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Original article

How far from naturalness: effects of anthropogenic landforms on spontaneous woodland vegetation in an urban ecosystem (Capital City of Rome, Italy)

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ABSTRACT

Urban landscapes in historical cities often result from long-standing stratification of human activities, which also affected natural morphology and, in turn, ecological settings and ecosystem dynamics. However, the specific impact of geomorphological alterations on woodland successional stages has remained poorly explored. This study aims at filling this gap, concomitantly providing useful knowledge for practical ecological restoration. Moving from the identification and mapping of anthropogenic landforms in Rome, the intensity of eventual disruption in spontaneous woodland dynamics was assessed across a disturbance gradient with respect to Potential Natural Vegetation (PNV) types, adopted as local reference systems. Vegetation was surveyed both inside (altered) and outside (low and high naturalness controls) anthropogenic landforms, and site-level shifts due to geomorphological alteration and/or alternative human disturbances were qualified and quantified. 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 from the urban context. Differences between woodland types were thus quantified, in terms of ecological distances from PNV, modelled and mapped across the city by 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 useful tools to inform ecological restoration in cities.

1. Introduction

As the global population increasingly inhabits cities, biodiversity in urban areas becomes a crucial element to be conserved and restored (CBD, 2022; Grimm et al., 2008). Urbanisation actually exerts negative impacts on sustainability and resilience of cities by affecting species survival, ecological functions, ecosystem service capacity, and human well-being (Adla et al., 2022). Protecting residual natural areas would not be sufficient to counterbalance these pressures, and a paradigm shift toward active enhancement and renaturalisation of human-modified habitats is urgently needed (EC, 2020; 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. Key NRR targets include No Net Loss of urban green spaces and tree canopy cover by 2030, as compared to 2024 levels, and a subsequent increase in their share from 2031 onward. These targets require effective planning and management practices, including the design of green infrastructure and implementation of restoration actions and nature-based solutions (EC, 2020). Bringing back nature to cities should thus be supported by scientific knowledge able to ensure the selection of the most suitable species and the establishment of self-sustaining ecosystems in the medium and long term (Kraft et al., 2014; Blasi et al., 2025). In particular, local Potential Natural Vegetation (PNV), i.e. the mature vegetation that could develop if human

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disturbance is removed (Tuxen, 1956), represents a proper reference model (Nelson et al., 2024) for reforestation and tree planting, successfully adopted also in urban areas (Miyawaki, 1998). Notwithstanding the recognition of PNV may be uncertain in strongly altered sites (Zerbe, 1998), scientific advances on ecological classification of land (Blasi et al., 2005; Montaldi et al., 2026) , along with the evidence of positive effects of near-naturalness on the stability, self-regulation capacity and biodiversity of urban green spaces (Dong et al., 2025), are making PNV baselines increasingly important for sustainable planning and management in urban ecosystems.

Still, especially within and around cities, several factors may lead to a deviation from the original PNV, with land uses and habitat changes that not only alter the composition, structure and functions of actual ecosystems, but also their dynamics (Sanderson et al., 2002). For example, the extent of residual natural systems determines the pool of species available, the ecological connectivity affects their dispersal (Valeri et al., 2021), the artificialisation facilitates the introduction and abundance of exotic and invasive species (Boscutti et al., 2022), and land use and management legacy is able to drive the direction of spontaneous renaturalisation in disturbed green spaces (McLauchlan, 2006; Bart & Davenport, 2015). This direction can thus be determined by local species extinctions, alterations of biogeochemical cycles and introduction of exotic species. Sometimes the result is the assemblage of "novel" or "emerging" communities, which can represent long-standing woodland stages (Hobbs et al., 2006), more or less distant from local PNV. However, while the effect of land use/cover changes was well-documented, and despite humans may act as a genuine physical process that determines persistent topographic changes in cities (Somodi et al., 2021), the shift due to anthropogenic alteration of geomorphology was poorly investigated. Anthropogenic geomorphological processes, like accumulation (valley filling, slope terracing, dumps, artificial hills) and erosion (excavation, mining, extraction activities), can overtake natural ones by re-shaping original landforms or entirely creating artificial ones (Szabó et al., 2010). These processes rapidly affect slope shape, drainage patterns, type of outcropping substratum (Mandarino et al., 2021; Pica et al., 2024), and edaphic conditions (La Vigna et al., 2016; Delchiaro et al., 2025), which in turn accommodate late-successional vegetation types that differ from local potential ones.

Accordingly, the present research was aimed at qualifying and quantifying the impacts of human-induced geomorphological alterations on PNV at the site level. The Municipality of Rome Capital was selected as a case study due to the historical and spatially widespread geomorphological alteration. An in-depth understanding of these impacts could inform effective restoration planning of self-sustaining ecosystems in the area, as well as assist in the fulfilment of current strategic and policy requirements, also in other cities.

2. Materials and methods

2.1. Study area

The municipality of Rome Capital extends over 1287 km^2 and has a population of 2.9 million inhabitants (Roma Capitale, 2022). The area has a notable ecosystem heterogeneity due to combined environmental and anthropogenic determinant factors (Celesti-Grapow et al., 2013; Blasi et al., 2005). The biogeographic location in the central Mediterranean Basin, the transitional bioclimatic and orographic setting between Tyrrhenian coasts and Apennine mountains, and a geomorphology characterised by wide volcanic plateaux, sedimentary hills and coastal plains, and the alluvial plain of the Tiber River (Funiciello and Giordano, 2008), together with long-lasting human activities determined an outstanding variety of occurring species and communities (Capotorti et al., 2013).

In synthesis, and according to an ecological classification based on distinctive combinations of natural bioclimatic, lithological and morphological features, the municipal landscape can be arranged into 16 Environmental Land Units (ELU), which support the development of as many types of local PNV (Blasi et al., 2005). Volcanic slopes and plateaux with *Q. cerris* forests and Sandy hills with *Q. pubescens* and *Q. suber* forests are the most widespread ELU/PNV types and have been considered for this research (Fig. 1). With respect to this background framework, three millennia of history transformed the abiotic environment with anthropogenic morphogenesis (Del Monte et al., 2016), so that Rome represents an illustrative example of how, in urban and peri-urban areas, natural and human altered/created landforms coexist (Burnelli et al., 2025). Moreover, a huge urban development occurred during the last 70 years and notably affected actual land cover and remnants of (semi-) natural vegetation (Frondoni et al., 2011; Fig. 1).

2.2. Methodological framework

Overall arrangement of the different phases of the research is shown in Fig. 2 and more specifically described in the following sections.

2.2.1. Sampling design and data collection

Sites for vegetation and soil surveys were selected from wooded areas, with an extent of at least 20 m× 20 m, in the two most widespread ELU of the Municipality. Wooded areas were recognised in a GIS environment (QGIS, version 3.34.4) by overlapping the Real Vegetation Map of the Province of Rome (CIRBFEP, 2013) with the European Forest Type and Small Woody Features high-resolution layers (EEA, 2018, 2021). Among this initial pool, three different woodland types were distinguished according to anthropogenic alteration of the geomorphology, surrounding landscape mosaic features, degree of naturalness, and importance for landscape ecological connectivity:

- Woodlands occurring on Anthropogenic Landforms (ALW), as identified following the methodological approach proposed by Del Monte et al. (2016) for urban geomorphological survey and mapping. These landforms represent human-driven landscape transformations and were mapped through the analysis of multi-temporal data (including historical topography, orthophotos, satellite imagery, and geognostic data), subsequently integrated with ancillary sources (such as archaeological reports, historical archives, and iconographical/artistic materials) and field observation (Supplementary material A). The resulting anthropogenic landform types, such as fills, terraced slopes and quarries, were organised into a spatial dataset that also records the period of artificialisation for each site (i.e. the time interval during which the alteration occurred).
- Control Woodlands with Low Naturalness (LNW), located in landscape contexts similar to ALW as for green space extent and fragmentation, but on sites without geomorphological alteration. Landscape context attributes were retrieved from a classification of 1 km x 1 km grid cells, proposed for biodiversity investigation across Italian urban systems (Dondina et al., 2025).
- Control Woodlands with High Naturalness (HNW), located in the core areas of mature-stage forests, with a crucial role for functional ecological connectivity (important nodes). Mature-stage forest patches were retrieved from the Real Vegetation Map, while respective core areas were identified by a Morphological Spatial Pattern Analysis, performed with the GUIDOS toolbox (Vogt et al., 2022). Among the same patches, important ecological network nodes were identified with Conefor 2.6, by calculating the percentage reduction in the Probability of Connectivity index (PC) due to node removal (Saura and Pascual-Hortal, 2007). The PC was calculated for the entire municipal territory using a threshold dispersal distance of 115 m, corresponding to the average dispersal capacity of dominant mature-stage species in the study area (i.e. *Quercus* spp.; Kurek et al., 2019).

The three woodland types were expected to guarantee a representative sample for the investigation of eventual shift in late successional

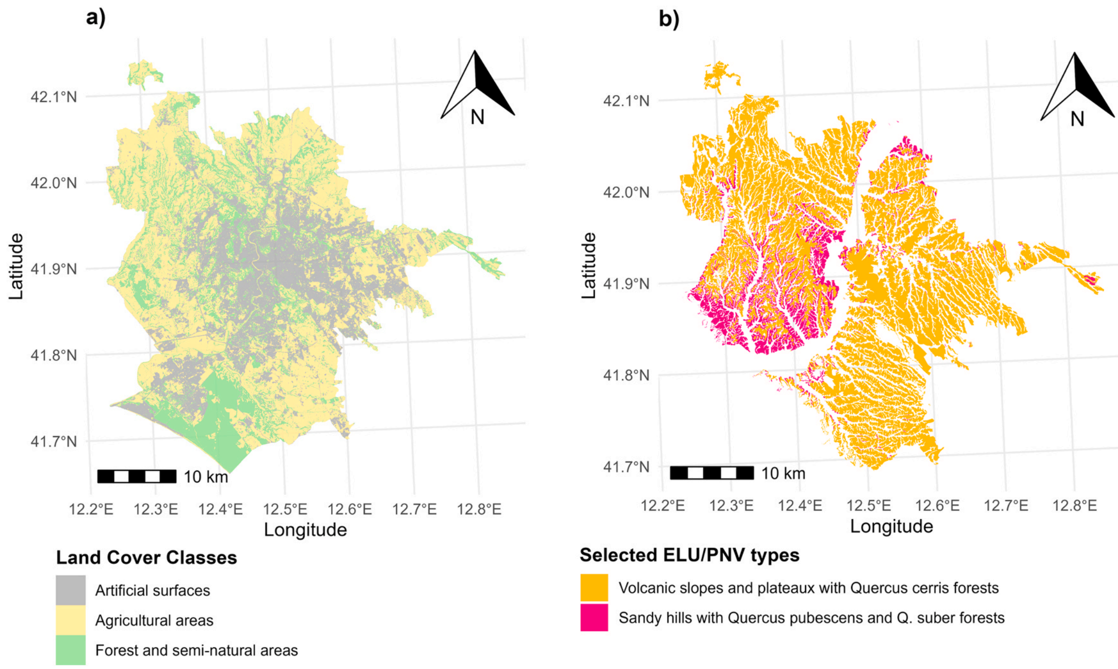

Fig. 1. Main land cover types in the Municipality of Rome Capital (a), reclassified from the Real Vegetation Map of the province of Rome and Environmental Land Unit/Potential Natural Vegetation (ELU/PNV) types selected for the research (b).

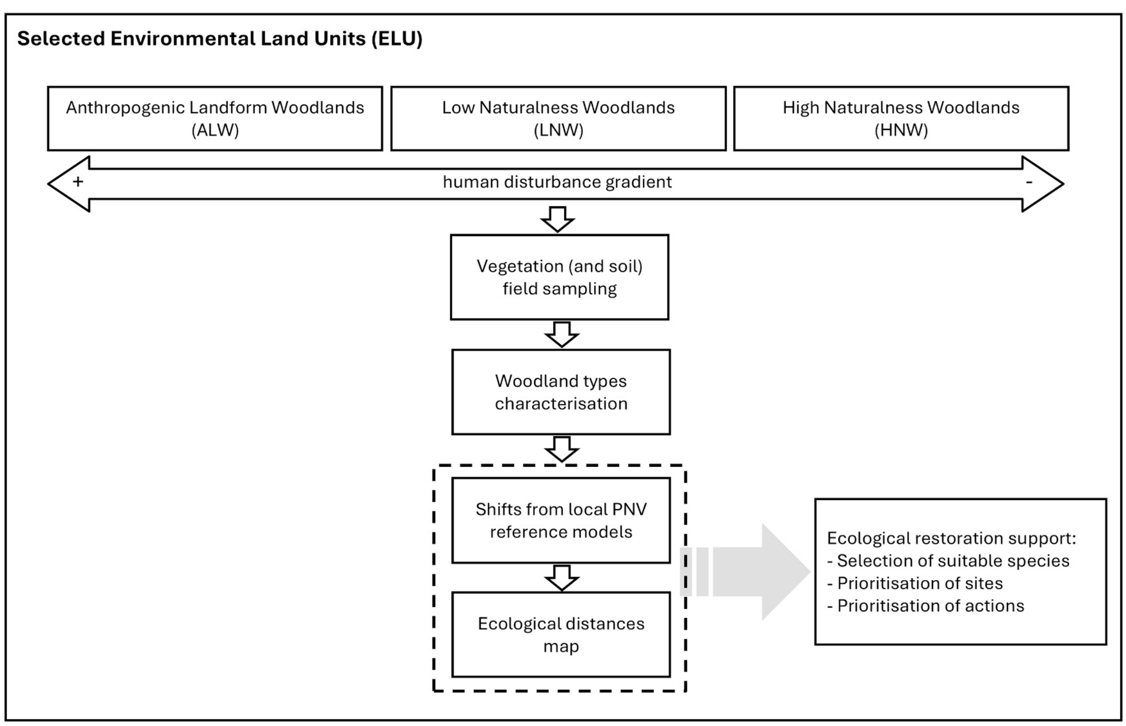

Fig. 2. Research workflow and potential application of the outcomes in urban restoration planning and design (bottom right corner).

stages due to geomorphological alteration (ALW), with respect to other disturbances related to the urban context (LNW) and to near-natural conditions (HNW). Plantations and artificial green urban areas, as reported in the source Real Vegetation Map, were excluded from the analysis, as they are supposed to fall under heavy human control of vegetation composition and natural processes. The spontaneous origin and dynamics of sampled woodlands was in any case assessed by cross-checking the information from the source Real Vegetation Map with direct field observation on eventual disrupting management actions. Thus, only woodlands shaped by multiple tree species, organised in multi-layered and uneven stands, without recent mowing, biomass removal or tree planting signs were surveyed. Surveys were performed on 10 m x 10 m plots, at least 10 m apart from woodland edges, and divided into four quadrants of 5 m x 5 m. All occurring plants were listed for each quadrant, and additional variables were retrieved/recorded for each plot. Environmental variables included: mean monthly temperature and precipitation for the period 2011–2020 at a resolution of 1 km x 1 km (Braca et al., 2021); lithological type (Regione Lazio, 2012);

elevation, derived from a 5 m Digital Elevation Model (DEM) (Regione Lazio, 2002; Hutchinson et al., 2011); heatload exposition index (McCune and Keon, 2002); slope and geographic coordinates (measured on field). Geomorphological alteration variables included: An_landform, reporting anthropogenic or natural landforms; Landform type (Szabó et al., 2010), indicating 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 surfaces at three different radii from plots (250, 500, and 1000 m); overall patch area containing the sampled plot and its Fractal dimension index, Shape-area and Perimeter-area ratio (McGarigal, 2015); age of vegetation cover since 1950, 1980, or 2001, based on multi-temporal land cover data (Frondoni et al., 2011). For a representative subset of woodland plots within a same ELU, ranging from HNW and LNW to some ALW across a range of different age of vegetation cover, top soils (30 cm) were also sampled according to the Lucas method (EC, 2022). Samples were subsequently analysed as for texture, pH, electrical conductivity, total calcium carbonate, active calcium carbonate, organic matter, total nitrogen, phosphorus, calcium, magnesium, potassium, sodium, C.E.C, base saturation, and C/N ratio. Resulting parameters were thus graphically represented for a preliminary interpretation of the relationships with 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 (IndVal index) per pooled groups, along with α- and β-diversity, chorological types, and life forms (Borcard et al., 2011; Midolo et al., 2024; Abbate et al., 2012). A Redundancy Analysis (RDA) was thus performed on Hellinger-transformed indicator species data to test the effect of anthropogenic landforms on woodland composition and explore the relationships with environmental and landscape features, considered as explanatory variables (Bisigato et al., 2016). A stepwise procedure was adopted to select significant variables, while a Variance Inflation Factor (VIF) analysis was performed to evaluate multicollinearity and remove highly correlated ones. Specifically, the effect of anthropogenic landforms was assessed by removing their contribution from the model and observing the decrease in total explained variance (Borcard et al., 2011).

To quantitatively measure the ecological distance between 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, Principal Components scores of HNW, for volcanic and sandy ELU separately, were averaged to generate reference PNV centroids and the Euclidean distance from them was calculated for each survey. Third, a Generalised Additive Model (GAM) was fitted to explain these distances according to environmental 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 30 m.

All the analyses were performed with RStudio (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*

A total of 80 woodlands were surveyed among ALW (20), LNW (20), and HNW (40), equally distributed between sandy and volcanic ELU (Fig. 3). Comprehensive species list and environmental and landscape variables for each survey are provided in Supplementary material B, while the degree of node importance and the spatial arrangement of core areas that guided the selection of HNW are provided in Supplementary material A. Soils were sampled in sandy ELU, from one HNW patch, one LNW patch, and eight woodlands on anthropogenic landforms (Fig. 3).

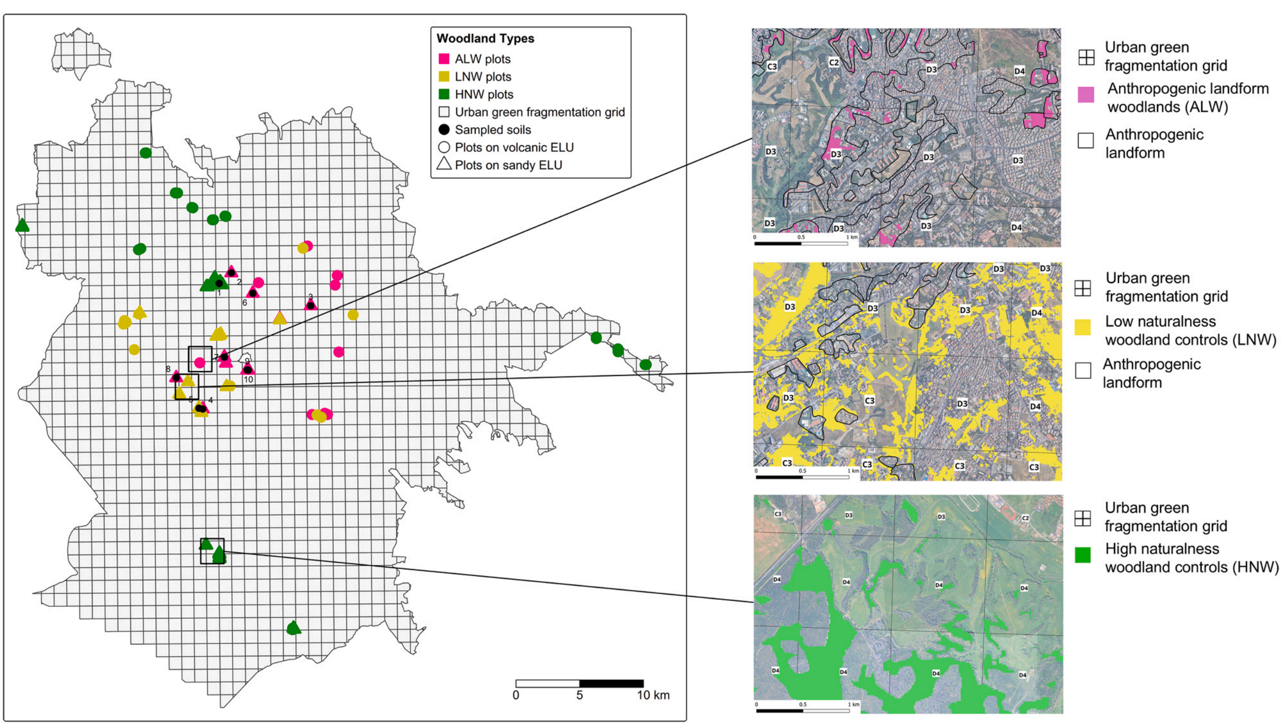

Fig. 3. Left side: sampled sites in the study area, represented upon the 1 km x 1 km reference grid. Right side: illustrative satellite views of woodland types, along with surrounding landscape mosaic and grid cell categories (coded from A to D according to increasing extent of green spaces, and from 1 to 4 according to decreasing degree of fragmentation; Dondina et al., 2025).

3.1.1. Anthropogenic landform characteristics

For the 20 ALW sampled sites, anthropogenic landforms resulted from three main processes (Table 1):

A. Accumulation of deposits and techno-soils due to human activities, such as filling surfaces, embankments, and ridges (often created for motorways and railways), with heterogeneous mixes of materials (artefacts, broken brick, concrete, reworked or redeposited natural sediment, and other debris). The resulting layer adds significant thickness, thus modifying local outcrop lithology and associated soils.

E. Anthropogenic erosion, involving large-scale removal of terrain due to e.g. industrial building excavations, quarrying/mining, and surface collapse over underground artificial cavities. These movements modify the type and continuity of outcrop lithology and can significantly alter the distance to the water table.

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

All these transformations 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 volcanic and sandy ELU.

3.1.2. Indicator species

Indicator species analysis (Table 2) revealed that physiognomic trees of local PNV, such as *Quercus cerris* and *Q. ilex* for volcanic, and *Q. suber* and *Q. pubescens* for sandy ELU, are typical for both LNW and LNW&HNW, along with many nemoral species. HNW resulted to be almost exclusively characterised by native species, with the only exception of *Lonicera japonica* occurring in the plots closer to the city centre. LNW were distinguished from HNW by ruderal and earlier successional shrubs and herbs, such as *Prunus spinosa*, *Torilis africana, Silene latifolia*, *Avena barbata* and *Hordeum murinum*. Among indicator species of ALW instead, no physiognomic trees of local PNV occurred, but many exotic and often invasive trees and shrubs, such as *Robinia pseudoacacia, Ailanthus altissima, Yucca gloriosa, Ligustrum lucidum,* followed by ruderal and nitrophilous herbs, such as *Parietaria judaica, Euphorbia peplus* and *Oxalis stricta.* Moreover, HNW and ALW, at the two ends of the disturbance scale, showed no indicator species in common. Results per distinct ELU are reported in the Supplementary material C.

Table 1
Characterisation of anthropogenic landform types for surveyed ALW. Process type: E = erosion landforms; A = constructional landforms; M = mixed erosion/constructional landforms. Artificialisation period: time span of morphological transformation occurrence, derived from multitemporal data (see details in the text and in Supplementary material A).

Process type	Landform type	Area [ha]	Artificialisation period
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

Table 2
IndVal multipatt analysis for ALW, LNW, and HNW types.

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

3.1.3. Chorological types and life forms

Chorological types (Fig. 4) in HNW and LNW showed a great occurrence of Atlantic, Mediterranean and Paleotemperate species. Both LNW and ALW were mainly characterised instead by a high proportion of exotic (see also Supplementary material C) and Turanian-Mediterranean species. Woodlands of the same type did not significantly differ between the two ELU, except for fewer Steno-

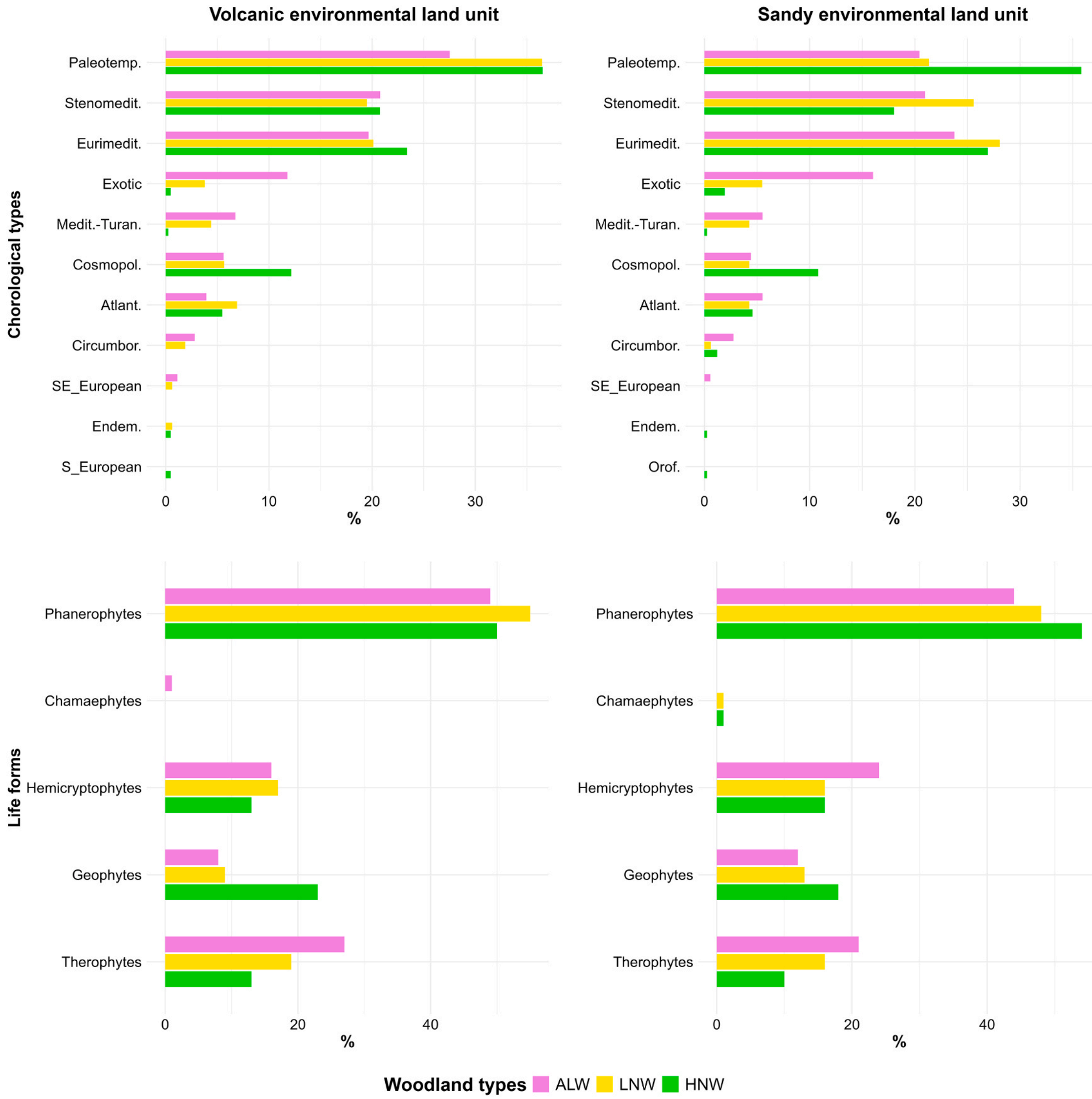

Fig. 4. Chorological types and life forms per woodland types on volcanic (left side) and sandy ELU (right side).

Mediterranean species in HNW than in LNW and ALW on sandy substrata. Life forms (Fig. 4) showed a high proportion of geophytes in HNW and of therophytes in LNW and ALW, with no marked differences between the ELU except for geophytes (more frequent on sandy than on volcanic substrata). The proportion of therophytes was generally lower across all woodland types on sandy than on volcanic substrata.

3.1.4. *Diversity metrics*

The α-diversity, measured as species richness (SR), evenness (EV) and Shannon diversity (SH), was higher for HNW in volcanic (SR: 22.00 ± 5.53; EV: 0.94 ± 0.03; SH: 2.87 ± 0.25) and sandy ELU (SR: 21.80 ± 5.17; EV: 0.80 ± 0.04; SH: 2.46 ± 0.29), followed by ALW but with a greater variability (SR: 19.00 ± 7.48, EV: 00.65 ± 0.08, SH: 1.89 ± 0.48 *vs* SR: 18.90 ± 7.98, EV: 0.60 ± 0.12, SH=1.73 ± 0.56), and LNW (SR: 16.90 ± 4.98, EV: 0.59 ± 0.07, SH: 1.67 ± 0.33 *vs* SR: 16.40 ± 6.38, EV: 0.59 ± 0.07, SH: 1.64 ± 0.41) (see also Supplementary material C). On the other hand, β-diversity denoted a higher species turnover in LNW, with no significant differences between the ELU (13.43 *vs* 13.84), than in ALW (11.94 *vs* 12.01) and HNW (10.31 *vs* 10.41).

3.2. *Ecological distance from local PNV in altered and not altered sites*

3.2.1. *Effects of anthropogenic landforms and other landscape/environmental variables on woodland successional stages*

A recognisable effect of anthropogenic landforms on woodland stages was confirmed by the RDA model, which explained 32% of the constrained variance (adjusted $R^2 = 0.24$). When the *An_landform* variable was removed from the model, the adjusted R^2 decreased to 0.14, with a 42% loss of the overall explained variance. The Global test of significance (P-value <0.05) rejected the null hypothesis according to which no relationships exist between response data (species composition) and explanatory variables, so that the latter represent meaningful drivers of community assembly. The RDA biplot confirmed a disturbance range among the woodland types, which was especially evident from the first/third axes biplot (Fig. 5). The first axis is characterised by landscape (*PLAND_art_1000*), legacy (*Veg_age*), perimeter-area ratio (*PARA*), along with environmental (precipitation and temperature), and geomorphological variables (*An_landform*). Higher values of artificial land cover and perimeter-area ratio, as well as lower values of forest cover and of *Veg_age*, are associated with LNW and ALW. The only

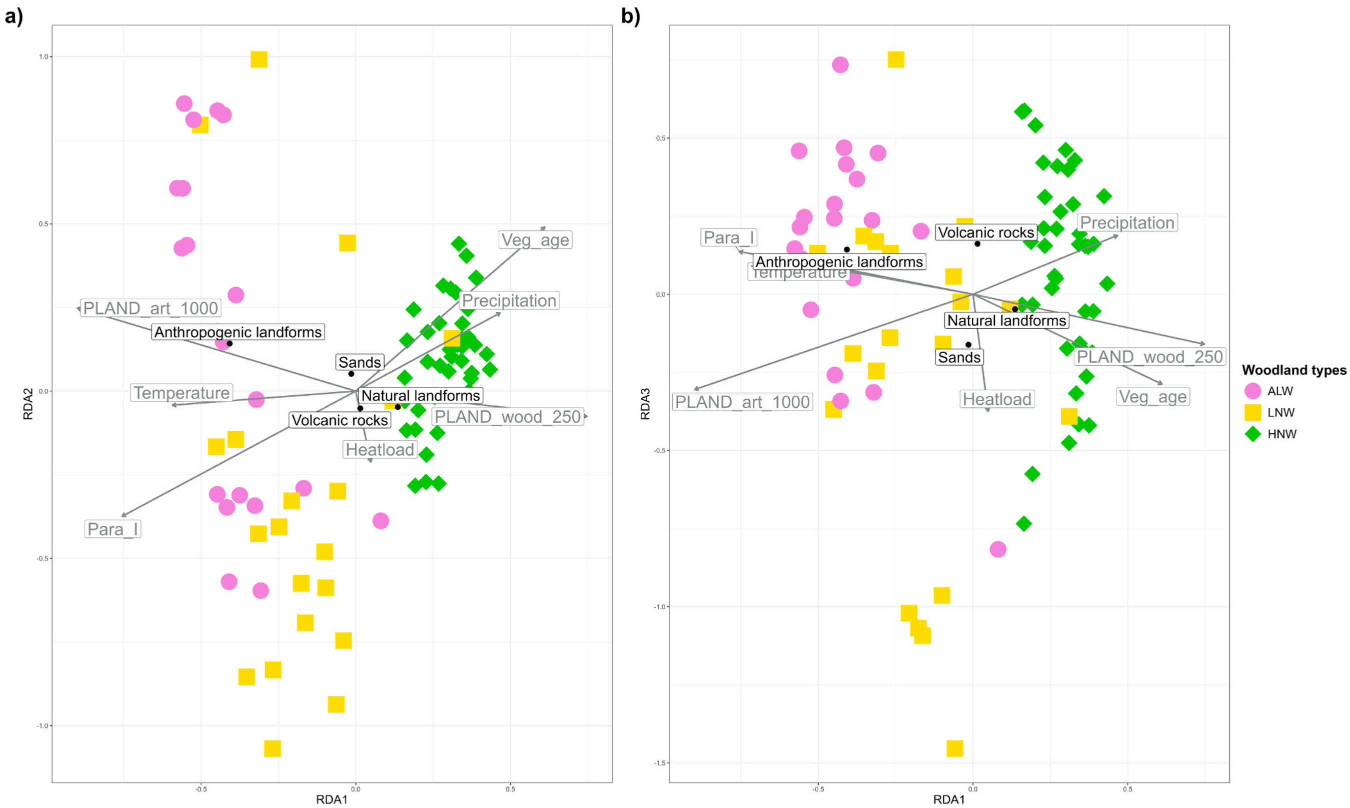

Fig. 5. RDA biplot with landscape/environmental variables (vectors for quantitative and points for qualitative variables) over the first and second (a) and first and third axes (b).

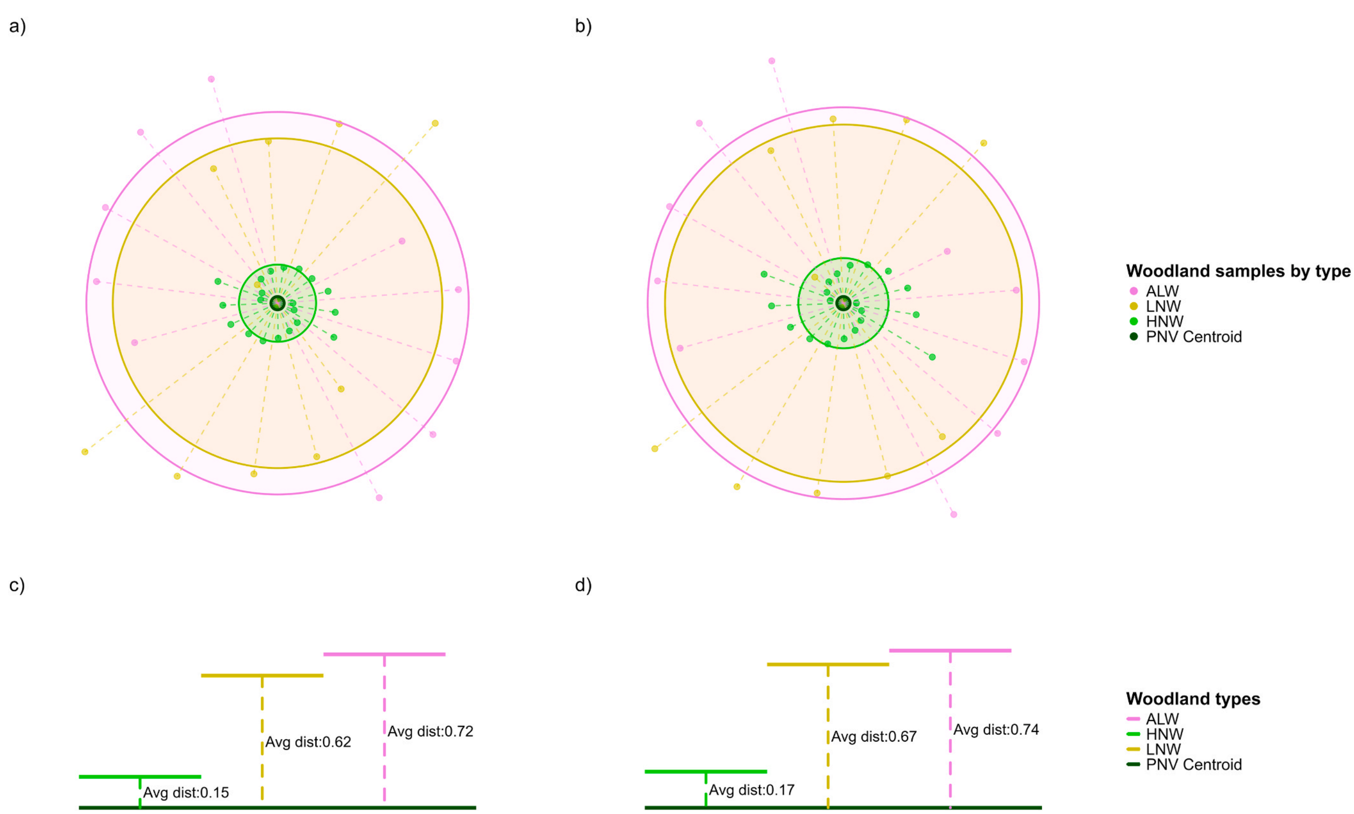

Fig. 6. Radial plots for volcanic ELU (a) and sandy ELU (b), showing the ecological distance (adimensional) between each sampled woodland and respective PNV centroids, with average distances represented as circles around the centroids. Linear plots for volcanic ELU (c) and sandy ELU (d), showing the average distance per woodland type from the reference baseline.

variable that seems to distinguish the ALW from LNW is *An_landform*, with ALW distributed around the "Anthropogenic landform" class. The second and third axis are instead mainly characterised by environmental variables alone, such as Heatload (representing the effect of topographic aspect) and lithology, with sands and volcanic substrata resembling the effects of the two ELU.

3.2.2. Measurement and spatial modelling of ecological distances from reference PNV

Ecological distances of sampled woodlands from PNV centroids, adopted as reference models for each ELU, are shown in Fig. 6(a and b). The average ecological distance from the reference baseline is also exemplified in Fig. 6(c and d). Both woodland types, more associated with the urban context, i.e. LNW and ANW, resulted in being 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, discernible and more marked for volcanic than for sandy ELU.

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*, modelled 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 explained 74.0% of the deviance (pseudo $R^2 = 0.67$) and indicated the years of naturalisation lead to a decrease in distance from the reference model, while the proportion of surrounding artificial land cover increased with ecological distances (Supplementary material C). Extended to the overall investigated ELU, the prediction allowed the expected ecological distance from PNV across the city to be shown, independently of actual land cover (Fig. 7). High ecological distances were primarily predicted at locations with recent naturalisation and in very artificialised contexts, especially in the city centre. Conversely, low ecological distances were modelled for residual natural and agricultural zones, especially in suburban sectors. These areas consistently exhibited low artificialisation of the surrounding mosaic, though the age of vegetation cover may vary, so that agricultural surfaces were also estimated to maintain moderate distances from natural baselines.

3.3. Soil characteristics

Soil properties for woodland plots in sandy ELU are shown in Fig. 8. Notwithstanding earthworks, almost all the soils in altered sites were found to be either sandy or sandy-loam as less disturbed ones (samples 1

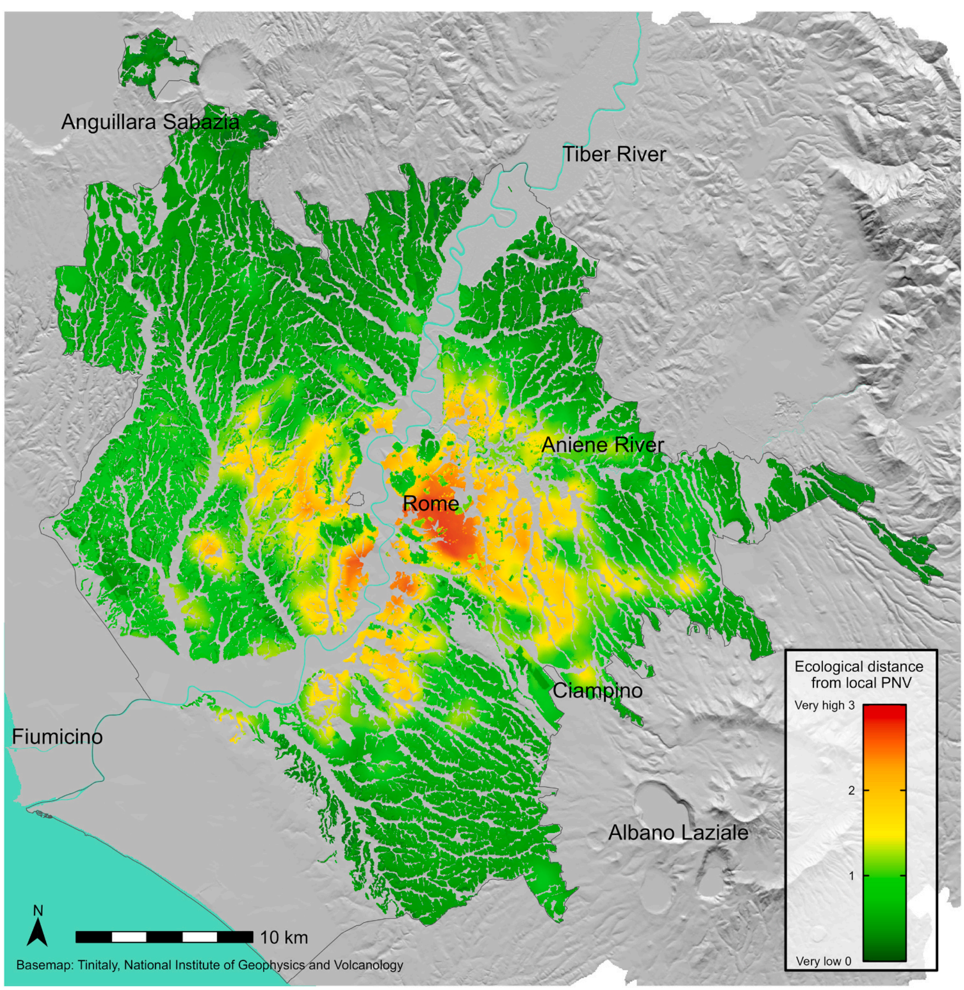

Fig. 7. Map of ecological distances (adimensional) from reference PNV in considered ELU. Independently from actual land cover, high values represent areas where eventual woodland successional stages are estimated to diverge most from local PNV, while low values correspond to higher estimated consistency with reference models.

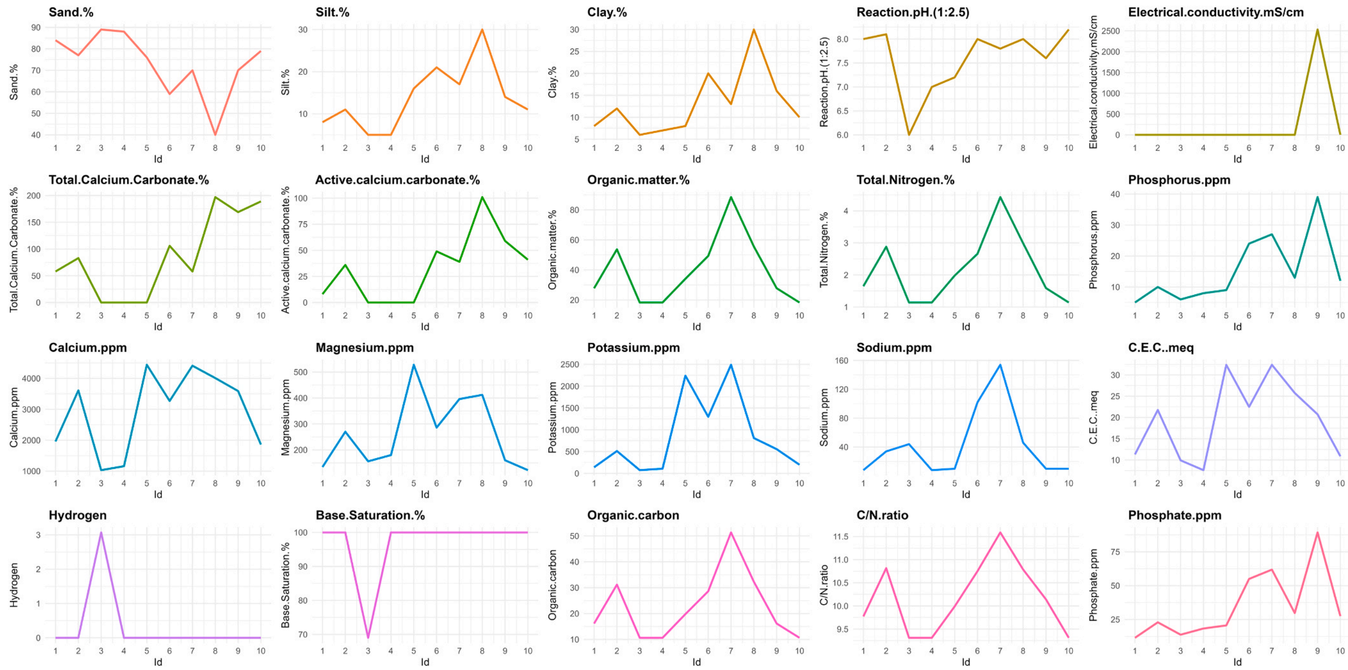

Fig. 8. Chemical and physical characteristics of soils underpinning HNW, LNW and ALW in sandy ELU (Id = sample number).

and 2), except for one sample (8) with high clay and silt proportion.

Less disturbed soils presented similar physical characteristics, even if nitrogen and organic matter were higher in LNW (sample 2) than in HNW (1). ALW soils showed marked internal variability, with some samples (3, 4, and 10) more similar to those collected in HNW, especially for physical characteristics. The remaining five samples showed quite different physico-chemical characteristics, with more clay and silt, as well as higher levels of organic matter, nitrogen, and phosphorus, probably due to fertiliser residue. Furthermore, two samples (5 and 7) stood out with high potassium levels, indicating probable contamination with volcanic soils. Another sample (9) exhibited high electrical conductivity by soluble anions, mainly sulfates (90%), due to high levels of gypsum and to probable construction wastes used as filling materials.

4. Discussion

Urban ecology is gaining significant momentum, with an increasing number of studies on biodiversity at different levels of complexity (Rega-Brodsky et al., 2022). The impact of land-use changes, habitat fragmentation, and biological invasions has been extensively documented on species within and around cities (Celesti-Grapow et al., 2006; Haddad et al., 2015). Many studies also focused on vegetation composition and condition along rural-urban gradients, but often lacked quali-quantitative assessment of alteration degree (Malkinson et al., 2018; Knapp et al., 2022). The present study contributed to understanding how and to what extent anthropogenic landform alteration affects actual woodlands with respect to local present PNV (*sensu* Somodi et al., 2021). Evidence confirmed that deep alterations of the substratum determine wider shifts from the potential natural state than other urban context disturbances alone. These differences have been qualified in terms of species composition and diversity along a disturbance gradient, and the ecological distance from PNV has been quantified and spatialised according to associated landscape/environmental variables.

Indicator species, chorological types, life forms, and α- and β-diversity revealed a clear differentiation among woodland types. As for more natural woodlands, selected among the core areas of most connected nodes, the good alignment with PNV models proved that node importance metric for identifying well-conserved forest patches is useful even in an urban context, and confirmed that ecological connectivity plays an important role in maintaining ecosystem integrity (Pascual-Hortal & Saura, 2008). Woodlands with a lower level of naturalness exhibited instead a more pronounced legacy from previous agricultural uses, with high rates of ruderal and light-demanding species typical of abandoned fields (Prach et al., 2001). Being more isolated than near-natural woodlands, these plant communities also showed a lower α-diversity, probably due to local extinctions (Ramalho et al., 2014). Nevertheless, some PNV typical species have still been found, suggesting the capacity to recover spontaneously over time has been maintained (Motta et al., 2009). On the contrary, PNV indicator species were never found in anthropogenic landform woodlands, likely due to combined effect of local extinction and long-term removal of the seed bank, resulting from landscape configuration and site alteration, respectively. In this case, the higher observed α-diversity was due to the entry of exotic species, typical of urbanised contexts (Boscutti et al., 2022), but also to the frequency of Mediterranean-Turanian species, facilitated by light availability, due to more open canopy cover and less complex structure of disturbed woodlands (Williams et al., 2015). Despite, HNW was mainly characterised by PNV and native species, also one exotic ones, *Lonicera japonica*, appeared as an indicator species due to the presence in sandy woodlands, more pronounced inside the 'GRA/metropolitan area', sign of the wide effect of Rome urban landscape. Consistently with previous research (Williams et al., 2005; Knapp et al., 2008), both disturbed woodland types showed few geophytes and many therophytes with respect to near-natural ones. The shift was, however, more pronounced on modified landforms, probably due to soil alteration, subsequent removal of subterranean geophyte regenerative structures (rhizomes or bulbs), and indirect facilitation of therophytes. Altogether, these indicators suggest passive recovery may not represent an effective solution for forest restoration on anthropogenic landforms, taking also into account the possibility of permanent successional diversion toward novel stable communities (Kowarik and von der Lippe, 2018), and hindered development capacity of root systems for native trees compared to more competitive and ruderal exotics (Gillini et al., 2025). The effects of different degrees of disturbance were confirmed by the RDA, whose reliability was in line with other multivariate ecological studies (Bisigato et al., 2016; Solé-Senan et al., 2018), and by the ecological distance from reference PNV, with compositional shifts due to site-level geomorphological alteration plainly discerned. Moreover, local environmental features, modelled by ELU, proved to play a definite role in the magnitude of the shift, with volcanic substrata more affected than sandy ones. Such preliminary evidence should, however, be further tested with additional samples per different extent/magnitude of landform alteration and per different environmental units, such as alluvial valleys and coastal plains.

Combined landscape-level and site-level effects on distance from PNV, resumed by the GAM, highlighted the positive importance of vegetation cover age, which probably relates to recovery time of soils after anthropogenic interventions (Bart and Devenport, 2015). On the other hand, surrounding artificial cover was confirmed as detrimental, often due to propagule pressure by exotic species (Heinrichs and Pauchard, 2015) and limitation in the dispersal of native ones (Lopez et al., 2018). Notably, results allowed the percentage of artificial cover to be recognised as a reliable proxy for anthropogenic landform alteration, contributing to viable spatial modelling of its effect on ecological distances from PNV. The same proxy has been used for calculating and interpreting the urban heat island (UHI) intensity in the city (Cecilia et al., 2023), so that the two phenomena widely overlap especially within the inner urban center. The eventual combined effect between landform alteration and UHI on local PNV could be explored with future research, even though in a transitional bioclimatic context (such as the sub-Mediterranean one of the investigated area) a few degree rise in mean temperatures could likely impact on the relative proportion between more and less thermophilous trees (e.g. facilitating evergreen with respect to deciduous oaks), without necessarily deviating from the current and already mixed natural models (Frigerio et al., 2023). Mapping of expected shifts from PNV baselines also provided a contribution to the investigation of the urban-rural gradient and its determinism on ecosystem condition (Kaminski et al., 2021). Practical implications include a better understanding of the deviation from reference models (Blasi et al., 2025) and a more indicative capacity on where passive (spontaneous recovery) *vs* active (replacement and forestation) woodland restoration should be pursued in the city. Moreover, this knowledge may effectively support site-level selection of suitable species for urban forestry and tree planting, allowing a better informed use of commonly adopted and plastic species (e.g. *Quercus ilex*) even outside their strict natural potential distribution, or of other natives belonging to early forest stages (e.g., *Ulmus minor, Laurus nobilis, Celtis australis*), but potentially embodying important cultural values (Caneva et al., 2020).

Notwithstanding the satisfying statistical significance of the results, their robustness may have been limited by the small number of observations, in turn determined by the restrictive sampling design and accessibility of suitable sites, especially in the case of woodlands on anthropogenic landforms. Namely, the potential sensitivity to outliers and underestimated consideration of the type and magnitude of morphological alterations may be overcome with additional geomorphological investigation. This could help identify a larger number of ALW for sampling, along with a more in-depth understanding of urban geo-morphodiversity (Burnelli et al., 2025), and an improved interpretation of the effects on spontaneous woodlands according to quantitative thresholds posed by transformation magnitude (e.g. slope shape changes, accumulation thickness, soil and drainage changes). Other aspects that should be explored further concern the recognition of spontaneous woodlands in the urban environment (Sikorska et al., 2021) and the relationships between soil quality and ecological distance from PNV, to be extended to a wider range of ELU and/or to additional indicators (e.g. heavy metals concentration; 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 historical and cultural values may override strict ecological integrity needs without important trade-offs.

CRediT authorship contribution statement

Emiliano Ceccarelli: Writing – review & editing, Validation, Methodology, Formal analysis. **Asja Iacopino:** Investigation. **Giulia Capotorti:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Alessandro Montaldi:** Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Michele Delchiaro:** Writing – review & editing, Methodology, Formal analysis. **Alessia Pica:** Writing – review & editing, Methodology, Investigation, Data curation. **Francesca Reame:** Investigation. **Maurizio Del Monte:** Writing – review & editing. **Duilio Iamonico:** Investigation. **Eva Del Vico:** Writing – review & editing, Investigation. **Camilla Frank:** Investigation. **Simone Valeri:** Writing – review & editing, Methodology, Investigation.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work, the authors used Grammarly and DeepL to check grammar and improve writing. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

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Supplementary material

Supplementary material A.
Supplementary material B.
Supplementary material C.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ufug.2026.129480.

Data availability

Data will be made available on request.

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