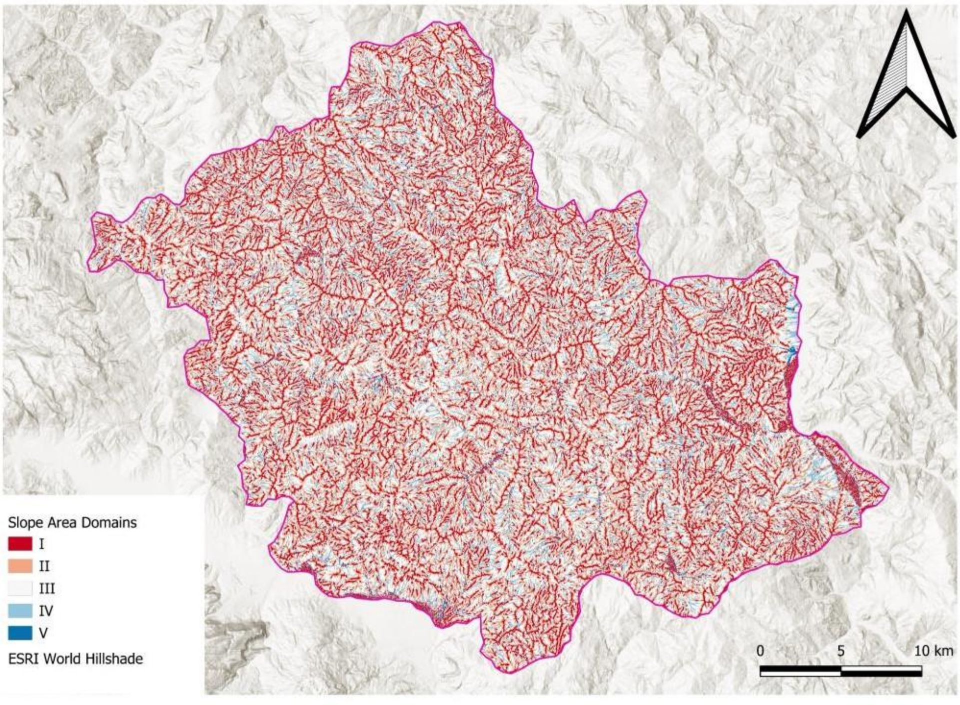


Figure S.12 s S.13: Slope area.

The geomorphological reclassification of the landscape into process domains is based on the widely recognized functional relationship between drainage area (A, in m²) and slope (S, in m m⁻¹). This S–A relationship enables the delineation of local geomorphic domains and supports the interpretation of process interactions (Delchiaro et al., 2023, 2025; Iacobucci et al., 2024; Valiante et al., 2024; Vergari et al., 2019; Wang, 2024) and it was computed using the MATLAB-based platform TopoToolbox (Schwanghart C Scherler, 2014). In this framework, slope values are empirically derived as a function of contributing area for each pixel within a drainage basin. To detect overarching patterns in the S–A distribution, slope and area data are grouped into 100 logarithmically spaced bins based on contributing area, following the approach of Ijjasz- Vasquez and Bras (1995). For each bin, the average slope is computed, resulting in a binned S–A diagram that reveals five distinct regions, each characterized by unique scaling behavior. Region I, associated with the smallest contributing areas, exhibits a positive S–A gradient, indicative of convex hillslopes. In Region II, the gradient turns negative, reflecting concave unchannelled topography. Region III continues the negative trend, but with a reduced slope, while Regions IV and V show a steepening negative gradient. These transitions illustrate a progression from convex hillslope domains (Region I), through concave colluvial and debris-flow- dominated zones (Regions II–IV), to fluvially dominated concave channels (Region V) (Vergari et al., 2019, and references therein). Thresholds between regions are identified by examining the first derivative of the S–A curve (δS/δA) and are supported by previous studies. The threshold between Regions I and II corresponds to the point where δS/δA changes from positive to negative, marking the shift from convex hillslopes to incipient channels, as described by Montgomery and Foufoula-Georgiou (1993) and Tarboton et al. (1992). The remaining thresholds draw on additional literature, such as McNamara et al. (2006). Region II's consistently negative gradient reflects dominance by wash and rill erosion. In Regions III and IV, δS/δA fluctuates near zero, consistent with lobate debris-flow features (Tucker C Bras, 1998). Finally, Region V exhibits a negative δS/δA following a power-law distribution, characteristic of fluvial concavity (Flint, 1974), indicating well-developed channel networks. These domain thresholds are plotted on each basin's binned S–A profile.


I-II
II-III
III-IV
IV-V
Slope (m m^{-1})
Slope derivative
Area (m^2)
10^{-1}
10^{-2}
10^1
10^2
10^3
10^4
10^5
10^6
10^7
10^8
10^9
0.02
0.015
0.01
0.005
0
-0.005
-0.01


Slope Area Domains
I
II
III
IV
V
ESRI World Hillshade
0
5
10 km

REFERENCES

1. Auslander, M., Nevo, E., C Inbar, M., 2003. The effects of slope orientation on plant growth, developmental instability and susceptibility to herbivores. Journal of Arid Environments, 55(3), 405-416. https://doi.org/10.1016/S0140- 1963(02)00281-1
2. Delchiaro, M., Della Seta, M., Martino, S., Nozaem, R., C Moumeni, M., 2023. Tectonic deformation and landscape evolution inducing mass rock creep driven landslides: the Loumar case-study (Zagros Fold and Thrust Belt,
3. Iran). Tectonophysics, 84c, 229655. https://doi.org/10.1016/j.tecto.2022.229655
4. Delchiaro, M., Vergari, F., Esposito, C., C Del Monte, M., 2025. The influence of anthropogenic topographic changes on geomorphological processes in the city of Rome (Italy): A case study of the Malagrotta area. Earth Surface Processes and Landforms, 50(3), e70033. https://doi.org/10.1002/esp.70033
5. Flint, J. J. (1974). Stream gradient as a function of order, magnitude, and discharge. Water Resources Research, 10(5), 969-973. https://doi.org/10.1029/WR010i005p00969
6. Iacobucci, G., Delchiaro, M., Troiani, F., C Nadali, D., 2024. Land-surface quantitative analysis for mapping and deciphering the construction processes of piedmont alluvial fans in the Anti-Lebanon
7. Mountains. Geomorphology, 453, 109148. https://doi.org/10.1016/j.geomorph.2024.109148
8. Ijjasz-Vasquez, E. J., C Bras, R. L.,1995. Scaling regimes of local slope versus contributing area in digital elevation models. Geomorphology, 12(4), 299-311. https://doi.org/10.1016/0169-555X(95)00012-T
9. Jasiewicz, J., C Stepinski, T. F., 2013. Geomorphons—a pattern recognition approach to classification and mappingof landforms. Geomorphology, 182, 147-156. https://doi.org/10.1016/j.geomorph.2012.11.005
10. Martinello, C., Delchiaro, M., Iacobucci, G., Cappadonia, C., Rotigliano, E., C Piacentini, D., 2024. Exploring the geomorphological adequacy of the landslide susceptibility maps: A test for different types of landslides in the Bidente river basin (northern Italy). Catena, 238, 107835. https://doi.org/10.1016/j.catena.2024.107835
11. McNamara, J. P., Ziegler, A. D., Wood, S. H., C Vogler, J. B., 2006. Channel head locations with respect to geomorphologic thresholds derived from a digital elevation model: A case study in northern Thailand. Forest Ecology and Management, 224(1-2), 147-156. https://doi.org/10.1016/j.foreco.2005.12.014
12. Montgomery, D.R. C Dietrich, W.E.,1994. Landscape dissection and drainage area-slope thresholds. In: M. Kirkby (Ed.) Process models and theoretical geomorphology, pp. 221–246. John Wiley and Sons Ltd.
13. Montgomery, D. R., C Foufoula-Georgiou, E., 1993. Channel network source representation using digital elevation models. Water Resources Research, 2S(12), 3925-3934. https://doi.org/10.1029/93WR02463
14. Schwanghart, W., C Scherler, D., 2014. TopoToolbox 2–MATLAB-based software for topographic analysis and modeling in Earth surface sciences. Earth Surface Dynamics, 2(1), 1-7. https://doi.org/10.5194/esurf-2-1-2014
15. Tarboton, D. G., 1997. A new method for the determination of flow directions and upslope areas in grid digital elevation models. Water resources research, 33(2), 309-319. https://doi.org/10.1029/96WR03137
16. Tarboton, D. G., Bras, R. L., C Rodriguez-Iturbe, I., 1992. A physical basis for drainage density. Geomorphology, 5(1- 2), 59-76. https://doi.org/10.1016/0169-555X(92)90058-V
17. Tucker, G. E., C Bras, R. L., 1998. Hillslope processes, drainage density, and landscape morphology. Water resources research, 34(10), 2751-2764. https://doi.org/10.1029/98WR01474
18. Valiante, M., Di Benedetto, A., C Aloia, A., 2024. A Comparison of Landforms and Processes Detection Using Multisource Remote Sensing Data: The Case Study of the Palinuro Pine Grove (Cilento, Vallo di Diano and Alburni National Park, Southern Italy). Remote Sensing, 1c(15), 2771. https://doi.org/10.3390/rs16152771
19. Vergari, F., Troiani, F., Faulkner, H., Del Monte, M., Della Seta, M., Ciccacci, S., C Fredi, P., 2019. The use of the slope–area function to analyse process domains in complex badland landscapes. Earth Surface Processes and Landforms, 44(1), 273-286. https://doi.org/10.1002/esp.4496
20. Wang, Y., 2024. Identifying geomorphic domains using hierarchically clustered drainage area-slope scaling. Earth Surface Processes and Landforms, 4S(6), 1762-1777. https://doi.org/10.1002/esp.5796
21. Smiraglia, D., Capotorti, G., Guida, D., Mollo, B., Siervo, V., C Blasi, C., 2013. Land units map of Italy. Journal of Maps, 9(2), 239-244. https://doi.org/10.1080/17445647.2013.771290.
22. Bucci, Francesco; Santangelo, Michele; Fongo, Lorenzo; Alvioli, Massimiliano; Cardinali, Mauro; Melelli, Laura; Marchesini, Ivan, 2022. A new digital lithological Map of Italy at 1:100.000 scale [dataset]. PANGAEA, https://doi.org/10.1594/PANGAEA.935673
23. McCune, B., and D. Keon, 2002. Equations for potential annual direct incident radiation and heat load index. Journal of Vegetation Science. 13:603-606.

24. ISPRA, Sinanet, SINAnet Groupware, last access on 13/05/2025
25. ‘Servizio WMS - CTR 5000’, accessed 2025-06-10, http://data.europa.eu/88u/dataset/r_molise-00013-20120413- 094513.

Supplementary material, section 2: clustering and environmental variable assessment results.

Figure S.1: Silhouette values of Partitioning around medoid (PAM) clustering.

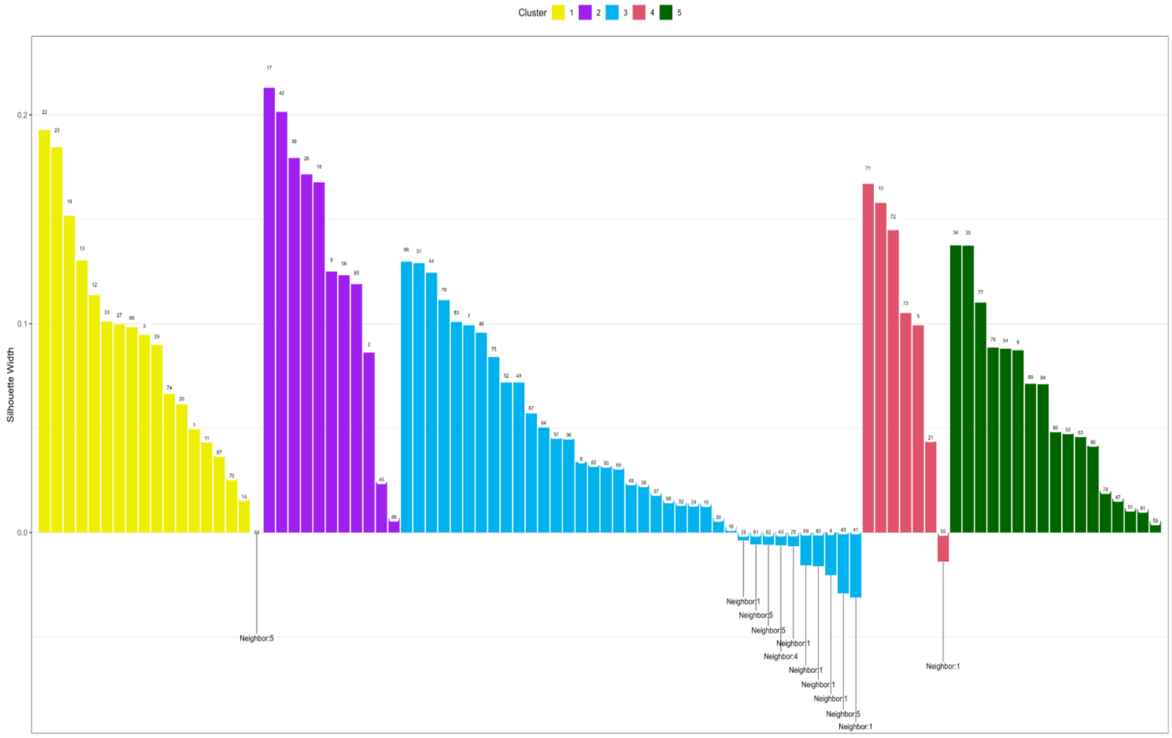


Figure S.2: Fuzzy clustering built upon PAM medoid values.

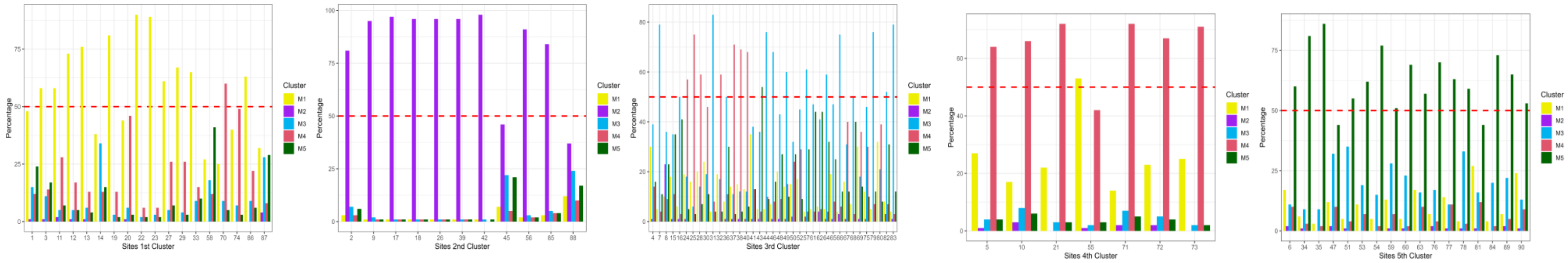


Figure S.3: Inside-group hierarchical clustering.

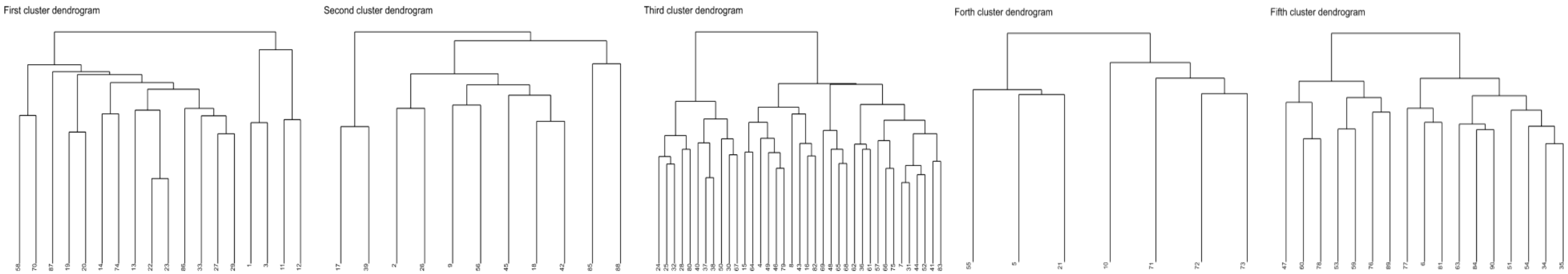


Figure S.4: PCA, projection on principal component 1 (PC1) and principal component 3 (PC3) with Environmental variables

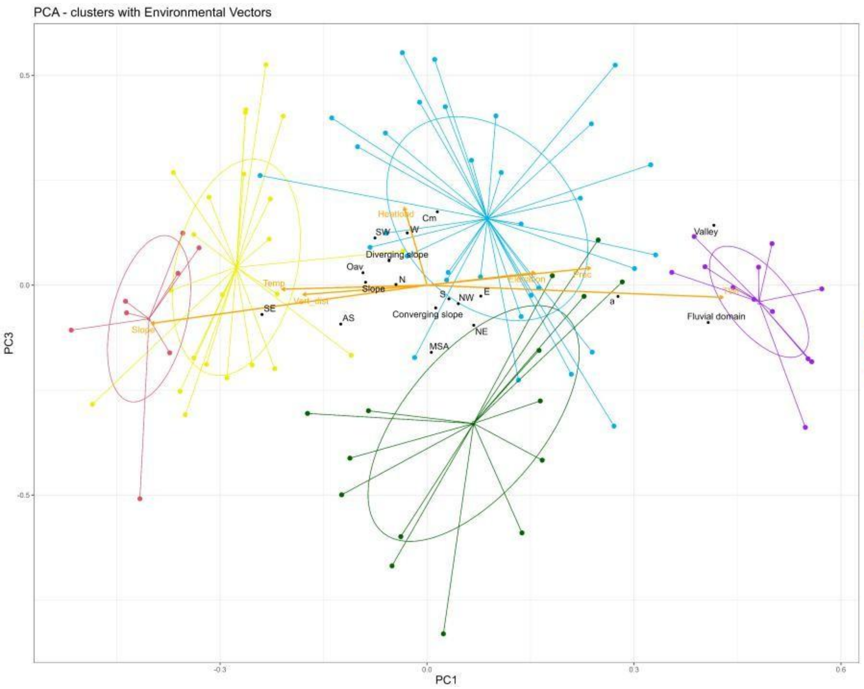


Figure S.5: PCA, projection on PC1 and PC2 with aspect factor classes and relative barplot.

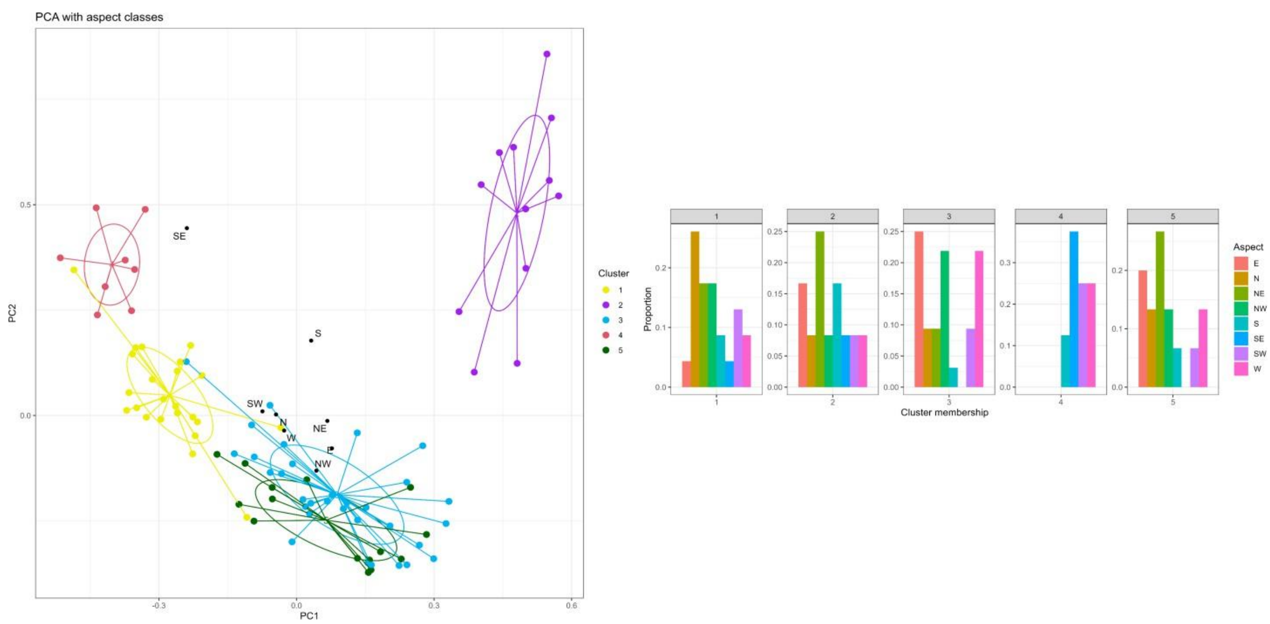


Figure S.6: PCA. Projection on PC1 and PC2 with slope-area classification factor classes and relative barplot.

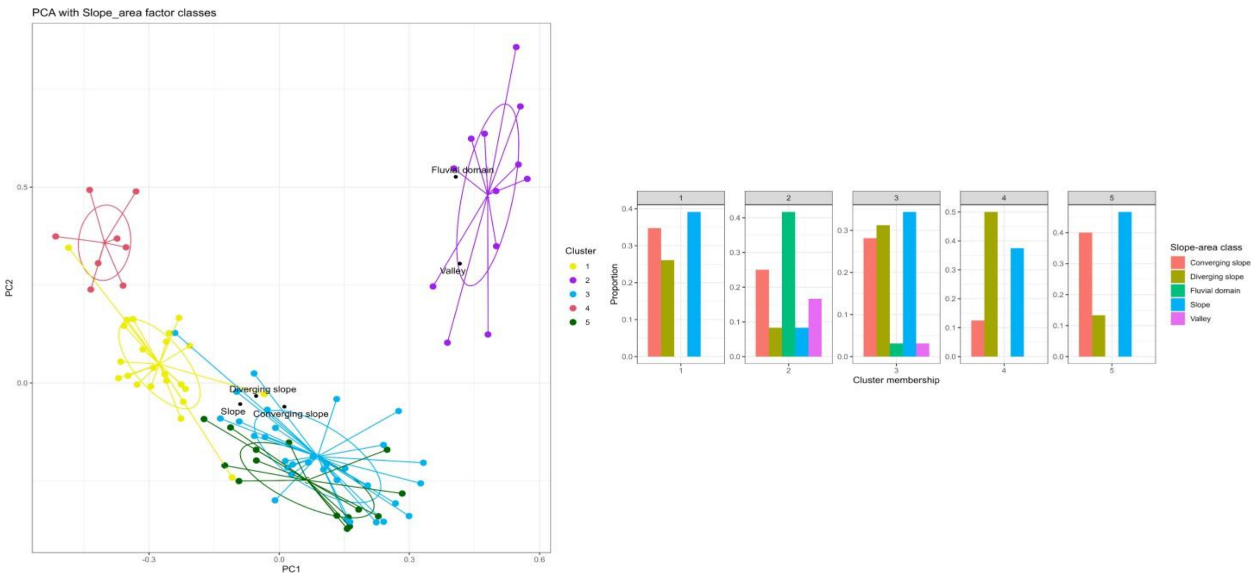


Figure S.7: PCA, projection on PC1 and PC2 with lithological factor classes and relative barplot.

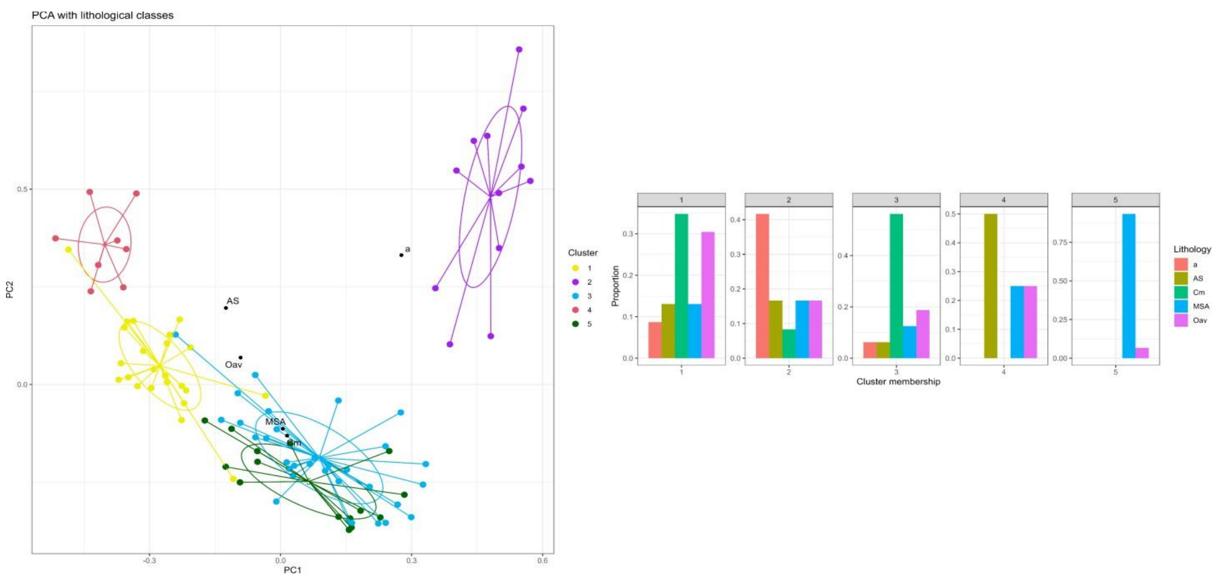


Figure S.8: PCA, projection on PC1 and PC2 with elevation ordisurf expressed in meters and boxplot.

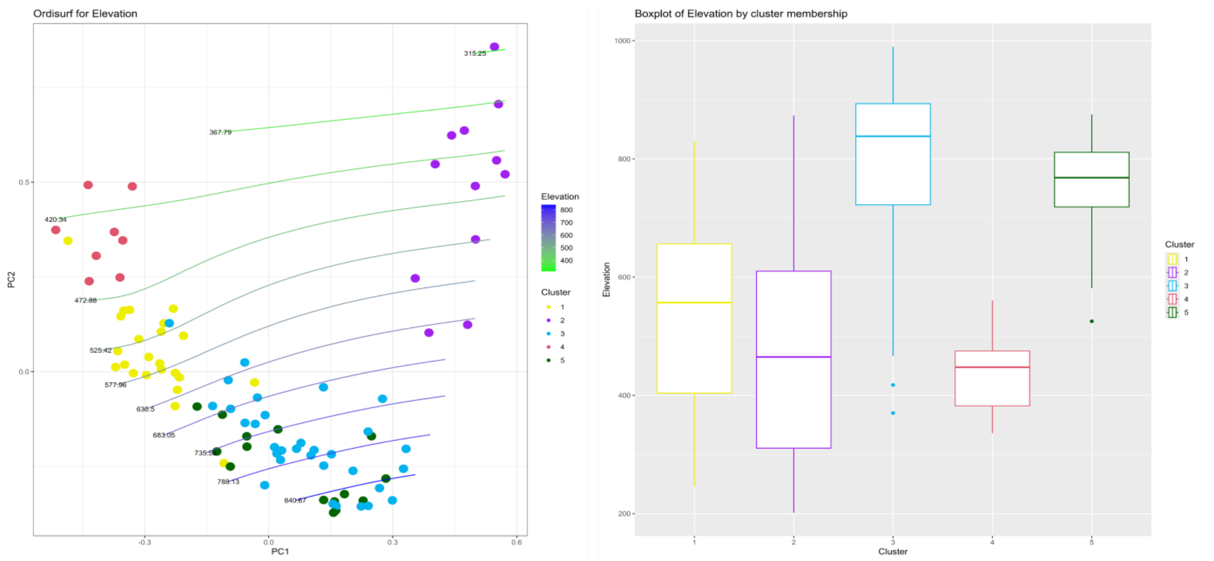


Figure S.9: PCA, projection on PC1 and PC2 with horizontal distance from channel network ordisurf expressed in meters and boxplot.

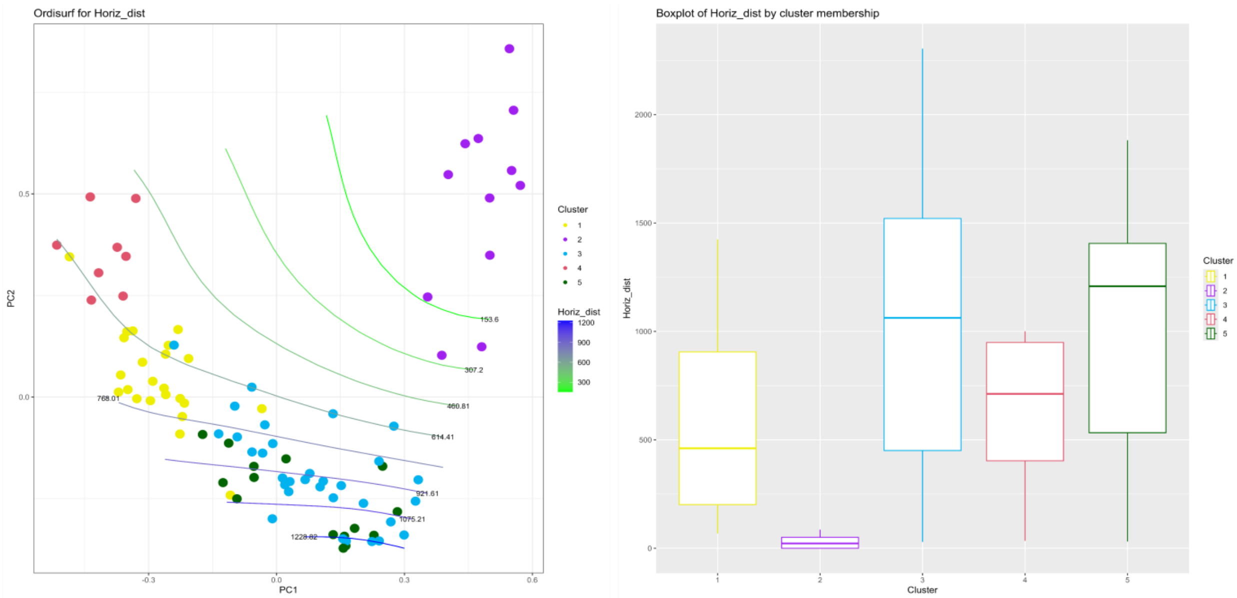


Figure S.10: PCA, projection on PC1 and PC2 with vertical distance from channel network ordisurf expressed in meters and boxplot.

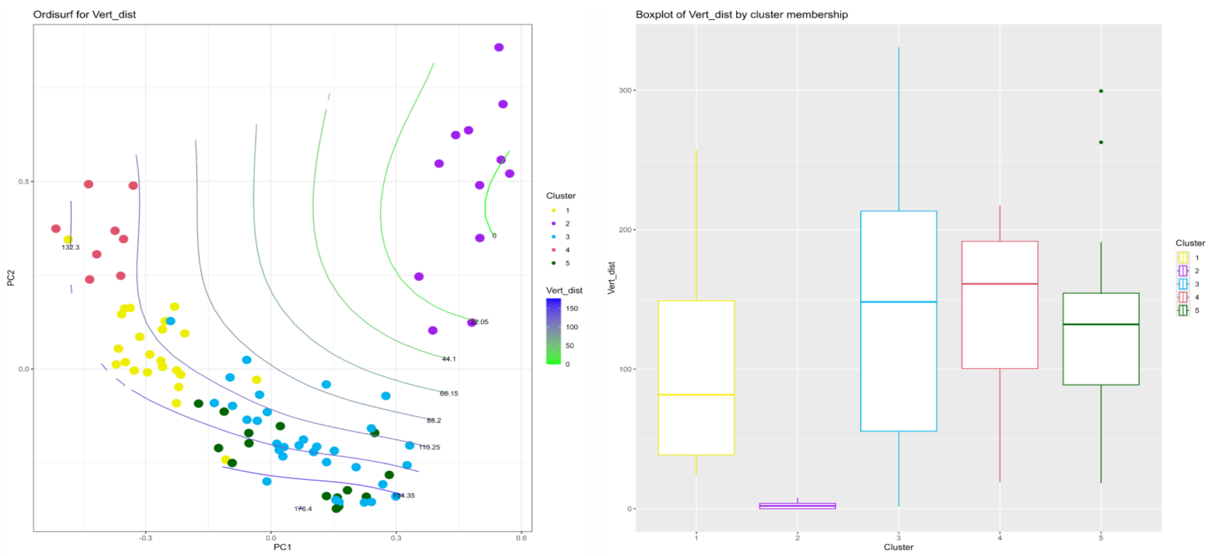


Figure S.11: PCA, projection on PC1 and PC2 with Topographic wetness index ordisurf and boxplot.

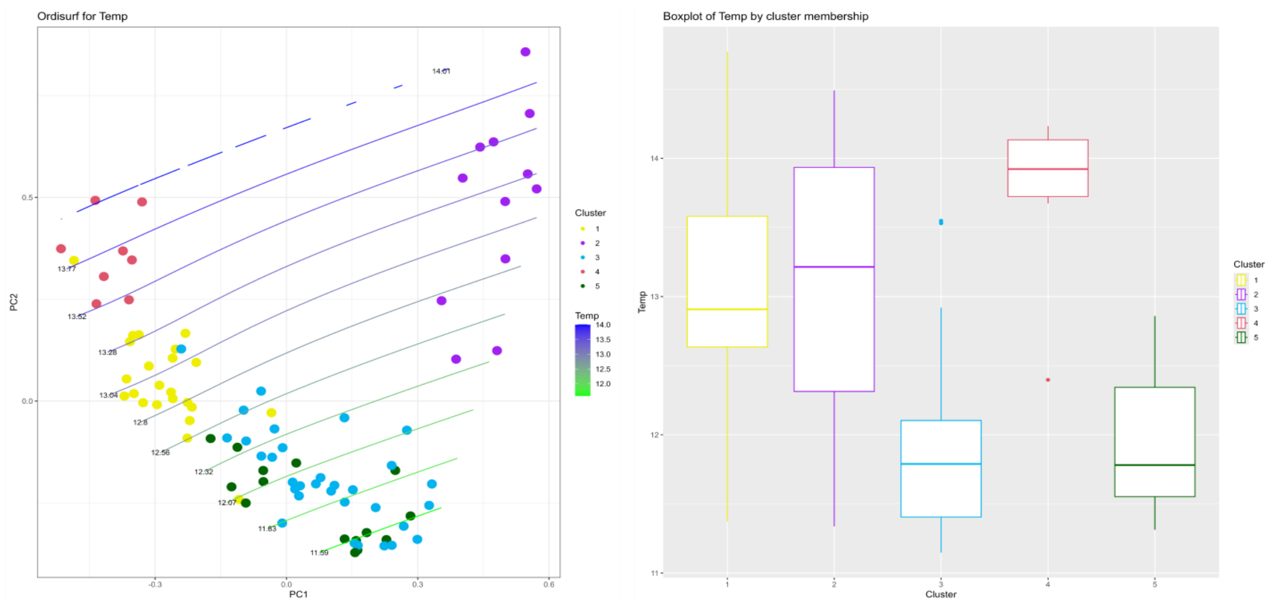


Figure S.12: PCA, projection on PC1 and PC2 with temperature ordisurf expressed in °C and boxplot

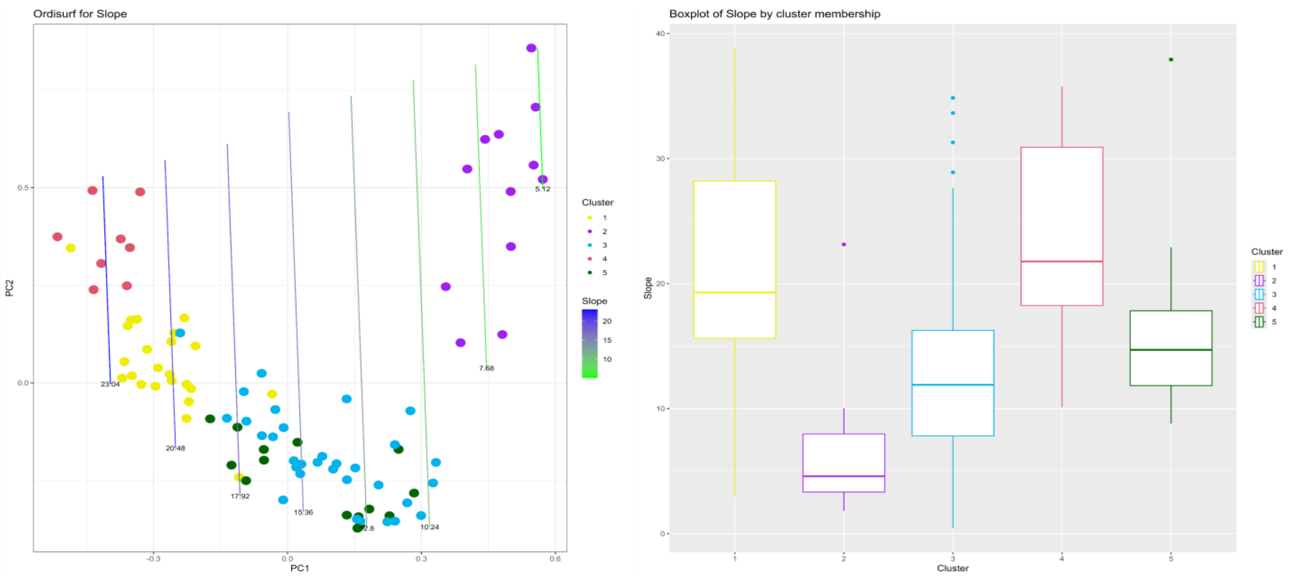


Figure S.13: PCA, projection on PC1 and PC2 with slope ordisurf expressed in degree ° and boxplot.

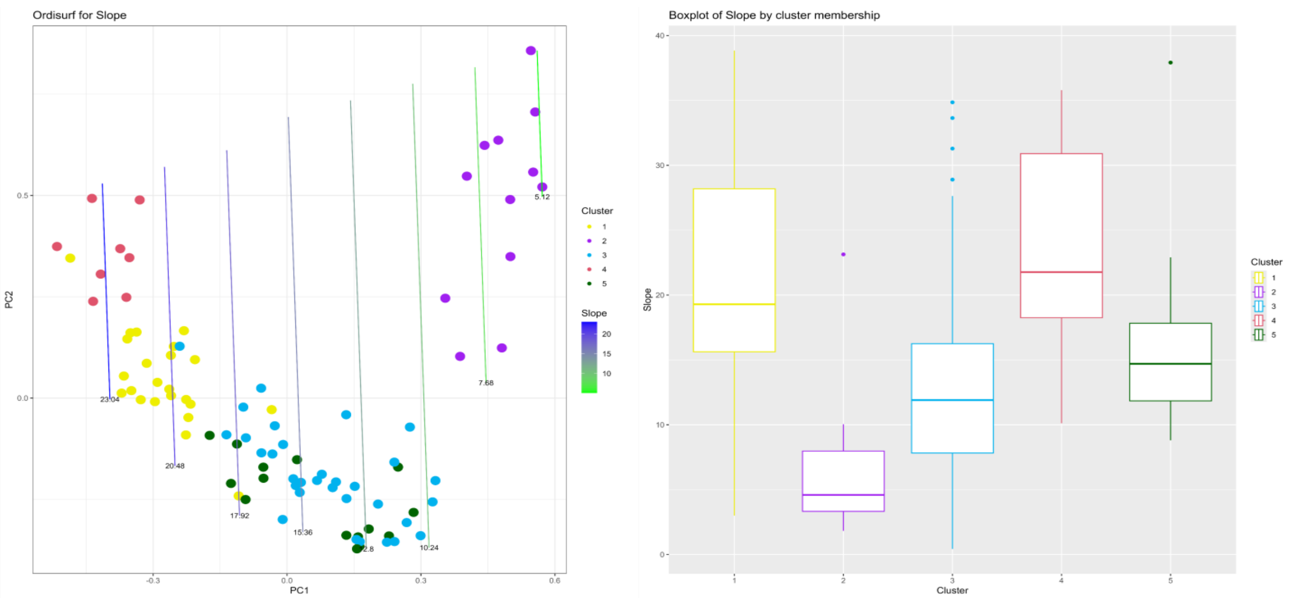
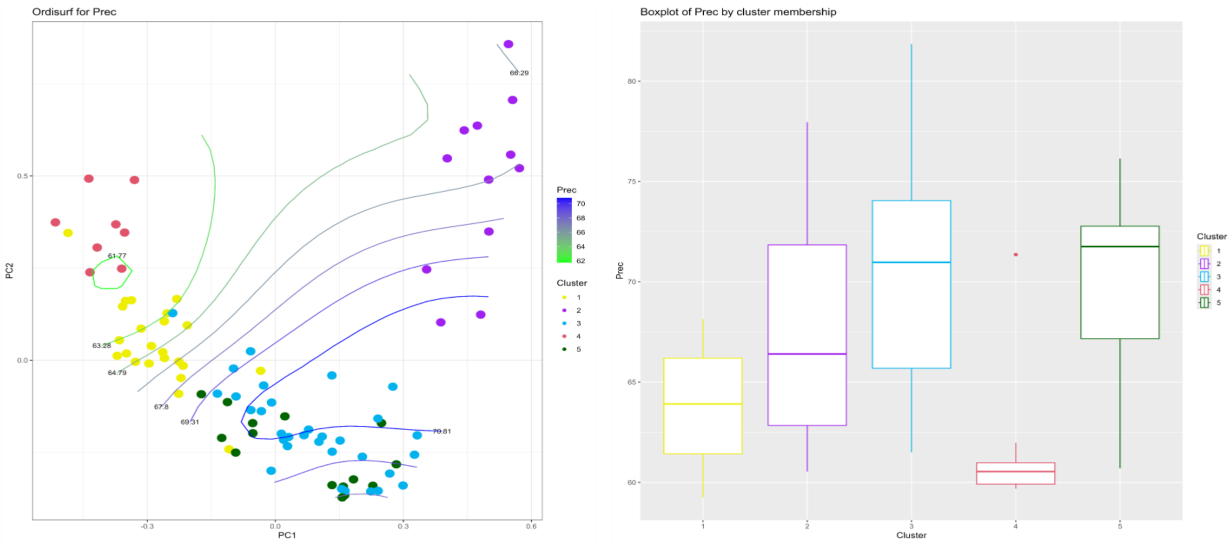


Figure S.14: PCA, projection on PC1 and PC2 with precipitation ordisurf expressed in mm and boxplot.



Supplementary material, section 3: Variance Inflation Factor VIF, correlations and significance tests tables and syntaxa cited in the text.

Table S.1: Complete significance test of environmental variables

Variable	Test	P-value
Precipitation	Kruskal-Wallis	5.34 E-06
Temperature	Kruskal-Wallis	8.35 E-08
Elevation	ANOVA	2.77 E-11
Topographic Wetness Index	Kruskal-Wallis	1.33E-06
Slope	Kruskal-Wallis	1.53 E-05
Heatload	Kruskal-Wallis	0.04 E-00
Vertical distance from the channel network	Kruskal-Wallis	2.01 E-06
Horizontal distance from the channel network	Kruskal-Wallis	4.52 E-07
Slope Area Domains	Chi-squared	0.01 E-01
Lithology	Chi-squared	0.05 E-02
Aspect*	Chi-squared	0.053 E00
Solar radiation aspect index	Kruskal-Wallis	0.075 E00
Tangential curvature	Kruskal-Wallis	0.579 E00
Profile curvature	Kruskal-Wallis	0.321 E00
Flow accumulation	Kruskal-Wallis	0.030 E00

Table S.2: Correlation between significant environmental variables

	Elevation	Slope	Temp	TWI	Prec	Vert_dist	Horiz_dist	Heatload
Elevation	1	-0.17418	-0.94917	-0.15173	0.449491	0.517595	0.6125	0.089124
Slope	-0.17418	1	0.21148	-0.5421	-0.11679	0.107971	0.026266	0.011158
Temp	-0.94917	0.21148	1	0.055347	-0.5069	-0.35936	-0.46217	-0.05821
TWI	-0.15173	-0.5421	0.055347	1	0.148643	-0.38799	-0.28771	-0.07879
Prec	0.449491	-0.11679	-0.5069	0.148643	1	-0.06217	0.057869	0.079448
Vert_dist	0.517595	0.107971	-0.35936	-0.38799	-0.06217	1	0.907137	0.101172
Horiz_dist	0.6125	0.026266	-0.46217	-0.28771	0.057869	0.907137	1	0.008104
Heatload	0.089124	0.011158	-0.05821	-0.07879	0.079448	0.101172	0.008104	1

Table S.3: VIF table of Cluster 1 level from the multinomial logistic regression model

Cluster 1-3	Environmental variables	GVIF	Df	GVIF^(1/(2*Df))
	Slope	2.187017	1	1.478856676
	Temp	2.502472	1	1.581920457
	TWI	5.71553	1	2.390717392
	Prec	2.993675	1	1.730224056
	SA_domains	6.041773	4	1.252118845
	Lithology	5.220457	4	1.229457631
	Vert_dist	9.408759	1	3.067370081
	Horiz_dist	8.710121	1	2.951291438
	Heatload	1.379518	1	1.174528922

Table S.4: VIF table of Cluster 2 from the multinomial logistic regression model

Cluster 2-3	Environmental variables	GVIF	Df	GVIF^(1/(2*Df))
	Slope	18.12397	1	4.257225924
	Temp	159.6438	1	12.6350219
	TWI	78.41404	1	8.855169961
	Prec	29.55601	1	5.436543947
	SA_domains	7998.384	4	3.075213584
	Lithology	80959.79	4	4.107084807
	Vert_dist	90.80436	1	9.529132273
	Horiz_dist	74.21577	1	8.614857714
	Heatload	15.23191	1	3.902808514

Table 5: VIF table of Cluster 4 from the multinomial logistic regression model

Cluster 4-3	Environmental variables	GVIF	Df	GVIF^(1/(2*Df))
	Slope	5.635411	1	2.373902096
	Temp	15.42826	1	3.927882696
	TWI	135.1952	1	11.62734635
	Prec	49.56112	1	7.039965877
	SA_domains	1294.167	4	2.449056413
	Lithology	330.1912	4	2.06464644
	Vert_dist	76.0815	1	8.722471149
	Horiz_dist	77.97043	1	8.830086409
	Heatload	6.754386	1	2.598920095

Table S.6: VIF table of Cluster 5 from the multinomial logistic regression model

Cluster 5-3	Environmental variables	GVIF	Df	GVIF^(1/(2*Df))
	Slope	13.55438	1	3.6816272
	Temp	3.840639	1	1.959754751
	TWI	8.865373	1	2.977477593
	Prec	7.645089	1	2.764975328
	SA_domains	41.3598	4	1.592473841
	Lithology	5.706336	4	1.243210489
	Vert_dist	31.54781	1	5.616743912
	Horiz_dist	17.94225	1	4.235829572
	Heatload	9.598036	1	3.098069707

Syntaxa cited in the text

Lauro nobilis-Quercenion virgilianae Ubaldi 1995; Lonicero etruscae-Quercetum cerridis (Taffetani et Biondi 1995) Terzi, Ciaschetti, Fortini, Rosati, Viciani C Di Pietro 2021; Populion albae Br.-Bl. ex Tchou 1948; Salicion albae Soò 1930; Daphno laureolae-Quercetum cerridis Taffetani et Biondi 1SS5 rosetosum arvensis Taffetani, Catorci, Ciaschetti, Cutini, Di Martino, Frattaroli, Paura, Pirone, Rismondo et Zitti ex Terzi, Ciaschetti, Fortini, Rosati, Viciani et Di Pietro 2021; Pulmonario apenninae-Carpinenion betuli Biondi, Casavecchia, Pinzi, Allegrezza et Baldoni ex Terzi, Ciaschetti, Fortini, Rosati, Viciani et Di Pietro 2021; Roso sempervirentis- Quercetum pubescentis Biondi 1986; Carpinion orientalis Horvat 1958; Crataego laevigatae- Quercion cerridis Arrigoni 1997; Echinopo siculi-Quercetum frainetto Blasi et Paura 1995.

Appendices II: How far from naturalness: anthropogenic landform effects on potential vegetation in a wide urban ecosystem (Capital City of Rome, Italy)

Supplementary material A: Supplementary material A.xlsx - Fogli Google

Supplementary material B

Indicator species analyses results

Table S.1: IndVal multipatt analysis of the six groups: volcanic high naturalness woodland control s (V_HNW), sandy high naturalness woodland controls (S_HNW), volcanic low naturalness control woodlands (V_LNW), sandy low naturalness woodland controls (S_LNW), volcanic anthropogenic landform woodland controls (V_ALW), sandy anthropogenic landform woodland controls (S_ALW).

Types	Species	indval	p-value
S_ALW	*Parietaria judaica*	0.60	0.00
	Oloptum miliaceum	0.46	0.03
	Acanthus mollis subsp. *mollis*	0.45	0.05
V_ALW	*Anthriscus sylvestris* subsp. *sylvestris*	0.53	0.01
	Sherardia arvensis	0.45	0.05
S_LNW	*Quercus suber*	0.69	0.00
V_LNW	*Torilis africana*	0.56	0.00
S_HNW	*Phillyrea latifolia*	0.68	0.00
	Lonicera japonica	0.53	0.01
	Euphorbia amygdaloides	0.45	0.04
	Quercus virgiliana	0.45	0.04
V_HNW	*Cyclamen hederifolium* subsp. *hederifolium*	0.71	0.00
	Polypodium interjectum	0.50	0.02
	Anthriscus nemorosa	0.45	0.03
	Asplenium trichomanes subsp. *quadrivalens*	0.45	0.04
S_ALW+V_ALW	*Robinia pseudoacacia*	0.68	0.00
	Ligustrum lucidum	0.66	0.00
	Urtica membranacea	0.64	0.01
	Arum italicum subsp. *italicum*	0.51	0.04
	Ailanthus altissima	0.51	0.02
	Anisantha sterilis	0.50	0.02
	Bryonia dioica	0.47	0.04
	Fumaria capreolata subsp. *capreolata*	0.46	0.05
	Oxalis stricta	0.45	0.04
V_ALW+V_LNW	*Ulmus minor* subsp. *minor*	0.59	0.02

	Avena barbata	0.47	0.03
V_ALW+V_HNW	*Viburnum tinus* subsp. *tinus*	0.51	0.03
S_LNW+V_LNW	*Quercus pubescens* subsp. *pubescens*	0.54	0.02
	Prunus spinosa subsp. *spinosa*	0.53	0.01
V_LNW+S_HNW	*Quercus robur* subsp. *robur*	0.62	0.01
V_LNW+V_HNW	*Quercus ilex*	0.64	0.00
	Acer campestre	0.56	0.02
S_HNW+V_HNW	*Ruscus aculeatus*	0.92	0.00
	Fraxinus ornus subsp. *ornus*	0.80	0.00
	Asplenium onopteris	0.73	0.00
	Stellaria media subsp. *media*	0.69	0.00
	Alliaria petiolata	0.68	0.00
	Smilax aspera	0.65	0.00
	Quercus cerris	0.58	0.03
	Cornus mas	0.58	0.01
	Ranunculus lanuginosus	0.55	0.01
	Anemone apennina	0.54	0.01
	Ligustrum vulgare	0.53	0.02
	Luzula forsteri	0.52	0.03

General Additive Model results

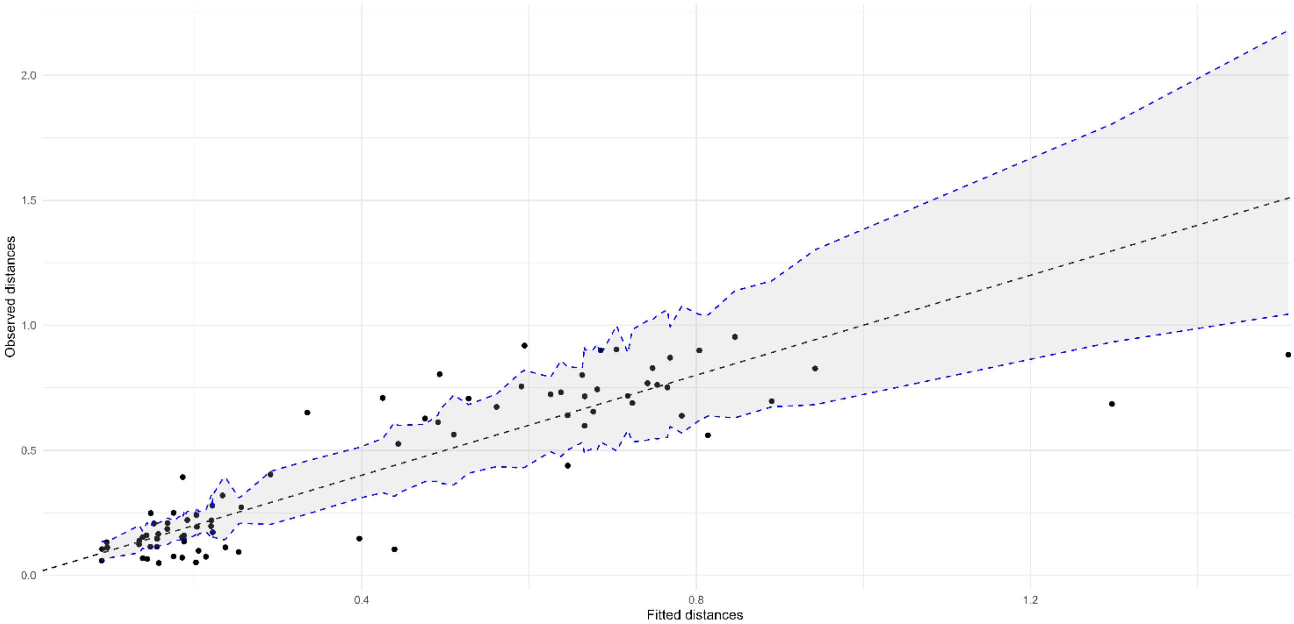


Figure S.1: Observed and fitted distances from PNV centroids. The dashed black line represents the1:1 relationship. The grey shaded area indicates the 95% confidence interval around the estimated mean response.

Appendices III: Green infrastructure design for the containment of biological invasions. Insights from a peri-urban case study in Rome, Italy

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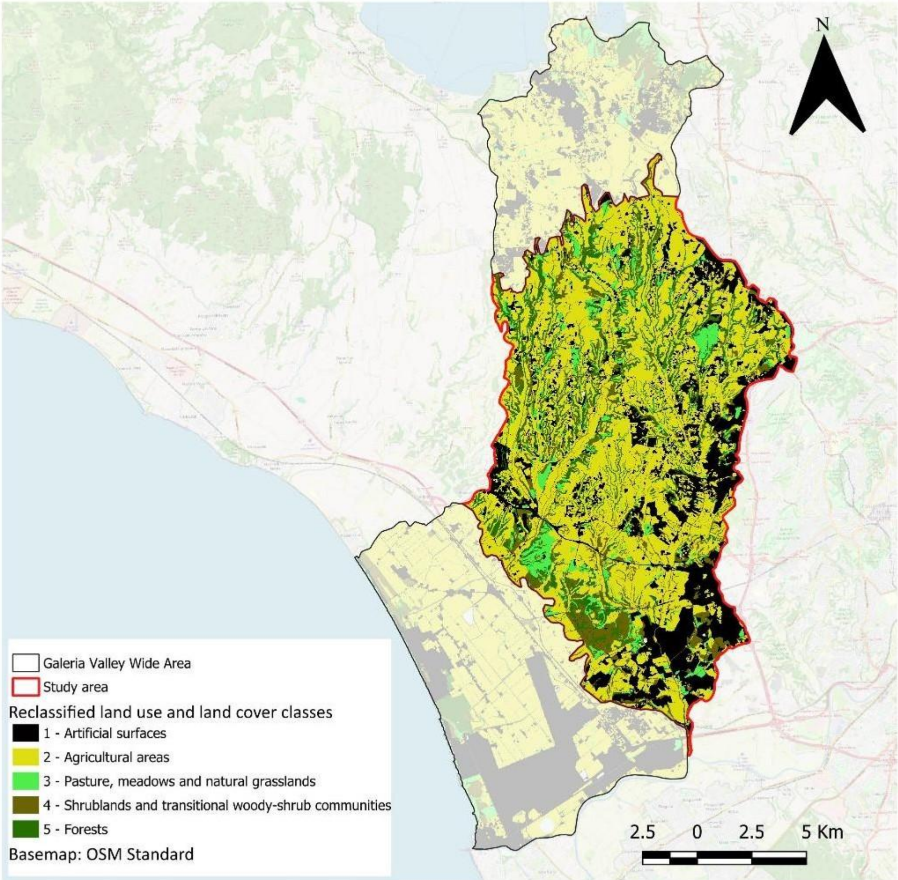
Supplementary material Figure S.1:



Supplementary material

Figure 1: raster version of the Land cover and land use map of the Galeria Valley simplified into 5 main classes:
1 - Artificial surfaces
2 - Agricultural areas
3 - Pastures, meadows and natural grasslands
4 - Shrublands and transitional woody-shrub communities
5 - Forests

Supplementary Table S.1: Supplementary Table 1.xlsx - Fogli Google