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Implementation of multispecies ecological networks at the regional scale: analysis and multi-temporal assessment

Abstract

Today, major landscape changes affect ecological connectivity exerting adverse effects on ecosystems. Connectivity is a critical element of landscape structure and supports ecosystem functionality. Landscape connectivity can be efficiently increased in landscape ecology by building ecological networks (EN) through models mimicking the interaction between animal and vegetal species and their environment. ENs are important in sustainable landscape planning, where they need to be studied both by applying landscape metrics and by performing multi-temporal analyses. This paper presents theoretical and practical evidence of an analysis of a multispecies ecological network in Calabria (Italy) and its changes over three decades. Landscape connectivity was modeled basing on 66 focal faunal species' requirements. Human disturbance (HD) was defined and assessed according to distance from different disturbance sources. This allowed for the definition of overall habitat quality (oHQ). Landscape permeability to the animal movement was focused as the main concept to measure landscape fragmentation. Landscape graph theory was applied to perform a spatial comparison of the ENs robustness. Many binary and probabilistic indices and landscape morphological spatial pattern analysis (MSPA) were used in this perspective. We obtained a set of ecological networks, including nodes, patches (i.e., habitat patches), linkages, and corridors, all intertwined in one giant component. The multi-temporal analysis showed many indices' stationary values, while MSPA yielded an increase of habitat quality and habitat patches in core areas. This methodological approach allowed for assessing the regional EN's robustness in the time-span considered, thus providing a reliable tool for landscape planners and communities.

KEYWORDS: Habitat patches; Landscape connectivity; landscape graphs model; multi-temporal assessment; landscape fragmentation; morphological spatial pattern analysis (MSPA).

1 Introduction

Worldwide, land use/land cover changes are widespread and progressively changed the urban-rural landscape arrangements, with a dramatic acceleration in the 20th century (Kienast et al., 2019), and significant adverse effects on ecosystem functionalities (Defries et al., 2004; S. Di Fazio et al., 2011; Modica et al., 2012; Vitousek, 1997). While certain land use/land cover modifications lead to landscape fragmentation (LF) - i.e., the process of subdivision of large habitats into smaller and more isolated patches (Battisti, 2003; De Montis et al., 2018; European Environment Agency (EEA), 2011; Fichera et al., 2015), other actions are conversely increasing landscape homogeneity (Farina, 1998; Forman, 1995). These dynamics are generating complex transitional landscapes, in which natural components interact with urban and rural ones in a continuous urban-rural-natural gradient (Vizzari et al., 2018; Vizzari & Sigura, 2015). Among land use/land cover changes trajectories, land abandonment and soil degradation represent two opposite phenomena that are progressively reducing landscape quality (Modica et al., 2012; Modica et al., 2017; Statuto et al., 2018). They both descend from intense progressive urbanization and population migration from rural to urban areas mainly occurred over the 20th century. A critical and common phenomenon in Italy - and some other southern European regions- is the so-called 'sprinkling' contributing to land take processes and LF (Romano et al., 2017; Saganeiti et al., 2018).

The loss of habitats, with consequent threats to biodiversity (Fahrig, 2003), is a major effect of the phenomena described above. In the last century, in Italy, as in most European countries, an ever-increasing depletion of natural resources and biodiversity loss, caused by unbalanced forms of land anthropization, has been paralleled by progressive growth in the number and surface of natural protected areas, as a matter of compensation (Di Fazio & Modica, 2018). The European Landscape Convention (CoE, 2000) has compelled the European governments to change their cultural vision and policies to integrate heritage and nature conservation in spatial planning and attribute primary importance to landscape (OECD, 2007). Methodological approaches to the definition of planning tools, either structural/general or specific and action-oriented, have been greatly influenced by this change of perspective. On the one hand, ecological approaches have gained significant consideration in view of sustainable landscape planning, which can be effectively pursued by taking into account the ecological network (EN) (Opdam et al., 2006). On the other hand, ENs, when set into the framework of spatial planning, need to be conceived in a different, more functional and usable way. It has been observed that the use of landscape ecological-based methodologies for the definition of an EN can reduce some shortcomings arising in a species centered approach. Since their focus is not on individuals but habitats, landscape ecology-based methodologies typically overcome the lack of information about the various animal and vegetal species' needs. This allows for spatial landscape metrics, based instead on data readily available (land use, land cover), and enables a multitemporal landscape analysis to detect habitat changes, in both quantity and quality (Fernandes, 2000; Botequilha Leitão & Ahern 2002).

In landscape planning, time is a variable as important as space. Since the landscape is a living entity (Steiner, 2008), it cannot be appropriately studied only by considering its present state as a static configuration. It must be seen from a historical perspective to investigate its ongoing change dynamics (Di Fazio & Modica, 2018).

The debate about the ENs' reliability in conservation and landscape planning is still open. Several researchers recently endorsed them as a useful framework for informing decision-makers in sustainable landscape planning (Babí Almenar et al., 2019; De Montis et al., 2016; Ersoy et al., 2019; Foltête, 2019; Tarabon et al., 2020). Nevertheless, spatial ENs implementation still deserves a finer focus. In this respect, there is a research gap in the studies concerning the mapping resolution grain most suitable to assess and design the landscape connectivity (LC) patterns at different scales (state, region, province) (Beier et al., 2011), the number of faunal species to be considered and the temporal analysis of the related habitat requirements.

To deal with the design of ecological corridors, some researches (e.g., Belote et al., 2016; Brodie et al., 2015) proposed the exclusive use of protected areas' boundaries, but this approach has been criticized, considering that in this way potential suitable patches could be excluded in the EN design (Beier et al., 2011).

Several studies modeled ENs taking into consideration one (Ehlers Smith et al., 2019; Hofman et al., 2018; S. Liu et al., 2018) or few focal species (Babí Almenar et al., 2019; Ersoy et al., 2019; C. Liu et al., 2018; Préau et al., 2020). The recent work of Tarabon et al. (2020) implemented the design of landscape connectivity modeling the requirements of twenty faunal species in Toulouse's conurbation (France).

In this research, we respond to the following questions, i.e. how to: i) construct ENs based on the habitat requirements (i.e., autecological needs) of numerous faunal species at the regional scale; ii) account for the dynamics occurring on a landscape and, therefore, on the EN design, in the framework of a sustainable landscape planning; iii) assess the robustness of an EN.

Our main contribution is to provide a methodological framework for integrating multiple data-sources into a consistent EN. Its novelty relies on the combination of: i) multispecies LC analysis at a regional scale, ii) multi-temporal assessment, and iii) robustness assessment.

In the region of Calabria (Italy), concerning three different years (1990, 2012, and 2018), it was defined an EN based on the choice of 66 representatives (i.e., focal or umbrella) faunal species, to cover the resource requirements of most wildlife species in the region. Moreover, we focused on the change of LC over time, developing a multi-temporal assessment that accounts for landscape evolution trends. In this respect, our primary objectives are to analyze ENs robustness and LF's role by interpreting the spatial change patterns of physical constituents (i.e., the different land uses) and qualitative constituents (distribution of habitats and landscape permeability).

Therefore, our research is the first that proposes the implementation of a multispecies EN at a regional scale based on the requirements of dozens of focal species and considering a diachronic assessment over nearly thirty years. Beyond the inclusion of integral natural reserves and Sites of Community Interest (SCIs) belonging to the so-called Natura 2000 European network (Council of Europe, 1992), we considered those areas with high suitability to the animal movement and acting as either habitat patches or corridors.

The paper is organized as follows. Section 2 presents a state of the art summary on landscape connectivity and multi-species EN. Section 3 illustrates the study area. Section 4 shows the input data (sub-section 4.1), explains step-by-step the rationale of the methodology adopted (sub-section 4.2), and describes the assessment and the diachronic comparison of the ENs (sub-section 4.3). In Section 5, we present the results that are discussed and interpreted in Section 6. In Section 7, we summarize the main messages of our work, presenting our conclusions and opening to future research directions.

2 Landscape connectivity, ecological networks and multi-species ecology studies: a state of the art summary

LC, the counterpart of LF, is a critical element of landscape structure and can be defined as the degree to which each component facilitates or impedes faunal species' movements among existing habitats or resource patches (Taylor et al., 1993). The role of LC for maintaining the landscape and ecosystem functioning is widely recognized (Chetkiewicz et al., 2006; Fahrig, 2003; Gaston et al., 2008). LC is species-specific, as different species have different habitat requirements and depend on scale and time (Kool et al., 2013; Rudnick et al., 2012). Moreover, connectivity analysis depends on the adopted spatial scale (Urban and Keitt, 2001). A multi-scale approach allows considering a network of habitat patches at a large-scale while, at a finer scale, the analysis focuses on individual habitat patches (Wildemeersch et al., 2019).

While in the case of LC fine-grained resolution, analyses concern site-specific interventions (i.e., linkage designs), coarse grained-resolution analyses (i.e., connectivity maps) constitute key-elements in decision support tools able to assess and design LC pattern of nations, regions or provinces (Beier et al., 2011). Moreover, According to Beier et al. (2011), this last type of LC analysis has rarely been developed in scientific literature.

The negative effects of habitat loss and LF on biodiversity and animal population persistence have been widely investigated by scholars that highlighted the positive effects of managing LC in reducing these impacts (Heller & Zavaleta, 2009). While LF hinders animal populations' ability to move and respond to external perturbations (Liu et al., 2018; Sawyer et al., 2011), LC increase is associated with situations where species range expands (Keeley et al., 2018). Additionally, LC improvement policies and actions are widely recommended as favorable issues for climate change adaption (Heller & Zavaleta, 2009). For instance, wildlife corridors play a significant role in improving species persistence and resilience to even severe climate changes (Keeley et al., 2018).

LC measurement methods are directed to i) assess structural connectivity through the spatial pattern of habitat patches or ii) gauge functional connectivity along with different species behavioral responses to physiographic conditions (Holyoak, 2008; Theobald, 2006). In this respect, many algorithms, tools and software packages are nowadays available and easily accessible to researchers and practitioners, as they are based on free and open-source software (Dickson et al., 2018; Foltête et al., 2012; Kool et al., 2013; McRae et al., 2013; McRae et al., 2008; Saura & Torné, 2009; Theobald et al., 2006).

Moreover, the inclusion of LC assessment in landscape planning and conservation programmes has been widely recommended (Botequilha Leitão & Ahern, 2002; De Montis et al., 2016), also in multi-actor planning processes considering different alternatives (Opdam et al., 2006). The EN provides analysts and practitioners with a compelling concept for modeling LC evolution in landscape ecology. An EN includes nodes, which stand for the habitat patches and links for the corridors representing functional (i.e., bidirectional) connections between the patches (Fall et al., 2007; Urban & Keitt, 2001). Since LC reflects the interaction between species and their environment, an EN's modeling is currently an essential issue for landscape planning and management (Gurrutxaga et al., 2010; Opdam & Wascher, 2004).

The concept of EN is increasingly accepted as an operational tool for i) improving the quality of natural ecosystems, ii) protecting biodiversity and iii) maintaining and improving LC (Damschen, 2013; De Montis et al., 2016; Forman, 1995; Gilbert-Norton et al., 2010; Jongman, 1995; Opdam et al., 2006). Besides, a key-role of ENs and LC is recognized in landscape planning and management policies and strategies (De Montis et al., 2016, 2019; Fichera et al., 2015; Keeley et al., 2018; Opdam et al., 2006; Termorshuizen et al., 2007; Xu et al., 2019).

Several studies have recently examined EN theory and practice (Battisti, 2013; Bennet, 1998; Boitani et al., 2007; Fahrig, 2013; Foltête, 2019; Gippoliti & Battisti, 2017; Opdam et al., 2006; Vimal et al., 2012). The assessment through field observations of the actual functional connectivity is not an easy task. Thus, many scholars -mainly involved in landscape studies at the regional scale- have proposed several methods to estimate potential functional connectivity (Adriaensen et al., 2003; Boitani et al., 2015; Calabrese & Fagan, 2006; Cook, 2002; Drielsma et al., 2007; McRae et al., 2008; Moilanen & Hanski, 2001; Tischendorf & Fahring, 1975; Urban & Keitt, 2001). To analyze the environmental interactions of structural and functional properties of ENs, the ecological network analysis (ENA) has been proposed by scholars (Fath et al., 2007).

The analysis and implementation of ENs are often based on graph theory, a branch of mathematics that deals with connectivity. In graph theory, each graph is a mathematical structure composed of points and lines that represent complex and interconnected ecosystems in landscape studies. Nevertheless, it is also a spatial graph, where each node (point element) represents a habitat patch with its position in space. One node is linked to another through a link (line element) that generally indicates dispersal potential and has length, direction, and orientation (Dale & Fortin, 2010; Urban et al., 2009). This is a universal spatially explicit model suitable for conservation planning issues (Galpern et al., 2011) and provides powerful tools to analyze network connectivity in terrestrial and marine landscapes (Appolloni et al., 2018).

Animal movement analysis refers to the dispersal capacity and implies assessing each patch's avoidance and attractive effects (Croft et al., 2008). Several authors develop single or two-species ecological corridors (Brodie et al., 2015; Hofman et al., 2018; Tarabon et al., 2019). In implementing multi-species LC (Brodie et al., 2015; C. Liu et al., 2018; Ersoy et al., 2019; Strimas-Mackey & Brodie, 2018, Tarabon et al., 2020), one of the most common approaches is to consider focal species with a wide-range dispersal capacity, so that the resulting spatial arrangement allows to support also smaller size species with low dispersal capacity (Baguette et al., 2013).

A crucial aspect of animal movements' analysis is the permeability of the surrounding landscape patches to the species dispersal. Thus, in graph-based approaches, the Euclidean distance is replaced by the least-cost distance.

This allows highlighting the least resistance connection between two suitable habitat patches based on landscape characteristics (Etherington, 2016; Fall et al., 2007; LaRue & Nielsen, 2008; Sawyer et al., 2011). Geographic Information Systems (GIS) and GIS-based tools play a crucial role in processing spatial analyses and designing the EN's components (Gurrutxaga et al., 2010; Landguth et al., 2012; Marulli & Mallarach, 2005; Theobald et al., 2006).

3 Materials and Methods

3.1 Selection of a case study in Italy

We applied our method to the implementation of the EN of Calabria, the most southern region of peninsular Italy. This region extends for about 15,600 km^2, is comprised between the Ionian Sea and the Tyrrhenian Sea, and shows a remarkably long coastline (738 km). The region's shape is slightly elongated, almost 250 km from North to South, and a width between 31 and 111 km (Figure 1).

Three national and one regional park cover a considerable extension of the region (a surface area equal to about 360,301 ha corresponding to 24% of the regional territory) and preserve critical habitat patches for flora and fauna species. From North to South, the first protected area is the Pollino National Park, which partly covers the Basilicata region. Founded as Regional Park in 1986 and upgraded to a national park in 1993, it extends for a surface of about 192,565 ha. Secondly, we find the Sila National Park, which was first established in 1968 as National Park of Calabria and restructured in 1997, and extends for a total surface of about 73,695 ha. Then, the Natural Regional Park of Serre, established in 1990, extends for 17,687 ha (www.parcodelleserre.it – last access 17/05/2020), and finally, the Aspromonte National Park, established in 1994, shows a total surface of 64,153 ha.

According to CORINE land cover (CLC) data (Table S3, supplementary material), urbanized areas occupied 2.59% of the territory of the Calabria region in 1990, a value increased to 3.58% in 2012, and that remained almost the same in 2018 (3.59%).

Because of its geographic position, Calabria has a typically Mediterranean climate at the center of the Mediterranean Basin. It is influenced by the Apennines mountains (Pellicone et al., 2018) stretching from North to South in the center of the region so that the Ionian side of it is mainly affected by winds from Africa and has a higher temperature and lesser precipitations than the Tyrrhenian one, where western winds prevail (Caloiero et al., 2011).

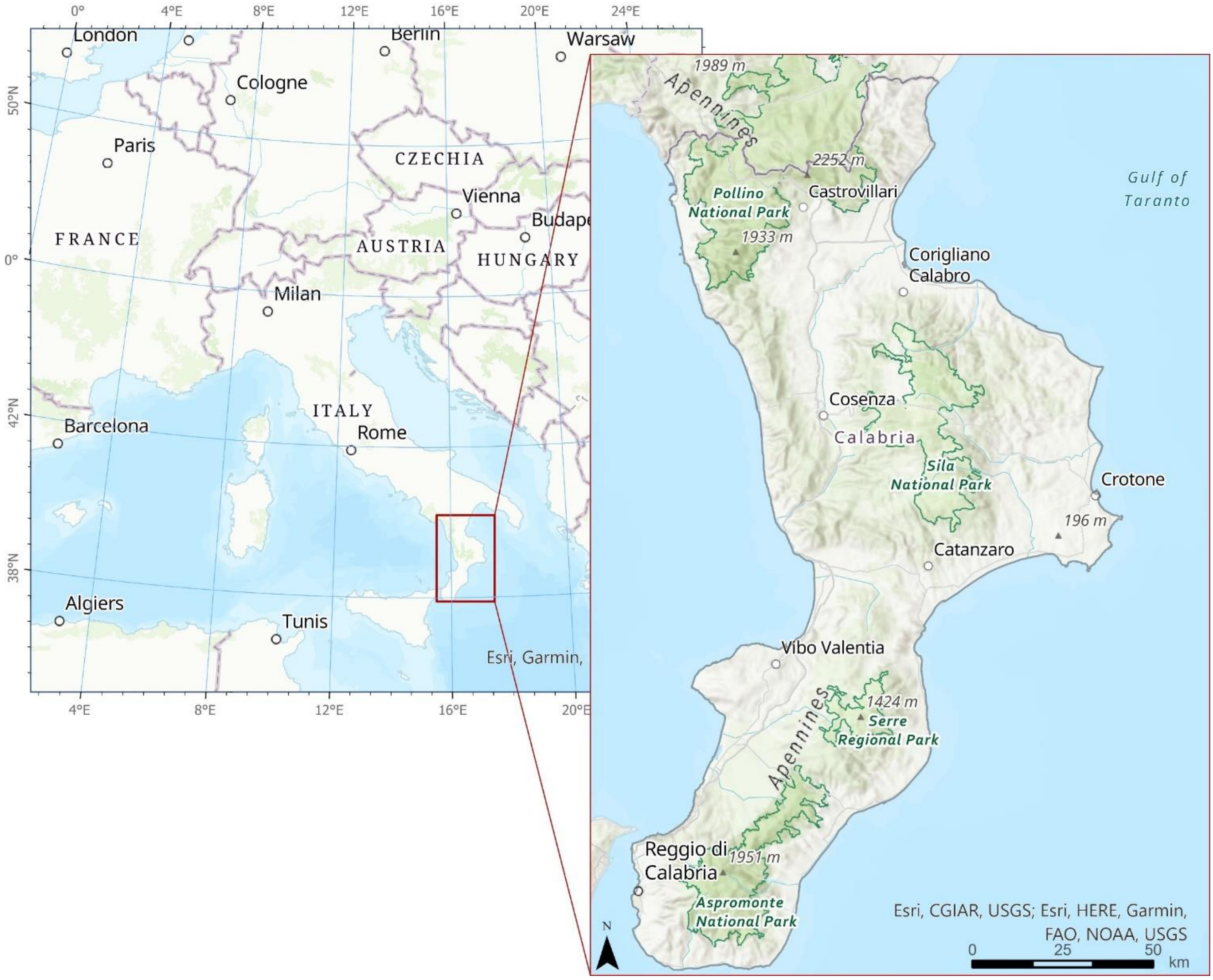


Figure 1. The geographical location of the study area, the Calabria region (Southern Italy).

3.2 Methodology for building and assessing multispecies and multitemporal ecological networks (ENs)

We structure the method in three major phases, as described in Figure 2. In Phase A, data are gathered and organized. Data reliability is relevant for describing the ecological characteristics (i.e. the foraging needs of the focal species) of the region of Calabria. In Phase B, spatial input data are processed in four steps to obtain ecological networks spanning through the entire region, and for years 1990, 2012, and 2018. In Phase C, these ecological networks (ENs) are analyzed to monitor their robustness in time. Each phase is detailed in the following subsections.

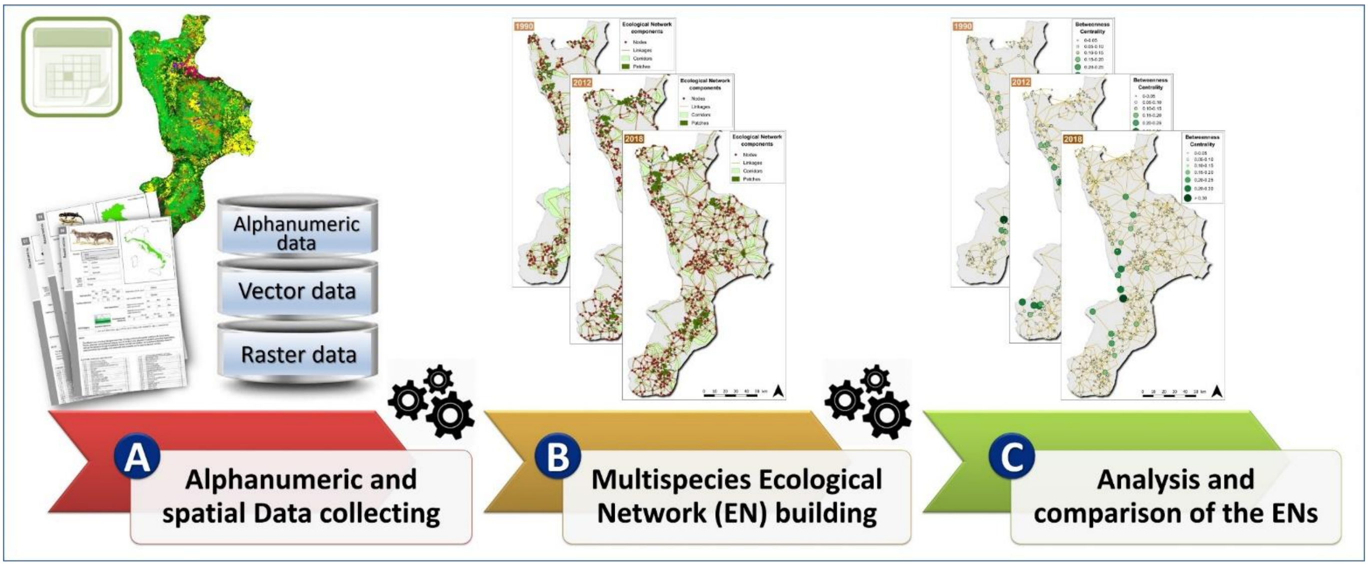


Figure 2. The workflow of the methodology including three phases: (a) collection of alphanumeric and spatial data; (b) building of the multispecies ecological networks (ENs); (c) ENs robustness analysis and comparison.

3.2.1 Base data

In Phase A, the spatial datasets reported in Table 1 were collected, organized, and analyzed. We used the CORINE land cover (CLC) inventory provided by the European Union with the Copernicus programme (see Table 1 and Fig. 3). The minimum mapping unit is 25 ha for areal phenomena while, for linear phenomena, minimum width of 100 m is established (https://land.copernicus.eu/pan-european/corine-land-cover, last access 15 September 2020). All spatial data were converted on a raster data model with 20 m x 20 m spatial resolution and projected in the WGS84 UTM zone 33N (EPSG code 32633) coordinate reference system (CRS).

Table 1. Description of the spatial datasets adopted by reference year and data source.

Data description	**Reference years**	**Data source**
Land cover - CORINE Land Cover (CLC) at the third level of detail	1990, 2012 and 2018	Copernicus Programme of the European Union (https://land.copernicus.eu/pan-european/corine-land-cover, last access 15 September 2020)
Boundaries of the urban built-up areas to integrate the CLC data	1995	Cartographic Centre of the Calabria region (CCR)
	2012 and 2018	Urban Atlas - Copernicus Programme of the European Union (https://land.copernicus.eu/local/urban-atlas, last access 15 September 2020)
Road and railroad networks data to integrate the CLC data	1995	Cartographic Centre of the Calabria region (CCR)
	2014	Tom Tom railroad database
	2018	OpenStreetMap data (©OpenStreetMap contributors)
Digital Terrain Model (DTM) with a geometric resolution of 5 m x 5 m.	2008	Cartographic centre of the Calabria region (CCR) (ftp://geoportale.regione.calabria.it/DTM5X5 - last access 15 September 2020)

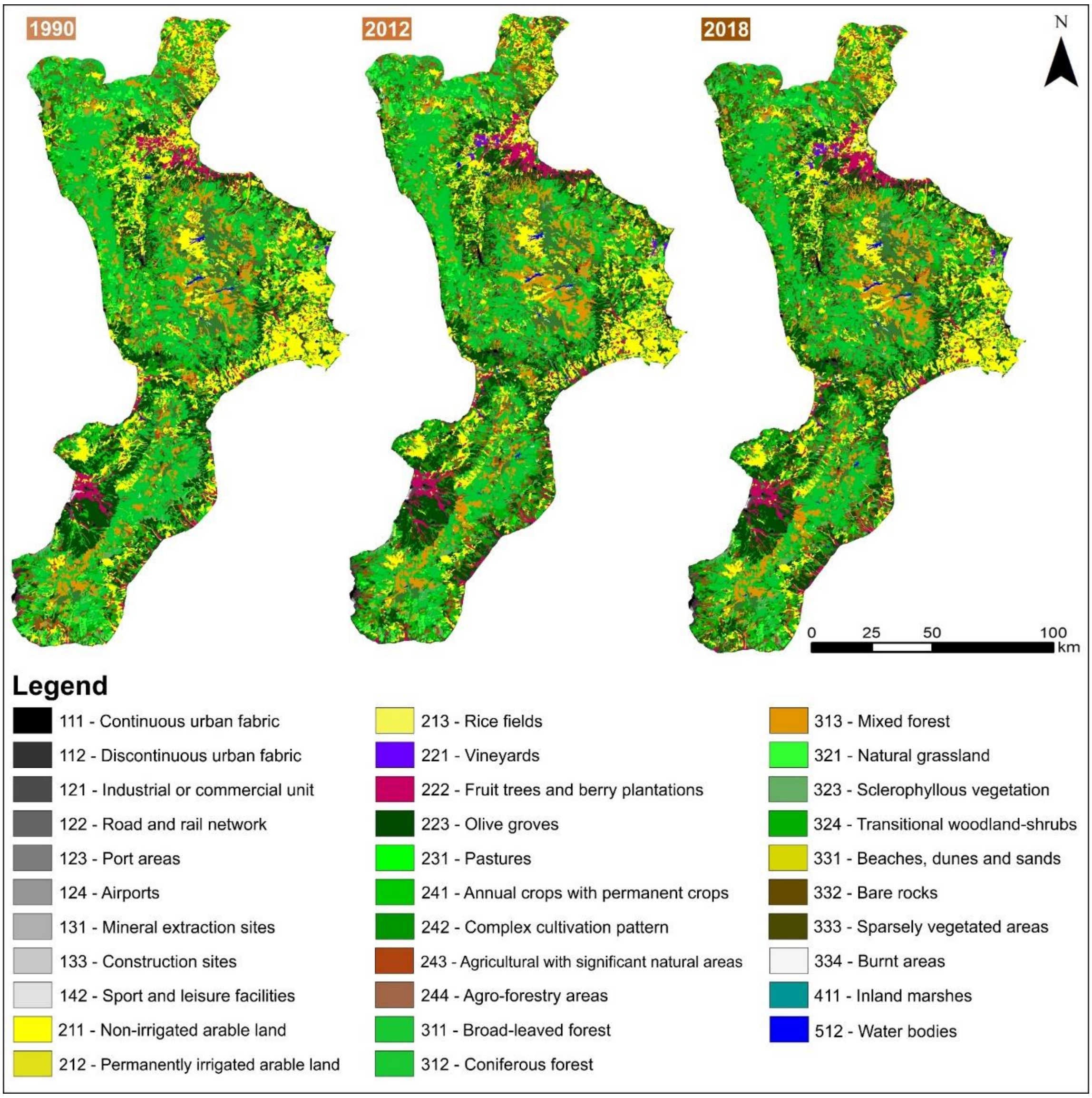


Figure 3. CORINE Land Cover (CLC) data for the years 1990, 2012 and 2012. The legend reports the III hierarchical level classes of the CORINE land cover project.

The assessment of the habitat-quality (HQ) is based on processing the autecological information of 66 terrestrial faunal focal species (Table S1, supplementary material). In Figure 4, we report on two exemplary cards drawn from the work of Boitani et al. (2003, 2007). Eleven focal species are included in the International Union for Conservation of Nature (IUCN) red lists of threatened species (IUCN, 2020).

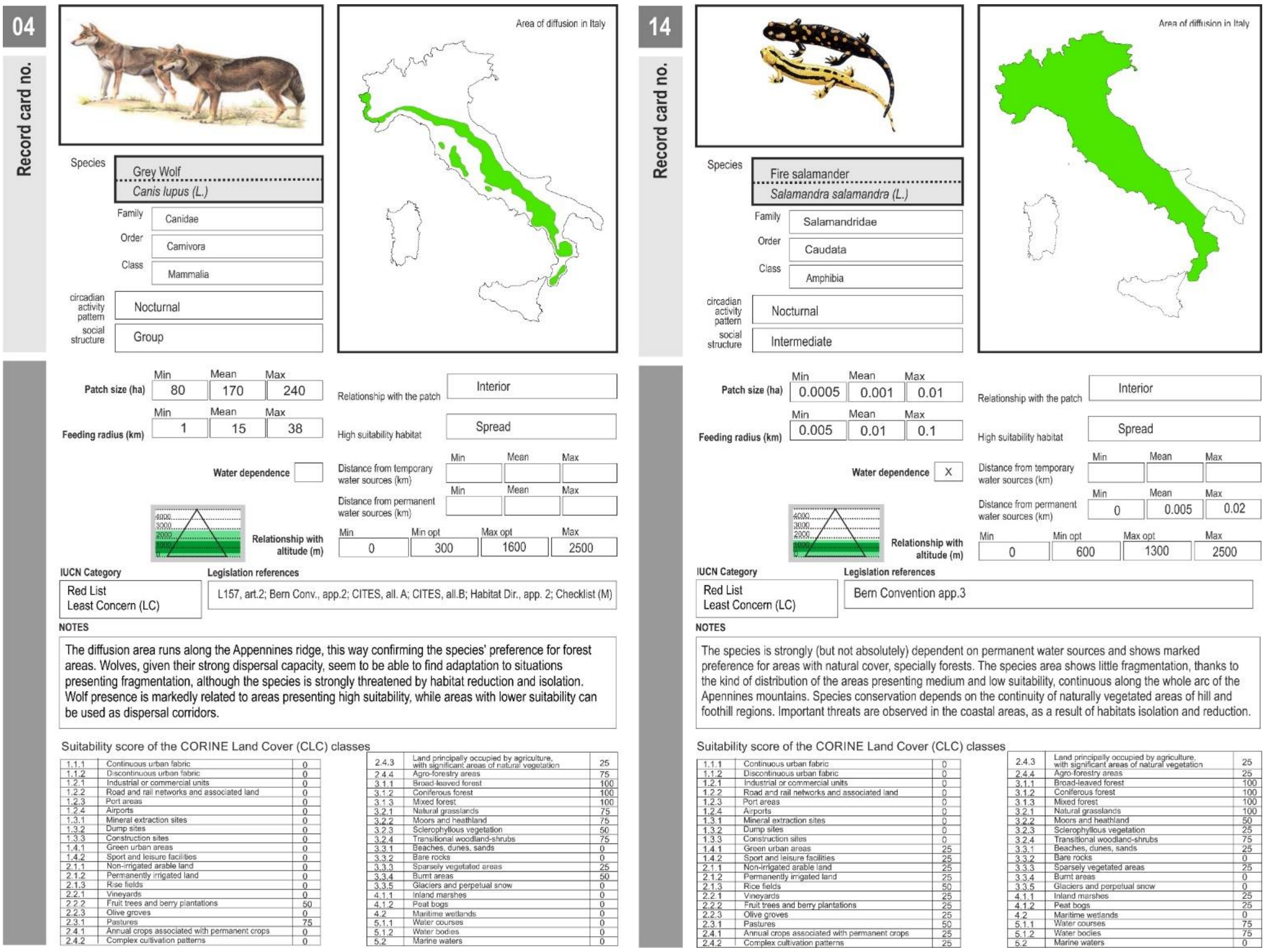


Figure 4. Two examples of the organized cards containing the autecological information of the considered 66 terrestrial faunal focal species and each of the CORINE Land Cover (CLC) classes' suitability score.

3.2.2 Building the multispecies ecological Network (EN)

In Phase B, we built the multispecies ENs for the years 1990, 2012 and 2018, following the workflow illustrated in Figure 5 and the approach proposed by Fichera et al. (2015). We used the Functional Connectivity (FunConn) model v1 (Theobald et al., 2006, 2011), a toolbox working on ArcGIS® environment that allows to identify the movement patterns and the LC for each single faunal species.

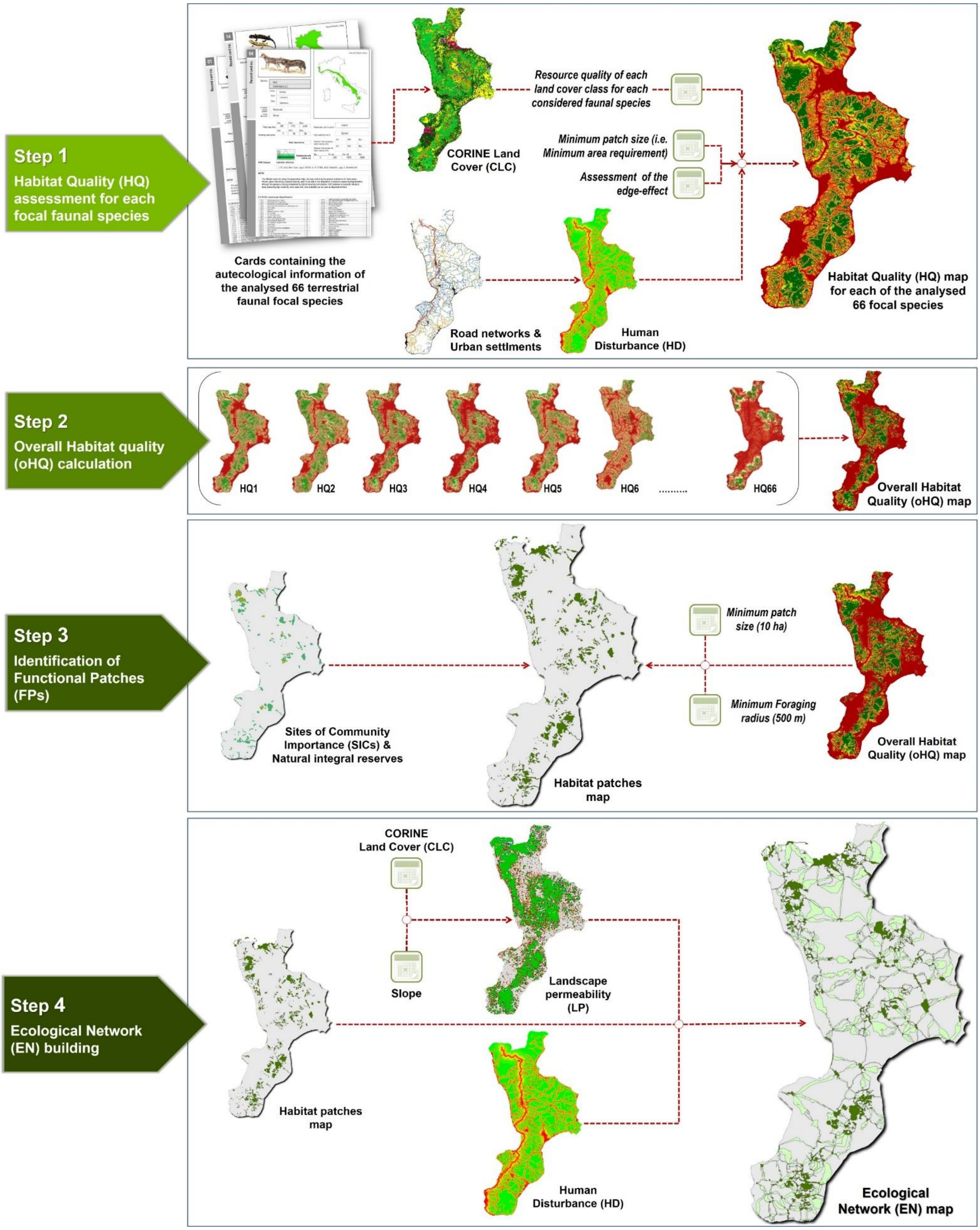


Figure 5. Methodologic workflow for building the multispecies ecological network (EN).

FunConn is a toolbox based on graph theory combined with the least-cost path analysis and permits to obtain a spatial and a topological EN model. FunConn was chosen for its reliability and flexibility, which lead to a much more informed identification of the most suitable habitats for a given species (Evangelista et al., 2012). Moreover, in this framework, the habits of the focal species and land use pattern are keys to the construction of ENs. Two specific datasets affect the spatial configuration and robustness of the obtained EN: i) Landscape permeability (LP), and ii) Human disturbance (HD). HD is calculated by processing spatial data on human settlements' surface and road and railway systems. As Figure 5 illustrates, the method develops in four steps (Theobald, 2006; Theobald et al., 2006), as detailed in the following subsections.

3.2.2.1 Step 1. Assessment of the Habitat-quality (HQ) for each focal faunal species

Step 1 consists of mapping in raster format the habitat-quality (HQ_i) for the focal species *i* at the periods considered (1990, 2012, and 2018). HQ_i ranges from zero (no habitat) to 100 (optimal habitat) and depends on the quality and availability of foraging resources. The HQ_i is first modeled by considering each focal faunal species' habitat preferences taking into account the following parameters: resource quality of each land cover class, minimum patch size (MPS), and patch structure.

The resource quality is obtained by reclassifying, for each species, the land cover maps into five classes of suitability score (0, unsuitable; 25, low; 50, medium; 75, high; 100, very high), according to the species' preferences for each land cover class (Figures 4 and 5). To avoid surfaces too small to guarantee species survival, we also considered the MPS parameter, representing the smallest surface area enough granting good functioning biological behavior to individuals belonging to the same species. It can be estimated by considering the relationships between the mass of the animal and the home range size or minimum foraging radius (MFR), i.e., a measure of how far target species move seeking out forage (Girvetz & Greco, 2007; Jetz et al., 2004).

In this application, we used MPS values provided by Boitani et al. (2003, 2007). The patch structure accounts for the so-called 'edge-effect' on the animal movements and consists of modeling the animal preference for the edges or core areas of the same quality and depends on the influence of functional patch structure (Theobald et al., 2006). The assessment is based on the evaluation of the distance from the patch edge. A species is defined as edge negative when a core area is more attractive than its edge, and edge positive, otherwise. If core areas and edges are equally attractive, the species is said to be edge neutral (Theobald et al., 2006).

The resulting map is then corrected, taking into account the typology and distance from the HD sources, describing the disturbance of some land cover classes (e.g., built-up areas, major roads, and railways).

Species like large carnivores are persisting outside protected areas (Chapron et al., 2014; Forrest et al., 2011) and capable of tolerating moderate levels of HD (Smith et al., 2019; Kimmig et al., 2020), being able to survive in human-dominated landscapes worldwide (Athreya et al., 2013; Carter et al., 2012; Chapron et al., 2014). Thus, we modeled HD sources' effect through a coefficient of reduction (HD_{coeff}) of the HQ value at a pixel level. HD_{coeff} values were implemented according to the different typology of disturbances and the distance in seven classes of HD (Table 2). The HD_{coeff} assumes values ranging from 0 (e.g., the maximum disturbance that makes a given pixel unsuitable) to 1 (e.g., no disturbance, therefore no effect on the HQ of a given pixel).

Table 2. Values of the human disturbance coefficient (HD_{coeff}) are defined according to the different classes of distance from the HD sources and to classes of HD (HDi, i = 1, 2, … 7) by built-up area extension (BAE) and infrastructure type. HD1: BAE > 2500 ha and motorways. HD2: 1000 ha<BAE< 2500 ha and highways. HD3: 500 ha<BAE< 1000 ha, provincial roads, and railways. HD4: 100 ha<BAE< 500 ha, and municipal roads. HD5: 50 ha<BAE< 100 ha. HD6: 10 ha<BAE< 50 ha. HD7: BAE< 10 ha.

Classes of distance [m]	**Human Disturbance coefficient (HD_{coeff})**						
	HD1	***HD2***	***HD3***	***HD4***	***HD5***	***HD6***	***HD7***
< 50	0	0	0.10	0.15	0.50	0.75	0.85
50-100	0	0.10	0.15	0.25	0.75	0.85	0.90
100-200	0.05	0.15	0.50	0.65	0.80	0.90	0.95
200-300	0.05	0.20	0.65	0.75	0.85	0.95	0.95
300-400	0.05	0.35	0.75	0.85	0.90	0.95	1
400-500	0.10	0.50	0.80	0.90	0.95	1	1
500-750	0.10	0.65	0.90	0.95	1	1	1
750-1000	0.25	0.80	0.95	1	1	1	1
1000-1500	0.40	0.85	1	1	1	1	1
1500-2000	0.50	0.90	1	1	1	1	1
2000-3000	0.60	0.95	1	1	1	1	1
3000-4000	0.75	0.95	1	1	1	1	1
4000-5000	0.85	1	1	1	1	1	1
> 5000	0.9	1	1	1	1	1	1

3.2.2.2 Step 2. Mapping the overall habitat-quality (oHQ).

In Step 2 of our methodology, we mapped the overall habitat-quality (oHQ) through a combination of all HQ_i according to the weighted mean aggregation rule obeying to the following Equation 1:

$$oHQ = \frac{\sum_{i=1}^{66} HQ_i \cdot w_i}{\sum_{i=1}^{66} w_i} \qquad (1)$$

Where w_i is a weight that accounts for the ecological importance of the focal faunal species i. For species included in the IUCN Red List, w_i = 3, while for those included in a Site of Community Interest (SCI), w_i = 2. For all other target species, w_i = 1. The complete list of weight values is reported in table S1 of the supplementary material.

3.2.2.3 STEP 3. DEFINING THE HABITAT PATCHES

In Step 3, we defined the habitat patches considering the minimum foraging requirements of each focal species and the possibility of animal movement among different patches (Girvetz & Greco 2007). Key information is provided by the oHQ map and the two main organism-specific parameters: the minimum foraging radius (in meters) and the MPS (in ha). Besides, we inserted the boundaries of integral natural reserves, and the SCIs, these last, only for 2012 and 2018 ENs, having them been designated and instituted in Italy since 1997, therefore not present in 1990.

All areas with an oHQ value above the threshold equal to 75 (very high suitability) were grouped according to a smoothing moving windows. Moving (or sliding) windows is a low-pass spatial filter used to smooth borders among different raster classes. In our case, we used the most common kernel of 3 x 3 pixels. The resulting surface represents the permeability (i.e., the inverse of resistance to the animal movement) across the analyzed landscape. While high-quality areas have high permeability (low resistance), cells grouped in low-quality areas have low permeability (high resistance) (Theobald et al., 2006; Zeller et al., 2012). According to the foraging radius, patches have grown outward from the core of the most suitable areas across the resistance surface to animal movement. As a result, we obtained an integrated network of habitat patches. In defining the three multi-species ENs (Fichera et al., 2015), the minimum foraging radius and the MPS were fixed at 500 m and, respectively, 10 ha, following the weighted mean aggregation of values of each faunal species (Table S2, supplementary materials).

3.2.2.4 STEP 4. CONNECTING THE HABITAT PATCHES

In Step 4, the method implies the completion of the EN through the identification of the ecological linkages - i.e., corridors- between the habitat patches. The analysis is based on LC's operationalization via the concept of permeability, implemented as a continuous gradient based on landscape permeability (LP). According to percolation theory (Sapoval & Rosso, 1995; Williams & Snyder, 2005) and directional and least-cost connectivity analysis (Theobald, 2006; Theobald et al., 2006), it is possible to obtain the LP map, which also combines information on land cover, HD, and slope. Step 4 includes the final generation of the ENs, where habitat (i.e., functional) patches (the nodes) are meant as source regions. At the same time, corridors with the least resistance to animal movement are adopted as edges. The spatial EN representation implies that each link is drawn as a segment with extreme points located on patch boundaries.

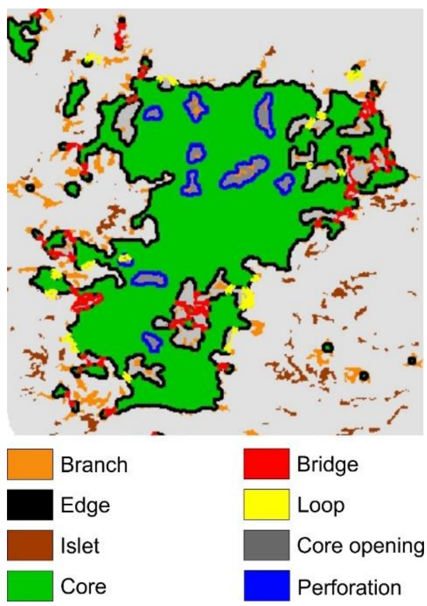
3.2.3 Analysis and diachronic comparison of the ecological networks (ENs)

In Phase C, we analyzed EN's spatial configuration by diachronically monitoring its characteristics through the three-time periods considered (1990, 2012, and 2018).

To investigate the fragmentation dynamics, we analyzed LF by using the FOSS software Guidos toolbox (Graphical User Interface for the Description of image Objects and their Shapes) (GTB) v2.9 (Vogt & Riitters, 2017). We apply the morphological spatial pattern analysis (MSPA) (Soille & Vogt, 2009; Vogt et al., 2007a; 2007b) that allows describing LF through the morphology of its different constituents: core, islet, perforated, core opening, edge, loop, bridge and branch (Table 3). We analyzed two different foreground elements: i) areas with oHQ>75 (i.e., areas presenting high suitability for the focal species and affected by low HD); and ii) habitat patches (i.e., the core areas of an EN). To this end, we reclassified the input data as binary data (i.e., background, value 1, and foreground value 2) and calculated LF as a function of spatial entropy (Shannon, 1948; Vogt & Riitters, 2017).

Table 3. The eight Morphological Spatial Pattern Analysis (MSPA) into which the foreground components of a landscape are subdivided (after Soille & Vogt, 2009; Vogt et al., 2007a). The provided ideogram showing how the MSPA can be spatially configured in a landscape.

Morphological class	Description	Graphical description
Branch	Connecting surfaces not belonging to any of the other classes. They emanate at one end from edge, loop, perforation, or bridge.	
Edge	Outer boundaries of core surfaces (i.e., transition zones between core areas of habitat patches and oHQ).	
Islet	Disjoint and too small surfaces to constitute a core area.	
Core	Interior area of the landscape, allowing a broad movement of faunal species.	
Bridge	Connecting surfaces of two different core areas. Movement outside a core area that connects to a different core area.	
Loop	Connecting surfaces emanating from the same area. Animal movement outside a core area returns to the same core area.	
Core Opening	Surfaces inside core areas and surrounded by perforation pixels.	
Perforation	Interior holes in a core area that constitute transition zones between core and non-core surfaces (dilation of the non-core areas). Movement in inner boundary adjacent to gaps in the core area.	



The analysis includes the calculation of two indices: foreground connectivity, which describes the adjacency pattern between pixels, and the edge width, that corresponds to a thickness threshold, under which it is not possible to discriminate core from non-core areas (Vogt & Riitters, 2017). As for the first index, MSPA was conducted using an 8-connectivity pattern: for each central pixel of a 3x3 pixel moving window, we included in the analysis the 8 surrounding pixels with a border and/or a corner in common. As for the second index, we set an edge width threshold at 10 pixels, corresponding (with 20 m x 20 m pixel resolution) to a circle with a 200 m radius. We performed a comparative EN robustness dynamic analysis using the free software Conefor v2.6 (Saura & Torné, 2009).

In this perspective, we calculate several network analysis metrics, both binary (B) and probabilistic (P), as detailed in Table 4.

Table 4. Network analysis metrics, by type (B, binary; P, probabilistic), description and ecological meaning, mathematical expression, and references.

Index	Type	Description and ecological meaning	Formula	References
Number of Patches (NP)	B	A patch is a habitat surface functionally defined by quality, size, and proximity. Topologically, a patch is modeled as a node located in the centroid of the polygon representing its boundary.	//	
Number of Links (NL)	B	A link stands as a connection (or corridor) between two patches. It is key to understanding the potential movement of faunal species. Since a more connected landscape has more links, this index is very useful in a comparative and/or diachronic analysis between landscapes.	//	(Fall et al., 2007; Urban & Keitt, 2001)
Number of Components (NC)	B	A component includes a set of nodes (patches) connected pairwise by at least one path. Disconnected (isolated) nodes constitute a component. The higher NC, the lower the connectivity. An NC value higher than 1 signals the rise of the insularization phenomenon.	//	(Urban & Keitt, 2001)
Harary index (H)	B	Half of the sum of reciprocals of topological distances between all pairs of nodes in a connected graph. The higher H, the higher LC.	$H = \frac{1}{2}\sum_{i=1}^{n}\sum_{j=1,i\neq j}^{n}\frac{1}{nl_{ij}}$	(Harary, 1969; Jordán et al., 2003; Pascual-Hortal & Saura, 2006; Ricotta et al., 2000)
Landscape Coincidence Probability (LCP)	B	The probability that two points randomly located within the landscape belong to the same component. The higher LCP, the higher LC.	$LCP = \sum_{i=1}^{nc}\left(\frac{a_i}{A_L}\right)^2$	(Pascual-Hortal & Saura, 2006)
Integral Index of Connectivity (IIC)	B	Habitat availability on a binary connection model. It equals to 1 in the hypothetical condition that the landscape equals to a habitat patch. Values range between 0 and 1: the higher IIC, the higher LC.	$IIC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i \cdot a_j}{1+nl_{ij}}}{A_L^2}$	(Pascual-Hortal & Saura, 2006, 2008)
Flux (F)	P	The probability that animal species disperse throughout the patches.	$F = \sum_{i=1}^{n}\sum_{j=1,i\neq j}^{n} p_{ij}$	(Saura & Pascual-Hortal, 2007)
Probability of Connectivity (PC)	P	The probability that two points randomly placed within the landscape falls into habitat areas interconnected	$PC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i\, a_j\, p_{ij}^{*}}{A_L^2}$	(Saura & Pascual-Hortal, 2007)

Index	Type	Description and ecological meaning	Formula	References
		(i.e., reachable from each other). Values range between 0 and 1.		
Betweenness Centrality (BC_k)	P	For any patch k, BC is the ratio between the number of shortest paths connecting two patches i and j (i, j ≠ k) and passing through that patch k and the total number of shortest path linking i and j in the network. BC_k gives a measure of how much a generic patch k serves as a stepping-stone for animal movements between pairs of proximal patches.	$BC_k = \sum_i \sum_j \frac{g_{ij}(k)}{g_{ij}}$	(Bodin & Saura, 2010; Estrada & Bodin, 2008; Freeman, 1977)
Betweenness Centrality-IIC (BC_k^{IIC})		This metric is derived from BC_k and also considers the surface area associated with each patch i and j and passing to the patch k (i, j ≠ k). The index allows focusing on the role of larger patches.	$BC_k^{IIC} = \sum_i \sum_j a_i a_j \frac{1}{1+d_{ij}}$	(Bodin & Saura, 2010)

nc, total number of habitat components.
a_i and a_j, sum of node attribute (i.e., patch area) for nodes (patches) i and j.
A_L, whole analyzed landscape area (i.e., comprising non-habitat areas).
n, total number of nodes in the landscape.
nl_{ij}, number of links in the shortest paths between patches i and j.
p_{ij}, probability of direct dispersal between patches i and j.
p^*_{ij}, maximum probability for all links (paths) between patches i and j.
$g_{ij}(k)$, all separate shortest paths between patches i and j, and passing through the generic patch k.
d_{ij}, *distance between* patches i and j

Betweenness Centrality (BC_k) metrics have an essential role, and these measures are provided at the node level (Saura & Torné, 2009). BC was introduced by Freeman (1977) to measure the inter-centrality of a node *i* as the share of shortest paths connecting a pair of nodes and passing through that node *i* and the total number of shortest paths connecting those nodes in the whole network. So, BC is a significant signal for a patch's bridging role, i.e., for identifying stepping-stones: the higher the BC_k, the higher the patch's importance as a stepping-stone for animal movement. We analyzed the classical BC_k and its generalization, the BC_k^{IIC}, introduced by Bodin & Saura (2010), that takes into account the area of those patches bridged by that node and giving more importance to those nodes that bridge larger (i.e., more important) patches.

4 Results

4.1 Land cover dynamics in the time period 1990-2018

From 1990 to 2012, we found a remarkable increase in urban settlements (+15,600 ha, +37.59%), while from 2012 to 2018, these surfaces are nearly the same. Most new urban areas occupied past agricultural areas that decreased by 30,000 ha (-3.82%). On the other hand, the surface occupied by forestry formations (CLC 311, 312 and 313) has slightly shrunk from 1990 to 2012 (-18,700 ha, -3.2%) while, in 2018, a little increase can be noticed (+0.4%) (Table S3, supplementary material).

4.2 Spatial configuration and indicators of ecological networks (ENs)

We obtained three types of results: i) the spatial configuration of the three ENs, ii) EN robustness analysis metrics, and iii) fragmentation dynamic analysis concerning oHQ and LPs. The surface covered by ENs accounted for 503,808 ha in 1990 (32.14% of the study area), 424,421 ha in 2012 (27.07%), reaching the largest surface in 2018 (610,285 ha, 38.93%) (Table S4, supplementary material). Referring to the CLC classes falling in the ENs, most parts of habitat patches are forestry areas, with the highest values in 1990 (98.32%), while in 2012 and 2018, they account for more than 80%. On the other hand, agricultural areas cover significant parts of corridors with the highest value in 2018 (47.66%), 46.36% in 2012, and 42.65% in 1990.

Figure 6 illustrates the dynamic spatial analysis of the ENs in 1990, 2012, and 2018. Maps include the ecosystem and topological representations of the ENs: patches and corridors are modeled through nodes and edges. Over time, a strengthening of ENs is evident with an increasing number of nodes and linkages from 1990 to 2012. This first visual appreciation is confirmed by the quantitative results of the network analysis reported in Table 5.

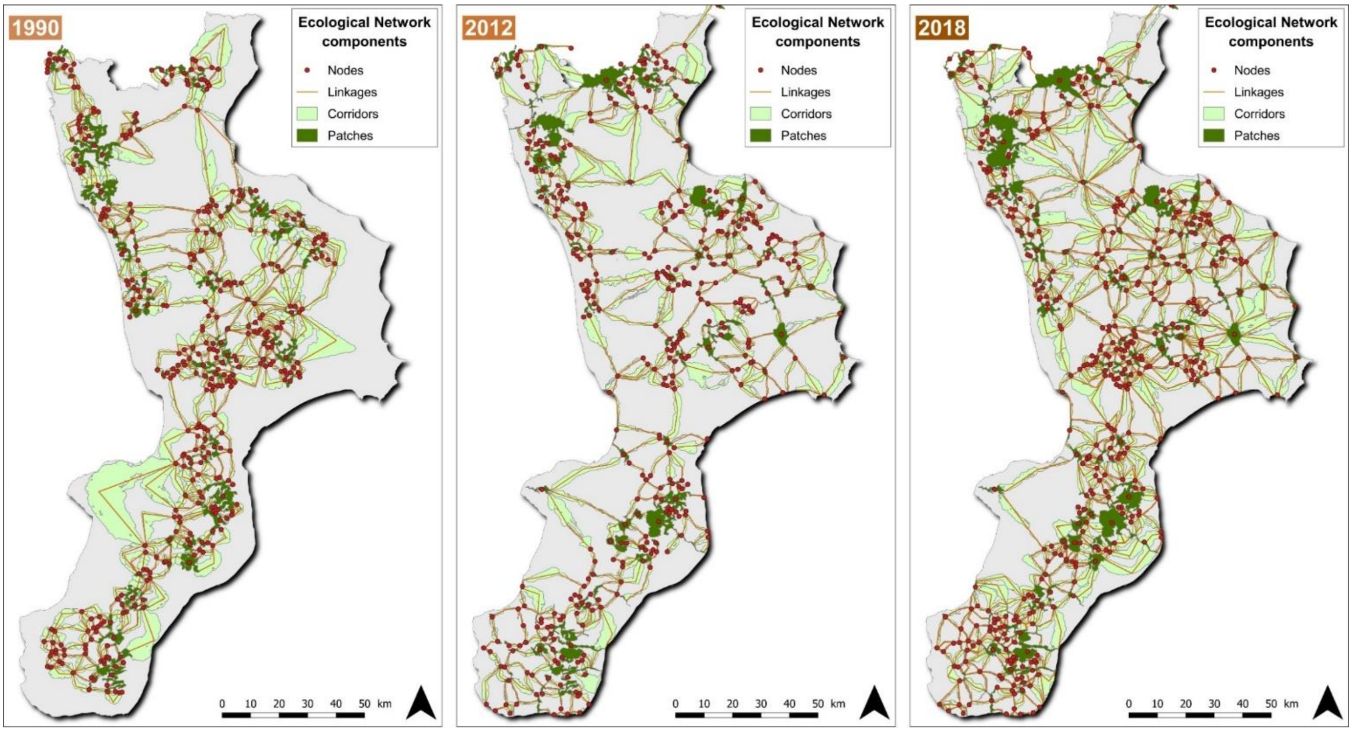


Figure 6. Spatial configuration dynamic analysis of the Ecological network (ENs) obtained for 1990 (left), 2012 (center), and 2018 (right). The EN components (corridors, in light green, and patches, in dark green) are reported together with the topological components (nodes, red dots, and linkages, orange lines) in each map.

Table 5. Diachronic analysis of EN spatial configuration using overall indexes - i.e., at the ecological network (EN) level – and node-level indexes (years 1990, 2012, 2018).

Overall Indices at EN level	1990	2012	2018
Number of habitat patches (nodes) (NP)	391	393	451
Number of Links (NL)	781	638	1078
Number of Components (NC)	1	1	1
Harary index (H)	9701.4	8947.9	14,503.1
Landscape Coincidence Probability (LCP)	0.39	0.32	0.47
Integral Index of Connectivity (IIC)	0.04	0.03	0.06
Flux (F)	1562	1316	2156
Probability of Connectivity (PC)	0.39	0.32	0.47
Betweenness Centrality (BC) metrics at the node level			
mean BC_k (median)	0.029 (0.005)	0.033 (0.004)	0.021 (0.002)
mean BC_k^{IIC} (median)	1.89 (0.408)	2.143 (0.363)	1.39 (0.182)

All values slightly decrease from 1990 to 2012, except the number of habitat nodes (NP), increasing from 391 to 393, and the number of components (NC), 1 for all the three ENs (Table 5). The number of links (NL) evidences the different rearrangements of the ENs, with a slight decrease from 1990 to 2012 (from 638 to 781) and a relevant increase toward the highest value reached in 2018. The Harary index (H) shows the same dynamics: it highlights the highest complexity in 2018 after a slight deceleration.

As NC stays equal to 1, LC metrics such as Landscape Coincidence Probability (LCP), Probability of Connectivity (PC), and Integral Index of Connectivity (IIC) show very similar and stabile trends. The Flux (F) index, the measure of dispersal probability, confirms what is reported for the other metrics: it first decreases, then increases, reaching the value of 2156. As for BC, in Figure 7, we mapped the BC_k of each node, differentiating the size and color of the symbol dots according to their different BC values.

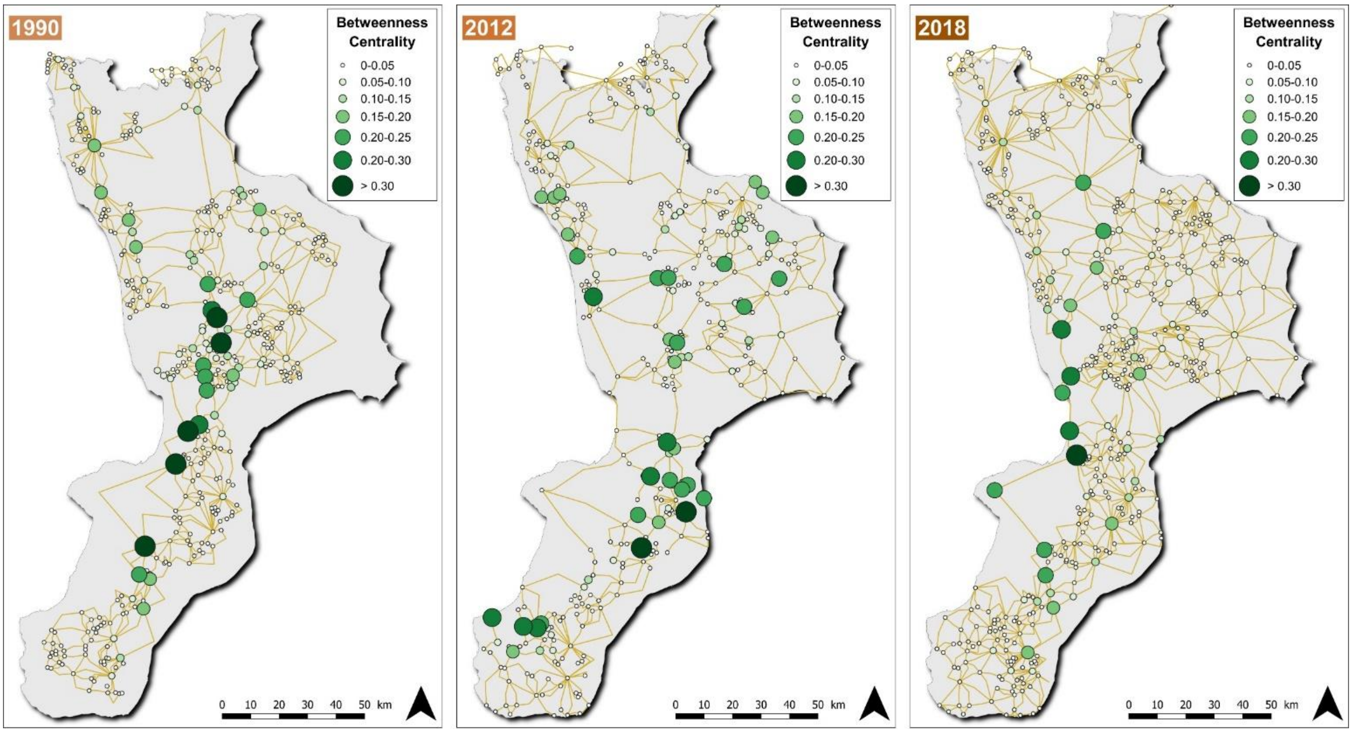


Figure 7. Maps of the betweenness centrality (BC_k) for the ENs in 1990 (left), 2012 (center), and 2018 (right).

In Figure 8, we reported the graphs showing the BC_k and the BC_k^{IIC} values of the best patches (i.e., those with the highest values). As for BC_k^{IIC}, the highest value is 21.36 in 1990, 19.41 in 2012, and 16.82 in 2018. In graphs of Figure 8, we reported the highest BC_k and BC_k^{IIC} values grouped into four classes. As for BC_k, most significant differences in the connectivity concern the 2012 EN, with 19 nodes (4.8% of the total) with high values of BC_k (class 0.20-0.30), while in 1990 and 2018, they are equal to 8 and, respectively, 9. The same insights can be found analyzing BC_k^{IIC}, with 6 nodes in 1990 falling in the class with the highest connectivity (>15), 3 in 2012, and 4 in 2018. The analysis of the BC_k^{IIC} at the node level shows good LC over time.

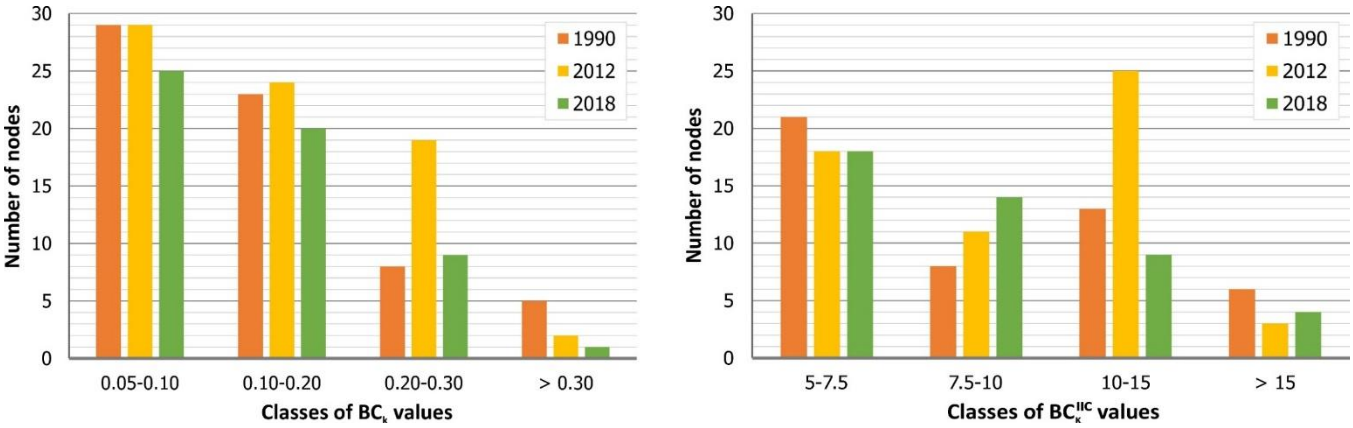


Figure 8. Graphs showing the number of nodes with the highest BC_k and BC_k^{IIC} values grouped into four classes and according to the three analyzed ecological networks (ENs).

4.3 Landscape fragmentation (LF) analysis

In Figure 9, we reported the Morphological spatial pattern analysis (MSPA) results by mapping the pattern of the fragmentation classes in each time period of the areas with oHQ higher than 75. As for oHQ, the core class passed from 89,541 ha in 1990 to 88,934 ha in 2012 and 109,199 in 2018. The edge class maintains the same surface, about 76,000 ha, in all years.

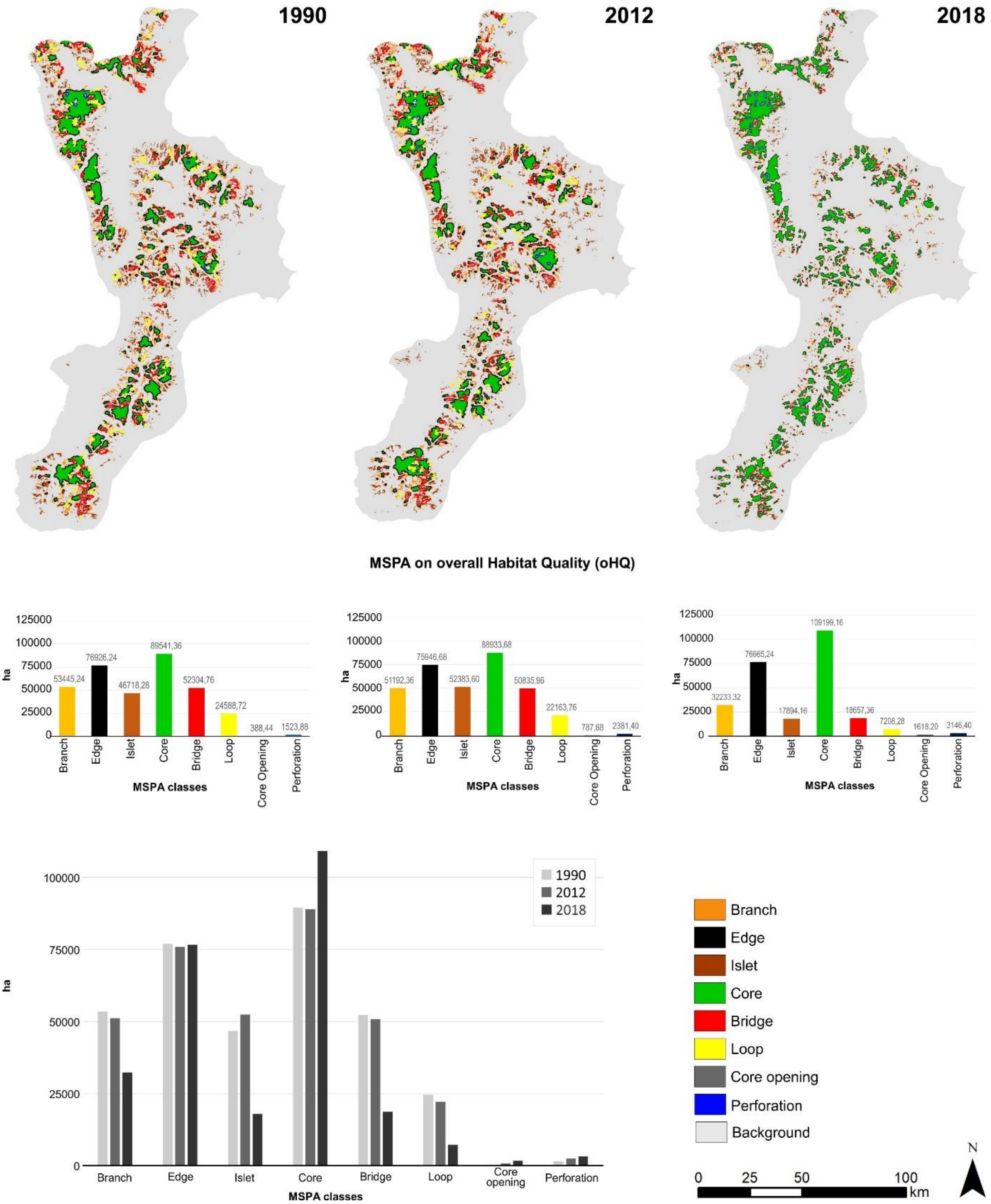


Figure 9. Morphological spatial pattern analysis (MSPA) of the areas with the overall Habitat Quality (oHQ) higher than 75 for the ENs in 1990 (top left), 2012 (top center), and 2018 (top right).

The islet class denotes the highest fragmentation of oHQ in 2012, with a surface of 52,300 ha, while it was 46,700 in 1990 and decreased significantly in 2018 (17,894 ha). The perforation class concerns limited oHQ largest areas (1523 ha in 1990, 2381 in 2012, and 3146 in 2018).

According to the MSPA of the habitat patches, EN shows a more robust configuration in 2012 and 2018 (Fig. 10).

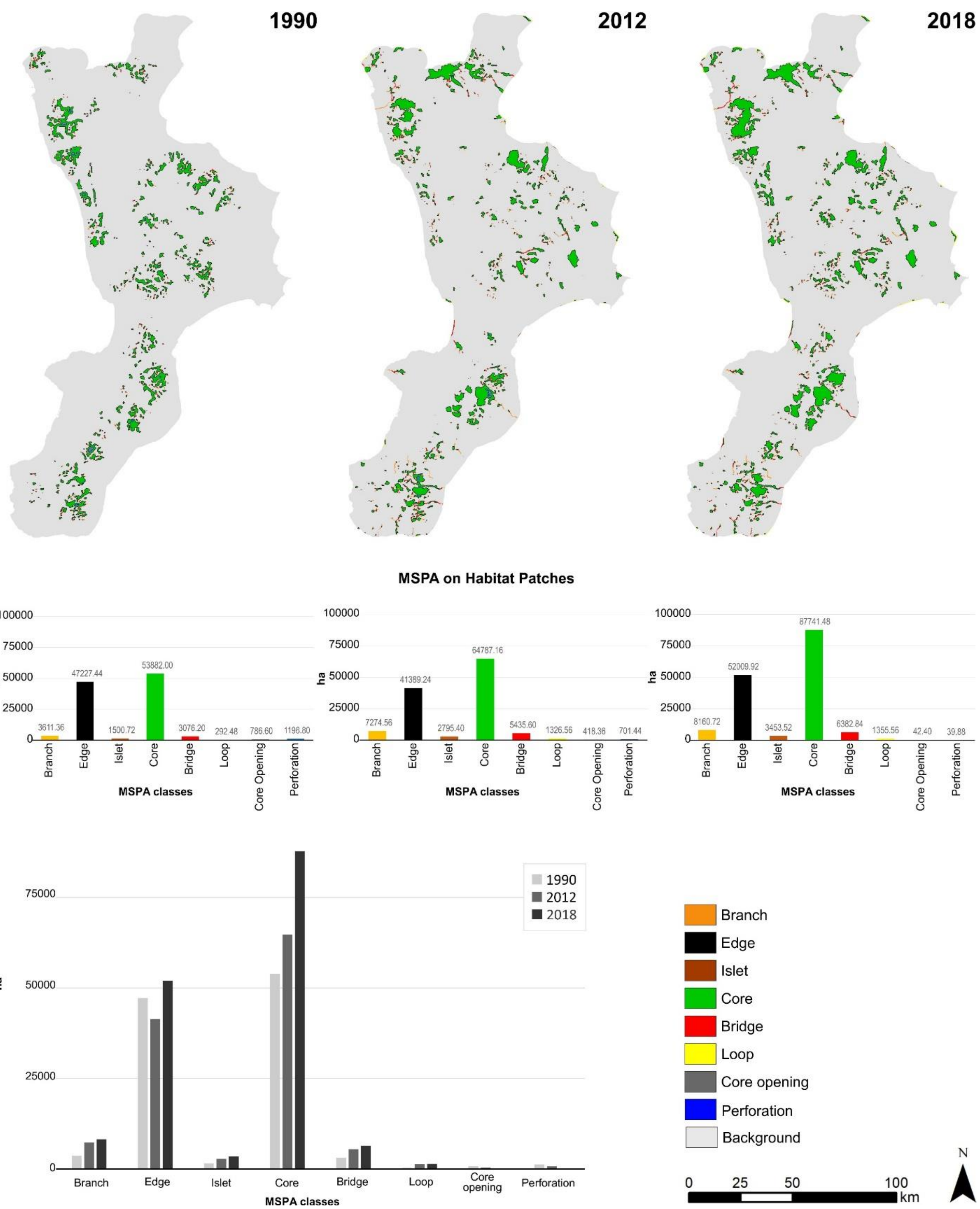


Figure 10. Morphological spatial pattern analysis (MSPA) of the Habitat Patches (HP) for the ENs in 1990 (top left), 2012 (top center), and 2018 (top right).

This is due also to the integration in the EN of SICs and integral reserves. The core class increased from 53,882 ha in 1990 to 64,787 ha in 2012 and 87,741 in 2018. The edge class analysis confirms this insight: its surface decreased from 1990 to 2018. As expected, the mean habitat patch surface increased from 282.6 ha in 1990 to 301.2 in 2012 and 353.1 in 2018. As for the bridge class, its surface covers a few thousand hectares (3076 in 1990, 5435 in 2012, and 6382 in 2018). Nevertheless, particularly in 2012 and 2018, the bridge class concerns significant surfaces acting as corridors while, in some cases, they allow the ENs to reach the coastal areas.

5 Discussion

In this section, we discuss the results concerning the research questions raised in the introduction. As for the first research question (how to structure ENs based on the habitat requirements of focal faunal species), we have studied ENs to implement the multispecies landscape corridors at the regional scale. In this respect, the definition of their optimum width is still an open question, as there is no agreement on a clear methodological framework (Beier et al., 2008; Sawyer et al., 2011). While previous studies have approached the issue by assessing the HQ by calculating the interplay between land cover dynamics and human impact (Dong et al., 2020; Tang et al., 2020), we have built a species-specific method. In our case, the study of HQ and LP is referred to the analysis of the autecological needs of a remarkable number of focal faunal species, unlike other studies that have focused only on a few species (Babí Almenar et al., 2019; Ehlers Smith et al., 2019; Ersoy et al., 2019; Hofman et al., 2018; C. Liu et al., 2018; S. Liu et al., 2018). Thus, our results provide the analysts with a more comprehensive picture of LC from the perspective of a tool aiding the design of multispecies ENs.

The second research question attains the dynamics occurring on a landscape and the adopotion of LC as a tool to support EN design. Evidence shows that the EN expands, as NP increases from 391 in 1990 to 451 in 2018, while the increment slows from 1990 to 2012. These dynamics are confirmed by the trend of NL, which decreases from 781 (in 1990) to 638 (in 2012) and jumps up to 1078 (in 2018). This signals a consistent general rearrangement of the components towards a more disjointed structure. Another relevant result is that the EN evolves, maintaining a unique component (NC=1), which is a condition that an EN should have to preserve LC effectively. In this configuration, no isolated patches are present, and two patches whatsoever can be reached through at least one walkable path. In ENs spanning through large areas, this holds for species with enough moving capacity (e.g., carnivores, artiodactyls, etc.) and does not for too small species having much shorter dispersal distances. This brings us to our third research question focusing on the assessment of EN robustness.

We have used landscape graphs suitable for computing connectivity metrics (Laita, Kotiaho, & Mönkkönen, 2011). However, we have demonstrated how those metrics do not merely gauge topological features but offer information on the robustness, a concept suitable to support EN planning and monitoring (Foltête et al., 2014; Rayfield et al., 2011). Typically, BC is a key indicator of robustness. The nodes with higher BC values represent stepping-stones for animal movements because they are interconnected with many pairs of nodes (Urban et al., 2009). The BC analysis shows that the set of elements with BC_k value higher than 0.30 includes just four nodes in 1990, two in 2012, and one in 2018. This can be justified by the increasing number of network nodes and edges, a phenomenon that makes the whole EN better interconnected and more robust in 2018, as a large pool of nodes show a moderate value of BC, i.e., acting as stepping stones in compensating the loss of high BC values with a broad set of shortcuts. Moreover, the BC_k^{IIC}, assigning a higher value to those nodes linking larger habitat patches, highlights quite good robustness of the 2012 EN, also compared to the 2018 EN. In fact, the mean value of BC_k^{IIC} in 2012 is the highest of the three ENs (2.143, st. dev. ±3.819). By the way, BC values of nodes (mean and median) (Table 5) and their distribution (Figure 7) confirm the major robustness of 2018 EN.

The MSPA analysis yields double-fold insights. On the one hand, it gives information on the past and the current state of LC. On the other hand, it allows detecting priority areas requiring active interventions to strengthening the EN. With reference to the results of the MSPA, two critical phenomena emerge: an increase of the total surface area of the patches (from 53,882 ha in 1990 to 64,787 ha in 2012, and to 87,741 ha in 2018) and a decrease of the core opening area. According to the same line, branch and bridge surface areas expand. This is an important sign of improvement and consolidation. The larger number and extension of patches can be explained by the increase of forestry areas from 1990 to 2018. The MSPA analysis on habitat patches confirms that integration of the SCIs in 2012 and 2018 ENs is able to significantly improve the compactness and the spatial distribution of the core areas. On the other hand, the increase of islet areas (Fahrig, 2003) from 1500 ha, in 1990, to 3453 ha in 2018 is a clear sign of an increase of LF in marginal areas. These figures suggest how a further strengthening of the current EN (2018) would be advisable and focused on reducing LF in marginal areas, improving the quality and distribution of forest and semi-natural habitats. Besides, the ENs spatial configuration highlights the need for an increase of riparian corridors, falling in the 'bridge' class of MSPA analysis, allowing faunal species to reach the coastal areas. On the opposite side, similar trends are also shown by the metrics used for studying the oHQ. The analysis conducted on oHQ focus on the most critical (i.e. with oHQ greater than 75) areas, upon which the EN should be built. Thus, this analysis is crucial to assess the vulnerability. The graphs with single year values of MSPA classes show a slight worsening of the overall scenario from 1990 to 2012, driven by increased LF. The core areas shrink from 89,541 ha to 88,934 ha, while the islet areas expand from 46,718 ha to 52,383 ha. On the other hand, in 2018, a positive trend is signaled by the increase of core areas and the decrease of islet areas. However, the increase in core areas is also accompanied by the rise in core opening areas, which expand from 388 ha in 1990 to 1618 ha in 2018, highlighting holes within the core areas. Also, these insights suggest the need for intervention of amelioration of the habitat quality in these areas.

6 Conclusions

In this study, we started by analyzing the current land-use dynamics in the Calabria region and scrutinized LC changes that occurred to multispecies ENs in the time shots 1990, 2012, and 2018. We clarified how the EN evolves over time by changing elements, without losing its potential for biodiversity conservation (Opdam et al., 2006). In a changing landscape, the dynamics of development and conservation can decrease one component of the EN that may be compensated by improving another (Opdam et al., 2006).

This work contributes to multi-temporal analysis that widely showed its significant role in sustainable landscape planning. The assessment of LC based on the requirements of a significant number of focal faunal species and the assessment of the robustness of ENs provide a reliable tool for landscape planners. Moreover, several future research directions are outlined: the analysis of intermediate years (2012 in this case), a deepening of the dynamics of fragmentation and their effects on landscape connectivity, the impact of new built-areas and the effects of climate change on biodiversity (Opdam & Wascher, 2004). In this respect, the implementation of a habitat quality monitoring system attaining the whole region (not only the protected areas) would provide the analysts with much more reliable outcomes. In our study, the CLC data present some limitations: relatively low (85%) thematic and geometric (100 m) accuracy (https://land.copernicus.eu/pan-european/corine-land-cover, last access 05 July 2020). Thus, additional sources of information should be considered. As highlighted by Beier et al. (2011), building an EN implies a variety of steps and the selection of several models, thresholds, and decision rules, for which there is no clear best option. Accordingly, stakeholders’ involvement since the first phases of the process can improve the quality and acceptance of the outcomes (Modica et al., 2013; Xu et al., 2020). An effective EN design requires a sufficient level of awareness of local communities and decision-makers (i.e., government agencies, policymakers) on the importance of protecting and improving LC (Brodie et al., 2016). Another key issue attains the need to deeply understand the impact of global changes, driven by human pressure, ecosystem functioning, and biodiversity conservation (Requena-Mullor et al., 2018). LC can be classified as a supporting ecosystem service for the biodiversity of plants and animals and, acting positively in reducing extinction rates, has a significant role in maintaining the other ecosystem services (Haddad et al., 2015).

The preservation of LC requires specific financial actions, i.e., as in the case of preservation of faunal species in national or regional parks. In this respect, the EN components (habitat patches and corridors) should be considered a structural invariant in planning tools. In other words, the planning of new settlements and other land-use changes leading to the HQ reduction should also be assessed, considering their impact on the robustness of the EN (De Montis et al., 2019). Moreover, to have planning reliability, ecological corridors should be clearly delineated (Hilty et al., 2020). It is also essential to highlight the replicability of the proposed method in other contexts, differing in topography, land cover, and human pressure, and the possibility of integrating it with further information on additional faunal species (birds, insects, etc.).

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Supplementary materials

Table S1. The 66 species selected as representative to build the three ecological networks (ENs). W_i is the multiplicative factor representing the different ecological importance of the species for the construction of the overall Habitat Quality (oHQ)

Card n°	Scientific name	Common name	w_i	Card n°	Scientific name	Common name	w_i
1	*Vulpes vulpes*	Red fox	*1*	34	*Capreolus capreolus*	Roe deer	*2*
2	*Martes foina*	Beech marten	*1*	35	*Bufo bufo*	Common toad	*2*
3	*Martes martes*	Pine marten	*2*	36	*Pseudepidalea viridis*	Green toad	*2*
4	*Canis lupus*	Wolf	*3*	37	*Mesotriton alpestris*	Alpine newt	*2*
5	*Lepus europaeus*	Brown hare	*1*	38	*Emys orbicularis*	European pond tortoise	*3*
6	*European hedgehog*	Common hedgehog	*1*	39	*Anguis fragilis*	Deaf adder	*2*
7	*Sciurus vulgaris*	Red squirrel	*1*	40	*Zamenis longissimus*	Aesculapian snake	*1*
8	*Salamandra salamandra*	Fire salamandra	*2*	41	*Neomys anomalus*	Miller's water shrew	*1*
9	*Hierophis viridiflavus*	Green whip snake	*3*	42	*Sorex samniticus*	Appennine shrew	*1*
10	*Elaphe situla*	Leopard snake	*2*	43	*Suncus etruscus*	Etruscan shrew	*1*
11	*Hemidactylus turcicus*	Mediterranean house gecko	*1*	44	*Talpa caeca*	Blind mole	*1*
12	*Podarcis sicula*	Ruin lizard	*1*	45	*Talpa romana*	Roman mole	*1*
13	*Podarcis muralis*	Wall lizard	*1*	46	*Lepus corsicanus*	Italian hare	*3*
14	*Lacerta viridis*	Green lizard	*2*	47	*Chionomys nivalis*	Snow vole	*2*
15	*Tarentola mauritanica*	Common gecko	*1*	48	*Myocastor coypus*	Coypu/Nutria	*1*
16	*Testudo hermanni*	Hermann's tortoise	*3*	49	*Mustela putorius*	European polecat	*1*
17	*Vipera aspis*	Asp viper	*2*	50	*Sus scrofa*	Wild boar	*1*
18	*Rana italic*	Italian frog	*2*	51	*Chalcides chalcides*	Three-toed skink	*1*
19	*Salamandrina terdigitata*	Spectacled salamander	*2*	52	*Natrix natrix*	Grass snake	*1*
20	*Dryomys nitedula*	Forest dormouse	*2*	53	*Crocidura leucodon*	Bicolored shrew	*1*
21	*Muscardinus avellanarius*	Hazel dormouse	*3*	54	*Crocidura suaveolens*	Lesser white-toothed shrew	*1*
22	*Felis silvestris*	European wildcat	*3*	55	*Neomys fodiens*	Eurasian water shrew	*1*
23	*Lutra lutra*	Eurasian otter	*3*	56	*Sorex minutus*	Pygmy shrew	*1*
24	*Bombina pachypus*	Appennine yellow-bellied toad	*3*	57	*Glis glis*	Edible dormouse	*1*
25	*Hyla intermedia*	Italian tree frog	*2*	58	*Myodes glareolus*	Bank vole	*1*
26	*Rana dalmatina*	Agile frog	*1*	59	*Apodemus flavicollis*	Yellow-necked mouse	*1*
27	*Triturus carnifex*	Italian crested newt	*2*	60	*Apodemus sylvaticus*	Wood mouse	*1*
28	*Lissotriton italicus*	Italian newt	*1*	61	*Mus domesticus*	West-European house mouse	*1*
29	*Coronella austriaca*	Smooth snake	*1*	62	*Rattus norvegicus*	Brown rat	*1*
30	*Elaphe quatuorlineata*	Four-lined snake	*3*	63	*Cervus elaphus*	Red deer	*2*
31	*Natrix tessellata*	Dice snake	*2*	64	*Mustela nivalis*	Least weasel	*1*
32	*Eliomys quercinus*	Garden dormouse	*3*	65	*Meles meles*	European badger	*1*
33	*Hystrix cristata*	Crested porcupine	*2*	66	*Rattus rattus*	Black rat	*1*

Table S2. Values of minimum foraging radius (MFR) [m] and minimum patch size (MPS) [ha] for each of 66 species selected. In the last row, the weighted values are reported.

Card n°	Scientific name	w_i	Minimum Foraging radius (MFR_i) [m]	Minimum Patch Size (MPS_i) [ha]	Card n°	Scientific name	w_i	Minimum Foraging radius (MFR_i) [m]	Minimum Patch Size (MPS_i) [ha]
1	*Vulpes vulpes*	*1*	2000	75	34	*Capreolus capreolus*	*2*	*750*	*5*
2	*Martes foina*	*1*	1000	90	35	*Bufo bufo*	*2*	*100*	*0.005*
3	*Martes martes*	*2*	1000	70	36	*Pseudepidalea viridis*	*2*	*200*	*0.005*
4	*Canis lupus*	*3*	2000	80	37	*Mesotriton alpestris*	*2*	*50*	*0.1*
5	*Lepus europaeus*	*1*	1000	3	38	*Emys orbicularis*	*3*	*250*	*0.001*
6	*European hedgehog*	*1*	200	5	39	*Anguis fragilis*	*2*	*20*	*0.0001*
7	*Sciurus vulgaris*	*1*	1000	2	40	*Zamenis longissimus*	*1*	*50*	*0.05*
8	*Salamandra salamandra*	*2*	5	0.0005	41	*Neomys anomalus*	*1*	*300*	*0.01*
9	*Hierophis viridiflavus*	*3*	100	0.05	42	*Sorex samniticus*	*1*	*200*	*0.01*
10	*Elaphe situla*	*2*	25	0.05	43	*Suncus etruscus*	*1*	*200*	*0.01*
11	*Hemidactylus turcicus*	*1*	5	0.0015	44	*Talpa caeca*	*1*	*3*	*0.15*
12	*Podarcis sicula*	*1*	25	0.0001	45	*Talpa romana*	*1*	*3*	*0.15*
13	*Podarcis muralis*	*1*	15	0.0001	46	*Lepus corsicanus*	*3*	*1000*	*1*
14	*Lacerta viridis*	*2*	5	0.002	47	*Chionomys nivalis*	*2*	*20*	*1*
15	*Tarentola mauritanica*	*1*	5	0.0005	48	*Myocastor coypus*	*1*	*50*	*2*
16	*Testudo hermanni*	*3*	100	0.001	49	*Mustela putorius*	*1*	*150*	*8*
17	*Vipera aspis*	*2*	15	0.005	50	*Sus scrofa*	*1*	*3000*	*100*
18	*Rana italic*	*2*	75	0.002	51	*Chalcides chalcides*	*1*	*20*	*0.0001*
19	*Salamandrina terdigitata*	*2*	5	0.001	52	*Natrix natrix*	*1*	*50*	*0.05*
20	*Dryomys nitedula*	*2*	1000	1	53	*Crocidura leucodon*	*1*	*200*	*0.1*
21	*Muscardinus avellanarius*	*3*	1000	1	54	*Crocidura suaveolens*	*1*	*200*	*0.1*
22	*Felis silvestris*	*3*	1000	70	55	*Neomys fodiens*	*1*	*50*	*0.002*
23	*Lutra lutra*	*3*	2000	10	56	*Sorex minutus*	*1*	*200*	*0.053*
24	*Bombina pachypus*	*3*	50	0.005	57	*Glis glis*	*1*	*500*	*1*
25	*Hyla intermedia*	*2*	100	0.005	58	*Myodes glareolus*	*1*	*10*	*0.05*
26	*Rana dalmatina*	*1*	50	0.002	59	*Apodemus flavicollis*	*1*	*400*	*1*
27	*Triturus carnifex*	*2*	50	1	60	*Apodemus sylvaticus*	*1*	*300*	*0.2*
28	*Lissotriton italicus*	*1*	50	1	61	*Mus domesticus*	*1*	*50*	*1*
29	*Coronella austriaca*	*1*	30	0.05	62	*Rattus norvegicus*	*1*	*200*	*1*
30	*Elaphe quatuorlineata*	*3*	50	0.05	63	*Cervus elaphus*	*2*	*750*	*75*
31	*Natrix tessellata*	*2*	50	0.05	64	*Mustela nivalis*	*1*	*500*	*1*
32	*Eliomys quercinus*	*3*	1000	1	65	*Meles meles*	*1*	*1000*	*25*
33	*Hystrix cristata*	*2*	5000	1	66	*Rattus rattus*	*1*	*200*	*1*

$$MFR = \frac{\sum_{i=1}^{66} MFR_i \cdot w_i}{\sum_{i=1}^{66} w_i} = 507.5\ m \qquad MPS = \frac{\sum_{i=1}^{66} MPS_i \cdot w_i}{\sum_{i=1}^{66} w_i} = 10.4\ ha$$

Table S3. Distribution of the surface of the study area according to the Corine Land Cover (CLC) classes in the three years under investigation.

	Surface [ha]			Variations [%]		
CLC class	**1990**	**2012**	**2018**	**1990-2012**	**2012-2018**	**1990-2018**
111	2,958.96	4,117.72	4,132.64	*39.16%*	*0.36%*	*39.67%*
112	30,296.68	41,193.80	41,564.84	*35.97%*	*0.90%*	*37.19%*
121	3,991.60	6,695.96	6,771.28	*67.75%*	*1.12%*	*69.64%*
122	309.28	332.32	749.08	*7.45%*	*125.41%*	*142.20%*
123	741.80	943.80	771.68	*27.23%*	*-18.24%*	*4.03%*
124	614.32	686.52	686.48	*11.75%*	*-0.01%*	*11.75%*
131	967.44	1,193.88	1,049.64	*23.41%*	*-12.08%*	*8.50%*
133	987.48	611.36	112.08	*-38.09%*	*-81.67%*	*-88.65%*
142	730.60	1,458.48	1,441.72	*99.63%*	*-1.15%*	*97.33%*
211	228,842.20	214,714.28	214,174.88	*-6.17%*	*-0.25%*	*-6.41%*
212	1,047.68	748.16	825.40	*-28.59%*	*10.32%*	*-21.22%*
213	0.00	734.00	733.84	--	*-0.02%*	--
221	1,329.44	4,482.60	4,481.88	*237.18%*	*-0.02%*	*237.13%*
222	44,760.80	52,479.96	54,199.80	*17.25%*	*3.28%*	*21.09%*
223	194,290.80	191,520.08	194,991.76	*-1.43%*	*1.81%*	*0.36%*
231	7,315.08	9,348.60	10,796.92	*27.80%*	*15.49%*	*47.60%*
241	91,895.40	61,706.32	54,680.52	*-32.85%*	*-11.39%*	*-40.50%*
242	44,742.20	104,830.76	105,525.24	*134.30%*	*0.66%*	*135.85%*
243	90,875.64	105,694.84	105,695.92	*16.31%*	*0.00%*	*16.31%*
244	71,010.84	183.24	31.68	*-99.74%*	*-82.71%*	*-99.96%*
311	383,730.36	371,005.64	372,003.24	*-3.32%*	*0.27%*	*-3.06%*
312	95,882.12	80,726.92	80,876.88	*-15.81%*	*0.19%*	*-15.65%*
313	111,470.56	120,647.76	121,587.80	*8.23%*	*0.78%*	*9.08%*
321	57,222.16	50,003.36	18,340.00	*-12.62%*	*-63.32%*	*-67.95%*
323	25,709.68	34,062.12	35,683.00	*32.49%*	*4.76%*	*38.79%*
324	45,569.36	79,485.16	77,181.56	*74.43%*	*-2.90%*	*69.37%*
331	14,528.88	14,115.28	13,881.56	*-2.85%*	*-1.66%*	*-4.46%*
332	2,691.32	1,896.24	1,932.76	*-29.54%*	*1.93%*	*-28.19%*
333	10,093.04	8,123.40	38,962.96	*-19.51%*	*379.64%*	*286.04%*
334	574.72	699.00	574.72	*21.62%*	*-17.78%*	*0.00%*
411	58.00	40.00	39.96	*-31.03%*	*-0.10%*	*-31.10%*
512	2,538.16	3,295.04	3,294.88	*29.82%*	*0.00%*	*29.81%*
Total	***1,567,776.6***	***1,567,776.6***	***1,567,776.6***			

Table S4. Distribution of the surface of patches and corridors in the three designed ecological networks (ENs) according to the Corine Land Cover (CLC) classes.

CLC class	1990		2012		2018	
	patches	*corridors*	*patches*	*corridors*	*patches*	*corridors*
	Surfaces [ha]					
111	0.00	121.60	0.00	167.68	0.00	110.80
112	0.12	2,350.56	131.24	2,374.28	234.52	3,968.96
121	0.00	59.80	19.16	144.20	65.68	324.28
122	0.00	5.92	0.00	21.20	0.00	14.32
123	0.00	12.56	0.00	3.32	0.40	0.00
124	0.00	62.44	13.16	18.76	25.24	10.24
131	0.16	125.64	7.72	291.16	10.36	180.88
133	0.00	72.88	0.00	75.68	0.00	1.04
142	0.00	60.00	70.16	139.92	95.80	231.32
Subtotal	***0.28***	***2,871.40***	***241.44***	***3,236.20***	***432.00***	***4,841.84***
211	6.56	47,903.84	2,890.72	36,361.52	3,603.32	53,473.92
212	0.00	42.64	0.00	21.72	0.36	82.20
213	0.00	0.00	215.96	22.48	225.52	141.40
221	0.00	44.44	0.00	652.76	0.00	845.12
222	0.00	7,941.92	780.64	6,967.68	1,069.64	13,463.60
223	18.64	50,496.44	2,374.84	39,351.84	2,772.16	61,290.72
231	5.40	2,351.96	498.28	1,651.20	532.68	2,729.52
241	18.44	27,820.96	545.64	13,272.92	671.88	16,698.36
242	3.40	7,295.40	759.40	22,251.64	1,103.76	33,672.24
243	326.44	23,711.64	1,701.92	19,896.52	2,052.44	33,848.40
Subtotal	***378.88***	***167,609.24***	***9,767.40***	***140,450.28***	***12,031.76***	***216,245.48***
311	103,797.44	104,354.60	81,725.36	71,626.72	108,925.60	100,650.88
312	626.48	35,584.68	7,692.56	16,360.80	8,889.68	27,819.80
313	4,499.28	42,677.80	9,455.00	18,179.24	11,293.24	42,645.72
321	201.84	14,061.40	2,916.32	10,184.16	1,937.56	4,463.12
323	445.80	8,273.16	1,640.48	20,614.28	1,946.88	13,353.36
324	787.00	12,034.36	2,993.32	17,298.40	3,762.52	26,524.60
331	46.52	2,263.40	3,251.12	2,123.32	3,681.24	3,153.28
332	0.08	895.16	155.64	493.64	134.20	515.52
333	3.20	1,688.60	981.28	1,450.84	2,778.64	12,054.60
334	0.00	0.00	25.16	185.28	0.00	180.52
Subtotal	***110,407.64***	***221,833.16***	***110,836.24***	***158,516.68***	***143,349.56***	***231,361.40***
411	0.00	0.00	37.72	0.00	39.12	0.76
512	0.00	707.20	562.04	773.40	662.04	1,321.68
Total	**110,786.80**	**393,021.00**	**121,444.84**	**302,976.56**	**156,514.48**	**453,771.16**
% of the Calabria region	7.07%	25.07%	7.75%	19.33%	9.98%	28.94%
	32.14%		27.07%		38.93%	