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Research article

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

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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.

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 (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). Peri-urban contexts, close to main urban input sources are significantly affected by urban sprawl in many European countries, a phenomenon that becomes somewhere informal

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and/or not planned and therefore difficult to predict (e.g. in Italy) (Romano et al., 2015; Salvia et al., 2020). The conversion of arable land into dispersed urban settlements makes these areas prone to exotic invasion, resulting in damages to residual native communities, alteration of spontaneous ecosystem dynamics, and detriment of ecosystem service provision (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 (Quintas-Soriano et al., 2022). 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 (Rejmánek et al., 2013). 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 as closed 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 (Carlucci et al., 2020; 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. The European Union devoted an ad-hoc regulation to prevent, minimise and mitigate the adverse impacts posed by alien species on native biodiversity and ecosystem services. The Regulation (1143/2014) indicates a set of measures to be taken (EU, 2014) and is useful for prioritising actions (Branquart et al., 2016). In addition, it requires Member States to facilitate the recovery of ecosystems degraded, damaged or destroyed by invasive alien species, provided that alien management plans are realistic and restoration costs for affected ecosystems are proportional to the expected benefits (Blaalid et al., 2021). These needs could be addressed by means of 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). Actually, besides being supposed to be multifunctional, GIs are strategically prompted to maintain healthy ecosystems and help stop the loss of biodiversity (Livesley et al., 2016; Threlfall et al., 2017). GIs explicitly devoted to the containment of biological invasions have already been proposed, but only at broad scales (Vallecillo et al., 2018), while no evidence has been yet provided as regards planning criteria at the local scale. On the other hand, some Authors promoted the use of well-performing but non-native plants to provide regulating ecosystem services (Cameron and Blanuša, 2016) and overlooked the risk of triggering or enhancing invasions while deploying GIs out of this strategic framework (Capotorti et al., 2019; Jacklin et al., 2021). In accordance with these premises the work presented here 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 mainly 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 (Fig. 1) embraces the median sector of the river basins of Arrone River and Rio Galeria and covers 22,900 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).

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).

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 1:10,000 scale with a minimum mapping unit of 0.01 ha (Interuniversity Research Centre 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, 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 1:25,000 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).

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

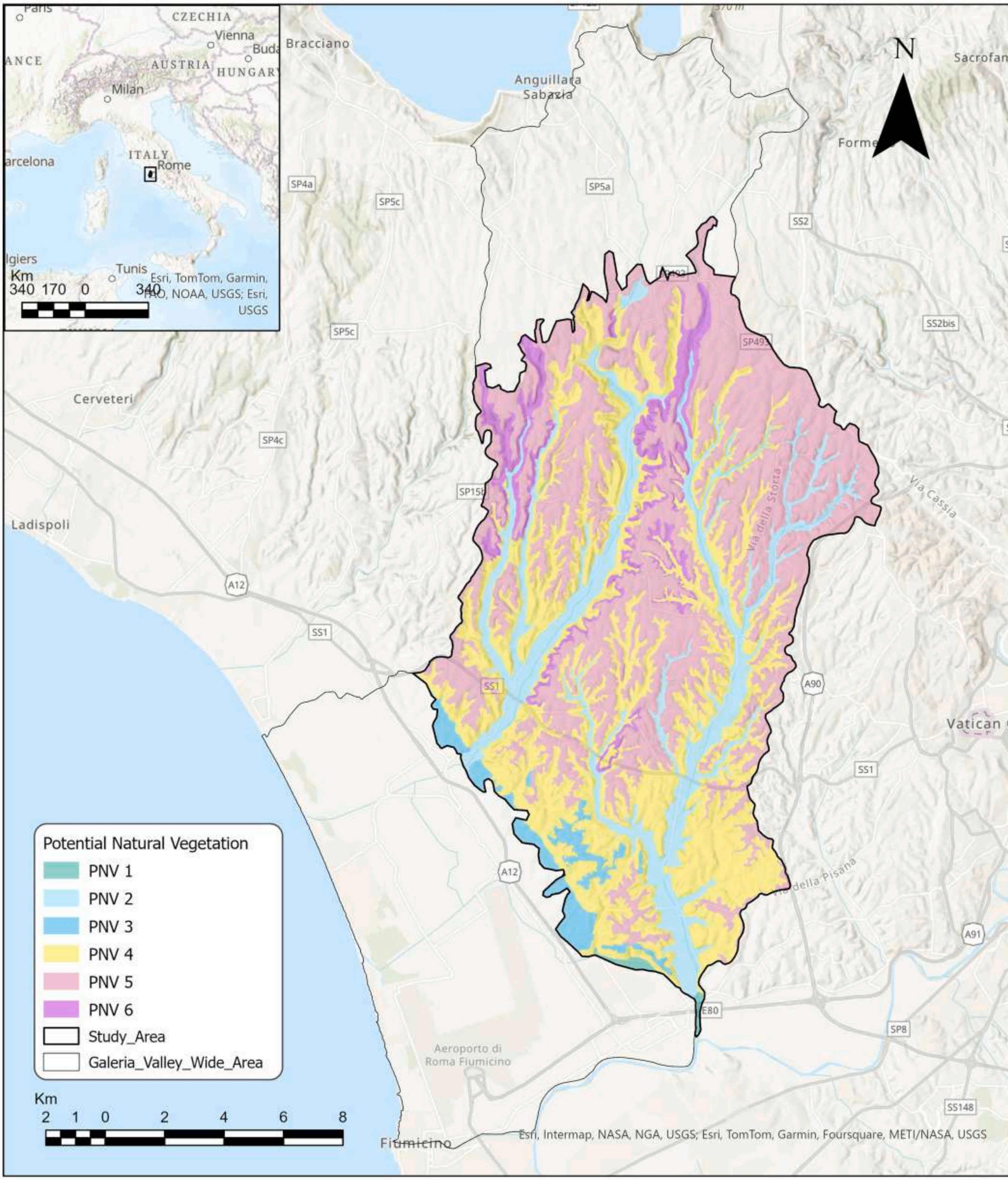

Fig. 1. Geographic location (upper left corner) and internal arrangement into Potential Natural Vegetation types of the study area (main picture): PNV 1 - *Quercus robur, Ulmus minor* and *Fraxinus angustifolia* subsp. *oxycarpa* forests of the Tiber River alluvial plain; PNV 2 - *Quercus robur, Q. cerris, 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 *Q.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).

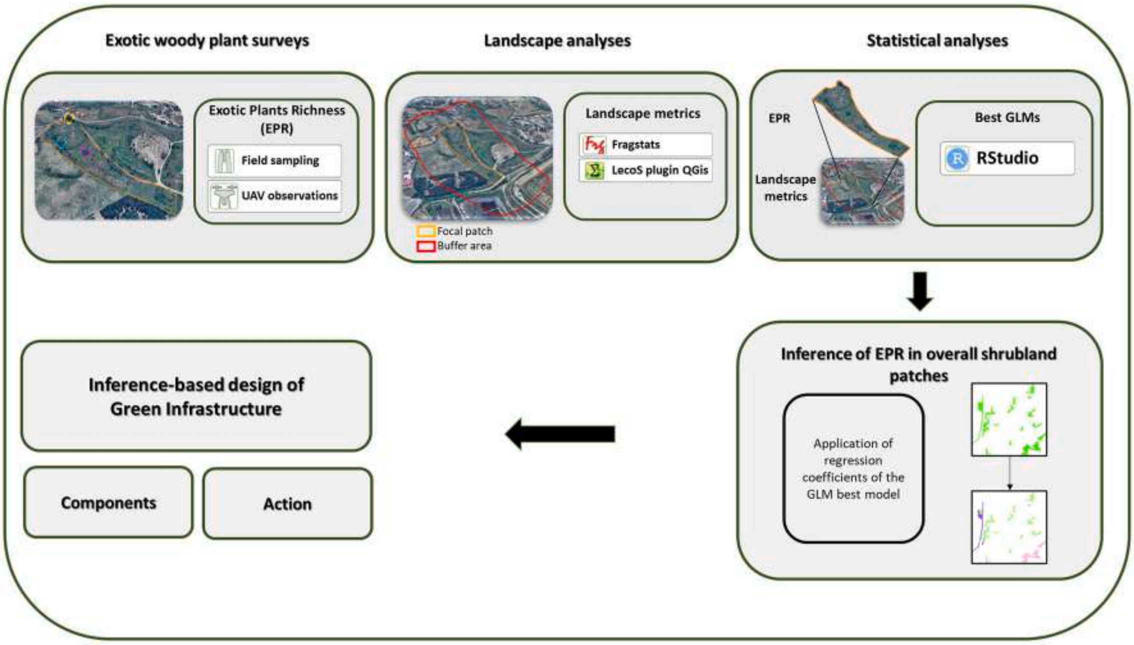

Fig. 2. Workflow for the design of GI proposal.

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 1), *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 artificial 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).

The effects of landscape metrics on EPR were thus statistically analysed and the most explicative variables identified. A generalised linear

Table 1
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)

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; Brundu et al., 2020).

Accordingly, a set of evidence-based practices for INNT 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 available knowledge and bibliographic evidence, with particular reference 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 2; 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 2). 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).

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 3).

Table 2
Exotic species sampled in the shrublands of Valle Galeria, comprehensive of the non-native status (N=Neophytes, A = Archaeophytes), 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

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

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

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

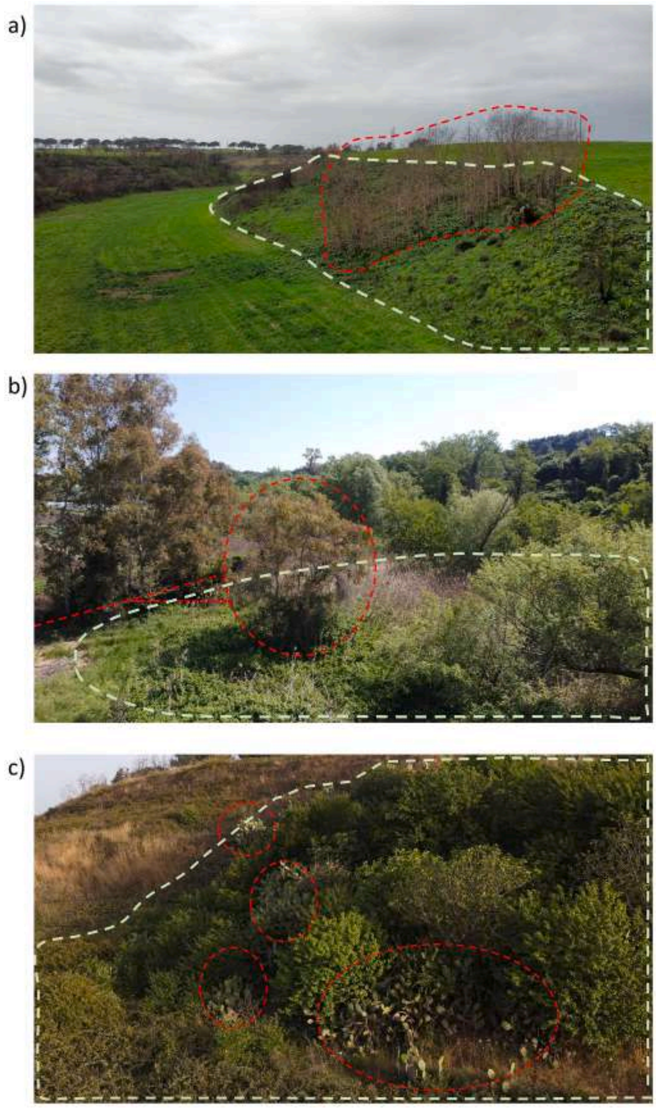

Fig. 3. Examples of invaded secondary shrublands: (a) on a sandy slope (*Quercus pubescens, Q. suber* and *Q. cerris* PNV), with a nucleus of *Ailanthus altissima* and a young individual of *Phoenix canariensis*; (b) on an alluvial plain (*Quercus robur* and *Ulmus minor* PNV), colonised by *Eucalyptus 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.*

Table 3
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			

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 number of individuals but a limited number of species, mainly represented by *Robinia pseudoacacia* and *Ailanthus altissima*.

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.

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 4).

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

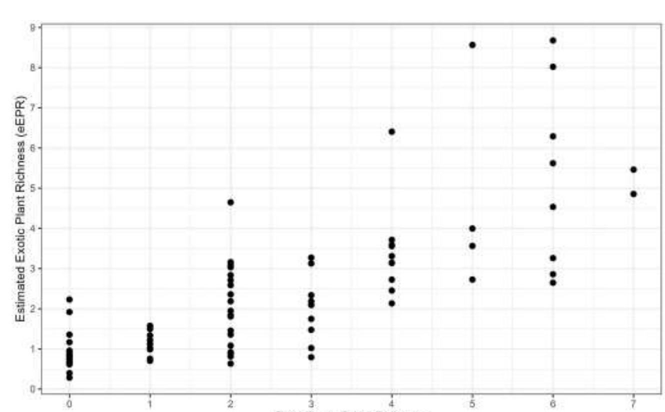

Fig. 4. Dispersion plot between estimated EPR (GLM coefficients) and real EPR in explored patches. Spearman's correlation = 0.86.

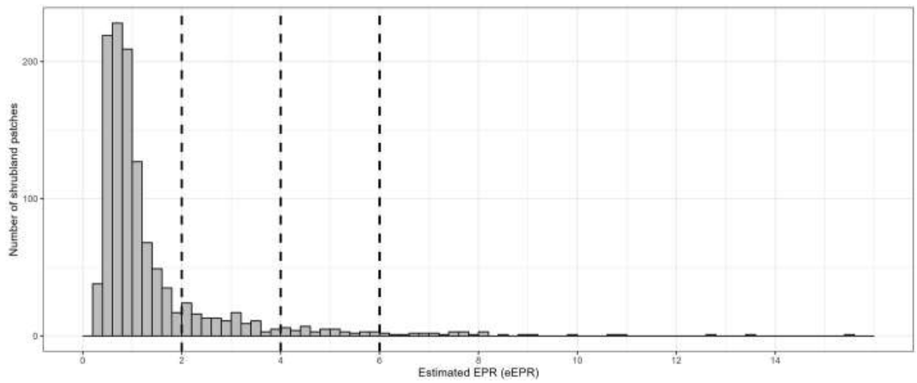

Fig. 5. Inferred invasibility for the shrublands in Valle Galeria.

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 and 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 (Arévalo 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,

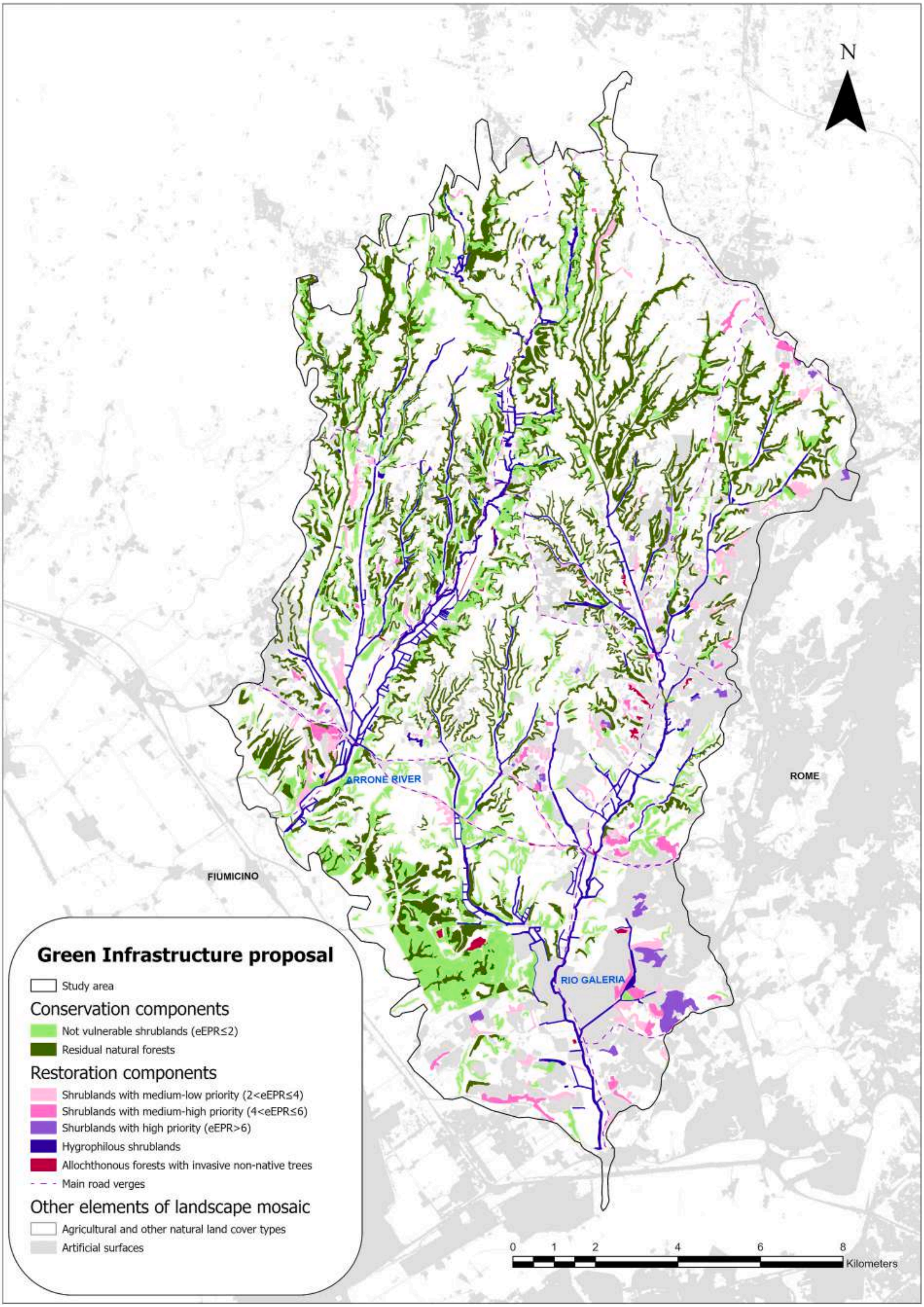

Fig. 6. Green Infrastructure proposal for the containment of plant invasion in Valle Galeria.

Table 4
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

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 cost-effective containment by widespread self-sustaining natural ecosystems. Several attempts to actively control invasive plants have had only moderate success, mainly due to overlooked revegetation with native species after removal, and limited time and spatial scope (Kettenring and Adams, 2011). These limitations are meant to be overcome by the proposed measures, since reforestation with native species, and a large and prioritised spatial scope have been embraced in the planning process. However, it is crucial to acknowledge that active containment may extend over time and determine significant costs according to site-specific conditions (Weidlich et al., 2020). Consequently, even though the GI deployment shall largely depend on ecosystems' natural recovery and prioritised actions, the cost-effectiveness of any active management of invasive plants should be assessed specifically, and eventually enhanced, for example, by biomass production (Carneiro et al., 2014).

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.

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CRediT authorship contribution statement

Alessandro Montaldi: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Duilio Iamonico:** Writing – review & editing, Investigation. **Eva Del Vico:** Writing – review & editing, Investigation. **Simone Valeri:** Writing – review & editing, Methodology, Formal analysis, Conceptualization. **Giovanna Jona Lasinio:** Validation, Methodology, Formal analysis. **Giulia Capotorti:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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