Local versus landscape-scale effects of anthropogenic land-use on forest species richness

G. Buffa*, S. Del Vecchio, E. Fantinato, V. Milano

Department of Environmental Sciences, Informatics and Statistics (DAIS), Ca' Foscari University of Venice, Via Torino 155, 30172 Venice, Italy

ABSTRACT

The study investigated the effects of human-induced landscape patterns on species richness in forests. For 80 plots of fixed size, we measured human disturbance (categorized as urban/industrial and agricultural land areas), at 'local' and 'landscape' scale (500 m and 2500 m radius from each plot, respectively), the distance from the forest edge, and the size and shape of the woody patch. By using GLM, we analyzed the effects of disturbance and patch-based measures on both total species richness and the richness of a group of specialist species (i.e. the 'ancient forest species'), representing more specific forest features. Patterns of local species richness were sensitive to the structure and composition of the surrounding landscape. Among the landscape components taken into account, urban/industrial land areas turned out as the most threatening factor for both total species richness and the richness of the ancient forest species. However, the best models evidenced a different intensity of the response to the same disturbance category as well as a different pool of significant variables for the two groups of species. The use of groups of species, such as the ancient forest species pool, that are functionally related and have similar ecological requirements, may represent an effective solution for monitoring forest dynamics under the effects of external factors. The approach of relating local assessment of species richness, and in particular of the ancient forest species pool, to land-use patterns may play an important role for the science-policy interface by supporting and strengthening conservation and regional planning decision making.

1. Introduction

Forest ecosystems are considered among the most important global repositories of terrestrial biodiversity (Liang et al., 2016). Forests contribute more than any other terrestrial ecosystems to climate relevant cycles and processes, at local, national, and global level. Forests also provide essential ecosystem services, like carbon storage, hydrological protection, air and water purification, improvement of urban and peri-urban living conditions and amenity values such as aesthetic enjoyment and recreation (Costanza et al., 1997; Pearce, 2001). Despite substantial efforts to support the preservation and sustainable use of forest biodiversity, in many industrialized countries, forests are often isolated patches embedded in an anthropogenic matrix, mostly represented by agricultural and built-up areas, which are the dominant elements of the landscape. Anthropogenic activities, such as agriculture, industry and urbanization, have been recognized as major drivers of biodiversity loss worldwide, exerting profound effects on the structure and function of remnant natural ecosystems (Del Vecchio et al., 2015; Guirado et al., 2006; Lindenmayer and Likens, 2011; Matson et al., 1997; Millennium Ecosystem Assessment, 2005; Newbold et al., 2015; Tilman et al., 2001).

Human-induced landscape-level transformations directly affect forest ecosystems by reducing available space for their development and permanence,

and fragmenting the remnant patches (Amici et al., 2015; Liu et al., 2003). Due to the adjacency of non-forest habitats, fragmentation has also indirect effects, such as changes in light availability (Cayuela et al., 2009) or wind influences (Svensson et al., 2010). Both direct and indirect effects lead to environmental deterioration and decreased habitat quality (Kinzig and Grove, 2001; Wei and Hoganson, 2005).

In recent decades, several studies have identified such spatial attributes of landscape elements as their size, shape, and extent as influencing population processes and the richness and composition of assemblages (Fischer and Lindenmayer, 2007; McKinney, 2008; Steffan-Dewenter and Tscharntke, 2000). Species richness, in particular, has been repeatedly linked to the spatial characteristics of ecosystems and has proved to be particularly sensitive to the influence of human land-use (Jentsch et al., 2012; McKinney, 2008), thereby being identified as an essential tool for biodiversity monitoring (Bitencourt et al., 2016; Del Vecchio et al., 2016; Janišová, 2014).

One problem that arises in such research is the scale dependence of both diversity and disturbance, namely they can depend on the scales at which they are sampled. The striking role of the scale has been well recognized, becoming one of the unifying concepts in ecology (Van Dobben and Lowe-McConnel, 1975; Wiens, 1989), concerning all organizational levels (from individuals to ecosystems). It represents a primary issue in the interpretation of environmen-

tal heterogeneity (Battisti and Fanelli, 2015; Levin, 1992; O'Neill et al., 1991), since different patterns can be revealed by different spatial scales of observation (Hanke et al., 2014). Moreover, such habitat properties as composition and structure may arise from the interaction of both coarse landscape- and fine local-scale filters (Buffa et al., 2007; Dale, 1999; Gigante et al., 2016b; Lortie et al., 2004), i.e. in addition to local physical and biotic factors, the response of a population to disturbance can also be influenced by larger-scale phenomena (Hanski, 1998). Thus, although no preferred scale exists (Levin, 1992), the scale at which environmental patterns are quantified influences the result (Turner et al., 2001; Wiens, 1989; Wu, 2004) and inappropriate scales may fail in detecting patterns (Li and Wu, 2004).

However, several studies evidence that the importance of environmental, external factors strongly differs not only across spatial scales but also among taxonomic groups (Laurance et al., 2002; Polyakova et al., 2016; Turtureanu et al., 2014). Dauber et al. (2003) further suggested that species richness patterns of different species groups are often not correlated and their dependence on the landscape pattern varies among groups. These studies consistently indicate that environmental changes both at local and landscape scales do not affect all species and taxa equally, but the effects depend on species traits. Arguably, different groups of species can respond in a different way to a given event, or respond to different events or to different intensities of the same event. Especially for species which do not possess evolutionary adaptations enabling them to cope with large scale disturbance, anthropogenic changes may have a much more dramatic effect relative to other species (Honnay et al., 2005). As for forest ecosystems, species such as forest floor plant species, and in particular the ancient forest species, suffer increased extinction probability compared to other forest species, due to their biological and ecological characteristics. Their common features are slow growth, long life-cycle, stress tolerant strategy, early and short flowering, a strong vocation for vegetative propagation, heavy seeds and transient seed bank (Verheyen et al., 2003), limited dispersal power in space and low colonization ability (Buffa and Villani, 2012). Being restricted to a narrow range of ecological requirements, ancient forest species are highly specialized, i.e. species strictly associated with the interior, more protected and undisturbed part of a forest (Hermy, 1994), and therefore indicative of more original forest conditions (Peterken, 1974). Ancient forest species thus meet qualitative (forest quality) as well as quantitative (diversity) conservation criteria (Hermy et al., 1999). Specialist species define the habitat identity, and have a prominent role in assuring the maintenance of its structure and functionality (Del Vecchio et al., 2016; Fantinato et al., 2016; Godefroid and Koedam, 2003). This is particularly important since their extinction may have consequences on the whole ecosystem, leading to ecosystems collapse (Keith et al., 2013), even if the number of specialist species is usually small compared to the total number of species in a habitat. Since specialist species are experiencing higher extinction risk relative to generalist species (Buffa and Villani, 2012; Clavel et al., 2011; Fantinato et al., 2017; Rooney et al., 2004; Slaviero et al., 2016), their presence can be used as a synthetic indicator of the status of a habitat, or fine-filter surrogate species which represent more specific habitat features (Jones et al., 2016; Rodrigues and Brooks, 2007). If attentively selected, specialist species can thus be more effective in describing the relationship between disturbance patterns and biodiversity than the use of total species richness.

Aim of the present paper was to investigate the effects of human-induced landscape patterns on species richness in forests. In particular, we analyzed the trends of total species richness and the richness of the ancient forest species pool and tested if total species richness and the richness of the ancient forest species pool respond in a different way to the disturbance generated by human activities. To evaluate the relationship between local plant species richness and landscape variables, while taking into account the effects of changing extent, we measured human disturbance at two spatial scales, at 'local' (500 m radius from the central point of each plot) and at 'landscape' scale (2500 m radius).

2. Methods

2.1. Study area

The study was conducted in the hilly sector of the Veneto Region (north-eastern Italy). The study area is a historically cultivated landscape, where human activities created a complex landscape where ecological, socioeconomic, and cultural patterns governed the presence, distribution, and abundance of wild species assemblages. Over the last 40 years, the human component has been increasingly dominating in space and time, and the area has faced dramatic land-use changes which include the conversion of complex natural ecosystems to simplified managed ecosystems, and the intensification of resource use, leading to the loss of biodiversity in pristine habitats and traditional, low-intensity agro-ecosystems. New disturbance regimes have transformed the traditional landscape, which is nowadays dominated by land-uses such as cereal cropping, vineyards, horticulture, and tree plantations. Natural and semi-natural habitats such as forests, grasslands and hedgerows, are interspersed with human settlements, roads, and trenches, making the landscape increasingly prone to the risks of rapid biological impoverishment. The remnant forest vegetation is mainly composed of mesophilous broad-leaved deciduous oak forest communities. The tree layer is dominated by *Quercus robur* L., *Q. cerris* L., *Fraxinus ornus* L., *Acer campestre* L., and *Carpinus betulus* L.. Locally, termophilous deciduous woods dominated by *Q. pubescens* Willd., and meso-hygrophilous deciduous forests dominated by *Alnus* spp. can also be found. Other traditional rural landscape components are mesophilous and dry grasslands.

2.2. Data collection

To allow inference about how landscape composition, i.e. the types of different land-uses and their relative proportion, influences species richness and composition in forest remnants, we surveyed 80 georeferenced plots with a fixed surface (8×8 m). A preliminary forest stand stratification was conducted according to elevation range (from 100 to 300 m a.s.l.) and bedrock types (neutro-basic limestone) which allowed the identification of 59 patches of mesophilous broad-leaved deciduous oak forest communities, with patch surface ranging from 0.08 to 350 ha. In order to achieve a homogenous and proportional distribution of the plots within the patches, we randomly created 80 points (function "*Create Random Points*" in ArcGIS 9.3), specifying a minimum of 1 and a maximum of 5 points per patch (see Appendix 1).

In each plot, all vascular plant species in each vertical layer were recorded.

For each plot we calculated total species richness and the richness of ancient forest species. Ancient forest species, characterizing the herb layer, were defined according to a previous study by Buffa and Villani (2012), in which they have been identified based on life history traits such as morphology (life form), life-cycle, floral and reproductive biology (type of reproduction – vegetative, sexual or both, pollen vector, dispersal mode), and ecological strategy (Grime, 1979; Pierce et al., 2017). Ancient forest species of studied forests have some common traits, such as a strong vocation for vegetative reproduction and ants as preferential dispersal agent. They occupy two different temporal and spatial niches: nearly 50% are small spring geophytes, CSR strategists, with entomophilous pollination (e.g. *Anemone* sp.pl., *Cardamine bulbifera* (L.) Crantz, *Lathyrus vernus* (L.) Bernh., *Polygonatum multiflorum* (L.) All.). The other 50% is represented by higher hemicryptophytes, with CS strategy (e.g. *Carex sylvatica* Huds., *Melittis melissophyllum* L., *Primula vulgaris* Huds., *Pulmonaria officinalis* L.).

The structure and composition of the landscape surrounding each plot was derived from a high-resolution land-use map (scale 1:10,000), extracted from the Corine Land Cover map level III (ISPRA, 2010). Human disturbance was defined as the proportion of land area converted by humans (Mayor et al., 2015), and was categorized as agricultural (e.g. pasture and croplands, tree planting areas), and urban/industrial (e.g. urban and rural settlements, roads and railways, green urban areas, leisure facilities). To quantify the effects of in-

creasing extent, for each plot, human disturbance was measured at two scales: 'local scale' and 'landscape scale', in two circular buffers, with a radius of 500 m and 2500 m from the central point of each plot respectively. The two scales were chosen among other intermediate possible scales because disturbance measured at these two scale resulted the least correlated (r = 0.487). In each buffer, the proportion of land area covered by forests was used as a proxy measure of the connectivity among natural patches.

For each plot, we also determined patch-based measures of pattern including the distance from the forest edge (measured as the distance in m from the nearest forest edge; function "*Near*" in ArcGIS 9.3), size and shape of the woody patch in which each plot was included. To figure out the shape of the patch we used a compactness index (Bosch, 1978; Davis, 1986), according to the formula:

$$K = \frac{2\sqrt{\pi A}}{P}$$

where A is the area (m^2) and P is the perimeter of the patch. This index provides a dimensionless number that represents the degree to which a shape is compact. When the value of K tends to 0, it indicates an elongated and irregular shape, while when it tends to 1, it indicates a circular and regular shape.

2.3. Data analysis

To investigate the effects of the landscape and patch-based variables on total species richness and richness of ancient forest species, we performed generalized linear models. The proportion of agricultural, urban/industrial, forested area and patch configuration measures (size, shape and distance from the edge) were used as predictor variables. Total species richness and richness of ancient forest species were used as response variables. The effects of disturbance were tested at both local and landscape scale. For each model we used the Poisson error distribution. To simplify the models we performed a stepwise algorithm of explanatory variables (backward selection). The Akaike Information Criterion (AICc; R package MASS; Venables and Ripley, 2002) was used to select the best model. According to the procedure, models explaining most variance with a lower number of predictors have smallest AIC and are considered 'best models' (Burnham and Anderson, 2002). The best fitting models procedure led to exclude the proportion of forested area since it resulted correlated to the proportion of agricultural land area (Pearson correlation; local scale: r = −0.81; Landscape scale: r = −0.73).

3. Results

In total, 239 plant species were recorded, of which 57 were classified as ancient forest species. Mean total species richness per plot was 16.8±5.2, while the mean richness of ancient forest species was 3.1±2.1. The most common ancient forest species were mostly geophytes such as *Allium ursinum* L., *Anemone nemorosa* L., *A. trifolia* L., *Cardamine bulbifera, Erythronium dens-canis* L., *Hepatica nobilis* Schreb., and *Polygonatum multiflorum*. Among hemicryptophytes, the most common were *Brachypodium sylvaticum* (Huds.) P. Beauv., *Carex sylvatica*, *Epimedium alpinum* L., and *Salvia glutinosa* L..

The average patch size was 57.94 ha±92, but many patches (46.83%) were small (surface < 15 ha). Most forest fragments (70%) had an irregular shape (shape index < 0.50), with an average value of 0.40±0.21 (median of 0.37). The distance of plots from the nearest forest edge ranged from 0.22 to 167.29 m (median of 22.26 m). Agriculture was basically the dominant land-use type at landscape scale, with a mean proportion of 72.9±15.5, ranging from 31.7% to 100%. Conversely, at local scale the agricultural land area exhibited the full continuum of 0–100 percent (mean proportion 66.6%±28.8). Urban/industrial surface ranged from 2.0%±7.4 at local to 6.9%±7.6 at landscape scale. The proportion of forested land area was always low and de-

The selection of the best fitting models (Δ_{AIC} total richness = −8.9; Δ_{AIC} ancient forest species richness = −6.06) indicated that, at local scale, none of the selected predictor variables (proportion of agricultural and urban/industrial, and patch-configuration measures) had a significant role in determining neither the total species richness nor the ancient forest species richness.

At landscape scale, the best models included the proportion of urban/industrial land area for both total species richness and the richness of ancient forest species. In both cases with a highly significant negative effect. However, if we consider the trendlines, which represent how the richness decreases with increasing urban/industrial land area, the models showed a stronger effect on the richness of ancient forest species (Table 1). Total species richness was also negatively influenced by the proportion of agricultural land area, while for ancient forest species richness, the model evidenced the significant positive role of a compact shape (Table 1 and Fig. 1).

4. Discussion

Our study demonstrates that patterns of local species richness in forest ecosystems are sensitive to the structure and composition of the surrounding landscape. Our results are consistent with several previous studies (e.g., Amici et al., 2015; Bennett et al., 2006; Collingham et al., 2000; Fahrig, 2003; Godefroid and Koedam, 2003; Murcia, 1995; Tilman, 1994) underlining the importance of the landscape context in shaping the richness and composition of local habitat species assemblages. Especially, according to Newbold et al. (2015), rural and urban areas, which constitute the two main land-use categories associated with human activities, can reduce species richness by more than half in the worst-affected habitats.

The observed pattern showed a similar negative trend of both total species richness and the richness of ancient forest species. Nevertheless, the best models evidenced a different intensity of the response to the same disturbance category as well as a different pool of variables for the two groups of species, i.e. the ancient forest species pool responded differently to small scale (patch-level measures) characteristics and large-scale (landscape) features. Although always present with a lower proportion relative to agricultural land-use, urban/industrial land area proved to have a strong negative influence on the richness of ancient forest species. Arguably, human settlements differ in their structural features compared to rural areas. Containing remnant patches of natural or semi-natural surfaces (Evans et al., 2009), rural areas can provide temporary surrogate habitats for some species, thereby favoring the permanence of the relationships between individuals, populations, and wild species assemblages. For example, man-made linear elements marking boundaries, like hedges, lines of trees, and grass strips, are particularly widespread in farmed landscape throughout temperate regions (Baudry et al., 2000), and may play a vital role in delivering ecosystem services (Lindborg et al., 2014). For example, Dainese et al. (2017) found that the presence of hedgerows in the landscape enhanced potential pollination, both in terms of visitation rate and seed set. Hedgerows can thus act as transitory surrogate habitats for forest species (Ernoult and Alard, 2011; Morelli, 2013). This assumption is confirmed by Masin et al. (2009) who reported the presence in the region of some ancient forest species (e.g. *Anemone* sp.pl., *Allium ursinum*, *Circaea lutetiana* L., *Polygonatum multiflorum*) outside the forests, along the oldest hedgerows, thereby strengthening the important role of spread hedgerows and other connecting elements in facilitating the movement of organisms. Conversely, the nature of urban/industrial land-use normally involves both a net loss of natural and semi-natural surfaces, thus reducing the area available for wild plants and animals (Auffret et al., 2015; McKinney, 2008), and a structural simplification of vegetation in many areas. Landscaping and the maintenance of residential, commercial and industrial areas typically entail removal of woody plants (e.g. hedges, small woods) and an increase in green urban areas such as lawns (Marzluff, 2001), resulting in the loss of surrogate habitats and increased isolation of ancient forest species populations. Further, forests directly bordering, or near to, human settlements are likely more prone to suffer higher intensity of disturbance due to open-air leisure activities such as walk-

Table 1
Summary table of the best fitting models and AIC values of each model, testing the effects of anthropogenic disturbance categorized as the proportion of agricultural and urban/industrial land areas, and patch-based variables (size, shape and the distance from the forest edge), on total species richness and the richness of ancient forest species.

Total species richness	Deviance Residuals:					
		Min	1Q	Median	3Q	Max
		−2.2216	−0.753	−0.023	0.578	1.951
	Coefficients:					
		Estimate	Std. Error	z	value	Pr(>|z|)
	(Intercept)	3.724	0.124	30.049	<0.000001	***
	Urban/industrial area landscape (2500m)	**−0.015**	**0.004**	**−3.811**	**0.000138**	***
	Agricultural land area landscape (2500m)	**−0.011**	**0.002**	**−6.633**	**<0.000001**	***
	(Dispersion parameter for poisson family taken to be 1)					
	Null deviance: 127.746 on 78 degrees of freedom					
	Residual deviance: 70.885 on 76 degrees of freedom (2 observations deleted due to missingness)					
	AIC Best fitting model: 441.83 (AIC Full model: 450.73)					
	Number of Fisher Scoring iterations: 4					
	Step	Df	Deviance Resid.	Df	Resid. Dev	AIC	Delta AIC
	1			71	69.78511	450.7255	−8.8999
	2-Patch size	1	0.000231	72	69.78535	448.7257	−6.9001
	3-Urban/industrial area landscape (500m)	1	0.013638	73	69.79898	446.7393	−4.9137
	4-Patch shape	1	0.087003	74	69.88599	444.8263	−3.0007
	5-Distance from the forest edge	1	0.356988	75	70.24297	443.1833	−1.3577
	6-Agricultural land area landscape (500m)	1	0.642283	76	70.88526	441.8256	0
Ancient forest species richness	Deviance Residuals:					
		Min	1Q	Median	3Q	Max
		−2.3003	−0.862	−0.051	0.536	2.735
	(Intercept)	0.894	0.152	5.881	<0.000001	***
	Urban/industrial land area landscape (2500m)	**−0.05**	**0.011**	**−4.386**	**<0.000001**	***
	Patch shape	**1.231**	**0.287**	**4.284**	**<0.000001**	***
	(Dispersion parameter for poisson family taken to be 1)					
	Null deviance: 109.574 on 78 degrees of freedom					
	Residual deviance: 72.345 on 76 degrees of freedom (2 observations deleted due to missingness)					
	AIC Best fitting model: 297.7 (AIC Full model: 303.76)					
	Number of Fisher Scoring iterations: 5					
	Step	Df	Deviance Resid.	Df	Resid. Dev	AIC	Delta AIC
	1			71	68.40971	303.761	−6.0647
	2-Patch size	1	0.4712536	72	68.88096	302.2323	−4.536
	3-Urban/industrial area landscape (500m)	1	0.8675621	73	69.74852	301.0998	−3.4035
	4-Agricultural land area landscape (500m)	1	0.6968999	74	70.44542	299.7967	−2.1004
	5-Agricultural land area landscape (2500m)	1	0.133221	75	70.57865	297.9299	−0.2336
	6-Distance from the forest edge	1	1.7663621	76	72.34501	297.6963	0

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1.

local direct human disturbance. All of these factors tend to increase with urbanization intensity (Hahs and McDonnell, 2006) and combine to reduce both habitat core area and the overall habitat quality.

Contrary to total species richness, for which no patch-level measure emerged as significant, ancient forest species responded positively to the patch shape, i.e. the richness of ancient forest species increased with increasing regularity of the shape. The positive effect of a regular shape on species richness has already been shown and related to the edge effect that in highly disturbed landscapes may become much more important than area and isolation (Harrison and Bruna, 1999; Gonzalez et al., 2010; Turner et al., 1996). For an identical area, an irregular forest fragment has a higher edge/core ratio than a compact forest fragment, namely a smaller undisturbed core area suitable for the persistence of specialist species (Dauber et al., 2003). Conversely, the best model did not retain the distance from the edge. This result is in accordance with Guirado et al. (2006) who found that forest species of small patches were not affected by the distance from the edge. This effect, somehow unexpected, could be due to either a non-linear responses of species richness and composition to the distance

Alaback, 2004) or to the configuration of patch edges linked to the forestry regimes, e.g. the presence of buffer zones that minimize the effect of a short distance from the edge.

The ancient forest species pool seems not to be affected by decreasing patch area. Several studies (e.g., Buffa and Villani, 2012; Dupré and Ehrlén, 2002; Kolb and Diekmann, 2005) suggested that patch area can become a secondary factor compared to species-specific life-history traits and habitat quality. Moreover, the effect of the patch area might change because of direct or indirect effects of human activities (Bennett et al., 2004; Honnay et al., 1999), especially in urban and peri-urban contexts (Guirado et al., 2006; Moffatt et al., 2004). Arguably, having short distance seed dispersal mechanisms, and spreading mostly vegetatively by stolons or rhizomes (Buffa and Villani, 2012; Honnay et al., 2002), ancient forest species likely benefit mostly from a high patch quality rather than from large areas per se. However, several studies (e.g. Buffa and Villani, 2012; Cousins, 2009; Ovaskainen and Hanski, 2002) evidenced that in long-lived species the negative consequences of fragmentation (e.g. reduced area and population size, and increased isolation) may become visible after a long time, caus-
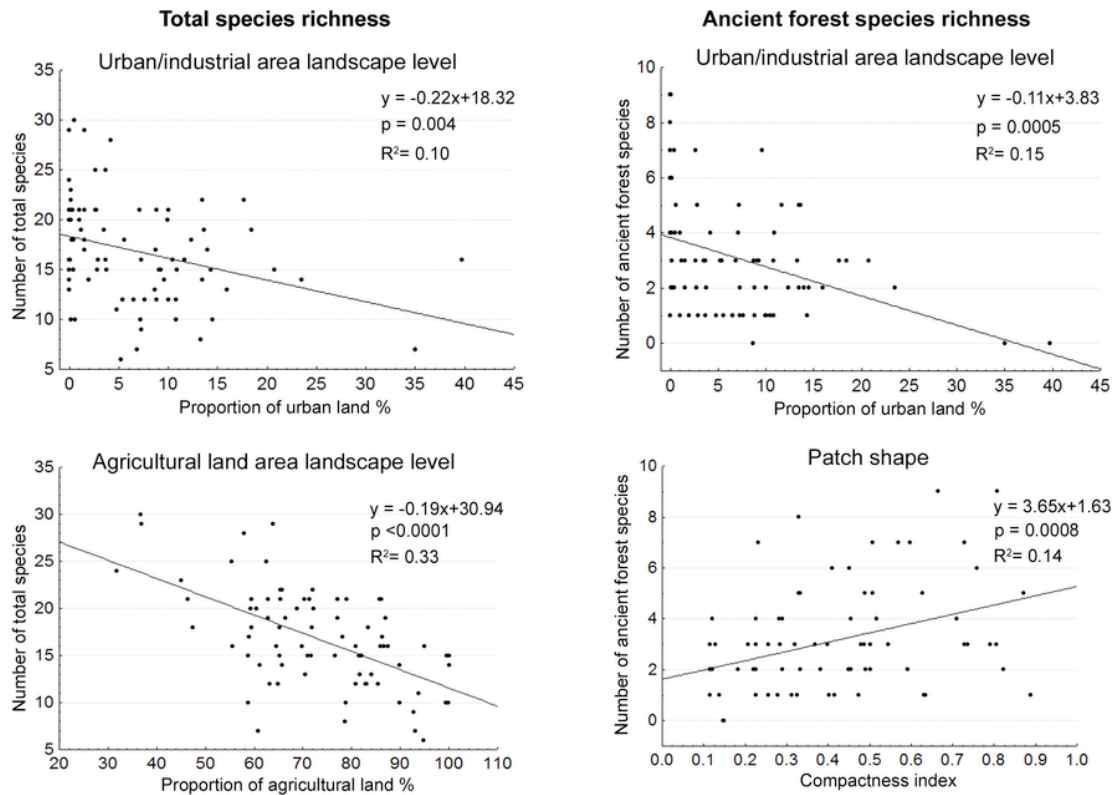
Fig. 1. Trends of total species richness and the richness of ancient forest species against urban/industrial and agricultural land areas at landscape scale (distance of 2500 m from the plot) and the patch shape.

habitats (see e.g. Kuussaari et al., 2009 for a review). In this regard, we have to consider that the ancient forest species richness we detected could not be in equilibrium and that species will go extinct until the ancient forest species richness reaches a new equilibrium.

The different behavior of ancient forest species compared to total species richness evidences that, although often used as indicator of ecosystem change, total species richness can be less able to reveal patterns underlying forests change. The use of groups of species, such as the ancient forest species pool, that are functionally related and have similar ecological requirements, may represent an effective solution for describing forest dynamics under the effects of external factors.

The use of groups of species as surrogates for the overall biodiversity has been given significant attention in recent times, due to its potential contribution to the simplification of data requirements for monitoring and conservation planning (Gigante et al., 2016a; Larsen et al., 2009; Leal et al., 2010). The assumption at the basis of this approach is that if threats faced by specialist species are effectively detected and managed, threats to all other species will also be effectively managed (Jones et al., 2016). Although the use of indicator species has been criticized (e.g. Lawton et al., 1998) for possible drawbacks such as the need of specific information for each site and the difficulty to standardize the survey method, the concept of indicator species remains an appealing one and potentially a powerful management tool because of the impossibility of monitoring everything (Lindenmayer et al., 2000). Moreover, as stated by Moffatt and McLachlan (2004), the presence of indicator species could be easily established at each site, thereby allowing managers to accurately assess the quality of forests without requiring an exhaustive description of the understorey. Indeed, understorey herbs have been already used as operative indicators of deciduous forest regeneration (McLachlan and Bazely, 2001), restoration of forest plantations (Honnay et al., 2002), long-term continuity of boreal forests (Ohlson et al., 1997), impact of forest management (Scolastri et al., 2017), and riparian forest disturbance (Moffatt and McLachlan, 2004).

Finally, biodiversity conservation policies and monitoring programs should take into account the crucial role of the landscape scale in predicting the trend

of local species assemblages. A lack of awareness of the close interrelation among patterns that occur at different spatial scales may cause flaws and failures in conservation strategies or actions implemented at the local scale. Such an approach of relating local assessment of ancient forest species richness to land-use patterns may thus play an important role for the science-policy interface by supporting and strengthening forest conservation and regional planning decision making.

Author contribution

GB conceived the idea and design of the study and wrote the manuscript, SDV, EF, conducted field work, data analysis and collaborated in the writing, VM collaborated in field work.

Appendix 1. Distribution of the 80 plots randomly generated by means of the function *Create Random Points*" in ArcGIS 9.3. To achieve a homogenous and proportional distribution of the plots within the 59 patches, points were randomly created specifying a minimum of 1and a maximum of 5 points per patch

patch id	patch area (ha)	n. of random plots
1	0.084	1
2	87.676	4
3	13.195	1
4	1.773	1
5	1.313	1
6	2.652	1
7	8.425	1
8	0.299	1
9	11.486	2
10	23.463	1
11	4.895	1
12	25.872	2

13	3.774	1
14	7.33	1
15	17.019	2
16	1.502	1
17	17.819	1
18	6.774	1
19	7.797	1
20	0.544	1
21	92.353	3
22	43.292	2
23	2.241	1
24	41.999	1
25	4.539	2
26	347.993	5
27	36.9	1
28	6.148	2
29	1.282	1
30	6.057	2
31	7.833	1
32	0.487	1
33	63.192	1
34	16.931	1
35	21.478	1
36	4.018	1
37	3.375	1
38	11.442	1
39	0.089	1
40	1.445	1
41	30.941	1
42	224.186	1
43	18.607	1
44	1.805	1
45	23.901	1
46	12.154	1
47	232.818	5
48	38.77	1
49	199.218	1
50	3.828	1
51	37.715	1
52	5.867	1
53	62.083	2
54	32.938	1
55	27.817	1
56	24.588	1
57	21.816	1
58	8.297	1
59	14.089	1

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