

Ecological thresholds in herb communities for the management of suburban fragmented forests

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ABSTRACT

Fragmentation is a major cause of biodiversity loss. The relationship between fragmentation and biodiversity might be non-linear, due to the presence of ecological thresholds; however, the application of the threshold concept on the conservation of herb biodiversity has received limited attention. We assessed the relationship between a pool of woodland indicators and features of forest fragments (area, isolation, shape), and we evaluated whether ecological thresholds might provide guidelines for conservation. We performed our study in a human dominated area of northern Italy. In order to obtain a group of floristic indicators, we chose perennial herbs characteristic of the phytosociological syntaxa *Fagetalia sylvaticae*, *Carpinion betuli* and *Erythronio-Carpinion*, with suitable Landolt L and H index, obtained from previous relevés in reference and high-quality forests. Subsequently, we analyzed the relationship between richness of indicators and patch features recorded in a highly fragmented landscape. Generalized linear models showed that the number of forest indicators was positively related to fragment size and connectivity. Generalized Additive Models showed that the relationship between indicator richness and fragment size was non-linear, with the presence of two thresholds. The number of forest indicators increased until 35–40 ha; above this size the richness remained constant. Fragments above this threshold should be the target of conservation. Furthermore, the relationship between area and number of indicators was less steep in patches smaller than 1–1.5 ha. Patches between these two thresholds can be optimal targets of management actions, as small size increases can greatly improve their value for biodiversity. The identification of two thresholds in the relationship between patch area and indicators suggests an abrupt response to changing landscape features and provide quantitative targets for conservation.

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1. Introduction

Fragmentation, i.e., the combination of habitat loss and increasing isolation occurring during landscape transformations (Forman, 1995; but see also Fahrig, 2003 for a different definition), is a major ecological problem and has particularly strong effects in human dominated landscapes (Levenson, 1981; Forman, 1995). Fragmentation has negative effects on ecosystems because of loss of habitat, isolation (Jacquemyn et al., 2001; Feeley and Terborgh, 2008) and edge effect, which can increase the abundance of weedy and alien species (Saunders et al., 1991; Honnay et al., 2002; Laurance et al., 2002) and can have negative consequences on

forest interior species (Brunet and von Oheimb, 1998; Verheyen and Hermy, 2004). Some forest herbs may be particularly affected by fragmentation, especially when their abundance is low, when they have a short dormancy period with no seed bank (Bierzychudek, 1982) or when they are not able to disperse over long distances (Whigham, 2004).

During fragmentation, the relationship between loss of habitat, patch size and isolation can be non-linear. When fragmentation reaches the so called "fragmentation threshold" an abrupt increase in isolation is possible (Andrén, 1994; With and Crist, 1995); when habitat cover falls below the fragmentation threshold, the risk of extinction of species living in patches can increase abruptly (Fahrig, 2002). The analysis of fragmentation thresholds can provide important information for the management of species: can assess the minimum habitat cover necessary for species persistence and can identify quantitative targets for conservation. In most of cases, the aim of management for biodiversity conservation is to maintain habitat cover above the fragmentation

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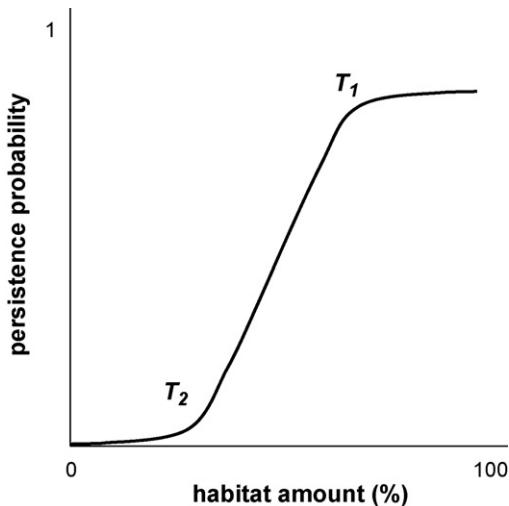


Fig. 1. Depiction of a theoretical ecological threshold (redrawn from Fahrig, 2002). When the habitat amount falls below the threshold point (T_1), the probability of the population survival abruptly decreases below 1. The probability of population survival is nearly zero if habitat amount is below T_2 .

threshold (T_1 in Fig. 1), because if cover falls below the threshold a relatively small loss of habitat can have disproportionate deleterious consequences (Huggett, 2005; Lindenmayer and Luck, 2005). Therefore, patches close to T_1 but above T_1 should be considered a priority for conservation. Furthermore, theoretical models (see With and King, 1999; Fahrig, 2001, 2002; Huggett, 2005) suggest the presence of a second fragmentation threshold (T_2 in Fig. 1). If habitat cover is below T_2 the landscape is completely unsuitable for the target species. Nevertheless, between T_2 and T_1 species survival can be strongly improved by small increases in habitat cover (Huggett, 2005). In other words, the management and improvement of patches between T_1 and T_2 can drive large changes in suitability; therefore these areas can be considered as a priority for landscape restoration. Nevertheless, this concept has been little explored in forest conservation.

In this study, we evaluate the effects of fragmentation on forest herbs in a human dominated landscape, linking a pool of selected indicators to three key features of forest patches: size, shape and isolation. First, we identified a group of herbaceous species describing the conservation status of the forest fragments (Honnay et al., 2005; Matlack, 2005). We analyzed large mesophylic forests (reference forests) to select *a priori* a pool of interior species. We used autoecological parameters and forest composition to identify a pool of species best describing the most natural forests available (hereafter: forest indicators), and to exclude weedy and edge species, or plants occurring occasionally. Subsequently, we evaluated the relationship between richness of indicators and the features of forest patches in a highly fragmented area to (1) identify the parameters most important to explain herbs distribution and (2) to evaluate the presence of ecological thresholds that can be used as management targets.

We performed this study at patch scale (Moilanen and Hanski, 1998; Dumortier et al., 2002; Lorenzetti and Battisti, 2006). The effects of fragmentation on forest ecosystems are evident at multiple levels (population, community, ecosystem and landscape; Forman, 1995) and the patch scale approach can allow a straightforward identification of targets for decision makers, because most of management in suburban areas is focused on the creation of new forest patches, or on the improvement of the existing ones (Van Calster et al., 2008). Management performed at the patch scale can have positive effects also at the other levels (Lindenmayer and Fischer, 2006).

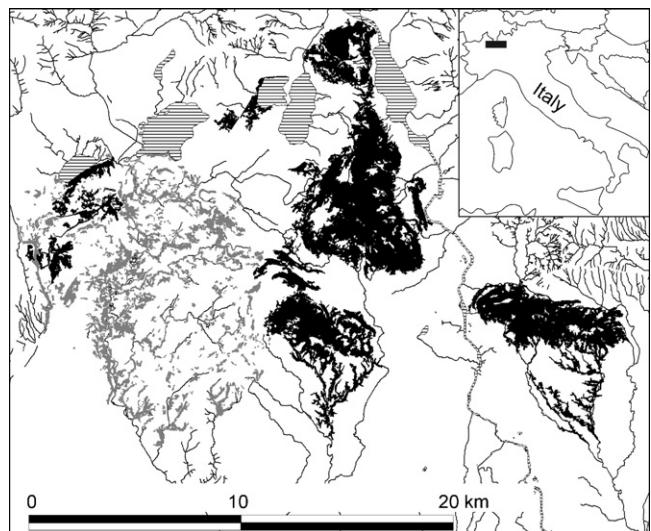


Fig. 2. Study area, showing the central part with small, highly fragmented forest patches (small fragments—grey patches), surrounded by larger and less fragmented forests (reference forest—black patches).

2. Study area

Our study area (about 240 km²) is in north-western Italy (Lombardy), close to the foothills of the Alps (Fig. 1). The study area is close to Milan, one of the largest urban areas in Europe. The substratum is homogeneous and consists of Würm morainic hills, laid from glaciers about 80,000 years ago. The landscape is dominated by croplands alternated to widespread urban settlements, roads and railways (Fig. 2). About 23% of the landscape is covered by forest patches, composed by oaks (e.g., *Quercus robur*, *Q. petraea*), hornbeams (*Carpinus betulus*) and maples (*Acer pseudoplatanus*, *A. campestre*). Some alien species are common, such as the black-locust tree (*Robinia pseudoacacia*), the black-cherry (*Prunus serotina*) and the red oak (*Quercus rubra*). The oak-hornbeam forests are classified into *Carpinion betuli* Issler 1931 (Digiovinazzo and Andreis, 2007), a phytosociological alliance that includes hills and plain mesophylic forests distributed in west Europe.

Within the study area, two forest typologies can be recognised, differing for landscapes features (Fig. 2). The central part (about 120 km²) contains small, highly fragmented forest patches (small fragments); it includes 610 forest fragments and is bordered by two natural parks largely forested. Forest patches have a wide range of size (0.1–136 ha; mean = 2.4 ha) and shape [Area (A)/Perimeter (P) mean ratio, using index $\gamma = (2\sqrt{\pi}A)/P$ in Forman, 1995: 0.11–0.93; mean = 0.62]. These small fragments are surrounded by larger and less fragmented forests (reference forest), including about 40 fragments (mean: 264 ha) (Fig. 2). Apart from differences in fragmentation and patch area, no major differences are present (e.g., same geological substrate, no large differences in altitude or water availability).

3. Methods

3.1. Study system

Our study can be described in two steps. First, we identified our target species (forest indicators) in the largest and less fragmented reference forests (Fig. 2). Then, we performed surveys to evaluate the distribution of forest indicators in 94 small fragments, selected as described below (see paragraph 3.3) to evaluate the effect of patch features on the distribution of forest herbaceous species.

3.2. Identification of forest indicators

We selected our indicators *a priori*; this approach is often used in applied conservation studies (Meffe et al., 2002; Padoa-Schioppa et al., 2006). Following Root (1967), we considered our pool of indicators as “a group of species (guild) that exploits the same class of environmental resources, in similar way”. We followed the criteria introduced by Hermy et al. (1999), modified as explained below, to identify forest herbs. First, we selected 126 relevés coming from previous researches performed in reference forests. The reference forests had: lower fragmentation; larger area (mean: 264 ha), similar phytosociological composition [*Quercus robur* and *Carpinus betulus* covering >10%; without *Quercus rubra* or *Prunus serotina* in the overstorey layer (Beltracchini, 2000; Digiovinazzo, 2000; Rovelli, 2000)]. From a set of 295 species (trees, shrubs and herbs) we selected the species characteristic of *Fagellalia sylvatica* Pawl. in Pawl. et al. 1928, *Carpinion betuli* Issler 1931 em. Oberd. 1957 or *Erythronio-Carpinion* (Horvat, 1958) Marinček 1993 (Oberdorfer, 1964, 1994; Mucina et al., 1993; Marinček and Čarni, 2000). Within this pool of species, we chose Raunkiaer (1934) life form geophyte and hemicryptophyte (Verheyen et al., 2003; Whigham, 2004) with frequency above 20%; we considered only shade-tolerant forest herbs (Landolt L index ≤ 2), or forest herbs associated to mature humus (Landolt H index ≥ 4). *Pulmonaria officinalis* was not included in the Landolt (1977) list, so we obtained autoecological informations by Ellenberg L index (Pignatti, 2005), while there is no Ellenberg correspondence for Landolt H index. This approach allowed us to focus on strictly forest species, excluding heliophilous and edge herbaceous species (Honnay et al., 2005).

3.3. Survey of fragments

We recorded the forest fragments in the south-western part of the study area (Fig. 2), overlapping the aerial photos with 1:10,000 D.U.S.A.F. digital land cover [Destinazione d'Uso dei Suoli Agricoli e Forestali (Classification of Agricultural and Forest Lands); E.R.S.A.F., 2002]. We clipped the forest patches if crossed by roads, railways or water courses, considered as fragmentation sources (Peterken and Game, 1984; Dramstad et al., 1996). To reduce the number of environmental variables, we considered patches with homogeneous substratum, similar soil and forest composition. Fragments were selected using a stratified random sampling, to better represent the whole range of surface areas (≤ 0.2 ha, 11 fragments; 0.2–0.5 ha, 10 fragments; 0.5–1 ha, 13 fragments; 1–2.5 ha, 17 fragments; 2.5–5 ha, 9 fragments; 5–10 ha, 13 fragments; ≥ 10 ha, 21 fragments). The number of patches of the last category was limited by the number of forests available in the study area. Then, we carried out phytosociological relevés according to Braun-Blanquet method in the 94 selected fragments.

3.4. Data analysis

For each fragment, we measured three features: size, shape [$\gamma = (2\sqrt{\pi}A)/P$: Forman, 1995] and connectivity. We measured connectivity using the nearest neighbour connectivity metric (McGarigal and Marks, 1995) calculated with ArcGis 9.2 and a distance of 500 m. Analyses performed using different distance values (e.g., 200 m) gave the same results (not shown).

We used Spearman's correlation to evaluate bivariate relationships between forest indicators and environmental variables. Subsequently, we used generalized linear models (GLMs) assuming a Poisson error distribution to evaluate the relationship between the number of indicators and the three patch features. We used an information-theoretic approach, based on Akaike's Information Criterion (AIC), to individuate the best model, given the data. Models explaining most variance with a lower number of

predictors have smallest AIC and are considered “best models” (Burnham and Anderson, 2002). First, we built GLMs using all combinations of the three independent variables (total: 7 models). As AIC may select overly complex models, we considered a complex model as “candidate model” only if it had a AIC less than the AIC of all its simpler nested models (Richards, 2008). We also calculated Δ -AIC, which equals the difference in AIC values between the candidate model and model with the lowest AIC (i.e., the best model). We calculated Nagelkerke's R^2 (R_N^2) as a measure of the proportion of variance explained by each model; furthermore, we used hierarchical partitioning to evaluate the independent and joint contribution of each variable to the explained variation (Chevan and Sutherland, 1991). The residuals of our models were not spatially autocorrelated, suggesting that spatial autocorrelation does not bias the results of our analyses (Monte Carlo procedure, $P > 0.05$; Lichstein et al., 2002).

Subsequently, we used generalized additive models (GAMs) to evaluate the presence of non-linearity in the relationship between patch area and species richness. In GAMs, increasing values for the effective degrees of freedom (edf) indicate an increased complexity and non-linearity of the response curve (Wood, 2006); we therefore considered $edf \geq 2$ as an evidence of a non-linear relationship. We performed GAMs using Poisson error, as Poisson error is best suited to model species richness (Quinn and Keough, 2002); analyses using normal error and log-transformed species richness yielded identical results. We subsequently identified the presence and position of threshold(s) by visual inspection of the plots of fitted values (Potvin et al., 2005; Ficetola and Denoël, in press). To improve the reliability of the estimate, the position of threshold was assessed by five independent people and averaged. Simulation studies showed that GAMs can reliably estimate the position of thresholds using this approach (Ficetola and Denoël, in press). We performed statistical analyses using R 2.5 (R Development Core Team, 2007).

4. Results

4.1. Identification of forest indicators

The analysis individuated a pool of 16 herbaceous indicators (Table 1). Among the perennial herbs selected, 10 are hemicryptophyte and 6 geophyte. The most frequent are *Anemone nemorosa* L., *Polygonatum multiflorum* (L.) All. and *Primula vulgaris* Hudson, while the less frequent species, but present in more than 20% of patches, are *Aruncus dioicus* (Walter) Fernald, *Circaeae lutetiana* L., *Salvia glutinosa* L., *Luzula nivea* (L.) Lam. et Dc.

4.2. Relationship between indicators and features of fragments

Spearman's correlation showed that the number of forest indicators was positively related to patch area and connectivity, and negatively related to shape (Table 2). Furthermore, environmental variables were significantly correlated among them. Large patches were the least isolated and showed a more isodiametric shape. The negative correlation between area and shape was particularly strong (Table 2).

The analysis of AIC showed that the best model included two environmental variables: patch area and connectivity (Table 3). The other candidate models with AIC close to the best model were simpler subsets of the best model (Table 3). The model including area but non-considering connectivity had slightly lower support than the best model (Δ -AIC = 1.47). Models non-considering the effect of area had very high Δ -AIC (Δ -AIC > 20) (Table 3), indicating that these models have low support and therefore the role of shape in determining the number of indicators is negligible. The best AIC model (Model 1 in Table 3) showed that the number of forest

Table 1

Selected forest indicators, a subset of 16 perennial herbs chosen from a total of 295 forest species (included stand, shrub and ground level).

Species	Frequency (%)	R.S.	I	S.D.	M.R.	P.T.	Phyt	L.F.	H index	L index
<i>Anemone nemorosa</i> L.	87.5	3	ME/MY	PZ/MYR	0.20	S/SR	F	G RHIZ	4	2
<i>Aruncus dioicus</i> (Walter) Fernald	23.6	UNKN	UNKN	UNKN	–	UNKN	C	H SCAP	3	2
<i>Brachypodium sylvaticum</i> (Hudson) Beauv.	34.3	2	AN	AN/PZ	0.77	S/SC	F	H CAESP	4	3
<i>Carex sylvatica</i> Hudson	40.7	2	AN	PZ/MYR	0.30	S	F	H CAESP	4	1
<i>Circaea lutetiana</i> L.	23.1	3	MY	PZ	0.43	CR	F	H SCAP	4	2
<i>Euphorbia dulcis</i> L.	51.4	2	MY	MYR	–	S/SR	F	G RHIZ	3	2
<i>Geranium nodosum</i> L.	60.2	3	ME	PZ	–	S/SC	E	G RHIZ	3	2
<i>Lamiastrum galeobdolon</i> (L.) Eh. et Pol.	68.5	2	ME	MYR	0.22	SC	F	H SCAP	4	1
<i>Luzula nivea</i> (L.) Lam. et DC.	20.4	2	AN	MYR	–	S/CSR	F	H CAESP	4	2
<i>Oxalis acetosella</i> L.	38.9	3	AU/ME	MYR/AUT	0.26	S/CSR	C	G RHIZ	4	1
<i>Polygonatum multiflorum</i> (L.) All.	78.7	3	ME	EZ	0.56	C/SC	F	G RHIZ	4	2
<i>Primula vulgaris</i> Hudson	72.7	2	ME	MYR	–	S	E	H ROS	4	3
<i>Pulmonaria officinalis</i> L.	43.1	2	ME	MYR	0.39*	SR	F	H SCAP	–	5*
<i>Salvia glutinosa</i> L.	23.1	2	ME	PZ	–	C/SC	C	H SCAP	4	2
<i>Sympyton tuberosum</i> L.	49.5	2	ME	MYR	–	SR/CSR	F	G RHIZ	4	2
<i>Viola reichenbachiana</i> Jordan ex Boreau	32.4	3	AU/ME	MYR	0.44	S	F	H SCAP	4	2

R.S.: Reproductive strategy. 2 = discrete vegetative reproduction capability; 3 = good vegetative reproduction capability; UNKN = unknown (Banfi and Galasso, 1998). I: Impollination. AN = anemophylous; AU = autophylous; ME = melittophylous; MY = miophylous; UNKN = unknown (Oberdorfer, 1994; Banfi and Galasso, 1998). S.D.: seed dispersal. AN = anemochores; AUT = autochores; EZ = endozoochores; MYR = myrmecochores; UNKN = unknown (Oberdorfer, 1994; Banfi and Galasso, 1998). M.R.: Migration rates. Average migration rates of forest species in recent deciduous woods, measured in m year^{–1} (Brunet and Oheimb, 1998). * = referred to *Pulmonaria obscura*. P.T.: Plant traits. C = competitive; S = stress tolerant; R = ruderal (Grime et al., 1988). UNK = unknown. Phyt: Phytosociology. F = *Fagetalia sylvaticae* Pawl. in Pawl. et al. 1928; C = *Carpinion betuli* Issler 1931; *Erythronio-Carpinion* (Horvat, 1958) Marinček 1993. L.F.: Life form. G = geophyte; H = hemicryptophyte; RHIZ = rhizomatose; SCAP = scapose; CAESP = caespitose; ROS = rosulate (Raunkjaer, 1934). H index: Landolt (1977) Humus concentration index: 3 = humus-middle soil; 4 = humus-rich soil. L index: Landolt (1977) Light index: 1 = high shade-tolerant species; 2 = shade-tolerant species; 3 = almost shade-tolerant species. *. Ellenberg L index for *Pulmonaria officinalis*: 5 = half-light (>10%) (Pignatti, 2005).

indicators was positively related to fragment size ($\chi^2_1 = 57.6$, $P < 0.0001$). Moreover, the number of indicators was positively related to connectivity, although the relationship was marginally non-significant ($\chi^2_1 = 3.5$, $P = 0.06$). This model explained a large proportion of deviance ($R^2_N = 0.649$). Following hierarchical partitioning, area was the variable with the largest independent contribution (47% of total variation accounted for); the independent effect of proximity accounted for 9% of explained variation, and the joint effect of proximity and area accounted for 9% (Fig. 3).

4.3. Analysis of threshold

The generalized additive model showed that the relationship between indicator richness and fragment size was non-linear ($\chi^2 = 68.9$, edf = 3.6, $P < 0.0001$). The number of forest indicators increased until 35–40 ha; above this size, the richness was almost

constant (Fig. 4). Moreover, the relationship between area and number of indicators was less steep in forests smaller than 1–1.5 ha (Fig. 4).

5. Discussion

5.1. Forest indicators

All the selected forest herbs have a discrete to good vegetative reproduction capability (clonality), while the impollination is mostly performed by Hymenoptera (Banfi and Galasso, 1998; Oberdorfer, 1994). Seed dispersal is mostly myrmecochore and, in a lesser extent, epizoochore (e.g., Grime et al., 1988; Hermy et al., 1999; Honnay et al., 2005; Oberdorfer, 1994). Most species are limited in dispersal. For example *Anemone nemorosa*, *Lamiastrum galeobdolon*, *Polygonatum multiflorum* have a low migration (<0.05–1.15 m year^{–1}) and colonization rate (0.2–0.77 m year^{–1}; Bossuyt et al., 1999; Verheyen and Hermy, 2001, 2004; Verheyen et al., 2003). This explains the positive relationship between

Table 2

Spearman's correlations between number of forest indicators and environmental variables. N=94 for all correlations.

		N indicators	Area	Proximity
Area	r_s	0.647		
	P	<0.0001		
Proximity	r_s	0.398	0.425	
	P	<0.0001	<0.0001	
Shape	r_s	−0.561	−0.860	−0.499
	P	<0.0001	<0.0001	<0.0001

Table 3

Candidate generalized linear models relating number of forest indicators to patch features. In parentheses, the sign of the relationships between richness and patch features. Models are ranked according to their $\Delta\text{-AIC}$; the model with the lowest $\Delta\text{-AIC}$ is the best AIC model. $\Delta\text{-AIC}$: difference between the AIC of each model and the AIC of the best model.

Rank	Patch features	$\Delta\text{-AIC}$
1	Area (+); connectivity (+)	0.00
2	Area	1.47
3	Shape (+); connectivity (+)	20.97
4	Shape (+)	21.27
5	Connectivity (+)	55.56

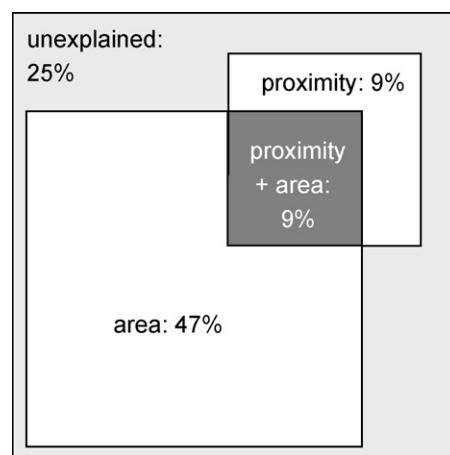


Fig. 3. Results of hierarchical partitioning of the influence (measured as percentage of explained variation) of patch area and proximity on the number of forest indicators. The area of each square is proportional to the variation explained by each variable.

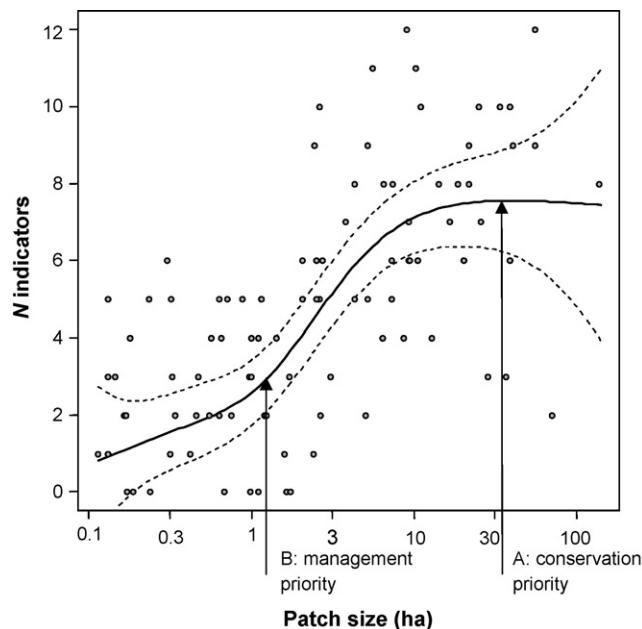


Fig. 4. Relationship between patch size (total habitat amount) and number of indicators. Bold line: regression line obtained from generalized additive models; dotted lines represent 95% confidence intervals. Fragments with area above the threshold A can be considered the optimum habitat where the maximum number of indicators can be found (conservation priority). Between A and B, species survival can be strongly improved by small increases in habitat cover.

species richness and proximity (our measure of connectivity), as dispersal-limited species can strongly suffer the negative effects of isolation (With and Crist, 1995). According to Grime's C-S-R plant functional types, which associate plant phenotypes (e.g., canopy, leaf height) with their role in community processes, most herbs have a 'S' strategy; they are stress tolerant and typical of the late-successional stages of forest dynamics (Grime et al., 1988). Therefore, this confirms that these species are characteristics of high-quality forests, and can explain their association with the largest, best conserved forest patches.

Several species were not included in the list of indicators because of their low occurrence in the references relevés (<20%). Among them, *Potentilla sterilis* may be a good indicator of forest status, as is included in the characteristic species of *Carpinion betuli* Issl. 1931, and is considered a good indicator for forest biodiversity (Dumortier et al., 2002).

5.2. Fragmentation: area and isolation

We have assumed that habitat loss, isolation and increased edge (measured here using shape) are cumulative consequences of fragmentation. Distinguishing the effect of these three processes can allow a better understanding of the effects of landscape alteration, and can improve management indications (Laurance, 2008). Several studies observed that habitat loss has the strongest effects, while the role of isolation remains debated (Fahrig, 2003). Our analysis suggests that patch size and isolation have partially independent effects; despite patch size is the factor explaining the largest amount of deviance, hierarchical partitioning showed that the independent effect of isolation is not negligible, and accounted for 9% of variation. Isolation may have negative consequences on the richness of indicators, because it negatively affects migration, and therefore hampers the colonization dynamics. Several fragments are not remnant of ancient woods, as they have been created in the 20th century (Sartori and Bracco, 1995); recently created forests can be colonized by forest herbs only if connectivity is high.

The relationship between patch area and species richness was the strongest one and explained most of variation. Many studies confirmed the positive relationship between patch area and biodiversity (Debinski and Holt, 2000; Whittaker and Fernández-Palacios, 2007), but several workers observed this pattern also when analyzing a guild of indicators, instead than the overall species richness (Yao et al., 1999; Laurance, 2002; Lorenzetti and Battisti, 2006). Our analysis is in agreement with these findings, and confirms that patch area is probably the most important feature to keep into account for management in the study area (see below).

5.3. Thresholds

The relationship between area and species richness is expected to be linear (on log-transformed axes or when Poisson regression is applied: Whittaker and Fernández-Palacios, 2007; Ficetola and Padoa-Schioppa, 2009). Deviations from the expected linear pattern suggest the presence of natural, geographic or human forces affecting species richness (Lomolino, 2002; Ficetola and Padoa-Schioppa, 2009). In this study, we observed a non-linear relationship between patch size and richness of forest indicators: richness was very low in patches below 1–1.5 ha, increased abruptly between 1.5 and 35–40 ha, and remained constant above 40 ha, where the curve seemed to reach an asymptote (Fig. 4). Despite this asymptote, the maximum species richness within a given patch was 12, and richness never reached the maximum possible value (16 species). Therefore, the presence of the asymptote cannot be explained by a limited pool of reference species. The value of 35–40 ha can be assumed as the patch size above which it is possible to find the maximum richness of forest herbaceous indicators. It is unlikely that our results are biased by the limited number of patches with larger area because, after the log-transformation, the frequency of large patches was not much lower than the frequency of small patches (see Fig. 4), and the confidence intervals of the GAM remained small until the threshold position (Fig. 4).

The presence of non-linearity in the species area relationship suggests the presence of thresholds that can be used for environmental planning. The preservation of fragments with area above the upper threshold (A in Fig. 4) can be considered a priority for conservation: these fragments are the only ones that can maintain the optimal richness of indicators. Indeed, the average richness of fragments above this threshold (approx. 8 species: Fig. 4) corresponds to the average richness of indicators in the reference forests (average 8.02 species). This further confirms the conservation value of patches above this threshold. Furthermore, fragments with area between the two thresholds (above B in Fig. 4) can be considered the optimal targets for restoration. When managing these fragments, small increases in size can quickly improve the richness of indicators.

6. Conclusions

Our pool of herbaceous indicators has been selected *a priori* on the basis of their ecological needs and Landolt indexes (see methods); their presence suggests that soil, humidity and humus correspond to the features of reference oak-hornbeam forests. Therefore, these indicators can help to identify the best conserved forest fragments. In human dominated landscapes, forest management is a key factor affecting forest regeneration (Godefroid et al., 2005; Van Calster et al., 2008). In Northern Italy, human activities strongly modified forest structure and floristic composition at least since the Middle Age (Giacomini and Fenaroli, 1958). For these reasons, ancient (*sensu* Peterken and Game, 1984), unmanaged forests are not present in the study

area; to identify reference forests we focused our attention on best conserved ones.

A major result of this study is the identification of two thresholds in the relationship between patch area and indicators, suggesting an abrupt response to changing landscape features. The exact position of threshold may be dependent on the context (Rhodes et al., 2008; Ficetola et al., 2009) or on the target species considered (Betts et al., 2007; Denoël and Ficetola, 2007). Nevertheless, our approach can be probably applied also to other fragmented landscapes, for the identification of management goals. In landscape planning, managers have to create new forest patches, or identify patches for restoration, improvement or enlargement. Using thresholds can allow a quantitative estimation of patch size, and can therefore help to identify the management priority or targets (Figs. 1 and 4). In many cases, afforestation projects endorsed by local environmental policies focus on improving existing patches to increase biodiversity, promote the restoration of agro-forested structures and support the connections between sinks and sources elements (Lassini et al., 2007). However, management is limited in resources, and limited quantitative tools are currently available to identify the optimal resource allocation and the ecological parameters needed to create woods with high value for biodiversity. The knowledge of factors affecting the distribution of forest herbs, and the analysis of thresholds, can provide key indications for planning.

To date, local afforestation interventions are mostly based on aesthetic or fruition, without taking into account ecological parameters such as forest structure, size and composition. We think that a correct environmental planning in suburban areas should evaluate both the aesthetic and the ecological aspects, for an optimal combination of citizens fruition and ecosystem preservation.

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