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FOREST ECOSYSTEMS FUNCTIONALITY EVALUATION BY MULTISCALE AND MULTITEMPORAL ANALYSIS OF ECOLOGICAL NETWORK

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The aim of this study was to model the relationships between forest spatial structure and potential forest vulnerability, resilience and resistance (FVRR) in Molise (Italy) on the basis of a multitemporal and multiresolution raster dataset. The FVRR index of forest ecosystems was quantified and mapped by two different approaches, such as a 'species-specific' and a 'species-unspecific' approach, modelling the proximity to a potential ecological optimum by fuzzy functions. The temporal trends between the summing up of index values of each model are similar. The 'species-unspecific' approach seems to be not scale dependent when applied for monitoring temporal trends, since trends of the index of low and high resolution datasets are nearly the same.

1. INTRODUCTION

Land is transformed by several overlapping spatial processes due to human activities or/and natural disturbances (Forman and Godron, 1984), which have varying effects on spatial pattern and ecological processes (Forman, 1995).

Although natural disturbances are usually pulse disturbances with characteristic magnitude and frequency, human activities tend to transform these disturbances into chronic ones that can reduce the recovery capacity of the ecosystem (Bengtsson *et al.*, 2003) influencing its resilience (Holling, 1973; Walker and Salt, 2006).

Resilience is one of the most important properties of forest ecosystems (Thompson, 2011), resulting from biodiversity at multiple scales, from genetic to landscape diversity (Thompson *et al.*, 2009). When minor disturbances over time occur, forests have a good capacity to resist (Van Andel and Grootjans,

2006). In general, ecosystems may be highly resilient, but have low resistance to a given disturbance (Thompson, 2011). However, most well-developed forests can be both resilient and resistant to changes (e.g., Holling, 1973, Drever *et al.*, 2006). The concepts of resistance and resilience are related to the stability of an ecosystem, that is the capacity to maintain a relatively constant state in spite of changes of environmental conditions (Van Andel and Grootjans, 2006). Forests are generally stable, and change little as a result of non-catastrophic disturbances (e.g. canopy gaps created by the death of individual or small groups of trees, quickly filled by new young trees) and may also be resistant to certain environmental changes, such as weather patterns over the time.

The habitat fragmentation is considered one of the major human-induced processes responsible for the loss of species diversity in forest landscapes (Wade *et al.*, 2003; Foley *et al.*, 2005). Fragmentation is a spatial process creating the 'breaking apart' of a habitat (Betts, 2000). Activated by several disturbance regimes at different temporal and spatial scales (White *et al.*, 1985; Turner and Dale, 1998), the fragmentation of forest ecosystems leads to a reduction in patch sizes, an increasing isolation of the remaining patches, and an increasing of the forest edges.

At European level, the problem of forest landscape fragmentation is recognized and explicitly addressed by several European policies (EC Biodiversity Communication 2006 on halting the loss of biodiversity by 2010 and beyond, with related Biodiversity Action Plan, EEA, SEBI2010, Ministerial Conference on the Protection of Forests in Europe - MCPFE), and correlated to a high number of specific indicators. One of the most relevant indicators for landscape fragmentation at European level is the 'fragmentation of natural and semi-natural areas' under the EU headline 'connectivity/fragmentation of ecosystems' (EEA 2007), which is complemented for forest ecosystems by the sub-indicator 'status and trends of forest spatial pattern per European bio-geographic region and per EU country' (Estreguil *et al.*, 2003; Estreguil *et al.*, 2007a), and the MCPFE indicator 4.7 'landscape pattern: landscape-level spatial pattern of forest cover' (MCPFE, 2007; Estreguil *et al.*, 2007a; Estreguil *et al.*, 2007b; Parvianen and Estreguil, 2007).

The amount of existing indicators shows the high relevance of forest spatial pattern in the context of European biodiversity policies and several efforts are ongoing to implement them in forest monitoring practices (Estreguil *et al.*, 2004; Estreguil *et al.*, 2007b). Among all spatial pattern indicators, an important parameter to be considered when assessing the forest fragmentation in Europe is the trend of forest 'core area', as it can provide a good description of the overall stability of the forest ecosystem under investigation (Estreguil and Mouton, 2009). In this study, the 'core' forests were mapped by the Morphological Spatial Pattern Analysis (MSPA), which was specifically developed through using the freeware software GuidosToolbox (Vogt *et al.*, 2007a; Vogt *et al.*, 2007b; Soille and Vogt, 2009), with forest/non-forest binary maps for both high and low resolution spatial scale (URL for download: http://forest.jrc.ec.europa.eu/download/software/guidos).

This software allows an automated *per* pixel classification and description of the geometry, pattern and connectivity of the forest landscape. The seven basic forest pattern classes provided by MSPA with a single edge width parameter(s) governing the entire classification process are described in Soille and Vogt (2009).

Taking into account the relevance of the forest fragmentation at European level and that no field data concerning the forest vulnerability, resilience and resistance (hereinafter FVRR) currently exists, this paper focuses on modeling the relationships between forest spatial structure and potential FVRR of a forest habitat, according to the correlation among vulnerability, resilience and resistance variables (as higher the level of resilience, higher the level of resistance and lower the level of vulnerability).

The relationships between spatial pattern information and FVRR were modeled by applying fuzzy membership functions (Van Ranst *et al.*, 1996; Groenemans *et al.*, 1997). Through the membership functions, we correlated the ecological factors to the optimal conditions, in order to produce a number of normalized maps of Habitat Suitability (HS).

The hypothesis is that FVRR of the forest habitat would resume the ecological distance between the real conditions (from maps) and the optimum ecological and biological forest conditions (found in literature review), depending on vulnerability, resistance and resilience-related factors.

The proposed aggregated FVRR index is thus tested by following a 'species-specific' approach, as well as a more general 'species-unspecific' one, which were oriented to describe the overall biological functionality of a forest habitat on the basis of general ecological rules. The set of selected ecological factors and fuzzy membership functions for each species are defined as ecoprofiles (or ecological profiles). Both tests were conducted in the Molise Region.

In the model, the extension and the spatial distribution of forest areas - where the overall ecological functionality of the habitat is high (minimum distance from the optimum) - are considered to potentially have a high possibility to both resist and recover after disturbances.

The values of the FVRR index were mapped by applying a geographical information system (GIS), in conjunction with a multi criteria analysis (MCA) and a multi criteria evaluation (MCE) (Eastman, 2006; Collins *et al.*, 2001, Jiang and Eastman, 2000).

Both 'species-specific' and 'species-unspecific' models were run for different dates and at two spatial scales, such as high (years 1936, 1954, 1992 and 2005) and low resolution (years 1990, 2000 and 2006). This was carried out in order to evaluate the scale dependency of the analysis.

Resulting values of the FVRR index for the 'species-specific' model (aggregated) were compared with resulting values of the index of the 'species-unspecific' model (for the different resolutions and years).

In order to take into account the influence of different moving windows filters applied in the model, all data are fully comparable just in a smaller area spatially defined at 100 km from the administrative boundaries of study areas.

2. METHODS

The present study was conducted in Molise Region (Southwestern Italy). The altitude ranges between the sea level (Adriatic sea) and 2,050 m a.s.l. of Monte Miletto (Matese Mountains). The mountainous areas (zones over 600 m a.s.l *sensu* L. n. 991/1952) cover the 78.68% of total area (349.149 ha).

From census data of 2015 from the Italian Institute of Statistics (ISTAT), the population of Molise is 313,348 inhabitants included in two districts: Campobasso (the regional capital, to the South-East) and Isernia (to the West) (Figure 1).
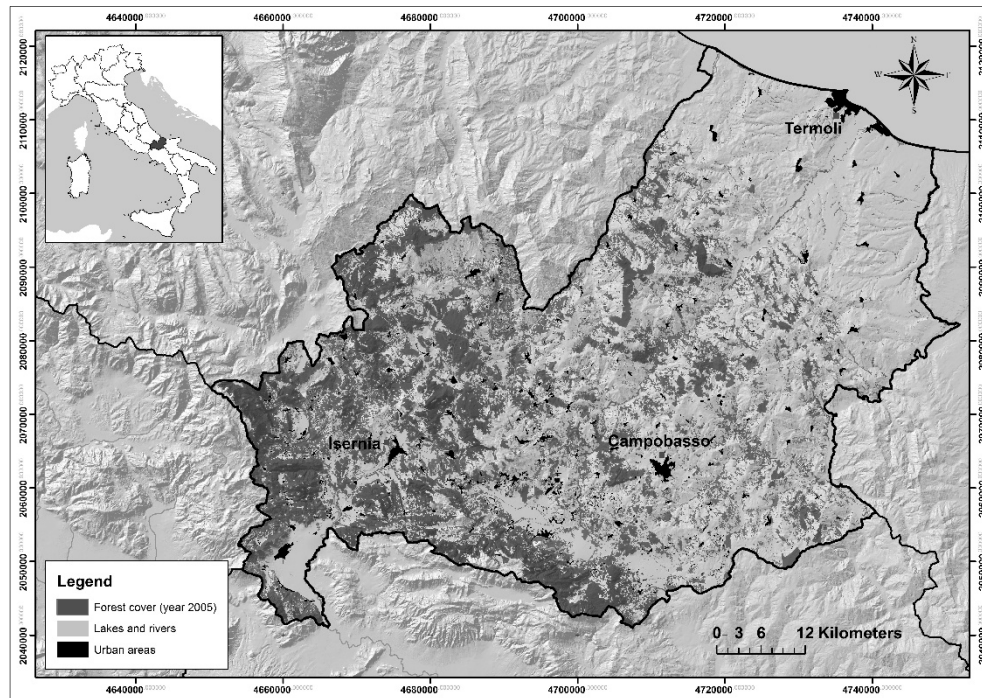

Figure 1 - Map of Molise Region showing forest cover, main urban areas and hydrographic network.

The regional climate, as consequence of the morphological difference between Apennine reliefs and the coast areas (to the East of the Region), presents a variety of meso-climatic conditions: average temperature of 14-16 °C in the close-to-sea zone (where negative average temperature is absent also in winter, and the rain is not very abundant) and temperature of about 10 °C (with, on average, two months with average temperature downgrading under 0 °C) in the mountainous zones of the Apennines (as in the Alto Molise zone, Matese and Mainarde mountains), where precipitation is abundant (average 1000 mm).

Forests are mainly located in mountainous and hilly zones. Their extension is favored by physiographic characteristics and by the recent abandon of agriculture. The district of Isernia is more forested than the district of Campobasso, where the landscape is dominated intead by agriculture (AppenninoVivoEuropa, 2002).

According to the Regional Forest types Map (Garfi and Marchetti, 2011), forests and other wooded lands coverage in Molise is 156,609 ha, representing about 35% of the regional area. The most represented forest categories are the mesoxerophil Turkey oak forest (about 19% of total forested area) and the mesophil Turkey oak forest (about 17% of the total forested area). Above 1000 m. a.s.l., forests are dominated by beech. They cover about 9% of the total forested area. A few represented but very important forest category for conservation purposes is the Silver fir forest. Pure stands of Silver fir cover 0.2% (343 ha), mixed with Turkey oak cover 0.4% (660 ha) and mixed with beech cover 0.02% (88 ha) of total forested area.

With regards to the management systems adopted, coppice forests are prevalent (53%). This system is mainly adopted in oak forest stands. High forests represent only 10% of total area, and are dominated by beech. The remaining part (37%) is represented by non-regular management systems.

Regional and national public forests are often subjected to conservation or protection policies. 36.3% of forest area (about 56,895 ha) are protected in Molise (Regional forests, Natura 2000 sites, Biosphere reserves and National Parks).

2.2. *Input datasets*

Forest/land cover maps were collected, implemented in a GIS (ESRI 9.10) (ESRI, 2010) and harmonized in terms of nomenclature system, projection and spatial resolution.

The nomenclature system was based on the main tree species composition and standardized accordingly to the European Forest categories (EEA, 2006). Data were projected in Lambert Azimuthal Equal Area projection system (ETRS-LAEA), according to the INSPIRE directive (15th May 2007). Forest/land cover maps were acquired as vector data, according to the spatial resolution, and two multitemporal raster datasets were created:

− 25 meters pixel for high resolution maps for the years 1936, 1954, 1992, 2005;
− 100 meters pixel for low resolution maps for the years 1980, 1990, 2000.

High resolution maps were available from different national and regional projects (Carta forestale del Regno d'Italia, year 1936; Progetto per il potenziamento del Sistema Informativo Territoriale, years 1954 and 1992; Regional Forest types Map, year 2005).

Low resolution multi-temporal datasets were derived from the Corine Land Cover (CLC) project. Both high and low resolution data were available for the entire Molise Region (about 443,758 ha).

Additional information were acquired, as follows:

− main road and railways networks (from http://www.pcn.minambiente.it/GN/);
− main towns and cities (available from CLC maps);
− boundaries of the main forest fires at European level at 1 km resolution (from EFFIS system: http://forest.jrc.ec.europa.eu/effis/);
− map of the Potential Natural Vegetation (PNV) at 1 km resolution (result of

the study "Forest habitat suitability distribution and future trends under Climate Change" - Tree species composition layer:
http://forest.jrc.ec.europa.eu/activities/climate-change/species-distribution/).

Each information was referred to the high and low resolution raster datasets, respectively.

2.3. *Evaluation of FVRR*

A potential ecological optimum was defined for both the five 'species-specific' approaches and the 'species-unspecific' approach (for a total number of six models), considering the relationships found in the literature review between forest spatial pattern, forest habitat vulnerability, resilience and resistance. At first, the found relationships were modelled using a specific logistic fuzzy membership function and to each forest pixel of the mapped FVRR indexes. Then, a value within the 0-255 range was given for representing the ecological distance of each forest pixel to the defined optimum. FVRR would vary from a value of 0 meaning that the site (i.e. forest pixel) is very far away from optimal ecological condition (low resilience and resistance, high vulnerability) to 255 (high resilience and resistance, low vulnerability), meaning that the forest pixel is optimal for the selected ecoprofile (species-specific or species-unspecific).

The 'species-specific' approach is based on the use of ecoprofiles species, belonging to the main species groups, with differing sensitivity to habitat fragmentation: marten (*Martes martes*) for small mammals, roe deer (*Capreolus capreolus*) and wolf (*Canis lupus*) for large mammals, lesser spotted woodpecker (*Picoides minor*) for birds and the butterfly lesser purple emperor (*Apatura ilia*) for insects. These models are different from traditional HS models because the final output variable cannot be considered a suitability index, but instead an aggregated FVRR index. The model output concerns a measure of the ecological distance between real and optimal conditions of forest habitats.

The 'species-unspecific' approach is based on the use of more general ecoprofiles. The model output is expressed in quantitative terms, as in the case of the 'species-specific' model, but in somehow referred not to a single umbrella species but to the general ecosystem functionality.

Models were run on the multitemporal dataset at the two different spatial resolutions. Forest spatial pattern factors were calculated *per* pixel (using circular moving windows) and/or *per* patch, and were the following: 1) forest and core forest patch sizes; 2) forest edge/core area ratio; 3) habitat percolation; 4) distance to neighbouring disturbing land uses (urban/settlements, roads, intensive agriculture) and from forest fires; 5) diversity and evenness of forest categories; 6) shape complexity; and 7) potential vegetation naturalness.

The MSPA applied to map core areas (forest habitats in this study) (by using GuidosToolbox) (Vogt *et al.*, 2007a; Vogt *et al.*, 2007b; Soille and Vogt, 2009) defines core pixels as those forest pixels whose distance to the non-forested areas is greater than the edge width parameter (*s*). All other forest pixels not corresponding to the core areas are assigned to one of the six other remaining pattern classes

(Soille and Vogt, 2009). The MSPA classification can be conducted with a 4- or 8-neighbourhood rule and we apply the default 8-neighbourhood rule, i.e. two pixels of the same class belong to the same landscape element if they share either one of their sides or vertices (Vogt *et al.*, 2007b).

For the 'species-unspecific' model, two edge width(*s*) of 25 m and 100 m, respectively, were used for high and low resolution input data. For the 'species-specific' model, the value of '*s*' was defined by taking into account the ecological profile of each considered species, such as: 25 meters from the forest patch border for *Apatura ilia*; 100 m for *Martes martes* and *Capreolus capreolus*; 100 m from the forest patch border of deciduous forest only for *Picoides minor*; and 1 km for *Canis lupus*.

The forest patch size, edge/core area ratio, diversity, evenness of forest categories, and shape complexity factors were calculated by using FRAGSTATS (McGarigal and Marks, 1995).

The habitat percolation factor was calculate per forest pixel according to the following 'percolation theory': when a forest pixel has in its surrounding a large area covered by forest pixels it is probable that the analyzed pixel is potentially connected with the other forest pixels, even if it is not physically adjacent (Farina, 1998). This theory was applied in order to emphasize the ecological role of stepping stones areas where forest patches are not physically connected but are sufficiently close to enable the movement of animals or the genetic dispersion.

The neighbouring landscape (to calculate the distance from disturbance sources) was defined by different size of circular moving window around each pixel: 5 km of *radius* for the 'species-unspecific' model and different size, on the basis of the home range of each considered species, for the 'species-specific' model (*Martes martes*: 4 km^2; *Capreolus capreolus*: 10 km^2; *Canis lupus*: 35 km^2; *Picoides minor*: 2 km^2; *Apatura ilia*: 0.8 km^2).

The potential vegetation naturalness of forests categories was calculated by contrasting forest/land use maps (the real condition for the model) with the PNV map. The ecological distance between real and PNV (the optimum conditions for the model) was based on the application of the phytosociological hierarchical system approach (Blasi *et al.*, 2000). In the resulting combination map, a value of 1 (high potential naturalness) was assigned to pixels of forest category falling in the PNV map category, and a value of 0 (low potential naturalness) to pixel not falling in the PNV map category and belonging to no-forest land use categories (e.g. urban categories).

2.4. *Species-unspecific model input factors*

In the species-unspecific model, the following nine factors were considered: 1) patch size; 2) core area size; 3) distance from neighbor disturbing sources; 4) forest edges/core area distribution; 5) potential percolation; 6) diversity and evenness of forest categories; 7) shape of patch; 8) disturbance by roads and settlements; 9) disturbance by forest fires and vegetation naturalness. Table 1 describes the species-unspecific model input factors factor, relative input data sets and fuzzy function used.

Table 1 - Factors used in the 'species unspecific' model.

FACTOR NAME	DATA INPUT	FUZZY FUNCTION		NOT APPLIED
		APPLIED		
		Type	*Thresholds*	
Patch size (per patch)	Forest maps	increasing logistic	0.5 ha and 100 ha	
Core area size (per patch)	Core/no core maps	increasing logistic	10 ha and 100 ha	
(¹) Distance from neighbor disturbing sources	Urban areas maps	increasing logistic	250 m and 1500 m	
	Roads and railways maps			
	Agricultural areas maps	increasing logistic	250 m and 500 m	
	Pastures areas maps			
Forest edges/core area distribution (per pixel)	Forest maps; Core/no core maps	Gaussian	potential optimal values = 50% presence of the core and 50% of edges	
Potential percolation (per pixel)	Forest maps	increasing logistic	50% and 100%	
Diversity (*Shannon Diversity Index*) & evenness (*Shannon Evenness Diversity Index)* of forest categories (per pixel)	EFC maps	increasing linear	one forest category and maximum number of forest categories in the biogeographical area	
Shape (Fractal dimension) (per patch)	Forest maps	linear increasing	1 and 2	
(²) Disturbance by roads and settlements (per pixel)	Urban and main routes maps	logistic decreasing	20% and 0%	
Disturbance by forest fires (per pixel)	EFFIS map	logistic decreasing	20% and 0%	
Vegetation naturalness (per patch)	EFC/land cover map; PNV map			To the resulting contrasting map between EFC/land cover and PNV map, was assigned: 1, to all pixel of forest category falling in the PNV map category (high potential naturalness); and 0 (low potential naturalness) to pixel not falling in the PNV map category (belonging to no forest categories of land uses e.g. urban categories).

(¹) A distance map was calculated from each disturb class and a different potential disturbing level was assigned: very high potential disturb (continuous urban areas and highways) = 4; high potential disturb (discontinuous urban areas, small settlements, roads and railway) = 3; moderate potential disturb (intensive agricultural areas) = 2; low potential disturb (pastures) = 1. The four disturbing factors were aggregated by weighted sum (adopted weights are equal to the relative disturbing factors) to obtain one single disturbing layer. In such a way, forest edges with pastures as neighbor have a very different potential ecological value than the same forest edge with urban area as neighbor.

(²) On the basis of urban areas, a distance buffer map of 250 m was created in order to prudentially take into account the effect on all the different species groups. On the basis of a circular moving window of 5 km radius, the percent amount of land impacted by anthropogenic disturbances was calculated for each forest pixel.

2.5. *Species-specific model input factors*

The approach adopted for the five species specific models is similar to that applied in the unspecific approach. The list and the description of the used factor are reported in Table 2.

Table 2 - Factors used in the five 'species specific' models.

MODEL NAME	FACTOR NAME	DATA INPUT	FUZZY FUNCTION APPLIED		NOT APPLIED
			Type	*Thresholds*	
1) Marten (*Martes martes*)	Forest core area in the home range (per pixel)	Core/no core maps	increasing logistic	70% and 100%	
	Forest patch size (per patch)	EFC maps	increasing logistic	150 ha and 300 ha	
	(¹) Anthropogenic disturbances (per pixel)	Urban areas maps; Roads and railways maps	increasing logistic	100 m and 2 km	
	(²) Type of forest	EFC maps			coniferous forest types equal to 1, broadleaf forest equal to 0.5
2) Roe deer (*Capreolus capreolus*)	Forest patch size (per patch)	EFC maps	increasing logistic	100 ha and 150 ha	
	Forest core area in the home range (per pixel)	Core/no core maps	Gaussian	minimum threshold of 40% of core forest, an optimum at 70% and a second minimum of 100%	
	Forest with open grassland/agriculture in the home range (per pixel)	Forest maps; Agricultural and pastures areas maps	Gaussian	minimum threshold of 20% open grassland/agricultural areas in the dispersal range, an optimum at 60% and a second minimum of 100%	
	Anthropogenic disturbances (per pixel)	Urban areas maps; Roads and railways maps	increasing logistic	100 m and 2 km	
3) Wolf (*Canis lupus*)	Forest patch size (per patch)	EFC maps	increasing logistic	10000 ha and 50000 ha	
	Anthropogenic disturbances (per pixel)	Urban areas maps; Roads and railways maps	logistic decreasing	minimum of 0.02% and maximum of 0.5%	
	Forest core area in the home range (per pixel)	Core/no core maps	increasing logistic	50% and 100%	
	Forest with open grassland/agriculture in the home range (per pixel)	Forest maps; Agricultural and pastures areas maps	Gaussian	minimum threshold of 5% of open areas in the dispersal range, an optimum at 20% and a second minimum of 50%	

(Continued)

(Tab. 2 - continued)

MODEL NAME	FACTOR NAME	DATA INPUT	FUZZY FUNCTION		NOT APPLIED
			APPLIED		
			Type	*Thresholds*	
4) Lesser Spotted Woodpecker *(Picoides minor)*	Forest patch size (per patch)	EFC maps	increasing logistic	10 ha and 100 ha	
	Forest core area in the home range (per pixel)	Core/no core maps	Gaussian	minimum threshold of 20% of core forest in the dispersal range, an optimum at 70% and a second minimum of 100%	
	Forest with open grassland/agriculture in the home range (per pixel)	Forest maps; Agricultural and pastures areas maps	Gaussian	a minimum threshold of 10% of open areas in the dispersal range, an optimum at 40% and a second minimum of 60%	
	Anthropogenic disturbances (per pixel)	Urban areas maps; Roads and railways maps	logistic decreasing	optimal minimum of 0.2% and a maximum of 5%	
	Distance from human activities (per pixel)	Urban areas maps; Roads and railways maps			Buffer area of 100 m around settlements, urban areas, roads and railways was considered not suitable
5) Lesser Purple Emperor *(Apatura ilia)*	Forest patch size (per patch)	EFC maps	increasing logistic	15 ha and 65 ha	
	Forest with open grassland/agriculture in the home range (per pixel)	Forest maps; Agricultural and pastures areas maps	Gaussian	minimum threshold of 10% of open areas in the dispersal range, an optimum at 40% and a second minimum of 60%	
	Forest core area in the home range (per pixel)	Core/no core maps	Gaussian	minimum threshold of 20% of core forest in the dispersal range, an optimum at 70% and a second minimum of 100%	
	Anthropogenic disturbances (per pixel)	Urban areas maps; Roads and railways maps	logistic decreasing	minimum of 50 m and an optimum at 500 m	

[1] On the basis of the distance from settlements, urban areas, roads and railways.
[2] Martens prefer old growth or mature coniferous forests. Especially in winter. The use of broadleaf forest is limited.

The model produced five different spatial analyses showing the biological distance between the real habitat environmental condition and the optimal habitat condition on the basis of each species ecoprofile. All resulting maps for each considered environmental factor and for each species are expressed, as for the 'unspecific approach', in the 0-1 range. A value equal to 0 means that the forest pixel is very far away from optimal ecologic condition, while a value of 1 express that the pixel is optimal for the selected species.

2.6. *Aggregation of factors*

For each model, ecological factors were combined in a MCE (Eastman, 2006; Collins *et al.*, 2001) based on a WLC (Eastman, 2006), currently adopted for various applications (e.g. Banai, 1993; Jankowski and Richard, 1994; Joerin, 1995; Barredo, 1996; Beedasy and Whyatt, 1999; Malczewski, 1999; Barredo et al., 2000; Mohamed et al., 2000; Bojorquez-Tapia et al., 2001; Dai et al., 2001; Joerin et al., 2001; Church, 2002), in order to have a sensibility analysis of the approach.

The MCE was based on a Weighted Linear Combination (WLC) (Eastman, 2006). Since the interpretation of the results is carried out in a comparative way (different resolutions and different years), the relative differences never vary if all the tests are carried out with the same methods.

All the output maps resulting from the MCE analysis were expressed in the 8 bit range between 255 and 0. A value of 255 means that the given pixel fulfills at 100% the biological requirements of local ecological optimum: for a given factor the level of resilience and resistance is maximum and the vulnerability is the minimum. At the opposite, a value of 0 represent a biological condition very far away from local ecological optimum: potential high vulnerability, low resilience, and low resistance of forest habitats (Figure 2).
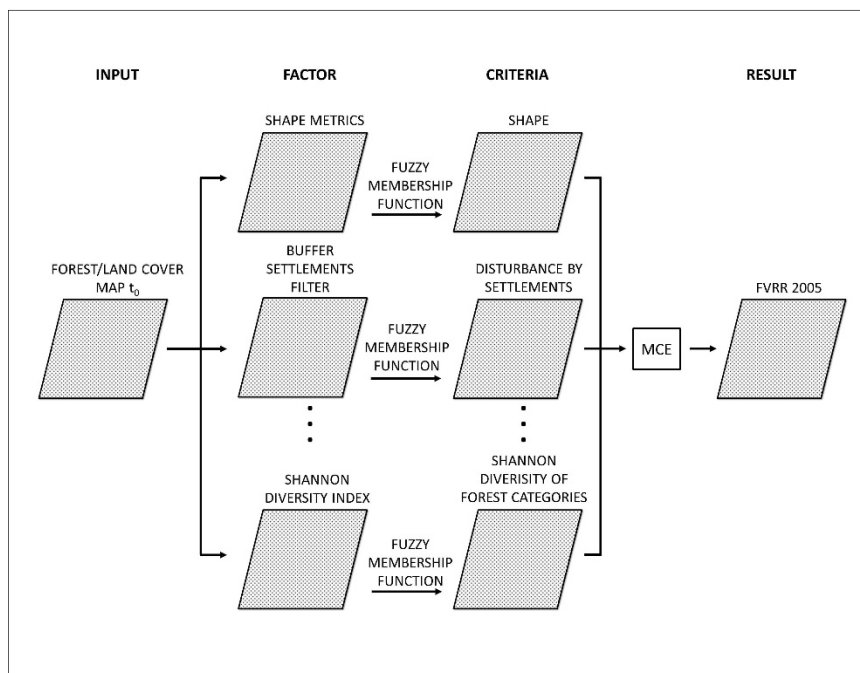

Figure 2 - Flowchart of the applied approach to model FVRR of forest habitats. Factors calculated for each multitemporal dataset and for both high (years 1936, 1954, 1992, 2005) and low (years 1980, 1990, 2000) spatial resolution data were normalized on the basis of fuzzy membership functions. Results were aggregated with MCE approach which results is one unique FVRR map with values expressed between 0 and 255. The complete procedure was repeated for both 'species unspecific' and 'species specific' models.

2.7. *Analysis of data*

Descriptive statistics of the FVRR indexes (extracted within a smaller area defined at 100 km from the administrative limit of study areas) are stratified in three different classes: forest area, core forest area (according to the MSPA) and no core forest area (all GuidosToolbox classes not belonging to core class). For both unspecific- and species-specific model and for both high and low resolution maps, the results of the FVRR index were presented as average and sum statistics. Average statistics were used to discuss the theoretical relationships between the models adopted, the scale of the analysis (high vs. low resolution maps), and the type of forest areas (core vs. no core). The sum statistics are used to describe the temporal trends since they are influenced both by the change in the forest area and the change in the results of the FVRR models.

3. RESULTS

3.1. *Forest cover trend in the study area*

The forest cover trend was analyzed for both high (years 1936, 1954, 1992, 2005) and low resolution (years 1980, 1990, 2000) maps. The two datasets are not fully comparable because none of the high/low resolution maps is acquired at the same date, but the difference can be estimated around 7,000 ha (about 6% of forest area). On the basis of high resolution maps, forest area changed from 239 km² in 1936 to 860 km² in 2005, with an increase of 260%, and the increasing was nearly linear over the time.

Forests increased in the period 1936-1954 of 3.4% yearly, of 1.9% in 1954-1992 and of 2.2% in the period 1992-2005. Low resolution maps confirmed similar trends with a yearly increase in forest area of 1.5% in the period 1980-1990 and of 2.0% in the period 1990-2000.

The trend of forest area in no core areas (Figure 3) is similar to that of general forest area. In low resolution maps, the trend for core areas tends to overestimate the estimated area. As a result, the ratio of core and no core areas is strongly affected by the maps resolutions. The increasing in forest area was meanly due to the abandon of marginal agricultural and pasture in mountain areas and these new forest areas are prevalently no core areas.

3.2. *Species-unspecific analysis*

The temporal trend of the FVRR index (Figure 4b) can be considered not scale-dependent, as revealed by the similar temporal trends between low and high resolution data, and as expected, the average value of the index (Figure 4c) is significantly higher in core areas than in no core areas, with an average difference of 23%.

The overall trend of the FVRR index in the high resolution maps in core areas is lower than in no core areas (Figure 4d). This can be explained by the strong

presence of no core areas. The opposite effect was revealed in low resolution maps, where core areas have an overall index value higher than in no core areas. At both resolutions, the temporal trend in core areas is weaker than in no core areas. This trend can be referred to the cumulative effect of the increasing forest area (especially of no core areas, at least in high resolution maps) and of the average values of the index.
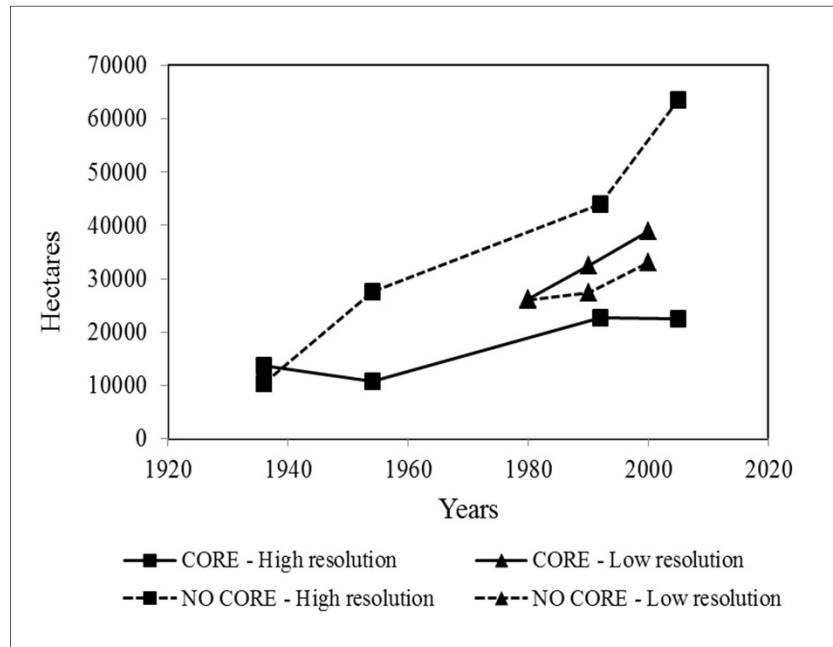

Figure 3 - Comparison of trends in forest area for core and no-core areas.

3.3. *Species-specific analysis*

As expected from the theoretical background in ecology, the average value of FVRR in core areas for *Picoides minor* is always much greater than in no core areas (Figure 5-1b), in both resolution maps, and the temporal trend is always increasing (Figure 5-2b). The trend in core and no core areas in high resolution maps is almost the same. In low resolution maps, since the area of no core forest is lower than in high resolution maps, the higher average value of core area determined a higher value in this class.

For the butterfly *Apatura ilia*, the relationship between the FVRR index and forest area type (core and no core) (Figure 5-1c) is not linear and in general the index is higher in no core areas than in core areas. The temporal trends of the butterfly index (Figure 5-2c) clearly show higher values of the FVRR index in forest no core area for both resolution maps. In core areas, the index has a stronger increasing rate moving from 1936 to 2005 than in no core areas, and this forest class seems therefore to have ecological conditions closer to the potential optimum for this species.
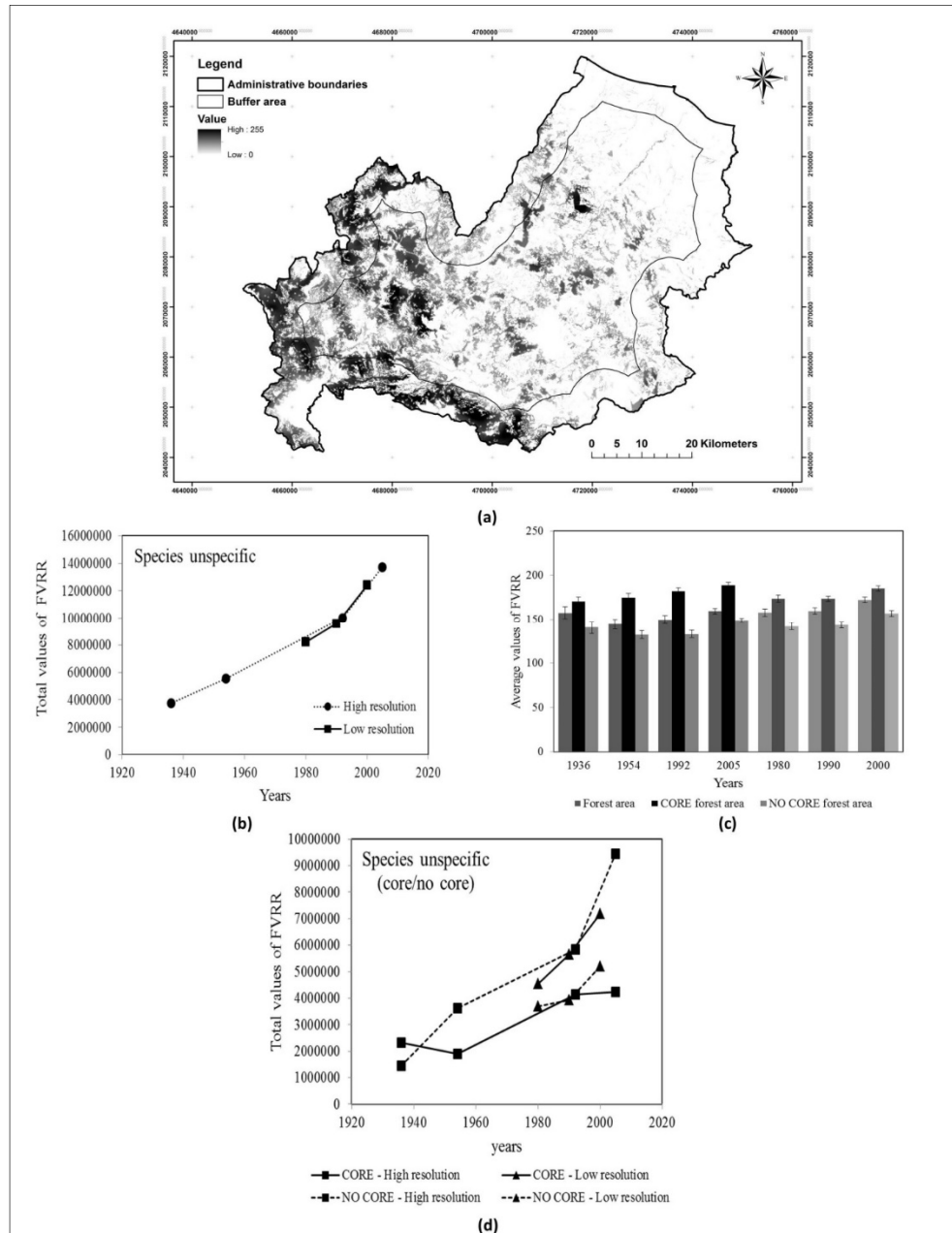
Figure 4 - Results of the 'species unspecific' model: a) example of the FVRR index map for the high resolution data (year 2005). The picture is presented in 8-bit range of values (0-255) where 0 express minimum value of FVRR index and high distance from local ecological optimum conditions (potential negative condition of forest habitats and potential high vulnerability, low resilience, low resistance) and 255 express the maximum value of FVRR index and low distance from local potential optimal conditions (potential optimum of forest habitats conditions and low vulnerability, high resilience, high resistance); b) comparison of the temporal trends of the FVRR index; c) comparison of average values of the FVRR index (with standard error) for both high (dark grey scale: years 1936, 1954, 1992, 2005) and low (lighter grey scale: years 1980, 1990, 2000) resolution data; d) comparison of the temporal trends of FVRR index in core and no-core areas.

For marten, the average values of the FVRR index in core areas (Figure 5-1d) are always higher. This result is clearly correlated to the ecological preferences of this species.

The differences between forest classes (core and no core) and their internal variability are similar for all the considered years and resolutions. The temporal trend of the FVRR index for this species (Figure 5-2d) is always increasing. The trend is very similar to that one of the woodpecker model (Figure 5-2b).

According with the ecoprofiles adopted in the model of roe deer, the average values of the FVRR index (Figure 5-1e) are always higher in core areas than in no core areas for all the years and resolutions. The temporal trends of the index (Figure 5-2e) are similar to those ones of woodpecker (Figure 5-2b) and marten (Figure 5-2d), and considerations previously made are valid also for the roe deer.

The wolf FVRR index is based on a different ecoprofile, in comparison with that used for the others. Since the wolf is very demanding for large undisturbed continuous core areas, the values of the index in the years 1936 and 1954, when the forest area were always very limited (Figure 5-1f). In the other years, the average values of the index are higher in core areas. In core areas from low resolution maps, the variability of the index is very high but the values are constantly increasing from 1980 to 2000. The temporal trends of the index (Figure 5-2f) are strongly increasing after the year 1980. The increasing rate is higher in core areas than in no core areas and the trends are similar for high and low resolution data.

The trend of the FVRR index calculated for all considered species and for all species aggregated is also reported in Figure 5g and shows a similar increasing trend for high and low resolution data. The FVRR index of wolf, compared to the other species is still lower, since the wolf is very demanding for large undisturbed forest areas.

3.4. *Comparison between models*

The comparison of the FVRR index values for both general and species-specific models and for both high and low resolutions datasets increase with the increasing of the forest area over time (Figure 6a). Although there is a difference in absolute value, the annual average percent change rate of the FVRR index belongs to the same distribution, thus demonstrating a linear relationship with the percent change rate in forest area (Figure 6b). Even the comparison between FVRR annual change rates for the four species models demonstrates a close linear relationship (Figure 6b).

The comparison between results shows that both all aggregate species-specific models and the unspecific model give the same information with regards to the annual change rate of the index as revealed by the distribution of the index values close to the bisector in the scatterplot (Figure 6c).
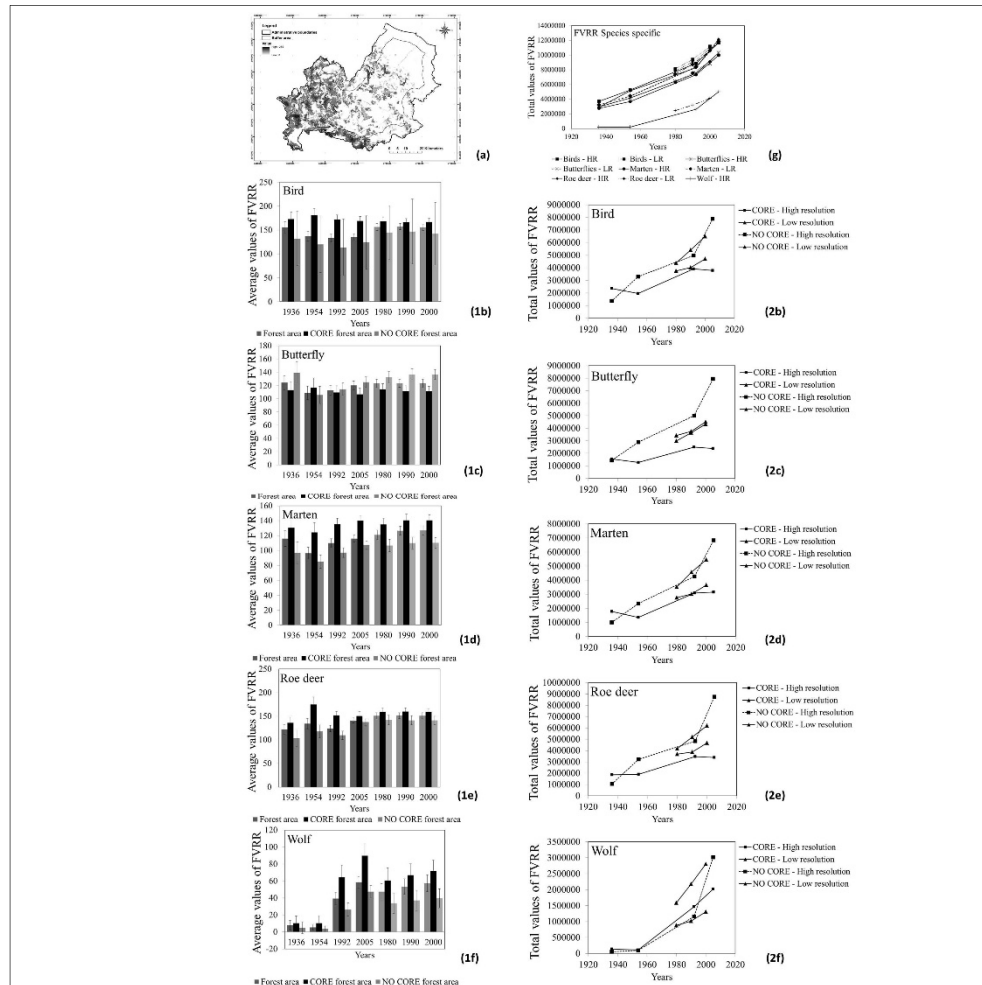
Figure 5 - Results of 'species specific' models: a) example of the FVRR index map summing up all the five umbrella species based on multitemporal high resolution forest maps (year 2005); 1b to 1f) comparison of average values (with standard error) of the FVRR index for each species specific model for both high (dark grey scale: years 1936, 1954, 1992, 2005) and low (lighter grey scale: years 1980, 1990, 2000) resolution data; 2b to 2f) comparison of trends in FVRR index; g) comparison of trends in FVRR index for all the umbrella species and summing up all species.

This can be explained as the consequence of the abandonment of mountain areas after the II World War (mainly located into the North-West of the study area), resulting in an intensive natural reforestation process creating new wooded lands. The newly formed landscape structures, as well as forest patches and vegetation corridors, facilitated also the connectivity between forest habitats and a general enhancement of the ecosystem functionality, as reveled by the models results.

Resolution of data influenced the MSPA and classification of core and no core areas in different ways. The small patches, often no core areas, are not detected in low resolution maps. On the other hand, high resolution maps present less compact patches that lead the MSPA analysis to identify more no core areas.
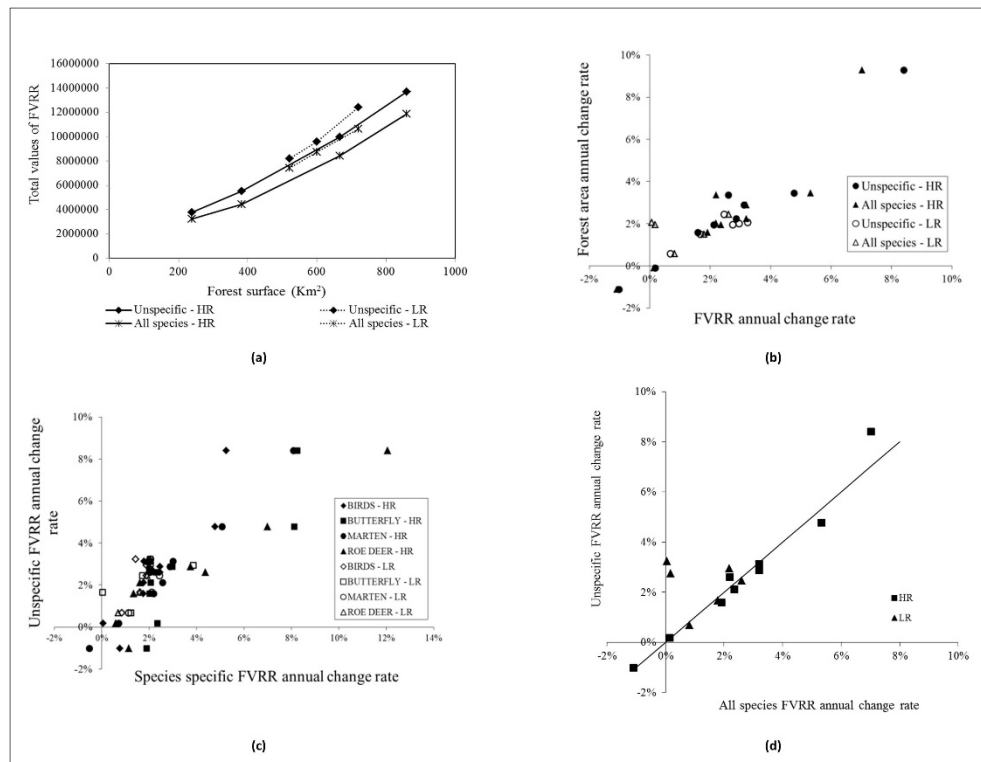
Figure 6 - Results of comparison between models: a) comparison of trends increasing forest area and the FVRR index (for species unspecific and all species specific models aggregated); b) relationship between the annual trend of FVRR index for all the considered models (species unspecific and all species specific models aggregated); c) relationship between the annual trend of FVRR index for four umbrella species and the annual trend of species unspecific FVRR index; d) scatterplot between all species and rate of unspecific annual change of the FVRR index.

The species-unspecific model seems to well designate the overall biological functionality of the forest habitats. Resulting average values of FVRR index are always higher in core areas than in no core areas for all the multitemporal and multiscale datasets. Since the trends of low and high resolution datasets are nearly the same, the general model applied in this area resulted not scale dependent when applied for monitoring temporal trends, overpassing the problem due to the scale dependency of other traditional spatial indexes.

Outcomes achieved from applied species-specific models are redundant, at least the models developed for woodpecker, marten and roe deer, since they show the same temporal trends. On the contrary, the models developed for butterflies and wolf are less redundant.

All these three species showed a positive temporal trend of the FVRR index related to the increasing of forest cover and a consequent modeled limited distance from their maximum ecological optimum (low vulnerability, high resilience and resistance) and they seem to be good indicators of the forest landscape status.

The increased FVRR values over the time for these three species designate an improved ecological condition as a consequence of the decreased forest fragmentation.

While woodpecker and marten FVRR indexes allowed monitoring mainly the changes in core forest area, roe deer model allowed analyzing also the increasing in forest connectivity, since this species requires large and heterogeneous landscapes with mixed forest and opening areas.

The results for butterfly *Apatura ilia* and wolf models showed more variability compared to the other species models, as a consequence of the different ecological preferences.

According to the selected ecoprofile, *Apatura ilia* prefers external forest borders and, as shown by the analyses, the higher FVRR average values are more related to no core forest areas both for high and low resolution data. As a consequence, the total FVRR index value showed a higher positive temporal trend in no core areas than in core ones. Furthermore, the slight decrease of FVRR index values in the high resolution contrasts with the positive trend in the same period in the low resolution dataset.

The wolf model was based on a different ecoprofile, with respect to the other species. Results showed a low value of the FVRR index in the years 1936 and 1954 when forest area was limited, but in the other years, average FVRR values strongly increased with higher values in core areas. For wolf, the values of FVRR index are lower respect to other species model. For this species, forest habitats in the study area seem to have high distance between real and optimal ecological conditions (high vulnerability, low resilience and resistance). However, in the analysis, the largest part of the suitable areas for wolf (in particular the areas from Matese relief to the South-West and from Mainarde relief to the North-West), were not considered because located out of the buffer area used for the analysis of results. The actual limitation of the moving window approach strongly limited the extension of the analyzed area (within the buffer zone). For this reason, important forest habitats were not considered in the analysis of the results. Nevertheless, the problem of moving windows influence could be avoided including in the analysis a larger area outside the defined test area.

The aggregation of all the species specific models produced an overall average value of the FVRR index. The comparison between the sum of the index values of the general species unspecific model and the sum of all the species specific models showed a high degree of correlation both for high and low resolution data. The temporal trends are consequentially very similar.

4. CONCLUSION

In conclusion, the approach we developed seems to be able to monitor the resistance, resilience and vulnerability of forest areas on the basis of their proximity to the potential ecological optimum for species unspecific and species specific models.

This approach can be potentially and operatively applicable on wider areas. In particular, the species-unspecific model demonstrates to be efficient and easier-to-be applied than the species-specific model.

The models based on high resolution data are able to track the temporal trends of forest areas in a more detailed way, especially when the two observations are near among each others (e.g., 10 years or less).

The obtained results confirm the following thesis: if results (in terms of temporal trends achieved at different resolutions with the same models) are similar, it can therefore infer that this monitoring approach is not scale dependent. The level of similarity expresses the level of scale independency of the analysis.

If implemented at pan-European level, these models may provide a cognitive analytic instrument supporting forest planning and biodiversity conservation measures.

Conservation strategies aiming to reduce forest landscape fragmentation and the loss of ecological functionality of forest habitats, cannot be limited to field-based assessment on sampling designs, but need to integrate the investigation of temporal dynamics at landscape level. In this way, the development of new operational methodologies to analyze forest landscape spatial pattern becomes an essential goal.

The evaluation of forest landscape patches and their spatial pattern can significantly contribute to support activities promoting the connectivity restoration between fragmented habitats and, consequently, mitigating the loss of biological communities, populations, and entities.

If the local indexes values are high (close to the potential maximum value), it is possible to infer that in the considered area forest habitats have higher potential resistance and resilience against external disturbing factors. If the values of the index instead are near to zero, the habitats are more potentially threatened.

All the here presented results were acquired on the basis of a MCA based on WLC procedure, through assigning the same weighs to all the considered factors (or criteria) (for both species unspecific an species specific models). Adopting the WLC procedure with different weights, assigned on the basis of a Saaty pairwise comparison matrix could be an alternative approach (Saaty, 1977). In the context of decision making methods, this process is known as the Analytic Hierarchy Process (AHP) (Eastmann *et al.*, 1995). This technique could be useful to compare the relative importance of each pair of factors for determining the FVRR indexes. As already applied in the study of Freire and Maseda (2008) concerning the analysis of the anthropogenic effects on dispersal pattern of wolf in Northwestern Spain, a variance analysis on the MCE outcomes can be carried out in future studies in order to evaluate the subjectivity inherent within decision-making processes for the determination of FVRR values.

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RIASSUNTO

Valutazione della funzionalità degli ecosistemi forestali mediante un'analisi muti-scala
e multi-temporale della rete ecologica

L'obiettivo di questo studio è quello di modellizzare le relazioni tra la struttura spaziale delle foreste e le loro caratterisitiche di vulnerabilità, resilienza e resistenza potenziali (*Forest Vulnerability, Resilience and Resistance*, FVRR) in Molise, utilizzando dati *raster* multi-temporali e multi-risoluzione. L'indice FVRR degli ecosistemi forestali è stato quantificato e mappato utilizzando due approcci diversi, uno "specie-specifico" e l'altro "aspecifico", simulando la prossimità rispetto ad un potenziale *optimum* ecologico attraverso funzioni *fuzzy*. Gli andamenti tra i valori di sintesi dell'indice FVRR di ogni modello risultano simili. L'approccio "aspecifico" sembra non dipendere dalla scala quando è utilizzato per monitorare gli andamenti dell'indice FVRR, considerando che questi ultimi risultano essere pressoché gli stessi relativamente ai dati a bassa ed alta risoluzione.

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