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How to design an ecological network for forest-dwelling species in a highly fragmented agro-ecosystem

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Contents

Chapter 1: Introduction, objectives and thesis structure

Chapter 2: A method to evaluate the combined effect of tree species composition and woodland structure on indicator birds

Chapter 3: Identification of ecological network elements by means of simulated species landscape perceptions

Chapter 4: An ecological network for forest mammals: a multi-species and multi-scale approach

Chapter 5: How to manage hedgerows as functional elements in an ecological network for forest mammals: a two-species approach

Chapter 6: Priority areas to plant new hedgerows to enhance functional connectivity for the Hazel Dormouse in fragmented landscapes

Chapter 7: General conclusions

Scientific papers and conference proceedings

Abstract

Forest fragmentation alters the structure and dynamic of wild animal populations, subdividing them into small and isolated sub-populations particularly prone to extinction. To achieve long-term persistence of fragmented populations, conservation strategies should be implemented at the landscape level, linking residual habitats by a network to guarantee their ecological connectivity. In this context, the appropriate landscape management tools are ecological networks, integrated landscape systems composed of nodes of suitable habitat, hosting animal populations, linked by connectivity elements.

The main aim of this research was to provide practical suggestions to design an effective ecological network for forest-dwelling species in an agro-ecosystem in northern Italy adopting a multi-species and multi-scale approach. The specific objectives of the study were: (i) to define how to correctly manage forest remnants; (ii) to define which land-covers can be used as secondary nodes and connectivity elements; and (iii) to identify priority areas where secondary nodes and connectivity elements should be located to increase landscape connectivity.

The first part of the research has been addressed to implement a new method to define forestry prescriptions by considering the combined effect of forest structure and tree species composition on four indicator bird species. This study showed the importance of maintaining forest remnants characterized by a high tree species richness, a high number of large trees and a high variety of stem diameters. Starting from the ecological requirements of the indicator species, quantitative forestry guidelines for singular plant species and diameter classes were provided for woodlands characterized by different exploitation degrees.

In the second part of the study a new method to identify secondary nodes and connectivity elements was implemented. The method allows inferring the species perception of land-covers, starting from the actual species distribution within the landscape and evaluating how it is affected by the configuration of different simulated landscapes. The method was tested on the Hazel Dormouse, the European Badger and the Roe Deer, three mammal species characterized by dissimilar forest specialization degrees and dispersal abilities, to evaluate how different agroforestry elements could be used in an ecological network. The results showed that hedgerows can play the role of nodes, while poplar cultivations, plantations for biomasses production and reforestations should be used as connectivity elements for the Hazel Dormouse.

For the European Badger poplar cultivations and hedgerows should be used as nodes and biomasses as connectivity elements; while for the Roe Deer poplar cultivations and reforestations should be used as nodes and biomasses and hedgerows as connectivity elements. These results revealed the key role of hedgerows in ecological networks for forest-dwelling species. Thus, which structural and floristic characteristics should be maintained to make hedgerows effective elements of ecological networks was investigated. The results suggested that hedgerows should be wide and continuous and should be managed to allow the growth of native species with a complex physical structure in the shrub layer. Furthermore, a study was addressed to define where to plant new hedgerows to improve the landscape connectivity for the Hazel Dormouse. The comparison of the increase of connectivity reached by planting new hedgerows along existing dispersal paths or by planting them in new strategic areas creating new corridors showed that this last strategy guarantees an increase of connectivity four times greater than the first one.

Chapter 1

Introduction, objectives and thesis structure

1.1 Introduction

1.1.1 Habitat fragmentation

1.1.1.1 Landscape changes and biodiversity decline: an issue of global concern

Global biodiversity is declining at an unprecedented rate (Pimm et al. 1995; Sala et al. 2000) as a complex response to multiple environmental human-induced changes (Vitousek 1994). In this context, habitat fragmentation due to land use changes is recognized as one of the most serious threats to wildlife populations' persistence (Debinski and Holt 2000; Fischer and Lindenmayer 2007; Mortelliti et al. 2010). The degradation of the natural environment is not a new phenomenon, but it is the rapidity and the wide scale at which these changes are now taking place that cause great alarm (Brown 1981; Myers 1986; Lunney 1991; Houghton 1994). Habitat fragmentation is so intense that most of the common landscape patterns worldwide are mosaics of human settlements, farmlands and scattered patches of natural habitat (Bennett 2003).

1.1.1.2 Effects of habitat fragmentation on wildlife populations

The phenomenon of habitat fragmentation is a dynamic process, which implies three main effects on habitat pattern: (i) the habitat loss and the consequent reduction in habitat amount within the landscape, (ii) the decrease in size and (iii) the increase in isolation of the remaining habitat patches (Bennett 2003).

A habitat is a "subset of physical environmental factors that permit an animal (or plant) to survive and reproduce" (Block and Brennan 1993) and the number of individuals of any species that a landscape can support is a positive function of the amount of habitat available to that species in the landscape (Fahrig 2003). It is then not surprising that habitat loss is the major cause of species loss all over the word (Fahrig 1999; Craig et al. 2000; Thomas et al. 2003). The most adopted tool to preserve the remaining habitats and their animal communities was the institution of protected areas (Pullin 2004). However, it is now widely recognized that in fragmented landscapes protected areas alone may not ensure an effective wildlife conservation (Cabeza 2013). Indeed, as mentioned above, beyond habitat loss, the process of habitat fragmentation also results in the decrease in sizes of the residual habitat patches and in the increase of their isolation.

Small and isolated populations inhabiting the residual habitat patches are more prone to extinction risk, respect to larger populations inhabiting continuous habitats, as they mostly suffer the deleterious effects of the genetic drift and inbreeding, two important genetic processes that can drastically decrease the vitality of natural populations (Caughley 1994; Crnokrack and Roff 1999). Genetic drift takes place when the frequency of alleles within a population increases or decreases by chance over subsequent generations. This process occurs more severely in small and isolated populations, where infrequently occurring alleles face a greater chance of being lost. It produces relatively quick random changes in allele frequencies over generations and, consequently, shrinks the population genetic diversity. Similarly, the phenomenon of inbreeding increases in small and isolated populations, because the mating probability between inbred is higher due to the low number of reproductive individuals and the lack of immigrants. This is particularly threatening because in natural populations recessive deleterious alleles are generally "hidden" by their dominant non-deleterious counterparts through the mechanisms of natural selection. An individual carrying a single recessive deleterious allele does not generally show any negative effect and can easily pass the deleterious allele into the next generation. When a natural population is large enough, it can carry many recessive deleterious alleles but they are rarely expressed (Frankham et al. 2002). However, in small populations, mating between relatives that likely carry the same recessive deleterious alleles is more common. When relatives mate, the offspring may inherit two copies of the same recessive deleterious allele and suffer the consequences of expressing it. This process decreases the individuals' ability to survive and reproduce over time leading to a phenomenon called inbreeding depression (Frankham et al. 2002). Although the mechanisms of genetic drift and inbreeding are different, the effects on small populations are the same: both genetic drift and inbreeding reduce genetic diversity, decrease population’s growth rate and negatively affect its potential for response to environmental changes (Caughley 1994; Bijlsma and Loeschcke 2012). Furthermore, in small and isolated populations typically inhabiting fragmented landscapes, the effects of demographic and environmental stochasticity are relatively high (Harmon and Braude 2010), contributing to a progressive reduction of the already small and isolated populations. Genetic drift, inbreeding and demographic and environmental stochasticity trigger a positive feedback mechanism known as extinction vortex (Gilpin and Soulé 1986). When this phenomenon leads the size of a population to fall below a minimum threshold known as *minimum viable population* (Caughley 1994), the population necessarily goes extinct. Other than at the local scale, i.e. the scale of local populations inhabiting small and isolated patches, habitat fragmentation has deleterious effects also at the landscape scale, i.e. the scale of the overall population. Wild animal populations can only persist if long-term reproduction outweighs long-term mortality and the extinction threshold occurs at the amount of habitat at which mortality balances reproduction over the landscape (Lande 1987). Any factor that affects how habitat amount influences this balance will affect the extinction threshold. Habitat fragmentation *per se* (residual habitat arrangement independently to habitat loss) has been suggested as one such factor (Fahrig 2002; Flather and Bevers 2002). Models predict that habitat fragmentation leads to a nonlinear increase of the extinction threshold, such that more habitat is needed for population persistence in more fragmented landscapes (Burkey 1995, Hill and Caswell 1999, With and King 1999). Indeed, at the landscape scale, habitat fragmentation results in a patchy distribution of the overall population, which is reduced to a set of local populations inhabiting residual habitat patches. In this context, the probability that individuals enter and spend time in the anthropogenic matrix, where reproduction is not possible and mortality rate is assumed to be higher, increases. This phenomenon produces a decrease in the overall reproduction rate and an increase in the overall mortality rate within the landscape, leading to a more rapid balance of mortality and reproduction in fragmented populations respect to continuous populations. This results in a decrease of the overall population size, causing an indirect decrease in the rate of patch immigration (Fahrig 2002). Reduced immigration reduces the rescue effect and the colonization rate of residual habitat patches where, as mentioned above, the local extinction rate is increased by genetic drift, inbreeding and demographic and environmental stochasticity. To achieve the long-term persistence of populations inhabiting fragmented landscapes it is thus crucial to mitigate the deleterious effects of fragmentation both at the local and at the landscape scale, by promoting the exchange of individuals among habitat patches. In other words, in fragmented landscapes, species require to be conserved as metapopulations (Soulè and Terborgh 1999; Hanski and Gaggiotti 2004; Hilty et al. 2006).

1.1.1.3 Metapopulations in fragmented landscapes

A metapopulation is a set of local populations which interact via moving individuals, where a local population is a set of individuals which interact with each other with a higher probability respect to what they interact with individuals of other local populations (Hanski and Gilpin 1991). In fragmented landscapes, individuals of metapopulations inhabit residual habitat patches spatially separated from each other. The single local populations have a considerable risk of extinction. Conversely, metapopulations are stable for long time, when they are in a dynamic equilibrium between extinction of local populations and colonization of currently empty suitable habitat patches. The long-term viability of a metapopulation depends on the dispersal of individuals among the habitat patches (Hanski 1999). In particular, a metapopulation depends on the viability of source patches (characterized by a positive population’s growth rate) able to sustain a roughly constant emigration rate of individuals that disperse toward sink patches (characterized by a negative population’s growth rate) affected by a remarkable risk of extinction (Lindenmayer and Fischer 2013).

Many species formerly characterized by a continuous distribution are being turned into spatially separated local populations by habitat fragmentation, and their long-term persistence relies on the possibility of individuals to disperse among local populations inhabiting habitat patches, in the form of an ecologically functioning metapopulation. However, the dispersal process is often critically hindered in fragmented landscapes where habitat patches are embedded in anthropogenic impermeable matrices (Fahrig 2003; Bani et al. 2016). Therefore, to counteract habitat fragmentation, wildlife conservation strategies should be implemented in order to maintain functional metapopulations at the landscape level, linking residual patches by enhancing the ecological connectivity (Noss et al. 1997; Soulè and Terborgh 1999).

1.1.1.4 The role of connectivity in maintaining viable metapopulations

Connectivity was originally defined as ‘‘the degree to which the landscape facilitates or impedes [individual] movement among resource patches’’ (Taylor et al. 1993). Connectivity is a major determinant of the dispersal rate of organisms between habitat patches and consequently has significant effects on metapopulation viability (Hanski 1999). The concept of connectivity is adopted to quantify how the spatial arrangement and the suitability of elements in the landscape affect the movement of individuals among habitat patches (Merriam 1991; Taylor et al. 1993; Forman 1995).

There are two main components which determinate connectivity – a structural component and a functional component (Bennett 2003; Wiens 2006). The structural component of connectivity disregards the response of individuals to landscape structure and only considers the spatial arrangement of different types of habitats in the landscape. It only refers to spatial features such as the amount and continuity of suitable habitat and the distance separating habitat patches. The functional (or ecological) component of connectivity, on the other hand, describe how the landscape composition and configuration influence the degree of movement or flow of individuals through the landscape, integrating structural connectivity with ecological information regarding the target species (Kadoya 2009).

Ecological studies aimed to the long-term wildlife conservation in fragmented landscapes, should therefore maintain and/or enhance the functional component of connectivity.

1.1.2 Fragmentation of forest habitats

Among all the human-induced phenomena leading to the decline of biodiversity, the process of forest fragmentation has been identified as one of the most important factor contributing to the decline and loss of species diversity worldwide (Noss and Cooperrider 1994).

Forest fragmentation occurs when a large extent of forest is modified by human activities, such as the conversion of forests into agricultural areas, logging and urban sprawl (Forman 1995), and the original cover becomes fragmented into a collection of smaller patches of forest habitat (Wilcove et al. 1986; Collingham and Huntley 2000; Fahrig 2003).

The severity of the negative impact of forest fragmentation on biodiversity firstly depends on the magnitude of the phenomenon of deforestation. Haddad et al. (2015) reported that deforestation led to the loss of more than a third of all forest cover worldwide. The authors highlighted that the effects of this process are more evident in the boreal biome. Indeed, forest fragmentation is not a random process and it has mainly affected most productive areas over the time (Bennett 2003; Lindenmayer and Fischer 2013). For example, in the last 2000 years, western Europe has been affected by a continuous spread of cultivated areas at the expense of the original broadleaved forests. Here, as in many other contexts, deforestation is particularly noticeable in lowland areas, where the original forest cover has been almost completely clear-cut (Darby 1956; Williams 2002).

Besides being a widespread phenomenon, the fragmentation of forest habitats has a major impact on wildlife, compared to the fragmentation of other habitat types, for different reasons. First, forest fragmentation originates landscapes characterized by remaining habitat patches (i.e.

forest remnants) associated with a degree of suitability for forest-dwelling species much higher than that of the anthropogenic matrix surrounding them (generally composed of intensive croplands and urban areas). The high difference between the degree of suitability of forest patches with respect to the matrix in which they are embedded, exacerbates the isolation between the patches and its negative effect on dispersal processes (Lindenmayer and Fischer 2013).

Another ecological phenomenon which is accentuated by the highly different suitability degree of forest patches and anthropogenic matrix is the edge effect (Laurence et al. 1997), which is also dramatically enhanced by the increase of the ratio of edge to interior habitats due to fragmentation (Paton 1994; Batáry and Báldi 2004). Edge effect results from the interaction between two adjacent habitats separated by an abrupt transition (Murcia 1995). In fragmented forest contexts, it leads to changes of biotic and abiotic conditions that may degrade the interior habitat of forest patches (Matlack & Litvaitis 1999; Fahrig 2003; Huhta et al. 2004). The portion of the residual patches under the influence of edge effect further reduces the effective extension of suitable habitat (Murcia 1995), exacerbating the negative effect of small patches on local animal populations. Forest degradation due to edge effects induced by fragmentation has an impact on forests ecosystems comparable to that of deforestation. The analysis of the global tree cover carried out by Haddad et al. (2015) revealed that nearly 20% of the world’s remaining forest is within 100 m of an edge and more than 70% is within 1 km, in close proximity to areas highly modified by humans. Thus, most of the world’s forests are located within the range where processes such as microclimatic alterations and interactions with non-forest species may influence and degrade forest habitats.

The integrity of forest habitat within remnants is not only indirectly affected by human activities through the edge effect, but also directly affected by forestry activities (Lindenmayer and Fischer 2013). Indeed, many forms of forestry ensure the preservation of forest cover, but significantly reduce its suitability for several species, by trivializing the structural complexity of forest remnants, which are usually more exploited than continuous forests because of their accessibility, causing long-term deleterious changes in forest composition and structure (Mueck et al. 1996; Ough 2001). Compositional and structural features which provide critical habitat components for several forest species include: mixtures of tree species and other plant taxa, trees from multiple age cohorts within stands, large living trees and standing dead trees, large-diameter logs on the forest floor, and vertical structural complexity crated by multiple canopy layers (Lindenmayer and Franklin 2002; Lindenmayer and Fischer 2013). Changes in these forest features can heavily affect taxa dependent on particular structural attributes (Gibbons and Lindenmayer 2002). For instance, old-growth forest-dependent species are particularly threatened by the alteration of structural features typical of old-growth stands (Scotts 1991; Matthysen 1998).

For all these reasons, forest-dwelling species are the most threatened by the phenomenon of habitat fragmentation. For their long-term persistence in fragmented landscapes, it is thus crucial to correctly manage forest remnants, to avoid an excessive alteration of stands’ structural complexity, and to maintain and enhance functional connectivity between spatially separated forest patches (Kramer-Schadt et al. 2004).

The management tool which integrates the protection of the structural integrity of forest patches and the maintenance of a high degree of connectivity among them, in order to maintain viable forest-dwelling species metapopulations, is the ecological network.

1.1.3 Forest-dwelling species conservation in fragmented landscapes: the role of the ecological network

1.1.3.1 The ecological network

The concept of ecological network has been developed over a number of decades in several European countries; beginning in Estonia and Czechoslovakia in the 1980s and in The Netherlands in 1990 (Jongman et al. 2004).

An ecological network is a management tool for wildlife conservation in fragmented landscapes. The main goal of an ecological network is to protect animal populations by promoting movement of individuals so that gene flow is maintained between local populations. Following the patch-matrix-corridor model proposed by Forman (1995), an ecological network is an integrated landscape system composed of nodes, that are patches of suitable habitats, linked by corridors which allow the animal dispersal process among nodes. Nodes and corridors are surrounded by the anthropogenic matrix, which dominates the landscape and hinders to a different extent individuals’ dispersal.

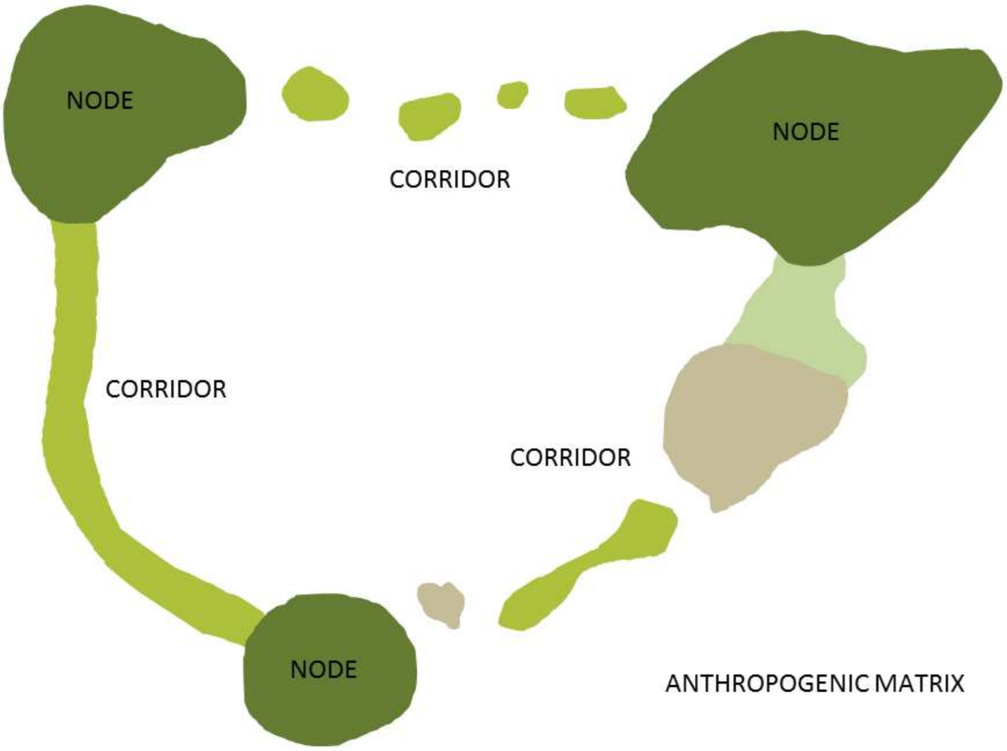
Forman (1995) originally defined corridors as strips of a particular land-cover type that differ from the adjacent land on both sided and connect two or more suitable patches. Subsequently, different definitions have been associated to the concept of corridor. Generally, three different types of corridor are mentioned: linear corridors, stepping stones corridors and landscape corridors. Linear corridors are long, uninterrupted strips of habitat; stepping stones corridors consist in a series of small, non-connected habitats used by species to find shelter, food, or to rest; landscape corridors consist of a mosaic of differently suitable landscape elements that offer sufficient cover for a safe passage from one habitat patch to another.

Despite differences in their definition, these three types of corridors do not differ so clearly between each other. They are landscape elements generally composed of sub-optimal habitats that, beyond their shape and structural continuity, can facilitate the animals' passage leading to the creation of paths preferentially traveled by dispersing individuals. This vision coincides with the more recent concept of corridors seen as least-cost paths (Knaapen et al. 2002, Adriaensen 2003). Least-cost paths are paths of minimum cumulative resistance, between each pair of residual patches. They are usually identified by mapping the matrix surrounding the residual patches in terms of a resistance surface, where different values of resistance to animal passage are associated to the land-cover types composing the matrix.

In this thesis, I will refer to corridor to indicate every landscape entity characterized by the presence of connectivity elements (every patch of permeable habitat, which allows animal passage despite its structure and composition). Corridors will then correspond to paths along which individuals preferentially disperse, regardless of whether it is a single linear element that completely connects two nodes, stepping stones embedded in the matrix, or a wide area separating nodes, characterized by different land-covers permeable to the passage of individuals (Fig. 1).

Figure 1

Example of an ecological network composed of nodes and different kind of corridors. In dark green are represented optimal habitats. The other colors represent sub-optimal habitats associated to different degrees of permeability to individuals' passage.



1.1.3.2 Designing an effective ecological network for forest-dwelling species

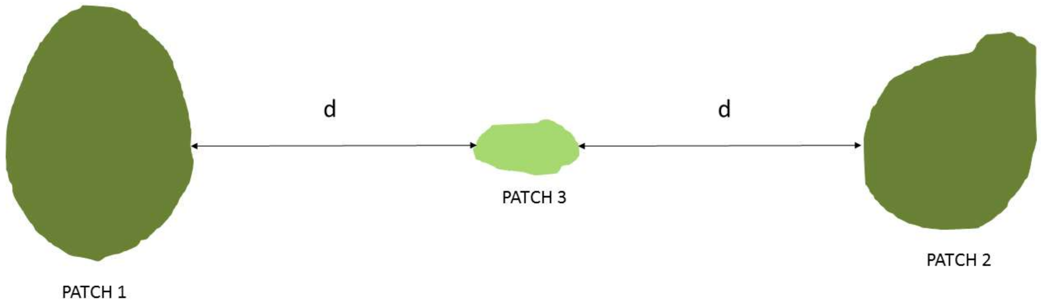
Many factors should be considered to design an ecological network for enhancing functional connectivity for forest-dwelling species in fragmented landscapes. Below four key steps that should be followed to designing an effective ecological network are summarized.

1.1.3.2.1 Adopting a multi-species and multi-scale approach

The effects of forest fragmentation, and of the consequent forest degradation, are strictly species-specific as the landscape composition and configuration are perceived differently by different species (Bennett 2003). First, different species have different requirements in terms of habitat quality, leading to dissimilar responses to the alteration of internal characteristics of forest remnants (i.e. of nodes). Moreover, landscape elements perceived as connectivity elements by some species can be perceived as impermeable by other species, for which they cannot be used as connectivity elements in an ecological network (Fig. 2). Finally, the degree to which the spatial arrangement of patches may provide connectivity for a species depends on its dispersal ability in the hostile matrix (Bowman et al. 2002; Cushman et al. 2013). Two patches may be perceived as connected by a species, but they can be simultaneously perceived as not connected by a species with a lower dispersal ability (Bennett 2003) (Fig. 2). The species dispersal ability among habitat remnants depends on the combined effect of species vagility and of the degree to which the species is negatively affected by the hostile matrix, which, in turn, depends on the degree of species specialization to forest habitat (Beier et al. 2008; Cushman and Landguth 2012). The resulting species dispersal ability determines the spatial scale at which different species suffer the negative effect of forest fragmentation.

Figure 2

Patch 1 and patch 2 are connected for a species that perceives as permeable (or suitable) the patch 3 and which has a dispersal ability in unsuitable habitat larger than *d*, but not for a species that perceives as impermeable the patch 3 and/or that has a dispersal ability in unsuitable habitat smaller than *d*.



Starting from these assumptions, ecological networks designed for a single target species hardly may ensure the long-term conservation of the entire forest community, or at least of a great part of it. In order to design an ecological network able to guarantee the conservation of a wide spectrum of forest-dwelling species, it is thus crucial to adopt a multi-species and multi-scale approach (Block et al. 2004). Particularly, it is important that management actions are not targeted on a single species but on multiple target species with different ecological needs (Cushman et al. 2013, Trainor et al. 2013), in terms of habitat requirements, degree of forest specialization and dispersal ability.

1.1.3.2.2 Ensuring high structural and floristic quality of forest remnants

To guarantee the functionality of an ecological network designed for the conservation of forest-dwelling species, it is firstly crucial that the remaining forest fragments, i.e. the primary nodes of the network, have internal characteristics adequately preserved. As mentioned in Section 1.2, the internal features of forest remnants, besides being altered by the unavoidable edge effect, are strongly degraded by forestry activities. However, even if forests have been managed for centuries, there is still a gap in knowledge of the quantitative relationships between forest structure and floristic composition alteration and biodiversity (Bani et al. 2006; Gil-Tena et al. 2007). To ensuring a sufficient degree of quality of forest remnants, it is thus important to identify target species particularly sensitive to structural and floristic alteration, to evaluate the effect of forest degradation on these species and to define which internal characteristics should be maintained and/or enhanced through sustainable forestry practices.

1.1.3.2.3 Identifying secondary nodes and connectivity elements

Once it has been defined which management strategies should be adopted to maintain high-quality forest remnants (i.e. primary nodes), it is fundamental to adopt specific analytical tools to identify which landscape elements could play the role of additional nodes (henceforth defined as secondary nodes). Moreover, it is crucial to identify which landscape elements could play the role of connectivity elements able to ensure the ecological connectivity among nodes, otherwise isolated by the anthropogenic matrix. In ecological networks designed for the conservation of forest-dwelling species in agricultural context, the landscape elements that could play the role of secondary nodes and/or connectivity elements promoting animal dispersal are typically represented by semi-natural elements. However, the suitability of these landscape elements is highly species-specific. In order to design an ecological network to protect the largest part of forest-dwelling species communities in agricultural landscapes, it is thus crucial to evaluate the suitability degree of these landscape elements for a set of target species with different ecological requirements. This way, information about which landscape elements can be used as secondary nodes or connectivity elements in a functional ecological network for different species can be obtained.

1.1.3.2.4 Estimating and enhancing the connectivity supported by the ecological network

After the identification of the landscape elements that can play the role of secondary nodes and connectivity elements for each target species, it is necessary to evaluate the current degree of landscape connectivity provided by their spatial configuration within the landscape. Different modeling methods have been adopted to quantify the overall landscape connectivity and to calculate the contribution of nodes and connectivity elements to the connectivity of the whole network (Gurrutxaga et al. 2011). These methods range from the calculation of simple structural metrics to considerably complex metapopulation models (Calabrese and Fagan 2004). Among these methods, graph-based models represent an effective, operational and flexible approach to analyze connectivity in heterogeneous landscapes (Carranza et al. 2012). Particularly, graph-based models handle the landscape as a set of nodes functionally connected to some degree by links (Urban and Keitt 2001; Saura and Pascual-Hortal 2007; Minor and Urban 2008; Vasas et al. 2009), corresponding to the pairwise Euclidean distances (e.g. Perotto-Baldivieso et al. 2009; García-Feced et al. 2011), least-cost distances between nodes (e.g. Watts and Handley 2010; Saura et al. 2011), or individual-based dispersal simulation (e.g. Morzillo et al. 2011).

Once estimated the connectivity associated to the current configuration of nodes and connectivity elements within the landscape, it is then crucial to provide information on how to enhance it by identifying priority areas where restoring and/or creating new nodes or connectivity elements in order to maximize the degree of connectivity supported by the ecological network. Graph theory is again one of the most suitable methods to reach this last crucial aim, as it allows calculating changes in connectivity associated to every possible landscape change caused by the addition of a node or a connectivity element in a specific site.

1.2 Objectives and thesis structure

1.2.1 Objectives

The main aim of this research was to provide practical suggestions to design an effective ecological network for forest-dwelling species in a highly fragmented landscape in northern Italy. Northern Italy is a highly representative area of the progressive fragmentation of broadleaved forest habitats, both on a temporal scale (original habitats replaced by intensive croplands in lowlands over the last centuries), and on a spatial scale (continuous mountain and hilly forests progressively fragmented from medium elevation to lowland by the agricultural matrix).

The specific objectives of this research were: (i) to define how to correctly manage residual forest fragments, i.e. the primary nodes of the network; (ii) to define which landscape elements can be used as secondary nodes and connectivity elements in an ecological network designed for forest-dwelling species; (iii) to identify priority areas where secondary nodes and connectivity elements should be located to enhance the current landscape connectivity provided by the network. Specifically, this research focused on the role that hedgerows, arboreal plantations (i.e. traditional poplar cultivations and arboreal plantations for biomass production, henceforth defined as biomasses), and reforestations could have in an ecological network for forest-dwelling species, trying to reconcile biodiversity conservation needs with economic interests.

1.2.2 Thesis structure

The structure of the central part of the thesis (Chapter 2-6) followed the four key steps to designing an effective ecological network summarized in Section 1.3.2; (i) adopting a multi-species and multi-scale approach; (ii) ensuring high structural and floristic quality of forest remnants, (iii) identifying secondary nodes and connectivity elements; (iv) estimating and enhancing the connectivity supported by the ecological network (Fig. 3).

In general, in this research, a multi-species and multi-scale approach was adopted, by considering different target species (characterized by different forest specialization degrees and dispersal abilities) depending on the specific objective pursued in each chapter. Specifically, the common approach that involves the planning of management actions aimed to improve habitat suitability for umbrella species (i.e. the most sensitive to a particular threatening factor, Lambeck

1997) was integrated with another strategy, according to which conservation actions should be directed also towards intermediate-sensitive species, with less restrictive ecological requirements in terms of forest structure and composition, habitat amount and landscape configuration, in order to maintain the largest part of the original biodiversity (Martensen et al. 2012).

In Chapter 2, a study carried out to define practical management actions aimed to guarantee a high-quality of forest remnants within the study area is reported. Specifically, a new method implemented to define forestry prescriptions by considering the combined effect of forest structure and tree species composition on indicator bird species, is described. Among the taxa most negatively affected by forest degradation, birds are considered good umbrella species, as they are particularly influenced by forest structure changes, and good indicators of the degree of forest alteration, because they rapidly respond to habitat disturbances (Hutto 1995; Bani et al. 2006; Sekercioglu 2006). The target species used to implement and test the proposed method were three secondary cavity-nesters characterized by a high degree of forest specialization (Marsh Tit *Poecile palustris*, European Nuthatch *Sitta europaea*, Short-toed Tree-creeper *Certhia brachydactyla*) and a more generalist species (Blue Tit *Cyanistes caeruleus*) in terms of integrity of floristic and structural features. The data used for the development of this study was obtained from a long-term monitoring program of breeding birds in Lombardy (Bani et al. 2009).

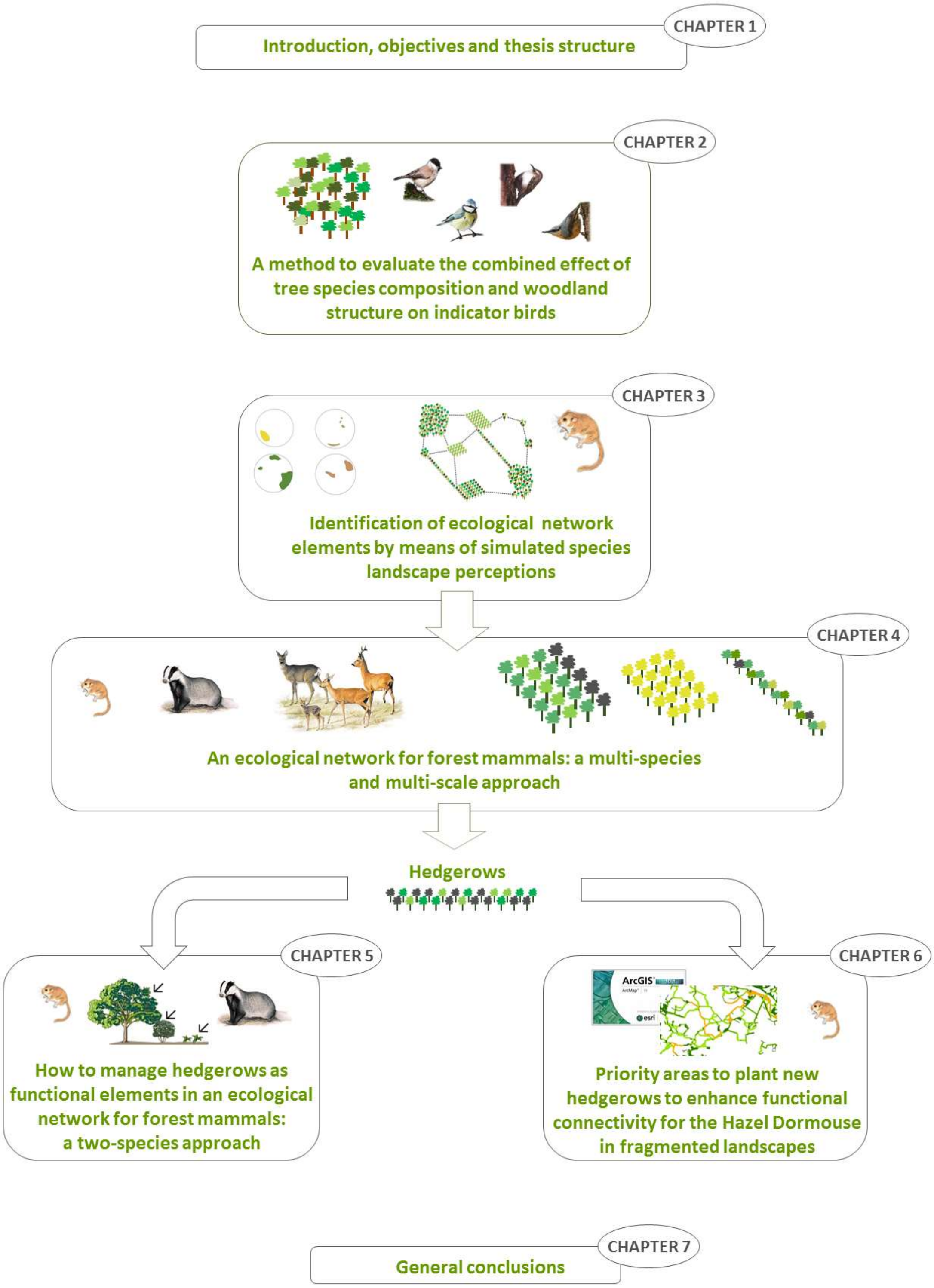
Once defined the management strategies to maintain high-quality forest remnants within the study area, Chapter 3-6 focused on the identification of secondary nodes and of connectivity elements for different mammal species, and on defining how they should be managed and where they should be located in the landscape in order to enhance functional connectivity. Mammals are often used as target species for designing ecological networks, because they are particularly sensitive to fragmentation dynamics (Gurrutxaga 2011) and because the nodes and corridors they require can be also used by multiple other species (Bruinderink et al. 2003; Beier et al. 2008). The target species selected for this research (Hazel Dormouse *Muscardinus avellanarius*, European Badger *Meles meles* and Roe Deer *Capreolus capreolus*) are characterized by a different size and vagility and by a different forest specialization degree. All the data used to carry out the studies described in these chapters were specifically collected for this thesis project.

In Chapter 3, a new method developed to objectively identify which land-cover types should be used as nodes and/or connectivity elements in an ecological network designed for a given species is described. The proposed method allows inferring the species perception of the different land-covers, starting from the actual species distribution within the landscape, and evaluating how it is affected by the configuration of different simulated landscapes (i.e. every possible combination of each land-cover within the landscape alternatively assuming the role of node or connectivity element). The new method was tested on occurrence data of the Hazel Dormouse collected within the study area in order to evaluate whether hedgerows, arboreal plantations and reforestations could play the role of secondary nodes or connectivity elements in an ecological network designed for this species. In Chapter 4, this new method was applied to identify whether hedgerows, arboreal plantations and reforestations could play the role of secondary nodes or connectivity elements for other two mammal species sensitive to forest fragmentation at different spatial scales: the European Badger and the Roe Deer. In this chapter, the results obtained for these two species and those obtained for the Hazel Dormouse in Chapter 3 were then compared in order to estimate at what extent the areas suitable and highly connected for the three species overlap. The application of the proposed method on the three target species revealed the key role of hedgerows in ecological networks designed for forest-dwelling species in agricultural landscapes, particularly for the Hazel Dormouse and the European Badger. In Chapter 5 and 6, I investigated how to correctly manage hedgerows by performing two different studies. In Chapter 5, I evaluated which structural and floristic characteristics make hedgerows suitable for the Hazel Dormouse and the European Badger in order to provide concrete suggestions for the correct management of these landscape elements as part of an ecological network designed for forest-dwelling species. In Chapter 6, I evaluated the landscape connectivity provided by hedgerows within a sample area and identified priority sites where new hedgerows should be located in order to enhance the landscape connectivity, using the Hazel Dormouse as target species. Specifically, by applying the graph theory approach, I compared the increase of connectivity reached by using the traditional strategy of habitat restoration along the existing corridors (Moqanaki & Cushman 2016), by simulating the plantation of new hedgerows along existing least-cost paths, with an alternative strategy, which involves the identification of new strategic areas where locating hedgerows in order to create new corridors within the landscape.

Lastly, in Chapter 7 some general conclusions regarding the results described in the previous five chapters were drawn and future perspectives were highlighted.

Figure 3

Flowchart of the thesis structure.



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Chapter 2

A method to evaluate the combined effect of tree species composition and woodland structure on indicator birds

The results of this chapter have been published in: Dondina, O., Orioli, V., Massimino, D., Pinoli, G., Bani, L., 2015. A method to evaluate the combined effect of tree species composition and woodland structure on indicator birds. *Ecological Indicators*. 55: 44–51. DOI: 10.1016/j.ecolind.2015.03.007.

Abstract

The study presented in this chapter focused on evaluating the effect of forest features' alteration on indicator bird species in order to provide information about how to correctly manage forest remnants. To reach this aim, data obtained from a long-term monitoring program of breeding birds in Lombardy were analyzed, and different kind of woodlands, characterized by different degrees of exploitation and distributed in whole Lombardy Region (northern Italy), were considered.

Traditional forest practices at the stand scale simultaneously alter both physical and floristic features with a negative effect on ecosystem processes. Thus, we tested and proposed a method to define forestry prescriptions taking into account the combined effect of woodland structure and tree species composition on the presence of four bird indicator species (Marsh Tit *Poecile palustris*, European Nuthatch *Sitta europaea*, Short-toed Tree-creeper *Certhya brachydactyla* and Blue Tit *Cyanistes caeruleus*). The study was carried out in Lombardy, from 2002 to 2005. By using a stratified cluster sampling design, Basal Area, one hundred tree trunk diameters at breast height (DBH) and tree species in 160 sampling plots, grouped in 23 sampling areas, were recorded. In each plot a bird survey using the point count method was also performed. We analyzed data using Multimodel Inference and Model Averaging on Generalized Linear Mixed Models, with species presence/absence as the response variable, sampling area as a random factor and forest covariates as fixed factors. In order to test the proposed method, we compared it with other two traditional approaches, which consider structural and tree floristic variables separately. Model comparison showed that the proposed method performed better than traditional ones, in both the evaluation and validation processes. In general, the results showed the importance of maintaining forest remnants characterized by a high tree species richness, a high number of large trees and a high variety of stem diameters. Based on our main results, in deciduous mixed forest where the exploitation demand is limited, we recommend maintaining at least 65 trees/ha with DBH>45 cm. In particular, we advise keeping 70 trees/ha with DBH>50 cm in chestnut forests and 300 trees/ha with DBH 20–30 cm in oak forests. Conversely, in more exploited forests, we advise maintaining at least 670 trees/ha with DBH 15–30 cm in chestnut forests and 100 trees/ha with DBH 10–15 cm in oak forests.

2.1 Introduction

Habitat degradation, meaning the gradual deterioration of habitat quality (Fischer and Lindenmayer 2007), is one of the most severe threats to global biodiversity (Foley et al. 2005). Forests represent one of the ecosystems most widely affected by this phenomenon, especially in fragmented landscapes. Indeed, intense human activity has strongly altered the original cover and the natural structure of woodlands in many regions of the world (Thiollay 1992; Berg et al. 1994; Clavero et al. 2011). However, even if forests have been managed for centuries, there is still a gap in knowledge of the quantitative relationships between forest structure and biodiversity, from wide to local scale (Bani et al. 2006; Gil-Tena et al. 2007). Certainly, management actions entail the loss of some physical features that alter forest structure (Lohr et al. 2002), thus affecting many animal groups linked to this habitat (Raphael and White 1984; Patton 1992; Newton 1994; Quine et al. 2007). Among these taxa, birds are considered good indicators of the degree of forest alteration because they rapidly respond to habitat disturbances (Hutto 1995; Bani et al. 2006; Sekercioglu 2006). Therefore, identifying good bird indicator species and understanding the influence of woodland characteristics on their presence is very important to reconcile conservation targets and the needs of economic exploitation of forest ecosystems (Franklin et al. 1997; Touihri et al. 2014). Vegetation structure is usually considered the most important factor locally affecting habitat suitability for temperate forest birds (Hildén 1965; Willson 1974; Hewson et al. 2011). The physical features that most positively influence the richness and abundance of the most sensitive bird species are logs, snags, and large-diameter trees, as well as canopy closure (Díaz et al. 2005; Fuller et al. 2012; Touihri et al. 2014). Gil-Tena et al. (2007) found that in the Mediterranean forests tree species diversity is another important variable explaining bird species richness. In fact, as highlighted in many other studies, the tree floristic diversity enhances the availability of specific resources provided by different plant species, such as food (seeds, fruits, invertebrates) and nest sites (Rotenberry 1985; MacNally 1995; Martin 1998; Díaz 2006). However, most of the previous research has independently considered the influence of structure and tree species on birds, although it has been proved that structure and floristic composition are intimately related (Hewson et al. 2011). Indeed, woodland structure, which is strongly influenced by tree species (Horn 1971), masks the pure effect of floristic composition on bird communities (Fuller 1995).

Therefore, developing research that takes into account the interaction between floristic composition and physical structure and evaluates its effect on the most sensitive bird species is fundamental to provide practical and detailed forestry guidelines in order to reach conservation goals.

In the present study, we investigated the relationship between the presence of four species of secondary cavity-nesting birds and forest features at the stand scale in order to provide information on how to correctly manage forest remnants to ensure the conservation of forest-dwelling species. We considered only indicator species, and not the whole bird community, as many studies have demonstrated how the use of requirements of the most demanding species is a good surrogate in supplying practical guidelines for conservation planning (Bani et al. 2006; Dunk et al. 2006; Roberge et al. 2008a, 2008b). We conducted the survey at stand scale because this is an appropriate scale at which to evaluate habitat relationships on territorial birds in managed temperate forests (Hewson et al. 2011) and also because the stand is the level at which management practices are generally conducted (Quine et al. 2007). The specific aims of this study were (i) to evaluate the independent effect of the structure and tree floristic diversity on the presence of indicator bird species; (ii) to evaluate the combined effect of these two variables; (iii) to assess the most effective among the two previous approaches in explaining the presence of our bird target species; and (iv) to provide practical management guidelines for different types of forest.

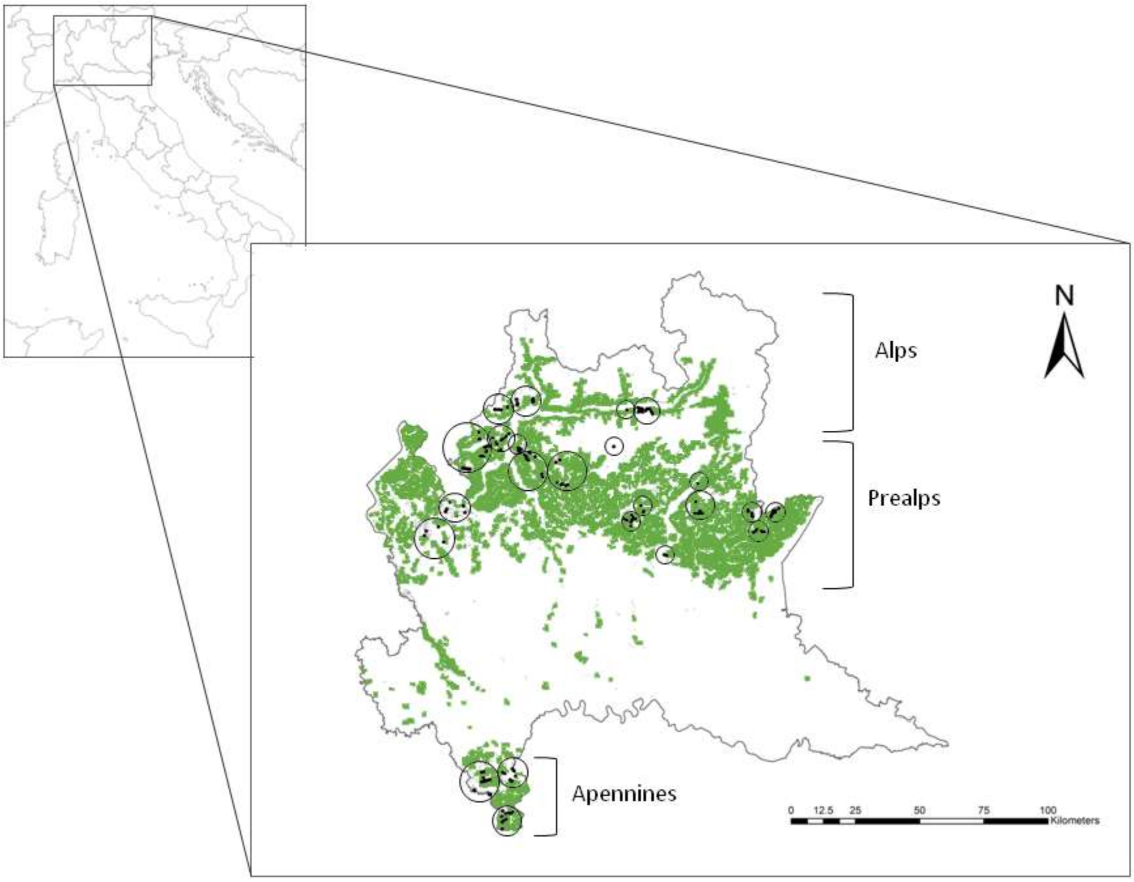
2.2 Methods

2.2.1 Study area

The study was carried out in Lombardy (northern Italy), a region 24,000 km^2 wide, with a forest cover of about 6,000 km^2. 63% of these forests are broadleaved, 15% mixed and 22% coniferous. Our study area is represented by three main forest sub-regions: the Alps, at the northern boundary; the Prealps; and the Apennines, at the southwestern corner of the region (Fig. 1). We decided to exclude floodplain forests (below 300 m) from our study area, because they are heavily fragmented, and the effect of isolation of forest patches on birds could mask those of forest structure and composition. In our study area, whilst almost all coniferous woodlands are managed as high forests, 94% of all broadleaved ones are managed as coppice, varying from simple coppice (coppice with standards of the same age class) to compound coppice (coppice with standards of several age classes) (Massimino et al. 2010). For this reason, in our context, most conservation concerns are linked to broadleaved forest ecosystems, which represent the case study of this research.

Figure 1

Lombardy region with sampling plots for forest and bird surveys (black dots) grouped in sampling areas (circles of varying sizes, 4 sampling areas in the Alps sub-region, 16 in the Prealps and 3 in the Apennines).



2.2.2 Forest data

Forest data were collected according to a stratified cluster sampling design. In the first stage, forests were stratified with respect to the three sub–regions of our study area. Within each stratum all accessible woodlands were identified as suitable sampling areas. In the second stage, from all suitable sampling areas a sample with size proportional to the percentage of broadleaved forests in each sub-region was randomly selected. As 15% of broadleaved woods in the study area are in the Alps, 75% in the Prealps and 10% in the Apennines, 4, 16 and 3 sampling areas were selected, respectively. In the last stage, within each sampling area, between four and 20 sampling plots, proportional to the size of the sampling area, were randomly selected. The plots were randomly selected using ArcGis (version 10.0, ESRI Inc.) forcing a minimum distance of 1 km between plots in order to collect independent data. 26 plots in the Alps, 103 in the Prealps and 31 in the Apennines were surveyed (160 sampling units on the whole, Fig. 1).

All 160 sampling plots were located in stands managed, during recent decades, as compound or simple coppice. From 2002 to 2005, in each sampling plot the Basal Area (BA) was estimated using a mirror relascope; the trunk diameters at breast height (DBH) of the first hundred trees, around the center of the plot where the BA was estimated, were measured; and the species of each measured tree was recorded. The number of logs and snags were not recorded as in our study area they are negligible due to heavy management practices that led to their removal.

2.2.3 Bird data

In each plot, it was simultaneously recorded the presence of four indicator species of secondary cavity-nesters: the Marsh Tit (*Poecile palustris*), European Nuthatch (*Sitta europaea*), Short-toed Tree-creeper (*Certhia brachydactyla*) and Blue Tit (*Cyanistes caeruleus*). We selected these indicator species as they proved to be the most affected by forest structure alteration, as highlighted in a previous study carried out in this area (Bani et al. 2006). Bird data were collected during the breeding season (from 10 May to 20 June) using the 10 minute unlimited-distance point-count technique (Blondel et al. 1981). Surveys were conducted from sunrise to 11 am, only in good weather (sunny to cloudy, without rain or strong wind). In the 160 surveyed plots, the Marsh Tit was found in 71 points (44.4%), the European Nuthatch in 42 (26.2%), the Short-toed Tree-creeper in 34 (21.2%) and the Blue Tit in 73 (45.6%).

2.2.4 Statistical analyses

In order to relate the bird species presence to woodland forest features, and simultaneously account for spatial dependence among sampling plots belonging to the same sampling area, we adopted Binary Logistic Regression Analyses (BLRA) (Rushton et al. 2004) with presence/absence of bird species as the response variable, sampling area as a random factor and forest covariates as fixed factors. We also took into account the temporal dependence by including the year as a random factor. However, the variance of the random factor year was not negligible only in the model C performed on the Blue Tit, therefore we re-ran all the other models without it. These analyses were performed by using Generalized Linear Mixed Models (GLMMs, McCulloch and Searle 2001), that allow the modeling of complex and realistic hierarchical biological systems and deal with non-normal response variables (Grueber et al. 2011). All the analyses were performed using R v. 3.1 (R Development Core Team, 2014) with the *lme4* package (Bates et al. 2012).

In order to assess if it is more effective considering the independent effect of the structure and tree floristic diversity or their combined effect on the presence of indicator bird species, we tested three different models.

Several studies usually take into account variables simplistically describing forest structure, such as the mean DBH and DBH variance, and the floristic richness separately (Laiolo, 2002; Díaz 2006). For this reason, in the first model (model A) we took into account, as covariates, (i) the tree species richness (S); (ii) the BA; and (iii) the mean value of DBH (DBH mean) and its variance (DBH var), considering all the tree species together.

Other studies use a more complex approach, describing forest structure quantifying the number of trees in diameter classes, but still considering independently the tree species composition (Díaz et al. 2005; Touihri et al. 2014). Accordingly, in the second model (model B) we took into account (i) the tree species richness; and (ii), considering all tree species as the whole, the number of trees per hectare in six diameter classes, whose limits were defined by the 10th , 25th, 50th, 75th and 90th percentile of the diameter variability distribution. Considering simultaneously all trees in all diameter classes, we discharged BA from this model.

Our approach aims to test the combined effect of forest structure and floristic composition. Hence, in the third model (model C), we used as covariates (i) the tree species richness; (ii) the

BA; and (iii) for each species, the number of trees per hectare in six diameter classes, defined by the above mentioned percentile values. In this last model, considering 23 surveyed tree floristic taxa and six diameter classes, the tree species variables amounted to 138. Among these, we considered only those that appeared in at least 10% of the sampling points. In this way, we selected 11 taxa (maples *Acer* spp. [Acespp], Birch *Betula pendula* [Betpen], Hornbeam *Carpinus betulus* [Carbet], Chestnut *Castanea sativa* [Cassat], Common hazel *Corylus avellana* [Corave], Beech *Fagus sylvatica* [Fagsyl], Ash *Fraxinus excelsior* [Fraexc], Manna Ash *Fraxinus ornus* [Fraorn], Hop Hornbeam *Ostrya carpinifolia* [Ostcar], oaks *Quercus* spp. [Quespp], Linden *Tilia platyphyllos* [Tilpla]) for a total of 45 tree variables. This high number of variables might affect modeling performance. In fact, as underlined by Massolo and Meriggi (1995) the number of covariates should not exceed one-fifth of the cases of presence. For this reason, in the last model, in order to identify the most important covariates affecting the response variable, we previously performed a covariates selection by means of the Wilcoxon test. Despite the high number of tests performed, we did not apply any correction for the significance level because we preferred to avoid a too restrictive approach in this preliminary covariates selection. Thus, we selected all the variables for which the mean values resulted significantly different among sites of species presence and those of absence (e.g. Treves et al. 2004). In addition, we excluded all the diametric classes of Beech from the model C performed on the data of European Nuthatch and Short-toed Tree-creeper because these two bird species never occurred in this kind of woodland in our study area.

For all models we standardized the covariates by centering and scaling each independent variable.

One of the most important assumptions of GLMMs is that predictors should not be strongly correlated among them (Quinn and Keough 2002; Zuur et al. 2009). For this reason we realized as many subsets as the combinations of independent variables whose pairwise correlation resulted < |0.5| using the Spearman method (Chiatante et al. 2013; Dondina et al. 2015). For each subset, we carried out a BLRA, obtaining several logistic models. In order to select the best models we used a Multimodel Inference approach. In particular, to evaluate the models in terms of goodness of fit, we used the percentage of explained deviance (ED), and, following an Information-Theoretic Approach, we ranked all the models using the corrected Akaike Information Criterion (AICc) and Akaike weights (w) (Anderson et al. 2000; Burnham and

Anderson 2002). In order to perform a Model Averaging, we retained all models with a ΔAICc≤2 (Burnham and Anderson 2002; Rayner et al. 2007). We assessed the relative importance (W) of the predictor variables by the sum of Akaike weights (w) of each model in which they appeared (Burnham and Anderson 2002). We validated the averaged models testing their effectiveness in correctly classifying presence and absence sites using the Area Under the Curve (AUC) of the Receiver Operating Characteristic (ROC) (Pearce and Ferrier 2000; Boyce et al. 2002; Fawcett 2006). In order to classify the accuracy of the diagnostic test we used the traditional academic point system (Swets 1988): 0.5–0.6=fail, 0.6–0.7=poor, 0.7–0.8=fair, 0.8–0.9=good, 0.9–1.0=excellent.

Finally, we performed univariate models for each significant tree variable from models B and C and we identified the minimum required number of trees per hectare as the values of each tree variable corresponding to the 0.9 probability of occurrence of the bird species (Roberge et al. 2008a; Touihri et al. 2014).

2.3 Results

2.3.1 Models

2.3.1.1 Model A

The Multimodel Inference selected six models and four independent variables for the Marsh Tit, only one model and two independent variables for the European Nuthatch, two models and three independent variables for the Short-toed Tree-creeper, and three models and three independent variables for the Blue Tit (Table S1 in Supplementary materials). The most important covariate positively affecting the probability of presence of all four bird species was the tree species richness (Table 1). In addition, the European Nuthatch was also significantly positively affected by the mean value of DBH, and the Short-toed Tree-creeper by the DBH variance (Table 1).

Table 1

Model A: Results of Model Averaging on logistic models with ΔAICc≤2 performed on presence/absence data of Marsh Tit (6 models), European Nuthatch (1 model), Short-toed Tree-creeper (2 models) and Blue Tit (3 models).

Bird Species	Predictors	Estimate	OR	SE	z	Pr(>|z|)	N	W
Marsh Tit	(Intercept)	−0.564	-	0.388	1.441	0.15	–	–
	S	0.533	1.704	0.238	2.226	0.026	6	1.00
	BA	0.178	1.194	0.236	0.753	0.452	3	0.52
	DBH var	0.090	1.094	0.185	0.482	0.630	2	0.33
	DBH mean	0.031	1.031	0.129	0.241	0.809	2	0.20
European Nuthatch	(Intercept)	−2.196	-	0.608	−3.610	<0.001	–	–
	DBH mean	1.393	4.027	0.395	3.524	<0.001	–	–
	S	1.260	3.525	0.396	3.184	0.001	–	–
Short-toed Tree-creeper	(Intercept)	−3.362	-	1.008	3.309	0.001	–	–
	DBH var	1.720	5.584	0.493	3.462	0.001	2	1.00
	S	0.926	2.524	0.464	1.981	0.048	2	1.00
	BA	0.370	1.447	0.406	0.908	0.364	1	0.61
Blue Tit	(Intercept)	−0.622	-	0.487	1.268	0.205	–	–
	S	0.744	2.104	0.275	2.68	0.007	3	1.00
	DBH mean	0.093	1.097	0.189	0.491	0.623	1	0.33
	DBH var	0.045	1.046	0.133	0.339	0.735	1	0.23

S = Tree species richness; BA = Basal Area; DBH mean = mean diameters at breast height; DBH var = variance of diameters at breast height; OR = odds-ratio; SE = unconditional standard error; N = number of selected models containing the independent variable; W = relative importance of the predictor variables.

2.3.1.2 Model B

By using the second set of variables, the Multimodel Inference analysis selected seven models for the Marsh Tit, with a total of six independent variables, three models and five independent variables for the European Nuthatch, two models and four independent variables for the Short-toed Tree-creeper and 10 models and seven independent variables for the Blue Tit (Table S2 in Supplementary materials). Even using this model, the most important variable positively affecting the probability of presence of all four target species was the tree species richness (Table 2). In addition, the European Nuthatch was also negatively affected by the number of trees per hectare with a DBH ranging from 8 cm to 15 cm, whereas the Short-toed Tree-creeper was positively influenced by the number of trees per hectare with a DBH larger than 45 cm (Table 2).

Table 2

Model B: Results of Model Averaging on logistic models with ΔAICc≤2 performed on presence/absence data of Marsh Tit (7 models), European Nuthatch (3 models), Short-toed Tree-creeper (2 models) and Blue Tit (10 models).

Bird Species	Predictors	Estimate	OR	SE	z	Pr(>|z|)	N	W
Marsh Tit	(Intercept)	−0.589	-	0.401	1.45	0.145	–	–
	S	0.545	1.724	0.243	2.22	0.026	7	1.00
	DBH30–45	0.270	1.309	0.306	0.87	0.380	4	0.60
	DBH8–15	0.260	1.296	0.288	0.89	0.369	4	0.60
	DBH4–8	0.065	1.067	0.182	0.35	0.722	1	0.18
	DBH<4	−0.034	0.966	0.120	0.28	0.779	1	0.12
	DBH15–30	−0.016	0.984	0.084	0.18	0.853	1	0.09
European Nuthatch	(Intercept)	−2.233	-	0.645	3.43	0.001	–	–
	S	1.155	3.174	0.392	2.92	0.003	3	1.00
	DBH8–15	−0.942	0.389	0.371	2.52	0.012	3	1.00
	DBH4–8	−0.630	0.532	0.472	1.33	0.183	2	0.73
	DBH<4	−0.262	0.769	0.467	0.56	0.575	1	0.27
	DBH>45	0.057	1.058	0.178	0.32	0.748	1	0.23
Short-toed Tree-creeper	(Intercept)	−2.767	-	0.741	3.70	<0.001	–	–
	S	0.902	2.464	0.416	2.14	0.032	2	1.00
	DBH >45	0.904	2.469	0.341	2.63	0.008	2	1.00
	DBH15–30	0.456	1.577	0.378	1.20	0.229	1	0.71
	DBH4–8	−0.166	0.847	0.315	0.52	0.598	1	0.29
Blue Tit	(Intercept)	−0.619	-	0.464	1.32	0.185	–	–
	S	0.783	2.188	0.268	2.90	0.004	10	1.00
	DBH30–45	−0.169	0.844	0.273	0.61	0.538	4	0.42
	DBH4–8	−0.174	0.840	0.281	0.61	0.537	3	0.37
	DBH15–30	0.071	1.073	0.167	0.42	0.671	3	0.25
	DBH>45	0.037	1.037	0.130	0.28	0.776	2	0.15
	DBH8–15	−0.023	0.977	0.108	0.20	0.835	1	0.09
	DBH<4	−0.010	0.990	0.072	0.14	0.886	1	0.07

S = Tree species richness; DBH… = number of trees per hectare with diameters at breast height included in classes expressed in cm (e.g. DBH30–45= number of trees per hectare with DBH ranging from 30 to 45 cm); OR = odds-ratio; SE = unconditional standard error; N = number of selected models containing the independent variable; W = relative importance of the predictor variables.

2.3.1.3 Model C

The analyses conducted on the third set of variables selected 30 models for the Marsh Tit, with a total of 15 independent variables, four models and seven independent variables for the European Nuthatch, 24 models and 12 independent variables for the Short-toed Tree-creeper, and five models and nine independent variables for the Blue Tit (Table S3 in Supplementary materials). Using this model, the tree species richness resulted the most important positive variable only for the Marsh Tit and European Nuthatch (Table 3). These two bird species, together with the Short-toed Tree-creeper, were also positively affected by the number of chestnuts per hectare with a DBH larger than 50 cm. In addition, for the European Nuthatch, the number of oaks per hectare with a DBH ranging from 20 to 30 cm positively affected the species presence; whereas, the number of maples per hectare with a DBH larger than 20 cm negatively influenced the presence of this species. On the other hand, the Short-toed Tree-creeper was negatively affected by the number of chestnuts per hectare with a DBH ranging from 5 to 10 cm. Finally, the number of maples per hectare with a DBH ranging from 15 to 20 cm, the number of chestnuts per hectare with a DBH ranging from 15 to 30 cm and the number of oaks per hectare with a DBH ranging from 10 to 15 cm were the most important independent variables with a positive effect on the presence of the Blue Tit. Conversely, the number of beeches per hectare with a DBH greater than 30 cm, the number of common hazels per hectare with a DBH ranging from 6 to 8 cm and the BA had a negative effect on the presence of this species (Table 3).

Table 3

Model C: Results of Model Averaging on logistic models with ΔAICc≤2 performed on presence/absence data of Marsh Tit (30 models), European Nuthatch (4 models), Short-toed Tree-creeper (24 models) and Blue Tit (5 models).

Bird Species	Predictors	Estimate	OR	SE	z	Pr(>|z|)	N	W
Marsh Tit	(Intercept)	−0.344	-	0.312	1.095	0.273	–	–
	S	0.498	1.645	0.227	2.174	0.030	30	1.00
	Cassat>50	0.717	2.048	0.243	2.928	0.003	30	1.00
	Cassat5–10	0.914	2.494	0.652	1.395	0.163	24	0.81
	Quespp10–15	0.424	1.528	0.378	1.118	0.264	22	0.72
	Acespp15–20	−0.194	0.824	0.303	0.639	0.523	13	0.45
	BA	0.088	1.092	0.167	0.522	0.602	9	0.29
	Quespp5–10	0.105	1.111	0.218	0.478	0.632	7	0.25
	Cassat10–15	0.136	1.146	0.314	0.431	0.667	6	0.19
	Acespp>20	−0.032	0.969	0.139	0.230	0.818	4	0.11
	Robpse10–15	0.020	1.020	0.080	0.245	0.806	4	0.11
	Fagsyl3–5	0.015	1.015	0.071	0.213	0.831	3	0.08
	Betpen20–30	0.007	1.007	0.048	0.143	0.887	2	0.05
	Fagsyl20–30	−0.004	0.996	0.048	0.084	0.933	1	0.03
	Tilpla10–20	−0.002	0.998	0.036	0.068	0.945	1	0.02
	Fagsyl10–20	−0.003	0.997	0.049	0.067	0.946	1	0.02
European Nuthatch	(Intercept)	−2.251	-	0.604	3.699	0.001	–	–
	S	0.984	2.675	0.372	2.622	0.009	4	1.00
	Cassat>50	0.830	2.293	0.287	2.874	0.004	4	1.00
	Quespp20–30	0.718	2.050	0.372	1.915	0.055	4	1.00
	Acespp>20	−1.830	0.160	1.105	1.643	0.100	4	1.00
	Cassat10–15	−0.317	0.728	0.344	0.916	0.360	3	0.68
	BA	0.074	1.077	0.216	0.341	0.733	1	0.20
	Tilpla10–20	−0.023	0.977	0.141	0.159	0.874	1	0.13
Short-toed Tree-creeper	(Intercept)	−3.243	-	1.015	3.173	0.002	–	–
	Cassat>50	1.088	2.968	0.333	3.239	0.001	24	1.00
	Cassat5–10	−0.959	0.383	0.658	1.447	0.148	24	1.00
	S	0.567	1.763	0.482	1.170	0.242	19	0.80
	Acespp15–20	−0.811	0.444	1.040	0.776	0.438	15	0.65
	Corave6–8	−1.836	0.159	2.778	0.657	0.511	13	0.55
	BA	0.075	1.078	0.219	0.339	0.734	4	0.18
	Quespp20–30	0.041	1.042	0.151	0.267	0.789	4	0.14
	Quespp15–20	0.018	1.018	0.106	0.166	0.868	2	0.07
	Quespp30	−0.016	0.984	0.105	0.155	0.877	2	0.07
	Acespp20	−0.061	0.941	0.355	0.171	0.864	2	0.06
	Robpse15–20	0.006	1.006	0.062	0.094	0.925	1	0.03
	Robpse3–5	0.005	1.005	0.061	0.079	0.937	1	0.03
Blue Tit	(Intercept)	0.009	-	0.786	0.011	0.991	–	–
	Acespp15–20	1.799	6.044	0.655	2.726	0.006	5	1.00
	Fagsyl>30	−1.239	0.290	0.459	2.680	0.007	5	1.00
	Cassat15–30	0.896	2.450	0.370	2.401	0.016	5	1.00
	Corave6–8	−1.169	0.311	0.544	2.132	0.033	5	1.00
	Quespp10–15	2.380	10.805	1.112	2.124	0.034	5	1.00
	BA	−0.675	0.509	0.332	2.020	0.043	5	1.00
	specie	0.303	1.354	0.355	0.849	0.396	3	0.60
	Carbet10–15	0.132	1.141	0.300	0.437	0.662	2	0.35
	Robpse15–20	−0.015	0.985	0.097	0.154	0.870	1	0.11

S = Tree species richness; BA = Basal Area; *Tree species…* = number of trees per hectare belonging to different tree species and diameter classes expressed in cm (for the full set of tree species names and codes see the Methods section); OR = odds-ratio; SE = unconditional standard error; N = number of selected models containing the independent variable; W = relative importance of the predictor variables.

2.3.2 Model comparison

In model A the explained deviance of each selected logistic model resulted quite low for the Marsh Tit (ED range=2.5–4.4%) and the Blue Tit (ED range=4.2–5.0%), while it was relatively higher for the European Nuthatch (ED=17.9%) and Short-toed Tree-creeper (ED range=18.4–20.6%). The ability of the averaged model to distinguish between occupied and unoccupied sites was excellent for the European Nuthatch (AUC=0.949, p<0.001) and poor for the Marsh Tit (AUC=0.600, p=0.015) and Short-toed Tree-creeper (AUC=0.666, p=0.001). For the Blue Tit, the averaged model did not classify occupied and unoccupied sites significantly better than a random model (AUC=0.546, p=0.160).

In model B the explained deviance of each selected model also resulted quite low for the Marsh Tit (ED range=2.5–5.8%) and the Blue Tit (ED range=4.2–6.9%) and higher for the European Nuthatch (ED range=15.1–16.3%) and Short-toed Tree-creeper (ED range=11.5–12.8%). The discriminant ability of the averaged model was poor for the Marsh Tit (AUC=0.621, p=0.004), European Nuthatch (AUC=0.663, p<0.001) and Short-toed Tree-creeper (AUC=0.676, p<0.001). Even in this case, the averaged model failed to correctly classify occupied and unoccupied sites for the Blue Tit (AUC=0.574, p=0.053).

Conversely, using model C, the explained deviance was generally higher for all the target species (Marsh Tit: ED range=10.6–14.1%, European Nuthatch: ED range=20.4–22.5%, Short-toed Tree-creeper: ED range=16.5–23.1%, Blue Tit: ED range=26.1–28.2%). Moreover, the accuracy of the averaged model was fair for the Marsh Tit (AUC=0.773, p<0.001), excellent for the European Nuthatch (AUC=0.942, p<0.001) and the Short-toed Tree-creeper (AUC=0.955, p<0.001), and good for the Blue Tit (AUC=0.810, p<0.001).

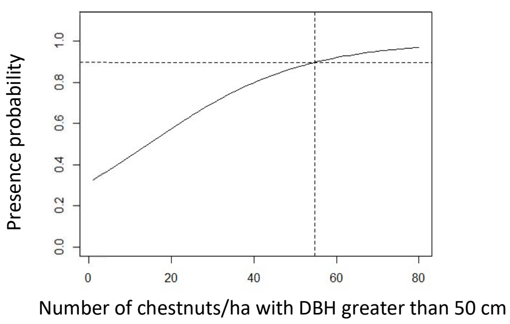
2.3.3 Habitat requirements

Based on the significant results of model B, the minimum required number of trees with DBH greater than 45 cm that should be maintained in a hectare to guarantee a Short-toed Tree-creeper occurrence probability higher than 0.9 was 65 (Fig. 2d). Using the results of model C, the minimum required number of chestnuts with DBH greater than 50 cm was 55 stems per hectare for the Marsh Tit (Fig. 2a), 70 for the European Nuthatch (Fig. 2b) and 61 for the Short-toed Tree-creeper (Fig. 2e). Oaks with a DBH ranging from 20 to 30 cm was statistically significant only for the European Nuthatch, with a minimum of 305 stems per hectare (Fig. 2c). The number of maples per hectare with a DBH ranging from 15 to 20 cm, the number of chestnuts per hectare with a DBH ranging from 15 to 30 cm and the number of oaks per hectare with a DBH ranging from 10 to 15 cm were statistically significant only for the Blue Tit, whose occurrence probability higher than 0.9 required a minimum of 69 (Fig. 2f), 672 (Fig. 2g) and 103 (Fig. 2h) stems per hectare, respectively.

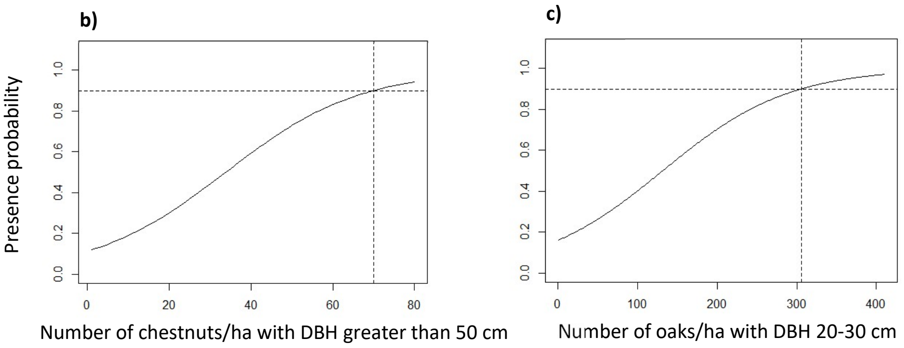
Figure 2

Effect size from univariate regressions of the number of trees (of different species and diameters classes) per hectare on the presence probability of Marsh Tit (a), European Nuthatch (b-c), Short-toed Tree-creeper (d-e) and Blue Tit (f-h). Vertical dashed lines identify the minimum required number of trees to guarantee a species presence probability greater than 0.9 (horizontal dashed lines).

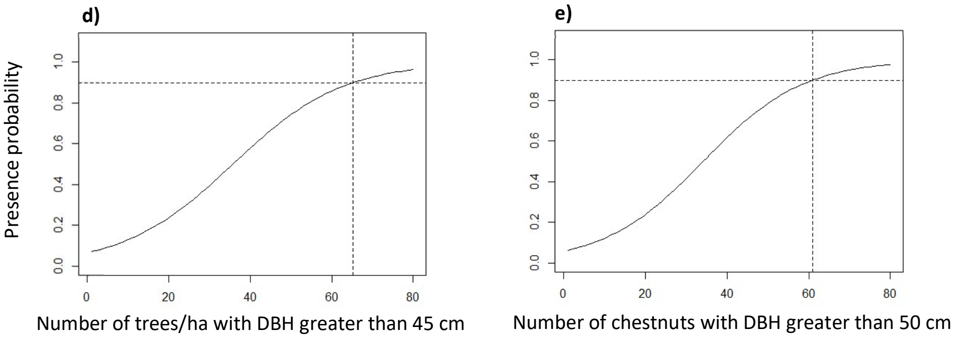
Marsh Tit



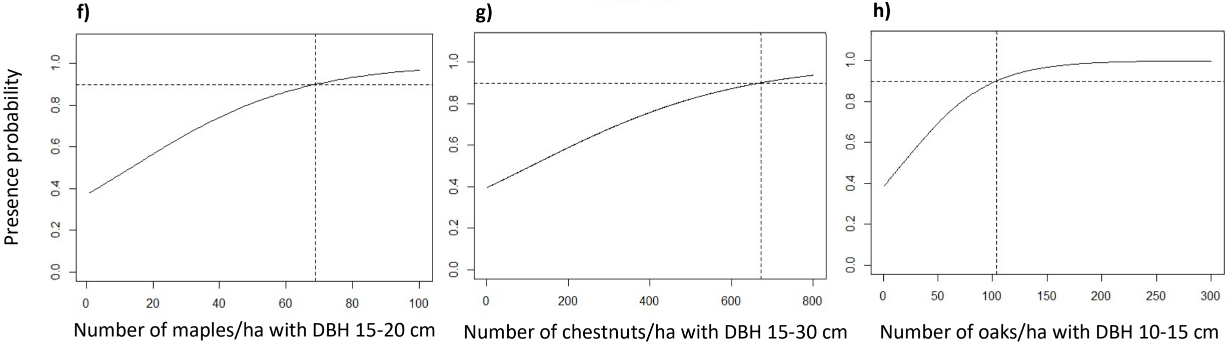
European Nuthatch



Short-toed Tree-creeper



Blue Tit



2.4 Discussion

2.4.1 Forest characteristics and bird species

One of the aims of this study was to evaluate both the independent and the combined effect of forest structure and floristic composition on four secondary cavity-nesters. Our analyses showed that the tree species richness was one of the most important environmental features positively affecting the probability of presence of all the bird target species using models A and B, and of the Marsh Tit and European Nuthatch using model C. Indeed, tree species richness enhances the availability of specific food resources for many forest bird species (Avery and Leslie 1990; Santos et al. 2002; DÍaz 2006; Fuller et al. 2012) and, in particular, for forest specialists (Gil-Tena et al. 2007).

The presence of European Nuthatch resulted positively influenced by the mean DBH using model A and negatively by the number of stems per hectare with a DBH ranging from 8 to 15 cm using model B. By means of this latter model, the presence of Short-toed Tree-creeper was positively affected by the number of stems per hectare with a DBH greater than 45 cm. All of these relations are due to the preference of these two species, for forests tending to mature stages with a relatively high number of large trees (Fuller et al. 2012). Indeed, second cavity-nesters are associated with old forests as they provide more food availability, a higher amount of dead wood as breeding and feeding substrate and more tree cavity for nesting (Camprodon 2001; Laiolo 2002; Machtans and Latour 2003). Using model A, Short-toed Tree-creeper, showed a significant preference for woodland characterized by a high diameter variance. This result was not surprising, as there is wide evidence that woods with a high variety of stem diameters will tend to be richer in bird species than woods where stem diameters are rather uniform (Fuller 1995; Symes and Currie 2005). Model C revealed that the presence of the Marsh Tit, European Nuthatch and Short-toed Tree-creeper were positively influenced by the number of chestnuts per hectare with a DBH greater than 50 cm. In our study area, high densities of large chestnuts are typically found in fruit production chestnut groves, whose structure could be considered as a surrogate of that of mature natural woodlands. Indeed, besides providing a high seed production, which is one of the most important determinants of survival during the autumn and winter (Nilsson 1987), they offer several opportunities for nesting (González-Varo et al. 2007).

Conversely, the Short-toed Tree-creeper resulted negatively affected by high densities of small chestnuts (DBH 5–10) typically characterizing early woodland stages (e.g. young coppices). Model C also highlighted how the presence of the European Nuthatch was positively influenced by the number of oaks per hectare with a DBH ranging from 20 to 30 cm. This is probably due to the high availability of invertebrates in the fissured bark of mature or medium-aged oaks, which could be used as food resource (Matthysen 1998). For the same species we found a negative effect played by maples (*Acer* spp.), a genus that is generally abundant in early or medium stages of woodland evolution.

On the other hand, model C, revealed how the Blue Tit has less strict ecological requirements, in respect to the other three most demanding bird species, allowing it to inhabit even more exploited forests (Bani et al. 2006). In fact, the Blue Tit resulted positively affected by medium sized maples (DBH 15–20), chestnuts (DBH 15–30) and oaks (DBH 10–15), and even negatively affected by BA. Nonetheless, the species was negatively affected by common hazelnuts, particularly abundant in heavily exploited coppices. Moreover, Blue Tit resulted negatively affected by the number of beeches per hectare with a DBH greater than 30 cm. This could happen because this bird species generally avoid beech forests, which host relatively less foliage invertebrates than others, such as native oak forests (Fuller et al. 2012).

2.4.2 Model comparison

Another crucial aim of this paper was to assess, among the three proposed models, (i) the most effective in explaining species occurrence, and (ii) the most practical in providing guidelines for conservation and management purposes.

First, considering the goodness of fit, the percentage of deviance explained by model C was always higher for all four bird species, with respect to models A and B. Moreover, considering the validation process, the discriminant ability in classifying sites of presence/absence of the species was always higher for model C with respect to models A and B, except for the European Nuthatch for which the AUC were very similar in models A and C. Secondly, the results of model A seem to be too general to be used as specific recommendations for forestry practices. The results of model B can provide some management suggestions, giving the requested number of trees per diametric class per hectare, but considering plant species on the whole. However, using only the results from model C we can provide guidelines for the management of specific kinds of forest.

In conclusion, model C showed a better performance in modeling bird data and resulted also more effective in providing information useful for forest management with respect to models A and B.

2.4.3 Management implications

To our knowledge, this is one of the few studies (e.g. Roberge et al. 2008a, 2008b; Touihri et al. 2014) that provided quantitative forestry guidelines for singular plant species and diameter classes, starting from the ecological requirements of bird indicator species.

The forest guidelines obtained in this study provided useful information for a correct management of both forest remnants in the lowland areas of Lombardy region and of the more continuous forests located in the Alps, Prealps, Apennines and along the Ticino River, which play the role of source areas for forest bird species in Lombardy.

In the context of less exploited woods, we considered the Marsh Tit, European Nuthatch and Short-toed Tree-creeper as indicator species. Their ecological specialization (Bani et al. 2006; Gonzáles-Varo et al. 2008) makes them suited to play the role of umbrella species (Roberge and Angelstam 2004) for many other taxa linked to mature forests. Our results suggested that forestry practices should promote a high tree species richness, a variety of stem age classes and the presence of large trees. In particular, the results of model B suggest 65 stems per hectare with DBH larger than 45 cm would be recommended in order to guarantee a 90% occurrence probability of the Short-toed Tree-creeper in broadleaved mixed forest. Regarding specific forest types, we propose to incorporate into forestry guidelines the result derived from model C applied to European Nuthatch, because it resulted the most demanding species from our analyses. In particular, for pure chestnut stands, we recommend maintaining 70 trees per hectare with a DBH greater than 50 cm. This result was similar to the one obtained for Short-toed Tree-creeper considering all tree species on the whole, pointing out the reliability of the performed models. For pure oak stands, we suggest maintaining about 300 trees per hectare with a diameter ranging from 20 to 30 cm. From a practical point of view, in non-pure stands, where Chestnut and oaks can co-occur, we propose maintaining a minimum number of trees proportional to the mixing percentage of these two tree species. As suggested by Touihri et al.

(2014), all the proposed thresholds should not be kept as strict guidelines but as optimal hypotheses for management. Indeed, these prescriptions are very conservative, as they refer to the 90% probabilities of occurrence of the indicator species. For heavily exploited woods that cannot reach an adequate structure to host the most sensitive secondary cavity-nesters, e.g. medium-aged compound coppices, we suggest to use the results obtained using the Blue Tit as indicator species. In particular, the results of model C suggest about 670 stems per hectare with DBH ranging from 15 to 30 cm would be recommended in order to guarantee a 90% occurrence probability of the Blue Tit in chestnut woodlands. Moreover, in broadleaved mixed forests we advise maintaining stands with a number of oaks with a diameter ranging from 10 to 15 cm and maples with a diameter ranging from 15 to 20 cm, corresponding to the proportion of occurrence of these two species, with respect to the number recommended in the case of an ideal pure stand (about 100 trees per hectare for oaks and 70 trees per hectare for maples). These recommendations are also useful in the case of non-permanent broadleaved reforestations that should increase the forest surface, providing potentially suitable habitats for the conservation of less demanding forest species in exploited landscapes.

On the whole, the prescribed densities of trees resulting from our analyses correspond to real cases encountered during the sampling.

2.5 Conclusions

In this study, we tested and proposed a method to define forestry prescriptions taking into account the interaction of woodland structure and tree species composition on bird indicator species. Based on our results, the provided management indications should be applicable in a wide geographical contexts different from Lombardy, where our indicator bird species are present, such as most of the European broadleaved forests. In general, our procedure can provide practical suggestions for the management of virtually every type of forest for which bird indicator species have already been identified and assessed.

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Supplementary materials

Table S1

Model A: Results of Multimodel Inference performed on presence data of Marsh Tit *Poecile palustris*, European Nuthatch *Sitta euopaea*, Short-toed Tree-creeper *Certhya brachydactyla* and Blue Tit *Cyanistes caeruleus* (only models with ΔAICc ≤ 2 are shown).

Bird Species	Predictors in the models	df	Log–Likelihood	AICc	ΔAICc	w	ED (%)
Marsh Tit	BA/S	4	−96.049	200.356	–	0.251	3.7
	S	3	−97.226	200.606	0.251	0.221	2.5
	BA/DBH var/S	5	−95.353	201.096	0.741	0.173	4.4
	DBH var/S	4	−96.528	201.314	0.959	0.155	3.2
	DBH mean/S	4	−96.936	202.131	1.775	0.103	2.8
	BA/DBH mean/S	5	−95.931	202.252	1.896	0.097	3.8
Nuthatch	DBH mean/S	4	−65.053	138.400	–	0.605	17.9
Short-toed Tree-creeper	BA/DBH var/S	5	−55.182	120.754	–	0.613	20.6
	DBH var/S	4	−56.707	121.673	0.918	0.387	18.4
Blue Tit	S	3	−91.758	189.670	–	0.440	4.2
	DBH mean/S	4	−90.997	190.252	0.582	0.329	5.0
	DBH var/S	4	−91.350	190.959	1.289	0.231	4.6

S = Tree species richness; BA = Basal Area; DBH mean = mean diameters at breast height; DBH var = variance of diameters at breast height; AICc = Corrected Akaike Information Criterion; w = Akaike weight; ED (%) = percentage of explained deviance of each selected logistic model.

Table S2

Model B: Results of Multimodel Inference performed on presence data of Marsh Tit *Poecile palustris*, European Nuthatch *Sitta euopaea*, Short-toed Tree-creeper *Certhya brachydactyla* and Blue Tit *Cyanistes caeruleus* (only models with ΔAICc ≤ 2 are shown).

Bird Species	Predictors in the models	df	Log–Likelihood	AICc	ΔAICc	w	ED (%)
Marsh Tit	DBH8–15/DBH30–45/S	5	−94.814	200.017	–	0.212	4.9
	DBH4–8/DBH8–15/DBH30–45/S	6	−93.915	200.378	0.361	0.177	5.8
	S	3	−97.226	200.606	0.589	0.158	2.5
	DBH8–15/S	4	−96.433	201.123	1.106	0.122	3.3
	DBH<4/S	4	−96.448	201.153	1.136	0.12	3.3
	DBH30–45/S	4	−96.464	201.186	1.169	0.118	3.3
	DBH8–15/DBH15–30/DBH30–45/S	6	−94.543	201.635	1.618	0.094	5.2
Nuthatch	DBH4–8/DBH8–15/S	5	−66.661	143.711	–	0.493	15.9
	DBH<4//DBH8–15/S	5	−67.254	144.898	1.187	0.272	15.1
	DBH4–8/DBH8–15/ DBH >45/S	6	−66.325	145.199	1.488	0.234	16.3
Short-toed Tree-creeper	DBH15–30/ DBH >45/S	5	−60.603	131.596	–	0.714	12.8
	DBH4–8/DBH >45/S	5	−61.516	133.422	1.826	0.286	11.5
Blue Tit	DBH4–8/DBH30–45/S	5	−89.581	189.551	–	0.164	6.4
	S	3	−91.758	189.670	0.118	0.155	4.2
	DBH4–8/S	4	−91.026	190.309	0.758	0.113	4.9
	DBH15–30/S	4	−91.147	190.552	1.001	0.100	4.8
	DBH4–8/ DBH8–15/DBH30–45/S	6	−89.121	190.791	1.240	0.088	6.9
	DBH30–45/S	4	−91.288	190.833	1.282	0.087	4.6
	DBH15–30/DBH30–45/S	5	−90.292	190.975	1.423	0.081	5.7
	DBH >45/S	4	−91.390	191.038	1.487	0.078	4.5
	DBH15–30/ DBH >45/S	5	−90.472	191.333	1.782	0.067	5.5

S = Tree species richness; DBH… = number of trees per hectare with diameters at breast height included in classes expressed in cm (e.g. DBH30–45= number of trees per hectare with DBH ranging from 30 to 45 cm); AICc = Corrected Akaike Information Criterion; w = Akaike weight; ED (%) = percentage of explained deviance of each selected logistic model.

Table S3

Model C: Results of Multimodel Inference performed on presence data of Marsh Tit *Poecile palustris*, European Nuthatch *Sitta euopaea*, Short-toed Tree-creeper *Certhya brachydactyla* and Blue Tit *Cyanistes caeruleus*.only models with ΔAICc ≤ 2 are shown)

Bird Species	Predictors in the models	df	Log–Likelihood	AICc	ΔAICc	w	ED (%)
Marsh Tit	Cassat5–10/Cassat>50/Quespp10–15/S	6	−88.169	188.887	–	0.063	11.5
	Acespp15–20/Cassat5–10/Cassat>50/Quespp10–15/S	7	−87.119	188.974	0.087	0.060	12.6
	BA/Acespp15–20/Cassat5–10/Cassat>50/Quespp10–15/S	8	−86.198	189.349	0.462	0.050	13.6
	Cassat5–10/Cassat>50/Quespp5–10/S	6	−88.461	189.471	0.584	0.047	11.3
	Acespp15–20/Cassat10–15/Cassat>50/Quespp10–15/S	7	−87.456	189.648	0.761	0.043	12.8
	BA/Cassat5–10/Cassat>50/Quespp10–15/S	7	−87.471	189.678	0.792	0.042	12.3
	Acespp15–20/Cassat5–10/Cassat>50/Quespp5–10/S	7	−87.536	189.808	0.921	0.040	12.2
	BA/Acespp15–20/Cassat5–10/Cassat>50/Quespp5–10/S	8	−86.473	189.900	1.013	0.038	13.3
	Cassat10–15/Cassat>50/Quespp10–15/S	6	−88.687	189.923	1.036	0.038	11.1
	BA/Cassat5–10/Cassat>50/Quespp5–10/S	7	−87.597	189.931	1.044	0.037	12.2
	Acespp>20/Cassat5–10/Cassat>50/Quespp10–15/S	7	−87.648	190.034	1.147	0.036	12.1
	Cassat10–15/Cassat>50/Quespp5–10/S	6	−88.833	190.216	1.329	0.032	10.9
	Acespp15–20/Cassat10–15/Cassat>50/Quespp5–10/S	7	−87.765	190.267	1.380	0.032	12.0
	Cassat5–10/Cassat>50/Quespp10–15/Robpse10–15/S	7	−87.823	190.383	1.496	0.030	11.9
	Cassat5–10/Cassat>50/Fagsyl3–5/Quespp10–1/S	7	−87.826	190.389	1.502	0.030	11.9
	Acespp15–20/Cassat5–10/Cassat>50/Fagsyl3–5/Quespp10–1/S	8	−86.735	190.424	1.538	0.029	13.0
	Acespp15–20/Cassat5–10/Cassat>50/Quespp10–15/Robpse10–15/S	8	−86.741	190.436	1.549	0.029	13.0
	BA/Acespp15–20/Cassat5–10/Cassat>50/Quespp10–15/Robpse10–15/S	9	−85.620	190.440	1.553	0.029	14.1
	Betpen20–30/Cassat5–10/Cassat>50/Quespp10–15/S	7	−87.965	190.667	1.780	0.026	11.8
	BA/Acespp>20/Cassat5–10/Cassat>50/Quespp10–15/S	8	−86.864	190.682	1.795	0.026	12.9
	Cassat5–10/Cassat>50/Fagsyl20–30/Quespp10–15/S	7	−87.973	190.683	1.797	0.026	11.8
	Acespp>20/Cassat5–10/Cassat>50/Quespp5–10/S	7	−87.993	190.723	1.837	0.025	11.8
	BA/Acespp15–20/Cassat5–10/Cassat>50/Fagsyl3–5/Quespp10–15/S	9	−85.779	190.757	1.871	0.025	14.0
	Acespp15–20/Cassat5–10/Cassat>50/S	6	−89.111	190.77	1.883	0.025	10.6
	Cassat5–10/Cassat>50/Quespp10–15/S/Tilpla10–20	7	−88.038	190.813	1.926	0.024	11.7
	Cassat5–10/Cassat>50/Fagsyl10–20/Quespp10–15/S	7	−88.040	190.817	1.930	0.024	11.7
	Acespp>20/Cassat10–15/Cassat>50/Quespp10–15/S	7	−88.043	190.822	1.935	0.024	11.7
	Acespp15–20/Betpen20–30/Cassat5–10/Cassat>50/Quespp10–15/S	8	−86.953	190.859	1.972	0.024	12.8
	BA/Cassat5–10/Cassat>50/Quespp5–10/Robpse10–15/S	8	−86.965	190.883	1.996	0.023	12.7
	BA/Acespp15–20/Cassat10–15/Cassat>50/Quespp10–15/S	8	−86.966	190.886	1.999	0.023	12.8

European Nuthatch	Acespp>20/Cassat10–15/Cassat>50/Quespp20–30/S	7	−61.948	138.634	–	0.347	21.8
	Acespp>20/Cassat>50/Quespp20–30/S	6	−63.118	138.785	0.151	0.322	20.4
	BA/Acespp>20/Cassat10–15/Cassat>50/Quespp20–30/S	8	−61.390	139.734	1.101	0.200	22.5
	Acespp>20/Cassat10–15/Cassat>50/Quespp20–30/S/Tilpla10–20	8	−61.809	140.572	1.938	0.132	22.0
Short–toed Tree-creeper	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/S	7	−54.161	123.059	–	0.081	22.1
	Acespp15–20/Cassat5–10/Cassat>50/S	6	−55.423	123.395	0.336	0.068	20.3
	BA/Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/S	8	−53.410	123.773	0.714	0.057	23.1
	Cassat5–10/Cassat>50/Corave6–8/S	6	−55.650	123.849	0.790	0.055	19.9
	BA/Acespp15–20/Cassat5–10/Cassat>50/S	7	−54.562	123.861	0.802	0.054	21.5
	Cassat5–10/Cassat>50/S	5	−56.863	124.116	1.058	0.048	18.2
	Acespp15–20/Cassat5–10/Cassat>50	5	−56.890	124.169	1.111	0.046	18.1
	Cassat5–10/Cassat>50	4	−58.005	124.269	1.210	0.044	16.5
	Cassat5–10/Cassat>50/Corave6–8	5	−57.000	124.390	1.332	0.042	18.0
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/Quespp20–30/S	8	−53.724	124.401	1.343	0.041	22.7
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8	6	−56.051	124.580	1.521	0.038	19.4
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/Quespp15–20/S	8	−53.831	124.615	1.557	0.037	22.5
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/Quespp>30/S	8	−53.685	124.685	1.627	0.036	22.5
	Acespp15–20/Cassat5–10/Cassat>50/Quespp20–30/S	7	−54.979	124.695	1.636	0.036	20.9
	BA/Cassat5–10/Cassat>50/Corave6–8/S	7	−55.039	124.815	1.757	0.034	20.8
	BA/Cassat5–10/Cassat>50/S	6	−56.164	124.878	1.819	0.033	19.2
	Acespp15–20/Cassat5–10/Cassat>50/Quespp15–20/S	7	−55.080	124.896	1.837	0.032	20.7
	Acespp>20/Cassat5–10/Cassat>50/Corave6–8/S	7	−55.084	124.904	1.846	0.032	20.7
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/Robpse15–20/S	8	−53.977	124.908	1.849	0.032	22.3
	Acespp15–20/Cassat5–10/Cassat>50/Corave6–8/Robpse3–5/S	8	−53.993	124.939	1.880	0.032	22.3
	Acespp15–20/Cassat5–10/Cassat>50/Quespp>30/S	7	−55.106	124.948	1.889	0.031	20.7
	Acespp15–20/Cassat5–10/Cassat>50/Quespp20–30	6	−56.223	124.994	1.936	0.031	19.1
	Cassat5–10/Cassat>50/Corave6–8/Quespp20–30/S	7	−55.134	125.004	1.945	0.031	20.7
	Acespp>20/Cassat5–10/Cassat>50/S	6	−56.239	125.027	1.968	0.030	19.1
Blue Tit	BA/Acespp15–20/Cassat15–30/Corave6–8/Fagsyl>30/Quespp10–15/S	10	−64.429	167.016	–	0.298	27.5
	BA/Acespp15–20/Cassat15–30/Corave6–8/Fagsyl>30/Quespp10–15	9	−70.766	160.732	0.339	0.244	26.1
	BA/Acespp15–20/Carbet10–15/Cassat15–30/Corave6–8/Fagsyl>30/Quespp10–15/S	11	−68.732	161.248	0.915	0.189	28.2
	BA/Acespp15–20/Carbet10–15/Cassat15–30/Corave6–8/Fagsyl>30/Quespp10–15	10	−70.059	161.595	1.261	0.159	26.8
	BA/Acespp15–20/Cassat15–30/Corave6–8/Fagsyl>30/Quespp10–15/Robpse15–20/S	11	−69.271	162.325	1.992	0.110	27.6

S = Tree species richness; BA = Basal Area; *Tree species…* = number of trees per hectare belonging to different tree species and diameter classes expressed in cm; AICc = Corrected Akaike Information Criterion ; w = Akaike weight; ED (%) = percentage of explained deviance of each selected logistic model.

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Chapter 3

Identification of ecological network elements by means of simulated species landscape perceptions

Abstract and links to other chapters

Once defined the management strategies to maintain high-quality forest remnants (Chapter 2), it is necessary to identify which other landscape elements could play the role of secondary nodes and connectivity elements in an ecological network. However, the traditional approaches aimed to define which elements can be used to design an ecological network present some limits. Ecological networks are usually designed adopting a physical approach or an ecological approach based on Habitat Suitability Models (HSMs). However, the physical approach only considers structural connectivity, which does not guarantee the necessary functional connectivity. On the other hand, HSMs have some limits in effectively identifying nodes and connectivity elements due to the difficulty to consider fragmentation metrics of all the land-cover types, without knowing which land-cover types are similarly suitable for the species and should thus be merged together before spatial metrics calculation. To overcome these issues, in this chapter, we proposed a new ecological approach in the design of an ecological network, which compares simulated species landscape perceptions (SSLPs) corresponding to every possible combination of land-cover types in the real landscape, alternatively assuming the role of a node, a connectivity element, or matrix. The method compares the ability of the fragmentation metrics calculated for each SSLP to explain the actual species distribution. The SSLP that performs better than the others will provide information about the land-cover types that should be used to design an effective ecological network for the target species, and their role. The method proved to be effective when applied to the Hazel Dormouse in a wide agricultural landscape in northern Italy, where it allowed to identify woodlands and hedgerows as nodes, and poplar cultivations, biomasses and reforestations as connectivity elements. These results were subsequently used in Chapter 4 to compare the extent of the study area suitable and well connected for the Hazel Dormouse, with those suitable and well connected for other two mammals, in order to evaluate the degree of overlap of suitable areas for species sensitive to fragmentation, characterized by very different ecological requirements. Moreover, this information was adopted to develop further analyses on how to manage hedgerows for the conservation of the Hazel Dormouse in Chapter 5 and in Chapter 6.

3.1 Introduction

Several studies showed that, to achieve long-term persistence, fragmented populations require protection as a metapopulation (Soulè and Terborgh 1999; Hanski and Gaggiotti 2004; Hilty et al. 2006). To this aim, wildlife conservation strategies should be implemented at the landscape level, linking residual habitats by a network to guarantee their ecological connectivity (Noss et al. 1997; Soulè and Terborgh 1999). In this context, the appropriate landscape management tool is the ecological network (Bennett 2003; Andersons and Jenkins 2006). Ecological networks are integrated landscape systems, composed of nodes linked by stepping stones or ecological corridors. Nodes are fragments of suitable habitat, often included in legally protected areas, hosting stable or ephemeral populations, while stepping stones and ecological corridors are, respectively, small and thin connectivity elements, composed of permeable habitat, which allow dispersal among nodes. Nodes and connectivity elements are surrounded by matrix, represented by land-cover types virtually impermeable to the passage of animals.

Ecological networks are sometimes designed using an “expert-based approach”, where nodes and connectivity elements are identified only using human-defined land-cover types suitable for a species. However, land-cover types considered *a priori* as suitable for a species often show a weak relationship with species ecology (Betts et al. 2014). In other words, this approach only considers the physical connectivity (or structural connectivity) and not the ecological connectivity (or functional connectivity; Wiens 2006) of the landscape. For this reason, the effectiveness of this approach for conservation purposes is strongly criticized (Franklin and Lindenmayer 2009).

To overcome this limit, several studies proposed a design of ecological networks based on *functional* connectivity through an “ecological approach”, which relies on Habitat Suitability Models (HSMs; Bani et al. 2002; Beier et al. 2008). These models estimate the relationship between the observed species occurrence or abundance and the amount of land-cover types, whereby species suitability maps for the whole landscape can be drawn. These maps are then used to identify both nodes and ecological linkages along which individuals may preferentially disperse in the landscape (Douglas 1994; O´ Brien et al. 2006; McRae et al. 2008; Cushman et al. 2013; Mateo-Sánchez et al. 2014). Nevertheless, species occurrence, that often represents the dependent variable in HSMs, is not only affected by the amount of land-cover types, but also by their spatial arrangement (e.g. whether or not a patch of habitat is connected with another). Not considering the spatial arrangement of land-cover types in HSMs leads to two key problems. First, HSMs which not consider the spatial arrangement of land-cover types may not identify some land-covers as suitable, i.e. as node, for the studied species. To give a practical example, if the patches belonging to a potentially suitable land-cover are mostly isolated in the landscape, a HSM could return that the considered land-cover is not selected or even avoided by the target species. However, the limited occurrence of a fragmentation sensitive species within this land-cover does not depend on its intrinsic characteristics, but on its own spatial arrangement and on the spatial arrangement of other suitable or just permeable land-covers within the landscape. Thus, the suitability of such a land-cover for the target species can be detected only by taking into account the spatial arrangement of all the non-matrix land-covers within the model. The second problem of HSMs relies on the difficulty of these models to identify which land-covers are used as connectivity elements by the studied species. Indeed, habitat suitability and landscape permeability are not necessarily synonymous (e.g., Spear et al. 2010; Zeller et al. 2012). Connectivity elements are mostly used by dispersing juveniles, but not by reproductive adults, and adults and juveniles select habitats and connectivity elements, respectively, in a very different way. Adults select habitats where they can establish home-ranges and breed, while juveniles' dispersal is mainly driven by permeable habitats that are often unlikely to be suitable habitats for foraging, sheltering, resting or breeding (Mateo- Sánchez 2015). The largest part of occurrence data, that are mainly represented by individuals belonging to stable local populations, is thus found within nodes (i.e. within habitats where individuals breed and local populations exist), while connectivity elements often result not occupied by the species. For this reason, HSMs, that only calculate the relationship between species occurrence and land-covers' amount, often do not detect the importance of connectivity elements for the studied species (see Mateo- Sánchez 2015). However, even if we do not find the species within connectivity elements, the species occurrence probability within nodes in fragmented landscapes is intimately related to the existence of connectivity elements. Indeed, we found stable local populations only within nodes sufficiently connected to other nodes. Nodes can be connected either because they are close to each other (i.e. the lay at a distance shorter than the dispersal ability of the considered species) or because permeable habitats able to support dispersal (i.e. connectivity elements) exist between them. However, connectivity elements surrounding nodes are not required to be large to sustain dispersal (they can sustain dispersal also if they do not occupy a large part of the landscape) and this is the reason why HSMs do not necessarily detect the relationship between the amount of connectivity elements and species occurrence, even calculating this metric within a wide buffer area surrounding nodes.

To overcome these problems, species occurrence should be regressed on the amount of nodes' coverage and the degree of connectivity provided by both nodes (i.e. spaces where connectivity exist; habitat availability concept, Saura and Rubio 2010) and connectivity elements (for which only the spatial arrangement have a significant influence on species occurrence).

Nevertheless, it is not easy to define how the amount and connectivity degree of land-cover types should be calculated. Even if HSMs could theoretically consider all land-cover types and their amount or connectivity degree (e.g. Bani et al. 2006), it is not correct to insert them as separate main factors in the model because, if different land-cover types are similarly perceived by a species (as nodes or connectivity elements), they should be merged in a unique land-cover characterized by its own amount and connectivity degree. To give a practical example, if two patches belonging to similarly permeable land-cover types connect two nodes and we calculate two connectivity indices, separately considering the two permeable patches, they both will reveal a low degree of connectivity. Instead, if the connectivity index was calculated considering the two patches as a unique hypothetical land-cover type, it will indicate a higher degree of connectivity. It happens because, in this example, connectivity is only guaranteed by the presence of both permeable patches, which jointly play the role of stepping-stones. Thus, before calculating habitat amount and connectivity, we need to know which land-cover types are similarly perceived by a given species (i.e. as nodes or connectivity elements), but to obtain this information we would need to develop models that already take into account habitat amount and connectivity.

To overcome this issue, in this study we propose a new method to objectively define the role of land-cover types (as nodes or connectivity elements) for a given species by comparing different simulated species landscape perceptions. We applied the proposed method to the Hazel Dormouse (*Muscardinus avellanarius*), a small-size arboreal rodent inhabiting deciduous forests.

3.2 Rationale of the method

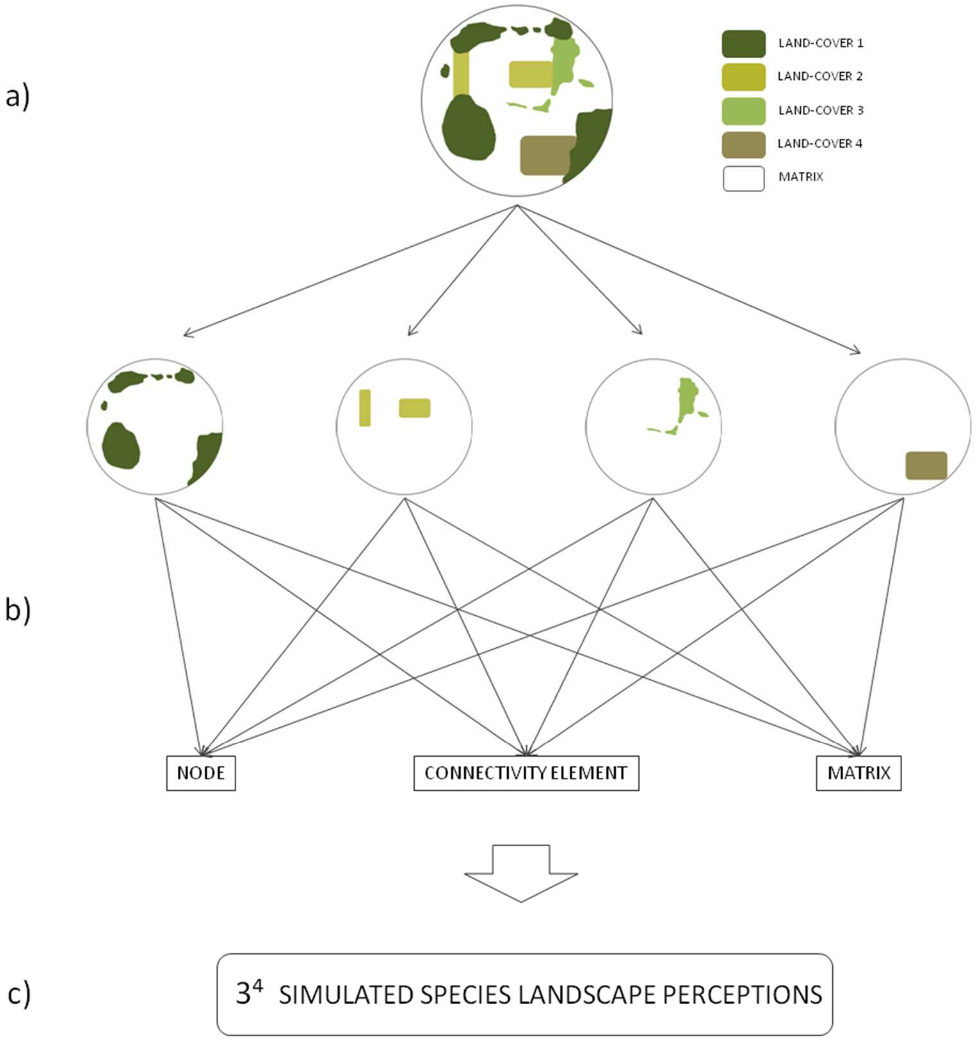
In a fragmented landscape, the current ecological status of a species particularly sensitive to habitat loss and fragmentation is the result of how the species itself perceives the different land-cover types (i.e. as suitable habitat, connectivity element or impermeable matrix) and of how their amount and spatial arrangement have affected the species over time. We thus propose a method to deduce the species perception of the different land-cover types starting from spatially-explicit ecological parameters (i.e. occupancy or demographic characteristics) which describe the ecological status of the species within the landscape. We assume that if a species perceives a land-cover type as suitable habitat (i.e. as node) its ecological parameters will be affected by both the amount and the connectivity provided by the spatial arrangement of the patches belonging to that land-cover type. Conversely, if the species perceives a land-cover type as connectivity element it will be affected only by the connectivity provided by that land-cover type. Finally, the land-cover types perceived as matrix are assumed to be completely unsuitable for the species so that they do not contribute neither to the amount of suitable habitat nor to the landscape connectivity for the target species.

Theoretically, the method can be adopted to identify the species perception for every land-cover type. However, in the first step of the method, according to the ecology of the target species or to specific research hypotheses, it is reasonable to previously select a subset of land-cover types that could play an effective role in an ecological network for the target species. In this case, all the other land-cover types would be considered *a priori* as a virtually impermeable matrix (Fig. 1a).

The second step of the method consists in setting up all the possible simulated species landscape perceptions (SSLPs) corresponding to all the combinations of the considered land-cover types alternatively assigning them the role of nodes, connectivity elements or matrix (Fig. 1b). Assigning the different roles to the land-cover types means to calculate the habitat amount of all those land-cover types considered as nodes, merged together in a unique land-cover, and the landscape connectivity provided by all those land-cover types considered as nodes or connectivity elements, merged together in a unique land-cover. For example, if we consider four land cover types (*a*, *b*, *c*, *d*), in order to build up the SSLP in which the land-cover *a* and *b* play the role of nodes, the land-cover *c* plays the role of connectivity element and the land-cover *d* plays the role of matrix we calculate the amount of habitat of a new artificially created land-cover type composed of the land-cover *a* and *b* merged together, and the landscape connectivity of a new land-cover composed of the land cover *a*, *b* and *c* merged together. The land-cover *d* is not considered in any calculation. Considering *n* land-cover types, alternatively assuming the three possible roles, we would obtain 3^n SSLPs (Fig. 1c).

Figure 1

Rationale of the method. Starting from the actual landscape, four land-cover types are considered, while all the others are treated as matrix (a). Subsequently, to each considered land-cover type it is alternatively assigned the role of a node, a connectivity element or matrix (b). Each combination results in a SSLP, making a total of 3^4 SSLPs (c).



The third step of the method consists in identifying the actual species landscape perception by comparing the ability of the different SSLPs in explaining the variability of the ecological parameters of the target species in the considered area. This can be achieved by performing statistical regression models where the dependent variable is the considered ecological feature and the independent variables are the habitat amount and the landscape connectivity calculated for each SSLP. Different types of regression models can be adopted, according to the sampling design (e.g. uniform or stratified) and the type of ecological data (e.g. occurrence, abundance or genetic characteristics), and consequently an appropriate measure of model goodness of fit can be calculated. This way, we will obtain a number of models equal to the number of SSLPs we built, and the model that performs better than the others will correspond to the SSLP representing the landscape effectively perceived by the target species. The SSLP associated to the best model provides information about the land-cover types that should be used in order to design an effective ecological network for the target species, and their role.

3.3 Case study: an ecological network for the Hazel Dormouse

We tested the proposed method on occurrence data of the Hazel Dormouse, a protected species (Annex IV of the Habitats Directive, Council Directive 92/43/EEC) particularly sensitive to habitat loss and fragmentation (Mortelliti et al. 2010; Keckel et al. 2012) because of its very low dispersal capability (Juškaitis 2008; Bani et al. 2016). We assumed that the variability of the species occurrence at a large-scale, where the influence of micro-habitat characteristics is less important, is mainly influenced by the amount of suitable habitats (nodes) and by the connectivity provided by nodes and other landscape elements that can be used as corridors or stepping stones by the species.

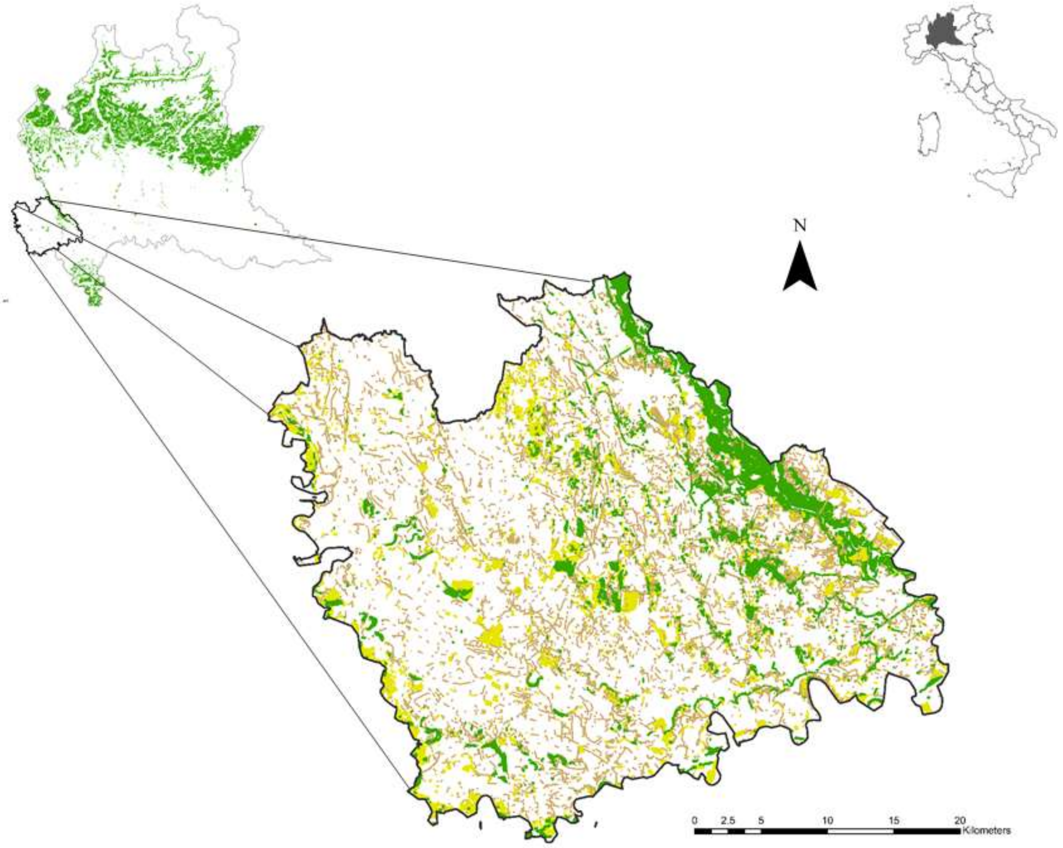
3.3.1 Study area

The study was carried out in the western part of Lombardy (northern Italy, 45°21′ N 8°80′ E), in an area of about 1,300 km^2 bordered by three main rivers, the Sesia, Po and Ticino (Fig. 2). This area shows the typical characteristics of a European lowland agro-ecosystem, with a prevalence of intensive cultivated crops, which represent 74% of the total surface and mainly consist of rice paddies. The 67% of the surface devoted to paddies in our study area is annually flooded from April to August, representing a very hostile matrix feature for wildlife movements (Ranghetti et al. 2015). The remaining area includes built-up areas (9%), reforestations and other arboreal cultivations (7%), hedgerows often distributed along the crop field borders and ditches (3%) and original forest remnants (7%, 99% of which are smaller than 1 km^2). Even though the study area shows a high degree of forest fragmentation, its eastern part falls within the boundaries of the

Ticino Natural Park, a 220 km^2 wide protected area with residual continuous forests, which represent the main source area for several forest-dwelling species (Bogliani et al. 2007).

Figure 2

Study area in Lombardy region (northern Italy, 45°21’ N 8°80’ E) with forests in dark green, traditional poplar cultivations, plantations for biomass production and reforestations in light green and hedgerows in brown.

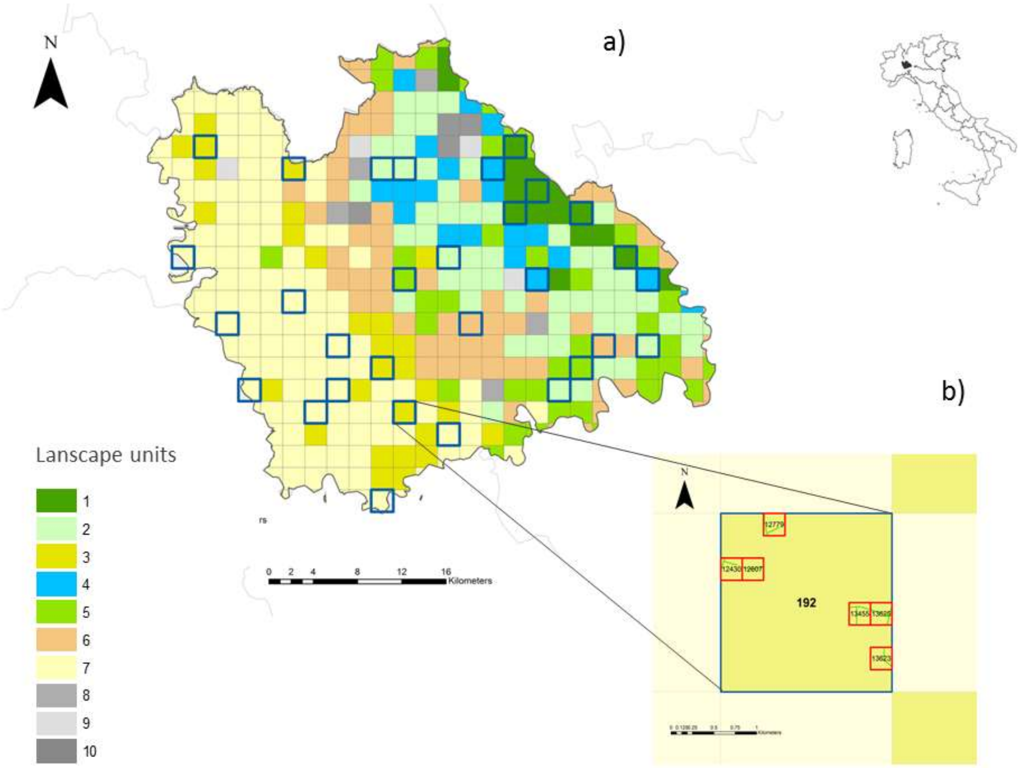


3.3.2 Sampling design and data collection

The data collection followed a stratified cluster sampling design. In particular, we overlapped a 2-km grid to the study area and, within each cell, we calculated five environmental variables: percentage of forest cover, distance from the source area (Ticino Natural Park), density of hedgerows, density of main roads and degree of habitat fragmentation calculated by means of a Modified Proximity Index (MPI; Bani et al. 2006) setting the proximity radius to 1 km. We calculated the environmental variables starting from the most recent regional land use map DUSAF 4.0 (ERSAF 2014). Subsequently, by means of a k-means cluster analysis, the 2-km cells were grouped in homogeneous Landscape Units (LUs) according to the five environmental variables considered, which led to a total of 10 LUs. The LUs were defined as follows: LU1, source areas; LU2, moderate fragmented areas near source areas; LU3, moderate fragmented areas far from source areas; LU4, highly fragmented areas near source areas; LU5, arable land near source areas; LU6, arable land with a high hedgerows density; LU7, arable land far from source areas; LU8, suburban areas; LU9, human infrastructures; LU10, urban areas. Among the 325 2-km cells of the study area, we randomly selected 30 cells (covering about 10% of the study area), allocated in each LU, excluding the LUs associated to urban areas (LU8, LU9 and LU10), in a number proportional to its size (Fig. 3a). The 30 2-km cells represent the primary sampling units in the sampling design. Within each primary sampling unit, we randomly selected six 250-m cells, which represent the secondary sampling units (Fig. 3b). In order to detect the presence of the Hazel Dormouse, we placed nest-tubes in every secondary sampling unit hosting land-cover types potentially suitable for the species. In order to maintain a constant sampling effort, we located a number of nest-tubes proportional to the extension of the potentially suitable land-cover within each secondary sampling unit, setting a maximum of eight nest-tubes per cell. The nest-tubes were placed at a distance of 50 m one from each other (Juškaitis 1997; Mortelliti et al. 2011). We placed 722 nest-tubes in 118 secondary sampling units, which were inspected in spring during two consecutive years (March-July 2014 and 2015). Overall, we detected the presence of the Hazel Dormouse in 63 secondary sampling units. A nest-tube was considered occupied if we observed the species in it at least on one visit, or if we found nests or feeding signs on the hazelnuts we left to attract it.

Figure 3

The ten landscape units characterizing the study area and the 30 primary sampling units (blue squares) (a); the six secondary units (in red) within each primary sampling unit (b).



3.3.3 Simulated Species Landscape Perceptions (SSLPs) setup

In order to set up the SSLPs for the Hazel Dormouse, we started from the most recent digital land-use cartography available for the study area (DUSAF 4). We also considered the Forest Management Plan of the Province of Pavia (Provincia di Pavia 2012) and the Agricultural Information System of Lombardy (SIARL-2013, ERSAF 2013), in order to obtain more detailed information regarding the distribution of arboreal plantations within the study area. By merging these three cartographies, we obtained 18 land-cover types in our study area. For this study, we considered *a priori* two land-cover types as a suitable habitat for the Hazel Dormouse, i.e. forests and riparian woodlands (Juškaitis 2008; Mortelliti et al. 2014), and four land-cover types as potentially suitable, i.e. three arboreal plantations (traditional poplar cultivations, plantations for biomass production, henceforth defined as biomasses, and reforestations) and hedgerows.

Starting from the spatial configuration of the considered land-cover types in the original landscape, we implemented the different SSLPs. In every SSLP, we assigned the role of node to the layer corresponding to woodlands (i.e. forests and riparian woodlands, merged together), while we alternatively assigned the role of node, connectivity element, or matrix to poplar cultivations, biomasses, reforestations and hedgerows for a total of 81 SSLPs. Layers arrangement (i.e. merging together the land-cover types playing the same role within each SSLP) was performed using the software ArcGIS 10.0 (ESRI 2011).

In order to evaluate the actual landscape perception of the Hazel Dormouse in our study area, we calculated the habitat amount and landscape connectivity for each SSLP using the software Fragstats 4.0 (McGarigal et al. 2002) working at the class level. Particularly, for each SSLP, we calculated the *Class Abundance* (*CA*) as a measure of habitat amount, and the *Connectance Index* (*CONNECT*) as a measure of landscape connectivity. While calculating fragmentation metrics for each SSLP, when a land-cover type was considered as a node it was taken into account both in the computation of *CA* and *CONNECT*, whereas when it was considered as a connectivity element it was taken into account only in the computation of *CONNECT*. If a land-cover type was assumed as matrix, it was not considered in any metrics. *CA* is the total area of the class (i.e. for each SSLP all the land-cover types playing the role of nodes, merged together) and equals to the sum of the areas (m^2) of all the patches belonging to that class, divided by 10,000 (for conversion to hectares).

$$CA = \sum_{j=1}^{n} a_{ij} \left(\frac{1}{10000}\right)$$

Where a_{ij} is the area (m^2) of the patch j belonging to the class i.

CONNECT is calculated by dividing the number of existing connections between all the patches of the class (i.e. for each SSLP all the land-cover types playing the role of nodes or connectivity elements, merged together), placing a threshold distance beyond which two patches are considered as not connected to each other, and the maximum possible number of connections between all patches.

The result is then multiplied by 100 to obtain the percentage value.

$$CONNECT = \left[\frac{\sum_{j=k}^{n} c_{ijk}}{\frac{n_i(n_i-1)}{2}}\right](100)$$

Where c_{ijk} is the existing connection between the patches *j* and *k*, belonging to the class *i*; and n_i is the total number of patches belonging to the class *i* within the investigated area. The value of this index ranges between 0, when there is a single patch or there are patches that are not connected to each other, and 100, when each patch is connected to the others. The threshold distance beyond which two patches are considered as not connected to each other should be set to the maximum dispersal capability of the target species in unsuitable habitats. For this study, we set the *CONNECT* threshold distance to 200 m. We obtained this value by averaging different documented values of Hazel Dormouse dispersal capability in unsuitable habitats (100 m, Bright 1998; 250 m, Büchner 2008; 100-300 m, Mortelliti et al. 2012).

We chose *CONNECT* as landscape connectivity index as Wang et al. (2014) highlighted that a good connectivity index should be independent of habitat abundance, in order to disentangle the effects of habitat amount and landscape connectivity and, at the same time, it should be able to differentiate landscapes with different spatial aggregation. The authors demonstrated that among the connectivity indexes calculated by Fragstats 4, *CONNECT* shows a very low dependency on habitat abundance, especially in landscapes where the habitat type is rare. Moreover, among all the indexes independent on habitat abundance, *CONNECT* is the only one that can distinguish between actual landscapes with a different spatial aggregation.

These two metrics were calculated using a Moving Window, i.e. a buffer area which moves from pixel to pixel of the whole landscape, circumscribing a sub-landscape where the values of *CA* and *CONNECT* are calculated. The Mowing Window should circumscribe an area corresponding to the scale of the target species perception of the fragmentation phenomenon. For this study, we used a circular Moving Window with a radius of 250 m, corresponding to the average dispersal distance covered by the Hazel Dormouse within suitable habitats (Juškaitis 1997). The results for each SSLP were two raster files, one for *CA* and one for *CONNECT*, in which each pixel takes the value corresponding to the metrics calculated within the Moving Window centred on it.

3.3.4 Statistical Analyses

We considered as Statistical Units (SU) the sets of nest-tubes within each secondary sampling units. We assigned to each SU a value of presence or absence, if the presence of the Hazel Dormouse was detected in at least one nest-tube, or if none of the nest-tubes were occupied by the species, respectively. We assume the absence of false negatives within our sample, as it is very unlikely that if the species occurs in a secondary sampling unit it does not colonize at least one among all the nest-tubes placed within the secondary sampling unit in two consecutive years. We excluded the presence of spatial autocorrelation in the Hazel Dormouse distribution data at different distance classes by means of the Moran's I test with 999 permutations (Cliff and Ord 1981) using the *spdep* package in R (Bivand et al. 2005).

Subsequently, for every SSLP, we assigned to each SU the average of the values of *CA* and *CONNECT* calculated for the pixels associated with the nest-tubes belonging to the correspondent SU.

In order to assess the influence of habitat amount and landscape connectivity on the presence of the Hazel Dormouse within each SSLP, we performed Generalized Linear Mixed Models (GLMMs) with a Binomial distribution. We used the presence/absence of the Hazel Dormouse as the response variable, the primary sampling units (2 km cells) as a random factor and *CA* and *CONNECT* as fixed factors. In order to account for the possible effect of the habitat within which we placed the nest-tubes belonging to the same SU, we included in each model an independent categorical variable with five levels: woodland, poplar cultivation, biomass, reforestation and hedgerow. This way, we built up 81 models, one for each SSLP. For each model, we checked diagnostic plots and we excluded spatial autocorrelation of model’s residuals at different distance classes by means of the Moran's I test with 999 permutations. We evaluated the goodness of fit of each model by the conditional R-squared. The model that performed better corresponded to the landscape effectively perceived by the Hazel Dormouse.

All statistical analyses were performed using R v. 3.1 (R Core Team 2014).

3.3.5 Results

The conditional R-squared values of the models performed for all the 81 SSLPs are shown in Table 1. We created a five-letter code for each SSLP. We adopted the capital letter to indicate the role of node (P, B, R, H), the lowercase letter to indicate the role of connectivity element (p, b, r, h) and 0 to indicate the role of matrix for each land-cover type. Each code always started with W (in capital letter) because woodlands were only considered as nodes. The rows in Table 1 show the 27 combinations of poplar cultivations, biomasses and reforestations as nodes, connectivity elements or matrix whereas the columns include hedgerows as nodes, connectivity elements or matrix.

The models referring to the SSLPs where hedgerows played the role of nodes (H) performed better than the models referring to the other two roles (h and 0) in 17 out of the 27 cases. When the hedgerows played the role of connectivity element (h) the models performed better in 8 cases, while when the hedgerows were considered as matrix (0) the corresponding model resulted as the best one in one case only. Moreover, in one case the models referring to the SSLPs where hedgerows played the role of nodes (W0bRH) and of connectivity elements (W0bRh) proved to be both the best ones. On the other hand, within each column corresponding to the three possible roles played by hedgerows, the first row represents the control condition. The SSLPs coded as W000H, W000h and W0000, indeed, are composed only of woodlands as nodes, and hedgerows considered, alternatively, as nodes (H), connectivity elements (h) or matrix (0). Of all the remaining 78 SSLPs, only 14 (Wp00H, Wp00h, Wp000, W0b00, W00rH, W00rh, Wpb0H, Wpb0h, Wp0rH, Wp0rh, W0brH, W0brh, WpbrH, Wpbrh) correspond to models that showed values of conditional R-squared similar or higher than those calculated for the corresponding control condition. Overall, the model that performed better than all the others correspond to the WpbrH ($R^2c=0.530$), where hedgerows, other than woodlands, play the role of nodes, while poplar cultivations, biomasses and reforestations are considered as connectivity elements. Other two models only showed an R^2 higher that the 0.5 threshold; they are associated to the SSLP Wpb0 ($R^2c=0.517$) and Wp0r ($R^2c=0.518$) which differ from the best model only for biomasses and reforestation, respectively, in the role of matrix rather than of connectivity elements.

Table 1

Conditional R-squared values of the models performed for all the 81 SSLPs. In bold the best performing model.

SSLPs[1]	Hedgerows as Node (H)	Hedgerows as Connective element (h)	Hedgerows as Matrix (0)
	R^2 c	R^2 c	R^2 c
W000	0.457	0.449	0.451
WP00	0.349	0.315	0.321
Wp00	0.493	0.449	0.452
W0B0	0.337	0.331	0.335
W0b0	0.383	0.445	0.451
W00R	0.336	0.337	0.323
W00r	0.477	0.452	0.449
WPB0	0.325	0.286	0.289
WP0R	0.284	0.289	0.281
W0BR	0.294	0.295	0.279
WPbr	0.357	0.324	0.312
WPb0	0.353	0.318	0.327
WP0r	0.360	0.326	0.316
WpBr	0.342	0.340	0.332
WpB0	0.338	0.335	0.335
W0Br	0.349	0.343	0.333
WpbR	0.329	0.332	0.324
Wp0R	0.326	0.331	0.323
W0bR	0.335	0.335	0.325
WPBr	0.329	0.290	0.280
WPbR	0.284	0.289	0.280
WpBR	0.281	0.283	0.277
Wpb0	0.517	0.460	0.450
Wp0r	0.518	0.467	0.450
W0br	0.478	0451	0.450
WPBR	0.259	0.263	0.256
Wpbr	**0.530**	0.473	0.448

[1] The five letters codes (four letters in row and the last one in column) of the SSLPs is created assigning to each land-cover type (poplar cultivations, biomasses, reforestations and hedgerows) the capital letter to indicate the role of node (P, B, R, H), the lowercase letter to indicate the role of connectivity element (p, b, r, h) and 0 to indicate the role of matrix. Each code always starts with W because woodlands were considered in the role of nodes only. In row the 27 combinations of poplar cultivations, biomasses and reforestations in the role of nodes, connectivity elements or matrix. In column, the hedgerows in the role of nodes, connectivity elements or matrix.

3.3.6 Discussion

Other than in the SSLP associated to the best model (WpbrH), the SSLPs in which hedgerows played the role of node performed better than the others in 63% of the cases. This result demonstrated that the role of hedgerows as nodes for the Hazel Dormouse is highly reasonable, and it is unlikely that it was included in the best model accidentally. In addition, in 100% of the 14 SSLPs associated to models performing better than the control conditions, poplar cultivations, biomasses and reforestations never played the role of nodes. This strong result allowed us to reject the hypothesis that these three land-cover types could be used as nodes in an ecological network designed for the Hazel Dormouse. Moreover, the 14 best models are associated to all the SSLPs in which poplar cultivations, biomasses and reforestations play the role of connectivity elements, considered alone or in combination with each other. Among those, the SSLP associated with the best model simultaneously considered these three land-covers as connectivity elements.

Starting from the SSLP associated with the best model, we were able to define how an ecological network for the conservation of the Hazel Dormouse should be designed in a highly fragmented agro-ecosystem, clarifying the perception of the different landscape elements by the species. This result has important practical consequences. The use of hedgerows as conservation tools to mitigate the effect of habitat fragmentation has been widely debated over the past two decades (Davies and Pullin 2007). On the one hand, it has been asserted that there is insufficient evidence to define if hedgerows act as connectivity elements or are ineffective in promoting species dispersal. On the other hand, some authors proposed the possibility that hedgerows represent not only effective connectivity elements, but also suitable habitats for small mammals (Henein et al. 1998; Laurence and Laurance 1999; Tattersall et al. 2002; Wolton 2009). Our approach highlighted that hedgerows should be used as nodes in an ecological network designed for the Hazel Dormouse and, thus, that they probably represent suitable habitats for the species in our study area. An adequate spatial configuration of hedgerows, and a correct management of their internal characteristics (Chapter 5), is thus crucial for guarantee functional connectivity for the Hazel Dormouse in agricultural landscapes. Moreover, the application of the proposed approach to the Hazel Dormouse allowed clarifying the role of reforestations in an ecological network for a small mammal particularly sensitive to habitat fragmentation. Our analyses revealed that reforestations could play the role of connectivity elements, but not of nodes. This does probably not depend on structural and floristic characteristics of reforestations, which, being an intermediate successional vegetation stage, represents the most suitable habitat for the Hazel Dormouse (Capizzi et al. 2002; Juškaitis 2008), but on the intrinsic short time permanence (generally 20 years in Italy) of this land-cover type in agro-ecosystems, which does not allow the establishment of a Hazel Dormouse population. Even if reforestations cannot be used as nodes in an ecological network for this species, they remain important connectivity elements, as suggested also by one of the “Pan-European Guidelines for Afforestation and Reforestation with a special focus on the provisions of the United Nations Framework Convention on Climate Change” (Forest Europe 2008), which recommends promoting reforestation activities in order to improve the ecological connectivity of the landscape (García-Feced et al. 2011). In addition, our approach showed that two other arboreal plantations, not specifically designed to increase connectivity, could play the role of connectivity elements in an ecological network designed for the Hazel Dormouse, as reforestations do. These are poplar cultivations and biomasses, which probably have an internal structure sufficiently adequate to perform the role of connectivity elements for our target species. This is the first study that highlights the importance and the role of these two arboreal plantations in an ecological network for this species, offering a great opportunity to integrate conservation measures and socio-economic interests.

3.4 Conclusions

In this study, we presented an innovative method to identify the elements of an effective ecological network for a given target species. The need to implement a new method relies on the necessity to overcome the limitations of the traditional approaches used in ecological network identification.

The proposed method still shows some practical constraints. Indeed, the procedure to identify the real landscape perception by the target species, starting from the SSLPs setup and ending with the best model selection, is still quite complex, and involves the use of three different software types. Moreover, the procedure could become even more complex if the ecology of the target species is scarcely known and it is necessary to hypothesize different threshold distances beyond which two patches are considered not connected to each other in the calculation of *CONNECT* and to test different Moving Window sizes.

Future developments of the method should involve the implementation of a new statistical package in order to integrate the SSLPs setup, the fragmentation metrics calculation and the statistical analysis. The new package should also be able to manage different hypotheses for the *CONNECT* threshold distance and the Moving Window size. The latter point would make the proposed method very useful also to identify the maximum dispersal distance in unsuitable habitats and the response scale to the fragmentation phenomenon for target species for which this information is unknown.

On the other hand, the proposed method has several great advantages. First, it allows overcoming all the main limitations of both the physical approach and the ecological approach based on HSMs. Moreover, the method that we proposed is an objective procedure aimed at designing an effective ecological network for virtually every species sensitive to fragmentation of potentially every kind of habitat. Achieving this result with a standardized method would have important practical implications. In fact, it would allow obtaining objective information that could be formally integrated in landscape management plans for wildlife conservation, developed at different territorial jurisdictions, from the local to the continental one.

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Chapter 4

An ecological network for forest mammals: a multi-species and multi-scale approach

Abstract and links to other chapters

In this chapter, the new method to design ecological networks proposed in Chapter 3 was applied on two other mammal species characterized by a different vagility and degree of forest specialization: the European Badger and the Roe Deer. Data collection was carried out in a wide agricultural landscape in western Lombardy (northern Italy) by following a stratified cluster sampling design and by walking 720 500-m linear transects. In order to test the role that four agroforestry elements (i.e. hedgerows, poplar cultivations, biomasses and reforestations) should play in an ecological network designed for the two species, we set up different simulated species landscape perceptions (SSLPs). To compare the ability of the habitat amount and landscape connectivity calculated for each SSLP in explaining species distribution, binary logistic models were performed by applying a ridge regression penalty. For both species, all the analyses were repeated setting up different scenarios assuming different species dispersal abilities through the matrix. The comparison of the performance of the models pertaining to the different dispersal ability scenarios, highlighted that a generalist species with an intermediate vagility (the European Badger) and a species with an extremely higher vagility, but with a higher degree of forest specialization (the Roe Deer), have the same maximum dispersal ability through the matrix (2 km). The model that better explained the variability of the European Badger occurrence corresponds to the SSLP where poplar cultivations and hedgerows, other than woodlands, play the role of node, biomasses play the role of connectivity element, while reforestations are perceived as matrix. Conversely, the best performing model for the Roe Deer corresponds to the SSLP where poplar cultivations and reforestations, other than woodlands, were considered as node, while biomasses and hedgerows as connectivity element. Starting from the best models, the occurrence probability of the two species within the whole study area was mapped and compared with the map of occurrence probability developed for the Hazel Dormouse starting from the results presented in Chapter 3. The very low percentage (4%) of the total extent of the suitable area for the three species, simultaneously suitable for all of them, confirmed that it is crucial adopting a multi-species and a multi-scale approach to ensure the conservation of a large part of the entire forest animal community. The results regarding the European Badger obtained with this study, was the starting point to develop further analyses on how to manage internal characteristics of hedgerows, displayed in Chapter 5.

4.1 Introduction

The continuous spread of human settlements and cultivated areas has dramatically decreased the extent of the original European forests reducing them to fragments scattered in human-dominated landscapes. Species sensitive to fragmentation perceive this kind of landscapes as composed of small suitable habitat fragments surrounded by an unsuitable matrix, which prevents movements to and from other fragments. The difficulty to cross the unsuitable matrix decreases animals' ability to reach declining populations in other fragments, to recolonize fragments where local extinctions have occurred, or to colonize new suitable habitats expanding the species range (Bennett 2003). The phenomenon of fragmentation is particularly evident in agricultural areas, where the original forest cover has been widely clear-cut to leave space for increasingly intensive crops (Darby 1956; Williams 2002). In order to counteract the effects of fragmentation in agricultural landscapes, it is crucial to ensuring the ecological connectivity among forest fragments (Noss et al. 1997; Soulè and Terborgh 1999). The ecological connectivity in modified landscapes depends on the availability and arrangement of suitable habitats, but also on sub-optimal habitats that may assist individuals to move through the inhospitable matrix, such as corridors or stepping stones (Bennett 2003). Managing the spatial arrangement of these habitats is thus a good strategy for reducing the negative effects of fragmentation (Lechner et al. 2015). Sub-optimal habitats for forest-dwelling species in agricultural landscapes are typically represented by semi-natural elements such as hedgerows, linear strips of shrubs and trees along the boundaries of fields (Bennett 2003), and arboreal cultivations. However, the degree of connectivity among forest remnants provided by the composition and the spatial arrangement of these semi-natural habitats is strongly species-specific (Tischendorf and Fahrig 2000; Trainor et al. 2013). First, different species perceive habitats in different ways. A habitat perceived as suitable by a species could be sub-optimal or even unsuitable for another species, according to their species-specific ecological requirements. Moreover, even the spatial arrangement of habitat fragments may affect the connectivity according to species-specific dispersal ability within the unsuitable matrix (Bowman et al. 2002; Cushman et al. 2013). For instance, two forest remnants may be perceived as connected by a species, but they can be simultaneously perceived as not connected by a species with a lower dispersal ability. The species dispersal ability among habitat remnants depends on the combined effect of the species vagility in suitable habitat and of how it is negatively affected by the unsuitable matrix that, in turn, depends on the species degree of forest specialization (Beier et al. 2008; Cushman and Landguth 2012). Therefore, when the final goal is the conservation of a great part of the forest species community, management actions aimed to increase landscape connectivity should not be targeted to a single species but to multiple species with different ecological needs (Cushman et al. 2013, Trainor et al. 2013).

As highlighted by Bennett (2003), forest fragmentation is not a random process, but "it is biased toward those areas that have the most fertile soils or are most accessible, such as plains and fertile river valleys". The lowland area in northern Italy (i.e. the plain of the Po river) is a typical example of a highly fragmented agricultural landscape, with small forest fragments scattered in an intensive cultivated matrix. In this area, semi-natural habitats that could serve as corridors or stepping stones for wildlife are represented by hedgerows and by three arboreal cultivations (traditional poplar cultivations, plantations for biomass production, henceforth defined as biomasses, and reforestations). Understanding how, and at what spatial scale, forest species perceive these four land-cover types is crucial in order to set up management actions addressed to mitigate the effect of fragmentation on forest-dwelling species in the highly fragmented lowland in northern Italy. In this study, we investigated if hedgerows, poplar cultivations, biomasses and reforestations are perceived as corridors or stepping stones, or even as suitable habitats, by two mammal species sensitive to forest fragmentation (Virgós 2001a, 2001b, 2002; Coulon 2004), but characterized by a very different vagility and a different degree of forest specialization: the European Badger (*Meles meles*) and the Roe Deer (*Capreolus capreolus*). The European Badger is a generalist forest species with a maximum dispersal ability in suitable habitats of about 20 km (Byrne et al. 2014). Conversely, the Roe Deer is a more specialist forest species with a very high dispersal ability in suitable habitats (ranging from 40 to 120 km) in colonizing populations (Gaudin 1993; Wahlström and Liberg 1995). This is the case of our study area, where the Roe Deer is re-colonizing lowland areas from the Ticino Natural Park in the central part of the study area, and from the forested Apennine hilly areas located in the southern part of the study area. By comparing different species' perceptions of simulated landscapes (see Chapter 3), we identified how these two species perceive hedgerows, poplar cultivations, biomasses and reforestations and which is their maximum dispersal ability through the unsuitable matrix in our study area. We then compared the obtained results with those obtained for the Hazel Dormouse, in a portion of the study area considered in this chapter, by means of the same method of analysis. The Hazel Dormouse is a small mammal with a high degree of forest specialization and a very low dispersal ability through the unsuitable matrix (about 200 m, Bright 1998; Büchner 2008; Mortelliti et al. 2012). In particular, starting from the amount of land-covers perceived as suitable habitats and the connectivity provided by the land-covers perceived as suitable habitats or connectivity elements, we realized a suitability map for the entire study area summarizing the occurrence probability of the three species. This way, we were able to identify how much of the landscape was suitable for each species and how much the areas suitable for the three species overlap.

4.2 Methods

4.2.1 Study area

The present study was carried out in an area of about 2,900 km^2 located in the western part of the Po plain (Lombardy, northern Italy, 45°11′ N 9°05′ E). Intensive cultivated crops, in particular paddies (41.4%) and other annual crops (wheat, maize, and alfalfa; 31.85%), cover most of the total surface. The remaining area includes built-up areas (10.3%), reforestations and other arboreal cultivations (6.8%). Original forests cover only the 4.9% of the total surface and they are mainly composed of pedunculate oak (*Quercus robur*), common hornbeam (*Carpinus betulus*), field maple (*Acer campestre*), field elm (*Ulmus minor*), and common ash (*Fraxinus excelsior*). Riparian woodlands developing along rivers are mainly characterized by common alder (*Alnus glutinosa*), poplars (*Populus alba, P. nigra* and *P. canadensis* cultivar), and willows (*Salix* sp.). Overall the study area shows a high degree of forest fragmentation (99% of forest remnants in agricultural areas are smaller than 1 km^2), however continuous forests are located along the Ticino River, which crosses the study area from North to South, and in the hilly area located in the southern part of the study area (Fig. 1).

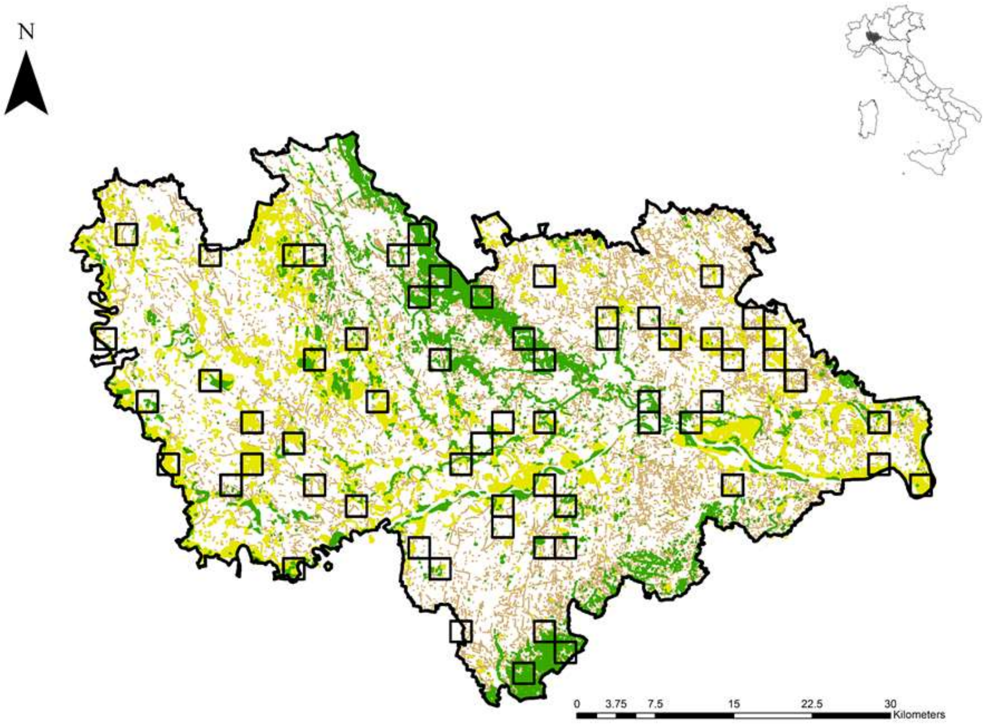
4.2.2 Sampling design and data collection

Data collection followed a stratified cluster sampling design (Krebs 1999; Barabesi and Fattorini 2013). Using ArcGIS 10.0 (ESRI 2011), we overlapped a 2-km grid to the study area, dividing the landscape in cells with a surface including more than one individual home range of both European Badger and Roe Deer (European Badger: 3.83 km^2 for a family group, Remonti et al.

2006; Roe Deer: 1 km^2 for single individuals or small groups in fragmented landscape, Cargnelutti et al 2002). Subsequently, the 2-km cells were grouped in 10 homogeneous Landscape Units (LUs) according to five environmental variables by means of a k-means cluster analysis (see Chapter 3 for more details). Among the 620 2-km cells of the whole study area, we randomly selected 62 cells (covering 10% of the study area), allocated in each LU proportionately to its extent (Krebs 1999) (Fig. 1). Within each of the 62 2-km cell, we randomly selected six 250-m cells. The 2-km cells represent the primary sampling units, while the 250-m cells represent the secondary sampling units in our sampling design. In order to detect the presence of the European Badger and the Roe Deer in our study area, we identified two 250-m linear transects within each secondary sampling unit. We collected European Badger and Roe Deer data between April and September 2014 along 720 linear transects spotting signs of presence (i.e. latrines, setts, and footprints for the European Badger and fraying barks, resting places and footprints for the Roe Deer).

Figure 1

Study area in Lombardy region (northern Italy, 45°21' N 8°80' E) with forests in dark green, traditional poplar cultivations, plantations for biomass production and reforestations in light green, and hedgerows in brown. The black squares are the 62 2-km primary sampling units.



4.2.3 Simulated Species Landscape Perceptions (SSLPs) setup

In order to investigate how, and at what spatial scale, the European Badger and the Roe Deer perceive hedgerows, poplar cultivations, biomasses and reforestations in our study area, we applied the method of the comparison of simulated species landscape perceptions (SSLPs, see Chapter 3 for more details). This method allows identifying the role that a land-cover type has for a given species, i.e. if the species perceives the land-cover type as a node (suitable habitat), as a connectivity element (typically a sub-optimal habitat which favour animals' movements among nodes of suitable habitat) or as unsuitable matrix. The procedure consists in setting up different SSLPs, which correspond to every possible combination of the land-cover types of interest, alternatively assuming the role of nodes, connectivity elements, or matrix. The role played by each land cover type in each SSLP is actually defined by calculating two fragmentation metrics. Particularly, for each SSLP we will calculate the habitat amount of all those land-cover types considered as nodes, merged together in a single land-cover, and the landscape connectivity of all those land-cover types considered as nodes or connectivity elements, merged together. Conversely, all those land-cover types playing the role of matrix in the considered SSLP are not included in the computation of any metric. Finally, the ability of the fragmentation metrics calculated for each SSLP to explain the actual species distribution detected on the field was calculated. The SSLP characterized by the fragmentation metrics that perform better than all the others in explaining the species occurrence will provide information about how the species perceives each considered land-cover type.

To set up the SSLPs for both the European Badger and the Roe Deer, we merged three recent digital cartographies available for the study area (DUSAF 4, ERSAF 2014; Forest Management Plan of the Province of Pavia, Provincia di Pavia 2012; Agricultural Information System of Lombardy, SIARL-2013, ERSAF 2013) by using ArcGIS 10.0. This way, we obtained detailed information regarding the distribution of forests, hedgerows, poplar cultivations, biomasses and reforestations within the study area. Starting from the pattern of these land-covers in the original landscape, we implemented 81 SSLPs for the European Badger and 54 for the Roe Deer. In every SSLP, we always assigned the role of node to the layer corresponding to woodlands (i.e. forests and riparian woodlands, merged together) for both species. Indeed, both for the European Badger (Virgós 2001b; Balestrieri et al. 2009a; Piza Roca et al. 2014) and for the Roe

Deer (Barančeková 2004) broadleaved woodlands are indicated as the optimal habitat. Subsequently, we alternatively assigned the role of node, connectivity element, or matrix to hedgerows, poplar cultivations, biomasses and reforestations for the European Badger, while for the Roe Deer we alternatively assigned the role of node, connectivity element, or matrix to poplar cultivations, biomasses and reforestations, and only those of connectivity element, or matrix to hedgerows. We tested the hedgerows in the role of node for the European Badger as the importance of this landscape element as suitable habitat for different mustelids was highlighted in several studies (Hilty and Merenlender 2004; Červinka et al. 2013; Šálek et al. 2009). Conversely, we did not test the hedgerows in the role of node for the Roe Deer, as we excluded that a hedgerow could be perceived as node by a species of that size. In order to define the habitat amount and the degree of connectivity for each SSLP, we calculated two class metrics using the software Fragstats 4.0 (McGarigal et al. 2002): *CA* (*Class Abundance*) and *CONNECT* (*Connectance Index*) (see Chapter 3 for metrics' description). Within each SSLP, *CA* was calculated for all the land-cover types playing the role of nodes, merged together, while *CONNECT* was calculated for all the land-cover types playing the role of nodes or connectivity elements, merged together. To our knowledge, any empirical study addressed to evaluate the maximum dispersal ability in the unsuitable matrix have been performed nor for the European Badger nor for the Roe Deer. We thus decided to tested three different hypothetical maximum dispersal abilities for each of the two species, in order to correctly calculate *CONNECT*. Indeed, simply assuming a single plausible dispersal distance from literature is hazardous, as dispersal ability plays a dominant role in affecting the degree of connectivity for a given species in fragmented landscapes (Cushman and Landguth 2012). For the European Badger, we tested as plausible maximum dispersal distance through the matrix 1 km, 2km and 4 km. We selected these distances because we excluded that a European Badger could be able to cross an unsuitable matrix for more than 4 km. Indeed, the average of the mean daily movement, in mosaics of suitable and unsuitable habitats, calculated for eight European Badger populations in Europe resulted 3.8 ± 0.7 km (Kowalczyk et al. 2006). Also Loureiro et al. (2007) suggested that the maximum distance travelled during a night by a European Badger corresponds to 4 km. In this study, it is stated that the distance more frequently walked by the European Badger in half an hour is 200 m, which, considering a mean summer night span equal to approximately 10 hours, corresponds to 4 km travelled during a night. As the European Badger makes its movements only during the night (while during the day it generally rests in woodland areas), it needs that two suitable or sub-optimal habitats lie no more than the distance it can cross during a night, i.e. 4 km. For the Roe Deer, we tested as plausible maximum dispersal distances in the matrix 2 km, 4 km and 8 km. We selected these distances because in most studied Roe Deer populations in fragmented landscapes, where the species is typically closely tied to forest remnants (Hewison et al. 2001, Cragnelutti et al. 2002), dispersal distances resulted rather short (Coulon et al. 2004). The authors reported that preliminary results showed a Roe Deer dispersal ability of about 3 km in a highly fragmented area in southwestern France. We thus decided to set the minimum hypothetical dispersal ability of this species to 2 km and to test other two hypothetical dispersal distances: 4 km and 8 km.

Both *CA* and *CONNECT* were calculated using a circular moving window. As the moving window should circumscribe an area corresponding to the scale of the target species perception of the fragmentation phenomenon, we used a moving window with a radius of 2 km in order to calculate *CA* for both the European Badger and the Roe Deer. Conversely, in order to calculate *CONNECT*, we used moving windows with different radius corresponding, from time to time, to the maximum dispersal ability we were testing, both for the European Badger and the Roe Deer. Combining the number of the land-cover types considered in the role of node, connectivity element or matrix, and the three tested hypothetical dispersal abilities, we obtained 243 SSLPs for the European Badger, and 162 for the Roe Deer (for which we did not consider hedgerows in the role of node). The output of the SSLPs set up were two raster files for each SSLP, one for *CA* and one for *CONNECT*, where each pixel takes the value corresponding to the metrics calculated within the moving window centred on it.

4.2.4 Statistical analyses

In order to identify how the two target species perceive hedgerows, poplar cultivations, biomasses and reforestations, we compared the ability of the two fragmentation metrics calculated for each SSLP in explaining the variability of the actual target species distribution within the study area. All the statistical analyses described below were performed using R v. 3.1 (R Core Team, 2014).

For both the European Badger and the Roe Deer we considered as Statistical Units (SUs) the primary sampling units. We assigned to each SU a value of presence or absence, if the presence of the species was detected in at least one secondary sampling unit within the SU, or if none of the secondary sampling units were occupied by the species, respectively. We assumed the absence of false negatives within our samples, as it is very unlikely that if the species occur in a primary sampling unit it would not be detected in at least one among all the six secondary sampling units (making a total of 12 transects). For both species, we excluded the presence of spatial autocorrelation in the presence data at different distance classes by means of a Moran's I test with 999 permutations (Cliff and Ord 1981) using the *spdep* package (Bivand et al. 2005).

Subsequently, for all the 243 SSLPs set up for the European Badger and the 162 for the Roe Deer, we assigned to each SU the average of the *CA* and *CONNECT* values calculated for all pixels falling within the primary sampling unit. In order to assess the influence of *CA* and *CONNECT* on the occurrence of the two species, for each SSLP, we performed a binomial logistic regression model with the presence/absence of the species within each SU as response variable and *CA* and *CONNECT* associated to that SU as independent variables. As we collected both the European Badger and Roe Deer presence data following a stratified sampling design, considering the landscape composed of 10 LUs defined at the primary sampling unit level, we decided to take into account the landscape context by including in each model an independent categorical variable with 10 levels, corresponding to the 10 LUs. For the European Badger, we also considered the distance of the SU centroids to streams and rivers as control covariate, as it was demonstrated that this species avoid to build dens closed to river in areas with a high risk of flooding (Hipólito et al. 2016), that is the case of our study area.

Considering the low number of presences (38 for the European Badger and 23 for the Roe Deer) compared to the number of independent variables (three continuous and one categorical variable with 10 levels for the European Badger and two continuous and one categorical variable with 10 levels for the Roe Deer) we performed the logistic models by applying a ridge regression penalty using the *rms* package (Harrell 2016). By means of the *pentrace* command we solved for the optimum penalty factor for each model and subsequently we fit the models using penalized maximum likelihood estimation. To obtain the bias-corrected R^2 we performed a resampling validation of each model by using the *validate* command and 500 repetitions.

This way, performed 243 models for the European Badger and 162 for the Roe Deer.

The SSLPs corresponding to the best models provided information about the more plausible dispersal abilities of the two species and about how they perceive the considered land-cover types.

4.2.5 Mapping habitat amount and connectivity

Starting from the statistical relationship between the habitat amount and the connectivity of the land-covers perceived as nodes and connectivity elements, resulted from the best model for each species, we produced two habitat suitability maps, one for the European Badger and one for the Roe Deer, that spatially represent the probability of the species occurrence within the whole study area. We also produced a third habitat suitability map, starting from the results obtained for the Hazel Dormouse in Chapter 3. The occurrence probability was predicted for each species starting from the habitat amount (*CA*) of all the land cover types playing the role of nodes and the connectivity (*CONNECT*) provided by all the land cover types playing the role of nodes or connectivity elements within the best performing SSLP. For the European Badger, we also took into account the LU to which each pixel belongs and its distance to streams and rivers, while for the Roe Deer we took into account the LU to which each pixel belongs only. Conversely, for the Hazel Dormouse, we ran a GLMM using the species presence/absence within secondary sampling units (see Chapter 3) as dependent variable, *CA* and *CONNECT* calculated for the best performing SSLP as fixed factors and the 2-km primary sampling units as a random factor, disregarding the variable describing the specific habitat where nest-tubes were located (see Chapter 3 for more information) as the habitat composition at local scale is unknown for the entire surface of the study area. The occurrence probability of the three species was then classified into two classes: scarce (0–0.6) and high (0.6–1.0). We finally joined the three maps in order to evaluate how much the suitable areas for the three species overlapped.

4.3 Results

4.3.1 European Badger

The R^2 values of the 243 models performed for the European Badger are shown in Table 1. The three columns correspond to the three hypothesized maximum dispersal abilities of the species in unsuitable habitat: 1 km, 2 km and 4 km. Each SSLP is associated to a five-letter code, where the capital letter indicates the role of node (P, B, R, H), the lowercase letter indicates the role of connectivity element (p, b, r, h) and 0 indicate the role of matrix for each land-cover type (poplar cultivations, biomasses, reforestations and hedgerows). Each code always started with W because woodlands were always considered as nodes in an ecological network for the European Badger. The rows in Table 1 show the 27 combinations of poplar cultivations, biomasses and reforestations as nodes, connectivity elements or matrix whereas the three columns, within each dispersal ability scenario, refers to hedgerows as nodes, connectivity elements or matrix. Among all the 243 models, those that generally fitted better the observed data belong to the set of models corresponding to the scenario of a maximum dispersal ability of 2 km. In particular, the best model belongs to this dispersal ability scenario and corresponds to the SSLP WPb0H (R^2=0.233), where poplar cultivations and hedgerows, other than woodlands, were considered as nodes, biomasses were considered as connectivity elements, while reforestation were considered as unsuitable matrix.

Table 1

R^2 of the binary logistic models performed for the three sets of 81 SSLPs hypothesizing a maximum dispersal ability of the European Badger in unsuitable habitat equal to 1 km, 2 km, 4 km. In bold the best performing model.

SSLP[1]	Dispersal ability: 1 km			Dispersal ability: 2 km			Dispersal ability: 4 km		
	H	h	0	H	h	0	H	h	0
W000	0.182	0.180	0.204	0.192	0.193	0.175	0.181	0.181	0.170
WP00	0.193	0.189	0.163	0.207	0.202	0.174	0.198	0.187	0.188
Wp00	0.180	0.189	0.167	0.193	0.188	0.170	0.175	0.173	0.172
W0B0	0.184	0.178	0.206	0.176	0.180	0.154	0.174	0.178	0.175
W0b0	0.184	0.191	0.209	0.196	0.195	0.174	0.191	0.180	0.195
W00R	0.174	0.176	0.198	0.188	0.186	0.168	0.191	0.198	0.163
W00r	0.184	0.187	0.206	0.191	0.192	0.165	0.190	0.192	0.173
WPB0	0.195	0.193	0.174	0.216	0.213	0.189	0.202	0.193	0.164
WP0R	0.196	0.200	0.176	0.200	0.193	0.173	0.195	0.192	0.184
W0BR	0.177	0.179	0.183	0.189	0.187	0.159	0.187	0.187	0.152
WPbr	0.218	0.218	0.175	0.224	0.214	0.198	0.216	0.212	0.180
WPb0	0.198	0.199	0.179	**0.233**	0.220	0.198	0.207	0.204	0.171
WP0r	0.203	0.203	0.191	0.207	0.197	0.176	0.198	0.194	0.194
WpBr	0.207	0.209	0.161	0.199	0.207	0.164	0.195	0.193	0.152
WpB0	0.192	0.194	0.158	0.199	0.201	0.174	0.178	0.176	0.158
W0Br	0.182	0.182	0.186	0.187	0.184	0.167	0.196	0.187	0.153
WpbR	0.207	0.207	0.172	0.204	0.203	0.165	0.197	0.200	0.164
Wp0R	0.192	0.184	0.163	0.185	0.183	0.179	0.174	0.180	0.171
W0bR	0.185	0.211	0.194	0.201	0.195	0.163	0.198	0.200	0.165
WPBr	0.201	0.213	0.168	0.214	0.213	0.182	0.202	0.196	0.171
WPbR	0.214	0.213	0.171	0.221	0.214	0.190	0.222	0.215	0.174
WpBR	0.207	0.208	0.157	0.203	0.198	0.160	0.187	0.185	0.143
Wpb0	0.199	0.199	0.199	0.202	0.211	0.189	0.178	0.198	0.166
Wp0r	0.193	0.194	0.175	0.185	0.181	0.164	0.177	0.177	0.181
W0br	0.194	0.192	0.194	0.198	0.194	0.166	0.204	0.196	0.167
WPBR	0.206	0.212	0.170	0.214	0.204	0.169	0.204	0.204	0.167
Wpbr	0.210	0.213	0.174	0.212	0.214	0.176	0.199	0.198	0.169

[1] The five letters codes (four letters in row and the last one in column) of the SSLPs is created assigning to each land-cover type (poplar cultivations, biomasses, reforestations and hedgerows) the capital letter to indicate the role of node (P, B, R, H), the lowercase letter to indicate the role of connectivity element (p, b, r, h) and 0 to indicate the role of matrix. In row the 27 combinations of poplar cultivations, biomasses and reforestations in the role of nodes, connectivity elements or matrix. In column, the hedgerows in the role of nodes, connectivity elements or matrix, at three different maximum dispersal distance.

4.3.2 Roe Deer

The R^2 values of the 162 models performed for the Roe Deer are shown in Table 2. The three columns correspond to the three hypothesized maximum dispersal ability of the species in unsuitable habitat: 2 km, 4 km and 8 km. As for the European Badger, each SSLP is associated to a five-letter code, where the capital letter indicates the role of node, the lowercase letter indicate the role of connectivity element and 0 indicate the role of matrix for each land-cover type. Each code always started with W because woodlands were always considered as nodes.

Among all the 162 models performed, those that generally fitted better the observed data belong to the set of models corresponding to the scenario of a maximum dispersal ability of 2 km. In particular, the best model belongs to this dispersal ability scenario and corresponds to the SSLP WPbRh (R^2=0.360), where poplar cultivations and reforestations, other than woodlands, were considered as nodes, while biomasses and hedgerows were considered as connectivity elements.

Table 2

R^2 of the binary logistic models performed for the three sets of 54 SSLPs hypothesizing a maximum dispersal ability of the Roe Deer in unsuitable habitat equal to 2 km, 4 km, 8 km. In bold the best performing model.

SSLP[1]	Dispersal ability: 2km		Dispersal ability: 4 km		Dispersal ability: 8 km	
	h	0	h	0	h	0
W000	0.271	0.263	0.262	0.279	0.265	0.273
WP00	0.341	0.314	0.308	0.283	0.297	0.310
Wp00	0.296	0.260	0.254	0.253	0.268	0.263
W0B0	0.259	0.250	0.242	0.248	0.245	0.244
W0b0	0.276	0.253	0.260	0.264	0.264	0.277
W00R	0.284	0.279	0.280	0.275	0.286	0.274
W00r	0.265	0.256	0.254	0.259	0.258	0.270
WPB0	0.346	0.311	0.300	0.289	0.291	0.293
WP0R	0.348	0.329	0.324	0.329	0.331	0.326
W0BR	0.271	0.257	0.260	0.257	0.258	0.257
WPbr	0.333	0.309	0.300	0.300	0.303	0.300
WPb0	0.355	0.328	0.311	0.302	0.300	0.294
WP0r	0.322	0.300	0.301	0.302	0.304	0.306
WpBr	0.279	0.242	0.236	0.243	0.243	0.240
WpB0	0.313	0.262	0.245	0.239	0.247	0.237
W0Br	0.250	0.247	0.245	0.244	0.242	0.240
WpbR	0.320	0.278	0.281	0.285	0.282	0.286
Wp0R	0.303	0.301	0.271	0.274	0.286	0.288
W0bR	0.286	0.278	0.282	0.269	0.286	0.277
WPBr	0.322	0.288	0.287	0.293	0.287	0.332
WPbR	**0.360**	0.339	0.324	0.341	0.328	0.320
WpBR	0.298	0.259	0.258	0.260	0.263	0.262
Wpb0	0.328	0.273	0.258	0.253	0.264	0.259
Wp0r	0.282	0.256	0.252	0.251	0.267	0.266
W0br	0.267	0.258	0.264	0.264	0.262	0.257
WPBR	0.342	0.315	0.327	0.324	0.315	0.308
Wpbr	0.299	0.259	0.257	0.262	0.266	0.266

[1] The five letters codes (four letters in row and the last one in column) of the SSLPs is created assigning to each land-cover type (poplar cultivations, biomasses, reforestations and hedgerows) the capital letter to indicate the role of node (P, B, R), the lowercase letter to indicate the role of connectivity element (p, b, r, h) and 0 to indicate the role of matrix. In row the 27 combinations of poplar cultivations, biomasses and reforestations in the role of nodes, connectivity elements or matrix. In column, the hedgerows in the role of connectivity elements or matrix, at three different maximum dispersal distance.

4.3.3 Mapping habitat amount and connectivity

For the Hazel Dormouse, we calculated the habitat amount of woodlands and hedgerows and the connectivity provided by woodlands, hedgerows, poplar cultivations, biomasses and reforestations. For the European Badger, we calculated the habitat amount of woodlands, poplar cultivations and hedgerows and the connectivity provided by woodlands, poplar cultivations, hedgerows and biomasses. Finally, for the Roe Deer, we calculated the habitat amount of woodlands, poplar cultivations and reforestations and the connectivity provided by woodlands, poplar cultivations, reforestations, biomasses and hedgerows. The single-species habitat suitability maps for the three species are showed in Figure 2, while the multi-species habitat suitability map that spatially represents the areas of the landscape with a high (>0.6) occurrence probability of the three species is shown in Figure 3. We found a limited overlapping area of the three species occurrence probability, as only the 4% of the total extent of the area suitable for at least one of the three species is simultaneously suitable for all the three species. In particular, the extent of the area simultaneously suitable for the three species represents the 42% of the total suitable area for the Hazel Dormouse, the 22% of the total suitable area for the Roe Deer and only the 4% of the total suitable area for the European Badger.

Figure 2

Single-species suitability maps for the Hazel Dormouse, European Badger and Roe Deer in a highly fragmented area in northern Italy. The degree of suitability (i.e. occurrence probability) was predicted for each species starting from the habitat amount (*CA*) of all the land-cover types playing the role of nodes and the connectivity (*CONNECT*) provided by all the land-cover types playing the role of nodes or connectivity elements within the best performing SSLP.

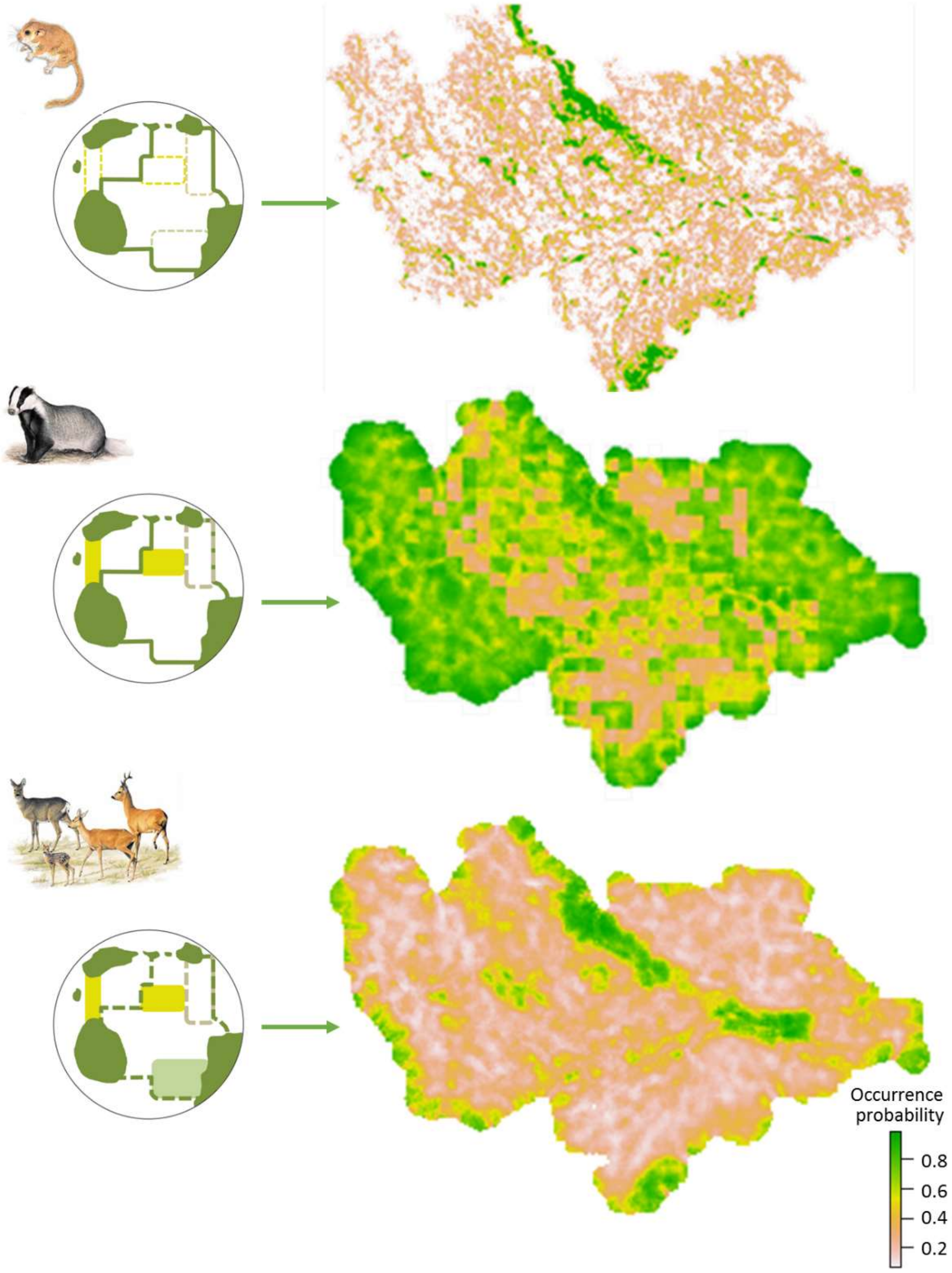
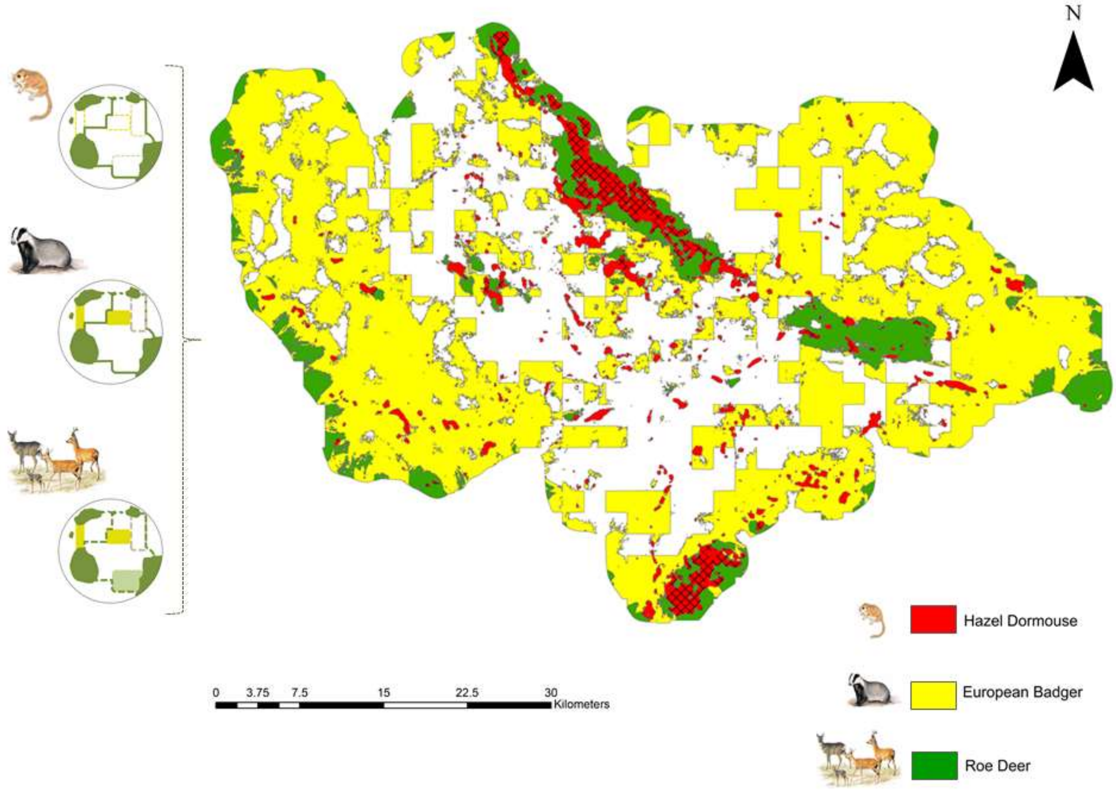


Figure 3

Multi-species suitability map in a highly fragmented area in northern Italy. The colors depict the areas with a high occurrence probability (> 0.6) for the Hazel Dormouse (red), the European Badger (yellow) and the Roe Deer (green). Dashed areas represent the areas simultaneously suitable for the three species.



4.4 Discussion and conclusions

To guarantee the functional connectivity for forest-dwelling species in agricultural landscapes, it is crucial to maintain and to manage the spatial arrangement of sub-forest habitats so that to facilitate the individuals' movements through the unsuitable matrix. The connectivity provided by the spatial arrangement of semi-natural habitats is strongly species-specific and mainly depends on the degree of suitability of each of these habitats for each species and on the species dispersal ability in the matrix, which strongly depends on the degree of species specialization for forest habitats (Beier et al. 2008, Cushman and Landguth 2012). Therefore, actions aimed to increase the habitat availability and landscape connectivity by managing sub-forest habitats should be targeted on multiple species with different ecological needs (Cushman et al. 2013; Trainor et al. 2013). In this study, we investigated if hedgerows, poplar cultivations, biomasses and reforestations are perceived as suitable habitats or connectivity elements by the European Badger and the Roe Deer in the highly fragmented Po plain in northern Italy. These two species are both sensitive to forest fragmentation, but they are characterized by a different vagility and a different degree of forest specialization.

One of the most interesting results of the present study was the identification of the maximum dispersal ability of both the European Badger and the Roe Deer through a matrix mainly composed of intensive crops. The comparative analyses highlighted that both species cannot cross the unsuitable matrix for more than 2 km in absence of a landscape element that can play the role of connectivity element. The maximum distance covered during a night by the European Badger in a mosaic of suitable and unsuitable habitat reaches 4 km and, as this species moves only during the night, it was reasonable hypothesizing that the maximum dispersal ability in the matrix in absence of connectivity elements was approximately half of the maximum distance covered in a mix of suitable and unsuitable habitat. This result agrees with a recent study carried out in southwest Britain (Woodroffe et al. 2016), which reported about 2 km as the maximum distance travelled by GPS-collared individuals during a night. On the other hand, 2 km is a maximum dispersal ability very low for a species with the vagility of the Roe Deer. This result, that is in agreement with what reported by Coulon et al. (2004), highlighted the strong specialization for forest habitats of this species in our study area (Bogliani et al. 2007), which, in turn, determine a stronger effect of forest fragmentation on the Roe Deer respect to the more generalist European Badger.

The method of the simulated species landscape perceptions applied on the European Badger highlighted that this species seems to use woodlands, poplar cultivations and hedgerows as nodes of suitable habitat, while the biomasses are only used as connectivity elements that allow the passage through the hostile matrix. As the European badger mostly feeds on earthworms living in the soil (Kruuk 1978; Balestrieri et al. 2004, 2009b; Cleary et al. 2009) the use of poplar cultivations as nodes could be due to the suitability of these plantations as foraging site. Indeed, the scarcity of the shrub layer typical of poplar cultivations, and their ground layer management (i.e. ploughing, mechanical weeding, etc.), increase the probability of food recruitment due to the easiness of digging in short grass surfaces (Piza Roca et al. 2014). On the other hand, in agricultural areas, hedgerows may offer coverage and suitable sett locations for the European Badger (Piza Roca et al. 2014), as it is confirmed by the finding of setts in hedgerows during our survey. Also O'Brien et al. (2016) highlighted the importance of hedgerows for the European Badger in areas with a very limited forest cover, suggesting that this species probably select hedgerows because they provide shelters and abundant and diversified food resources, such as invertebrates (Thomas and Marshall 1999; Facey et al. 2014), berries, small mammals and birds (Gelling et al. 2007). From these results, it seems that in agricultural landscapes, where forests, but also natural meadows, are very scarce, poplar cultivations and hedgerows are able to sustain European Badger populations by ensuring suitable sites for foraging and for building setts. On the contrary, biomasses are only used as connectivity element by the species probably because they are not managed, leading to the development of the shrub layer. European Badger, in fact, tend to avoid areas with an excessive developed shrub layer (Virgós 2001b) as it prevents both digging dens and searching for food in short grass. It is probable that this species avoids reforestations, even as simple connectivity elements, for the same reason. Indeed, in our study area reforestations have a very short time of permanence (generally 20 years), that make them an intermediate successional vegetation, typically hosting a very well developed shrub coverage, maybe excessive even for the simple movement of badgers.

The comparison of the models performed for the Roe Deer highlighted that this species seems to use woodlands, poplar cultivations and reforestations as nodes and biomasses, other than hedgerows, as connectivity elements. It is not surprising that the Roe Deer perceives reforestations as suitable nodes. Early stage reforestations characterized by a well developed and diversified shrub layer are, in fact, optimal habitats for this ungulate (Gill et al. 1996), both in providing important food resources, such as fresh buds, and in ensuring a complex vegetation structure able to offer suitable shelters (Gaillard et al. 1993). Reforestations are probably particularly important for this species during the vegetation period in late spring and early summer, which coincides with the period of births, when females choose places rich in understory to give birth and hide fawns. Similarly to the European Badger, the Roe Deer probably use poplar cultivations as foraging sites. Indeed, Roe Deer tends to frequent open areas closed to woodlands to grazing, and poplar cultivations are optimal grazing sites in agricultural landscapes because of the scarcity of the shrub layer typical of these plantations and because poplars offer a not negligible degree of canopy cover during the vegetation season. Conversely, biomasses are simply used by the species as connectivity element as they can provide shelters, but not adequate food resources for the species. Indeed, being characterized by a well developed mono-specific shrub layer, they prevent grazing activities, but also the possibility to find diversified fresh buds. Finally, the role of hedgerows as connectivity elements for this species was strongly confirmed by the models comparison under the 2 km dispersal ability scenario, where models corresponding to SSLPs where hedgerows play the role of nodes performed better than those where they play the role of matrix in 100% of the cases. This result highlighted the importance of hedgerows for Roe Deer's movements in highly modified agricultural landscapes and, thus, their crucial role in facilitating the re-colonization of the lowland areas of northern Italy by the species.

Starting from these results, we could define which land-covers should be used as nodes and connectivity elements, and at what distance patches should be located in the landscape, in order to design a functional ecological network for the European Badger and the Roe Deer. Moreover, from this information, we defined the suitability of the investigated landscape for the two species. Consequently, we were able to identify areas where habitat amount and landscape connectivity are not adequate for each of the two species and, thus, where it would be necessary to set up management actions. The map showed in Figure 2 clearly highlights significant differences in the extension and distribution of areas suitable and well connected for the two species. This is due to the different perception of poplar cultivations, biomasses, reforestation and hedgerows by the two target species, but also to the different degree of sensitivity to fragmentation characterizing the two species. From the map, it even seems that the areas suitable and well connected for the Roe Deer has an extension and distribution more similar to that of a much smaller species, the Hazel Dormouse. This depends on the different specialization degree for forest or sub-forest elements of the three species. Considering how much the area simultaneously suitable for the three species overlaps the areas suitable for each of the three species, we were able to order them for forest specialization and fragmentation sensitivity, in decreasing order Hazel Dormouse (42%), Roe Deer (22%) and European Badger (4%). The low degree of specialization of the European Badger is the cause of the lower value of the R^2 of the best model obtained for this species (R^2=0.233), compared to the values of the best models selected for the Hazel Dormouse (R^2=0.530) and for the Roe Deer (R^2=0.360). The low specialization of the European Badger is also evident and in Figure 2 and Figure 3, where the suitable area for this species is often mainly influenced by the landscape context (corresponding to the LUs represented as 2-km squares) rather than by the amount and connectivity degree of forest or sub-forest elements. Overall, as only the 4% of the total extent of the suitable area for the three species is simultaneously suitable for all the three species, it is arduous that one of the three considered species could play the role of umbrella species for the other two. Indeed, from one hand the areas suitable for the more specialists Hazel Dormouse and Roe Deer cover a very small part of those suitable for the European Badger (7% and 16% respectively), but from the other hand, the areas suitable for this last species go excessively beyond the borders of those suitable for the Hazel Dormouse and the Roe Deer.

Considering the grater overlap between the suitable areas for the Hazel Dormouse and the Roe Deer, we investigated if one of these two species could be used as umbrella species for the other. However, it resulted that the areas suitable for the Roe Deer cover only the 47% of those suitable for the Hazel Dormouse, leaving out all the small areas scattered in the matrix that are certainly crucial for the maintenance of the metapopulation of this small mammal. On the other hand, the suitable areas for the Hazel Dormouse cover only the 24% of those suitable for the Roe Deer, leaving out most of the potential range of the ungulate within the study area.

In conclusion, all these considerations demonstrated that, in order to identify areas where it would be necessary to mitigate the effects of fragmentation by setting up management actions, it is crucial to adopt a multi-species approach considering species with different specialization degrees but also with different dispersal ability in the unsuitable matrix.

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Chapter 5

How to manage hedgerows as functional elements in an ecological network for forest mammals: a two-species approach

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Abstract and links to other chapters

The application of the method to design ecological networks comparing simulated species landscape perceptions, described in Chapter 3, on three mammal target species, revealed the key role of hedgerows in ecological networks for the conservation of forest-dwelling species, particularly for the Hazel Dormouse (Chapter 3) and the European Badger (Chapter 4). Thus, in this chapter the internal characteristics which make hedgerows suitable for these two species were evaluated in order to provide concrete suggestions for the correct management of these landscape elements. Indeed, hedgerows vary dramatically in their internal structure and quality and their effectiveness as ecological networks' elements depends both on their physical features, such as width and continuity, and internal habitat conditions. Moreover, the ecological requirements related to hedgerow structure are strongly species-specific. In this study, we evaluated which characteristics make a hedgerow suitable for the European Badger and the Hazel Dormouse, two mammal species characterized by different ecological requirements and sensitive to forest fragmentation at two very different spatial scales. The study was carried out in a wide lowland area of northern Italy, where, following a stratified cluster sampling design, 55 hedgerows were surveyed. For each hedgerow, we collected both structural and floristic variables and we evaluated how differently they affect hedgerows use by the European Badger and Hazel Dormouse. Our results suggested that, in order to simultaneously increase landscape connectivity for both mammal species, hedgerows should be wide and continuous. Moreover, they should be managed to allow the growth of native species with a complex physical structure in the shrub layer and to promote shrubs development by preventing an excessive tree canopy closure. Part of the information obtained with this study have been used to correctly setting up the landscape resistance map for the Hazel Dormouse in Chapter 6.

5.1 Introduction

Fragmentation alters the structure and dynamic of populations, subdividing them into smaller and isolated sub-populations, and making them particularly sensitive to the negative effects of genetic, demographic and environmental stochasticity (Gilpin and Soulé 1986; Fahrig 2003; Fischer and Lindenmayer 2007). This is particularly noticeable in European lowland areas, where natural or semi-natural elements suitable for wildlife, such as woodlands and hedgerows, dramatically decreased during the past decades due to the wide conversion of original habitats to intensively cultivated areas (Arnold 1983; Darby 1956; Williams 2002). One of the management tools designed to mitigate the negative effects of forest fragmentation consists of ecological corridors, i.e. linear structures that could restore and enhance the connectivity among forest remnants (Forman and Gordon 1986; Šálek et al. 2009). In particular, they should facilitate the gene flow through sub-populations enabling individual dispersal (Červinka et al. 2013; Mech and Hallet 2001). In lowland agro-ecosystems, ecological corridors are traditionally represented by hedgerows, defined as continuous or closely spaced lines of shrubs and trees. Although the effectiveness of hedgerows and other ecological corridors in mitigating the effect of forest fragmentation has been widely debated over the past two decades (Davies and Pullin 2007; Tattersall et al. 2002), some authors clearly pointed out their importance in providing shelter, breeding sites and food resources for wildlife (Bennett 2003; Hilty and Merenlender 2004). In particular, several studies highlighted that hedgerows could represent not only effective ecological corridors, but also suitable habitats for different species, especially for birds and small mammals (Arnold 1983; Gelling et al. 2007; Hinsley and Bellamy 2000; Laurance and Laurance 1999; Silva and Prince 2008; Wolton 2009). Moreover, recent studies have demonstrated that hedgerows are also preferentially used respect to the agricultural matrix, both as corridors or habitats suitable for feeding and breeding (Hilty and Merenlender 2004), by several carnivore species, in particular by mustelids (Červinka et al. 2013; Šálek et al. 2009). However, hedgerows can have different origins and structures. They may be residuals of native woodlands or new plantations, and they may suffer different management strategies. This leads to a dramatic dissimilarity in their internal structure and quality (Gelling et al. 2007) and, thus, in their suitability for wildlife. Indeed, the effectiveness of hedgerows as ecological corridors or suitable habitats depends on their structural features, such as width, continuity and internal habitat conditions (Šálek et al. 2009, Silva and Prince 2008). For examples, poor-quality and discontinuous hedgerows proved detrimental to some farmland birds (Hinsley and Bellamy 2000) and small mammals (Bright 1998), while narrow hedgerows without trees seem to be unsuitable for carnivores (Hilty and Merenlender 2004). Therefore, it is important to define which structural features should be preserved in order to ensure the effectiveness of hedgerows in providing additional suitable habitat and in restoring or enhancing landscape connectivity. Since the ecological requirements related to hedgerow structure are strongly species-specific, it would be appropriate, where possible, to apply a multi-species approach.

In this study, we evaluated which internal characteristics make a hedgerow suitable for two mammal species particularly sensitive to fragmentation in northern Italy, the European Badger (*Meles meles*) and the Hazel Dormouse (*Muscardinus avellanarius*). The European Badger is a widespread species considered as least concern by IUCN and it is not so sensitive to fragmentation in areas where agro-ecosystems are composed of forest remnants and open areas (e.g. pasturelands) which form mosaics that may even be favourable to the species (Kruuk 1989). However, this species is strongly negatively affected by forest loss and fragmentation where the agricultural matrix is characterized by intensive crops (Virgós 2001, 2002a, 2002b). On the other hand, the Hazel Dormouse is a protected species included in the Annex IV of the Habitats Directive (Council Directive 92/43/EEC) and it is widely recognized as a species particularly sensitive to habitat loss and to the disruption of connectivity (Mortelliti et al. 2014). We chose to investigate the ecological requirements of these two species, in terms of structural and floristic characteristics of hedgerows, because of their sensitivity to forest fragmentation at two very different spatial scales. Indeed, the dispersal magnitude of these two species through unsuitable habitats is strongly different (European Badger: 500 m - 8 km, Kowalczyk et al. 2006, Macdonald and Barret 1993; Hazel Dormouse: 150 m - 300 m, Mortelliti et al. 2012). Moreover, it is well known that both species are particularly linked to hedgerows (Ehlers 2012; Elliot et al. 2015; O'Brien et al. 2016; Wolton 2009). Within our study area, it was demonstrated that they perceive hedgerows not only as ecological corridors but also as preferential habitats (Chapter 3 and 4). The information provided by this two-species approach could generate important suggestions for favorable management of hedgerows, which might guarantee the conservation of the two target species and, virtually, of any other species with analogous ecological requirements and that responds to fragmentation at similar spatial scales.

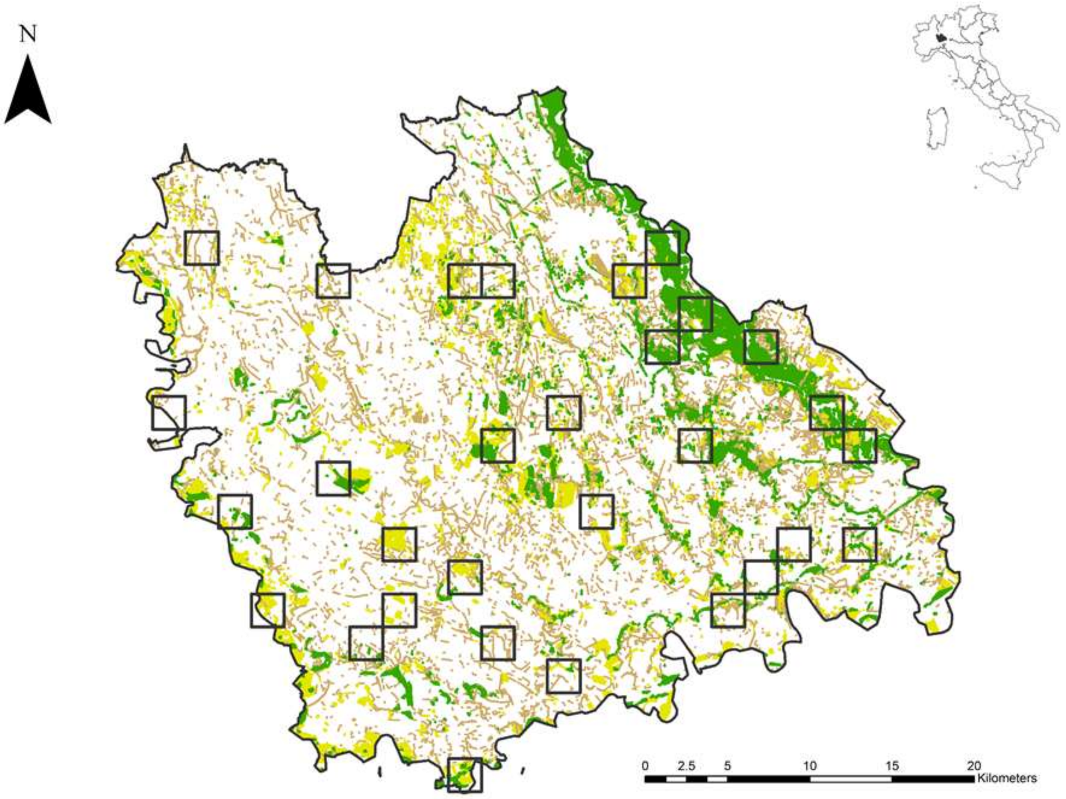
5.2 Methods

5.2.1 Study area and sampling design

The study area and the sampling design followed for data collection (Fig. 1) are described in detail in Chapter 3. Specifically, for this study, we considered every hedgerow falling within a secondary sampling unit, i.e. a 250-m cell. If two hedgerows were contiguous, we considered them as the same hedgerow, for a total of 55 hedgerows within the study area.

Figure 1

Study area in northern Italy (45°21’ N 8°80’ E). The brown color represents hedgerows, the dark green color shows the original broadleaved forest remnants, the light green color represents arboreal plantations. The black squares are the 2-km primary sampling units, each of which includes six secondary sampling units.



5.2.2 Environmental variables

In order to evaluate which internal characteristics of hedgerows would make them suitable for the European Badger and the Hazel Dormouse, we collected both structural and floristic variables. For each hedgerow, we considered physical continuity, average width (calculated averaging three measures taken at both ends and in the middle point of the hedgerow portion included into the secondary sampling unit), average diameter of 20 trees randomly selected within the hedgerow, average shrubs height, percentage of shrub cover, average grass height and percentage of grass cover as structural variables. The last four variables were visually evaluated by the same person (O.D.) to avoid differences in the estimates (Červinka et al. 2013). Moreover, in order to characterize each hedgerow from a floristic point of view, we identified all shrub and tree species, within the hedgerow portion included into the secondary sampling unit. Each hedgerow was visited in spring during two consecutive years (March-July 2014 and 2015), and the environmental variables were measured during the first year of data collection.

5.2.3 Mammal data collection

In order to evaluate the use of hedgerows by the European Badger, we collected data between April and July in 2014 and 2015 performing surveys along linear transects (Krebs 1999). We walked alongside each hedgerow and noted all signs of species presence (i.e. latrines, setts, and footprints) (Sadlier et al. 2004). A hedgerow was considered used by the European Badger if any signs of presence were detected on at least one visit. In order to detect the presence and abundance of the Hazel Dormouse, we placed nest-tubes in every hedgerow. In order to keep a constant sampling effort, we used a number of nest-tubes proportional to hedgerow length within each secondary sampling unit, for a maximum of eight nest-tubes per cell. The nest-tubes were placed 50 m apart from each other, in order to include each of them in an individual's home-range (Juškaitis 2008, 1997; Mortelliti et al. 2011). We placed 173 nest-tubes in 38 secondary sampling units, which were inspected during European Badger surveys. A nest-tube was considered used if we observed any individuals inside it on at least one visit, or if we found any nests or feeding signs on the hazelnuts that we left to attract dormice. The relative abundance of the Hazel Dormouse within each hedgerow was calculated as the maximum number of simultaneously occupied nest-tubes detected during the first or the second survey year, and it was related to the total number of nest-tubes placed within that hedgerow. During the survey period, four hedgerows were completely or largely cut. In those cases, we considered only the data collected during the first year for both species.

5.2.4 Statistical analyses

We considered the 55 hedgerows as statistical units for our analyses on both the European Badger and the Hazel Dormouse. All analyses were performed using R v. 3.1 (R Core Team 2014). For the European Badger, we performed a presence model. We considered as independent variables only those concerning hedgerow structure, thus excluding the categorical variable describing the physical continuity of hedgerows, as it is unlikely that this variable might affect the use of a hedgerow by a species that usually crosses open areas. In order to account for the possible effect of the fragmentation degree of the landscape in which each hedgerow was located (Červinka et al. 2013), we also considered three other variables, i.e. the total surface occupied by hedgerows in a 2-km buffer surrounding each hedgerow, the abundance of suitable habitat (i.e. woodlands, poplar cultivations and hedgerows) in hectares, and the Connectance Index (Fragstats 4; McGarigal et al. 2002) calculated on those landscape elements (i.e. woodlands, poplar cultivations, hedgerows and plantation for biomass production) used as connectivity elements by the European Badger (Chapter 4). The value of the last two variables for each hedgerow was calculated using a moving window with a 2-km radius (a buffer area that moves from pixel to pixel in the whole landscape) and averaging the values calculated within each primary sampling unit. The mean and ranges of all the environmental variables considered for the European Badger presence model are listed in Table 1. As we collected data following a hierarchical sampling design (Crawley 2007), we ran a Generalized Linear Mixed Model (GLMM) with a binomial error distribution by using the *lme4* package (Bates et al. 2012). We used the presence/absence of the European Badger as the response variable, the primary and secondary sampling units as nested random effects, and the environmental variables as fixed. However, since the random effects explained a very low percentage of the variance of the dependent variable, we removed them and re-ran the model as a GLM with a binomial error distribution. As the Moran's I Test performed with 999 permutations (Cliff and Ord 1981) using the *spdep* package (Bivand et al. 2005) revealed spatial autocorrelation in both the response variable and model residuals, we added the spatial coordinates of the centroid of each hedgerow to the model as covariates. In this way, we were able to remove the residuals spatial autocorrelation, as confirmed by the following Moran's I Test. All continuous independent variables were standardized using an autoscaling procedure. We selected the variables following an Information-Theoretic Approach by means of the Akaike Information Criterion (AIC) (Burnham and Anderson 2002). We considered only the best model, i.e. the model with the lowest AIC. The explanatory power of the best model was evaluated by means of the explained deviance D^2 (Boyce et al. 2002), and its ability to distinguish between occupied and unoccupied hedgerows was tested using the area under the curve of the Receiver Operating Characteristic plot (ROC curve) (Fawcett 2006; Pearce and Ferrier 2000) by using the *verification* package (NCAR - Research Applications Laboratory, 2014).

For the Hazel Dormouse, we performed both a presence and abundance model. In both cases, we considered both structural and floristic features as independent variables, as we hypothesized that floristic characteristics could have a great influence on a species which typically feeds on berries and builds nests using vegetal products, such as moss and leaves (Juškaitis 2008). We also took into account the total surface occupied by hedgerows within a 250-m buffer (considering continuous and discontinuous hedgerows separately), the amount of suitable habitat (i.e. woodlands and hedgerows) in hectares, and the Connectance Index calculated on those landscape elements (i.e. woodlands, hedgerows, poplar cultivations, plantations for biomass production and reforestations) used as connectivity elements by the Hazel Dormouse (Chapter 3). The last two variables were calculated using a moving window with a 250-m radius and averaging the values calculated in correspondence to the nest-tubes located in the same hedgerow. The mean and ranges of all the environmental variables considered for the Hazel Dormouse presence and abundance models are listed in Table 1. In order to perform the presence model, we ran a GLMM following the same procedure used for the European Badger, however since the random effects did not explain any percentage of the variance of the dependent variable, we applyed a GLM without considering any random effect. We excluded the presence of spatial autocorrelation in the Hazel Dormouse presence data and model's residuals by means of the Moran's I Test, and we performed the model selection, evaluation and validation following the same procedure used for the European Badger presence model.

Finally, in order to perform the abundance model for the Hazel Dormouse, we modelled the proportion of occupied nest-tubes, considered as a relative index of abundance. Once again, we did not run a GLMM, since the random effects did not explain any percentage of the dependent variable variance, and we applied a GLM using a two-vector response variable (occupied versus non-occupied nest-tubes placed in the hedgerow) with a binomial error distribution, without taking into account any random effect. We excluded the presence of spatial autocorrelation in the Hazel Dormouse abundance data and in the model's residuals by means of the Moran's I Test and we performed the model selection and evaluation following the same procedure used for the European Badger and Hazel Dormouse presence model.

For the best models including both plot variables (i.e. structural and floristic variables) and landscape variables (the amount of suitable habitat, the Connectance Index and the spatial coordinates of the centroid of each hedgerow), we used variance decomposition (using the ''varpart'' function in the *vegan* package in R; Oksanen et al. 2013) to assess the amount of variance explained independently and jointly by the two groups of covariates (i.e. plot and landscape variables). For the three best models, we checked for residual normality by means of the Kolmogorov-Smirnov test (Legendre and Legendre 1998) and we checked for variables collinearity by means of the Variance Inflation Factor (VIF), by the *usdm* package (Naimi 2015). For the two presence models, we checked for residuals independence using the Durbin-Watson test (Crawley 1993; Savin and White 1977) by the *lmtest* package (Zeileis and Hothorn 2002), while for the abundance model we just checked diagnostic plots, as the abovementioned test do not support weighted regressions.

Table 1

Environmental variables considered to evaluate the use of hedgerows by the European Badger and Hazel Dormouse in northern Italy.

Environmental variables	Species for which the variable influence was considered	Variable type	Mean	Min-Max
<u>Hedgerow structure</u>				
Continuity	Hazel Dormouse	Categorical	-	0-1
Width	Both	Continuous	7 m	1-30 m
Average diameter of 20 trees	Both	Continuous	17 cm	7-66 cm
Average shrubs height	Both	Continuous	186 cm	0-300 cm
Percentage of shrub cover	Both	Continuous	56 %	0-100 %
Average grass height	Both	Continuous	48 cm	0-170 cm
Percentage of grass cover	Both	Continuous	80 %	0-100 %
<u>Shrub layer</u>				
Alnus glutinosa	Hazel Dormouse	Categorical	-	0-1
Corylus avellana	Hazel Dormouse	Categorical	-	0-1
Crataegus monogyna	Hazel Dormouse	Categorical	-	0-1
Populus sp.	Hazel Dormouse	Categorical	-	0-1
Robinia pseudoacacia	Hazel Dormouse	Categorical	-	0-1
Rubus ulmifolius	Hazel Dormouse	Categorical	-	0-1
Sambucus nigra	Hazel Dormouse	Categorical	-	0-1
<u>Tree layer</u>				
Alnus glutinosa	Hazel Dormouse	Categorical	-	0-1
Populus sp.	Hazel Dormouse	Categorical	-	0-1
Quercus robur	Hazel Dormouse	Categorical	-	0-1
Robinia pseudoacacia	Hazel Dormouse	Categorical	-	0-1
Salix sp.	Hazel Dormouse	Categorical	-	0-1
<u>Landscape context</u>				
Total hedgerows cover (buffer 2 km)	European Badger	Continuous	1 ha	0-3 ha
Continuous hedgerows cover (buffer 250 m)	Hazel Dormouse	Continuous	0.5 ha	0-1.8 ha
Discontinuous hedgerows cover (buffer 250 m)	Hazel Dormouse	Continuous	0.4 ha	0-2 ha
Suitable habitat amount (buffer 2 km)	European Badger	Continuous	212 ha	47-659 ha
Suitable habitat amount (buffer 250 m)	Hazel Dormouse	Continuous	2 ha	0.3-9 ha
Connectance Index (buffer 2 km)	European Badger	Continuous	65 %	58-78 %
Connectance Index (buffer 250 m)	Hazel Dormouse	Continuous	53 %	0-100 %

5.3 Results

5.3.1 European Badger presence model

Overall, 18 out of 55 hedgerows were found to be used by the European Badger within the study area. The best presence model selected by the Information-Theoretic Approach explained 39% of the deviance (Table 2). The model highlighted a significant positive effect of hedgerow width (Fig. 2a) on the probability of occurrence of the European Badger, while the percentage of grass cover played a negative effect (Fig. 2b). The analyses also showed a negative effect of the hedgerows centroids longitude. The ROC plot analysis showed that the discriminatory ability of the model was good (AUC=0.872, p<0.001). Model variance decomposition analysis indicated that the total variance explained by covariates in the best model is equally distributed between the plot and landscape group of variables (Fig. 3a). Model residuals were normally distributed (Kolmogorov-Smirnov test, D=0.166, p=0.097), independent (Durbin-Watson test, DW=2.30, p=0.779) and there was no collinearity between variables (VIF<3; Zuur et al. 2009).

Table 2

Best presence model for the use of hedgerows by the European Badger in northern Italy. East: longitude (UTM, WGS84_32N) of each hedgerow centroid. The model explained 39% of the null deviance.

Predictors	Estimate	SE	Odds ratio	z	Pr(>|z|)
(Intercept)	-1.335	0.475	-	-2.808	0.005
Width	2.496	0.869	12.134	2.872	0.004
Percentage of grass cover	-1.878	0.797	0.153	-2.357	0.018
Total hedgerows cover (buffer 2-	0.712	0.453	2.038	1.574	0.115
East	-3.208	1.136	0.040	-2.825	0.005

5.3.2 Hazel Dormouse presence model

Overall, 21 out of 55 hedgerows were found to be used by the Hazel Dormouse within the study area. The best presence model selected by the Information-Theoretic Approach explained 21% of the deviance (Table 3). The most important variables among those selected in the best model were the percentage of shrub cover (Fig. 2c) and the total surface occupied by continuous hedgerows within a 250-m buffer (Fig. 2d), both with a positive effect on the probability of occurrence of the Hazel Dormouse, and the presence of oaks, with a negative effect. The discriminatory ability of the model was good (AUC=0.809, p<0.001). Model variance decomposition analysis indicated that the independent effect of plot variables accounted for a large part of the total variance explained in the best model, with the remaining variance approximately equally distributed between the independent effect of landscape variables and the effect shared between the two variables' groups (Fig. 3b). Model residuals were normally distributed (Kolmogorov-Smirnov test, D=0.155, p=0.147), independent (Durbin-Watson test, DW=1.80, p=0.194) and there was no collinearity between variables (VIF<3; Zuur et al. 2009).

Table 3

Best presence model for the use of hedgerows by the Hazel Dormouse in northern Italy. The model explained 21% of the null deviance.

Predictors	Estimate	SE	Odds ratio	z	Pr(>|z|)
(Intercept)	0.649	0.843	-	0.770	0.441
Poplar spp._Tree layer	1.593	1.145	4.918	1.390	0.164
Oak spp._Tree layer	-2.477	1.328	0.084	-1.865	0.062
Locust tree_Shrub layer	-1.354	0.929	0.258	-1.457	0.145
Percentage of shrub cover	0.898	0.414	2.455	2.173	0.029
Continuous hedgerows cover (buffer 250 m)	0.765	0.446	2.149	1.714	0.086
Discontinuous hedgerows cover (buffer 250 m)	-0.605	0.469	0.546	-1.291	0.197

5.3.3 Hazel Dormouse abundance model

The Hazel Dormouse relative abundance within the occupied hedgerows ranged from 1 to 3 nest-tubes (mean 1.33 ± 0.12). The best abundance model explained 28% of the deviance (Table 4). The most important variables in the model with a positive effect were the percentage of shrub cover (Fig. 2e), the presence of poplars in the tree layer and the presence of common alders (*Alnus glutinosa*) in the shrub layer. Conversely, the presence of oaks in the tree layer and the presence of poplars in the shrub layer had a negative effect on the abundance of the species. Even in this case, model residuals were normally distributed (Kolmogorov-Smirnov test, D=0.135, p=0.276) and the collinearity between the selected variables was negligible (VIF<4).

Table 4

Best abundance model for the use of hedgerows by the Hazel Dormouse in northern Italy. The model explained 28% of the null deviance.

Predictors	Estimate	SE	Odds ratio	z	Pr(>|z|)
(Intercept)	-0.550	0.636	-	-0.865	0.387
Discontinuity	-1.189	0.828	0.305	-1.435	0.151
Poplar spp._Tree layer	1.767	0.908	5.853	1.945	0.052
Oak spp._Tree layer	-1.902	0.848	0.149	-2.244	0.025
Locust tree_Shrub layer	-1.122	0.704	0.326	-1.594	0.111
Common alder_Shrub layer	2.082	0.951	8.020	2.189	0.029
Poplar spp._Shrub layer	-2.720	1.241	0.066	-2.191	0.028
Percentage of shrub cover	0.661	0.307	1.937	2.154	0.031

Figure 2

Scatterplots showing the effects of hedgerow structural covariates on the European Badger presence probability (a, b) and the Hazel Dormouse presence probability (c, d) and relative abundance, measured as the ratio of the number of occupied nest-tubes to the total number of nest-tubes placed in a hedgerow (e). Solid lines represent the effect of a covariate obtained from GLMs when all the other independent variables equal their average value. Dotted lines are the 95% confidence intervals. Black dots show the observed data (overlapping data are present).

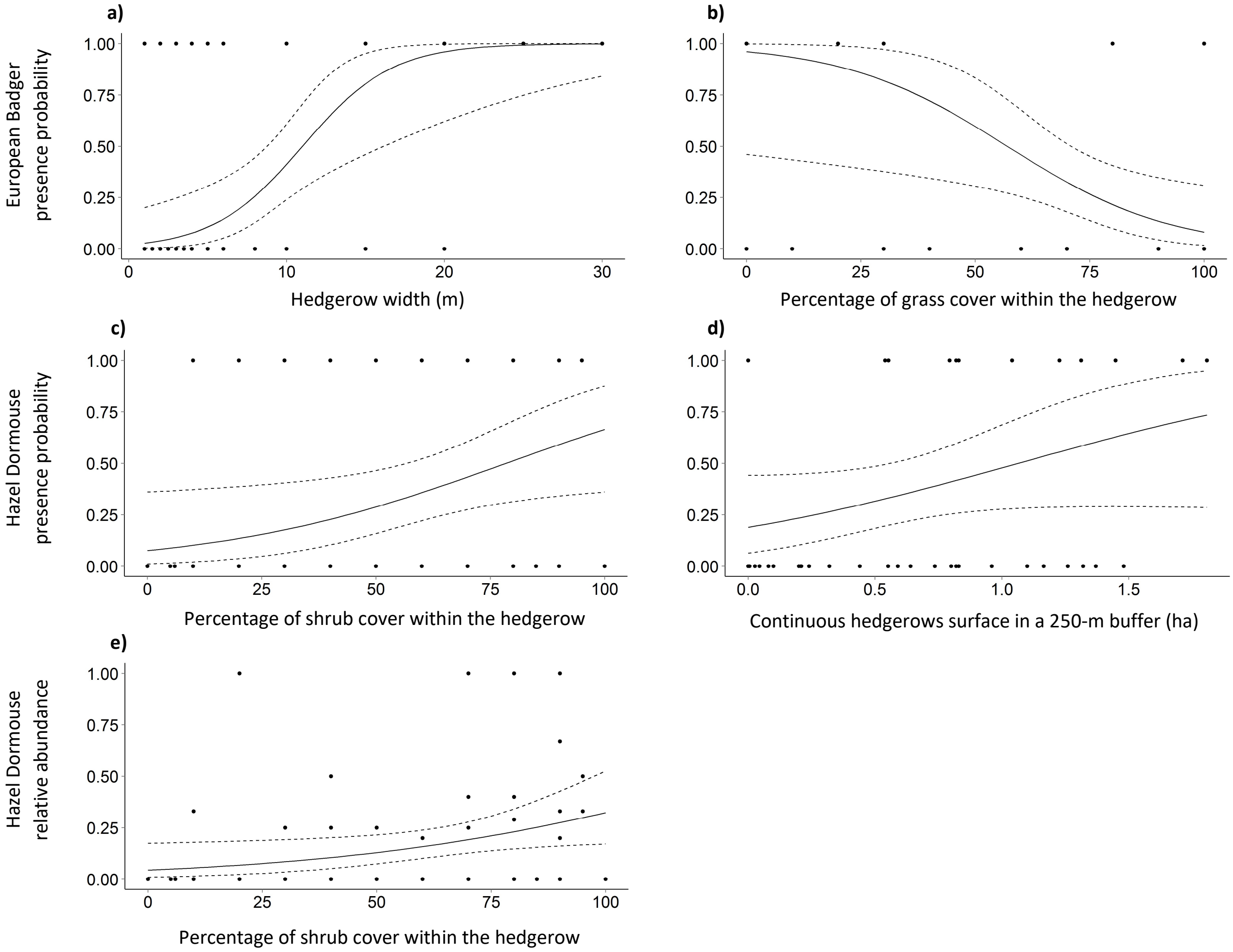
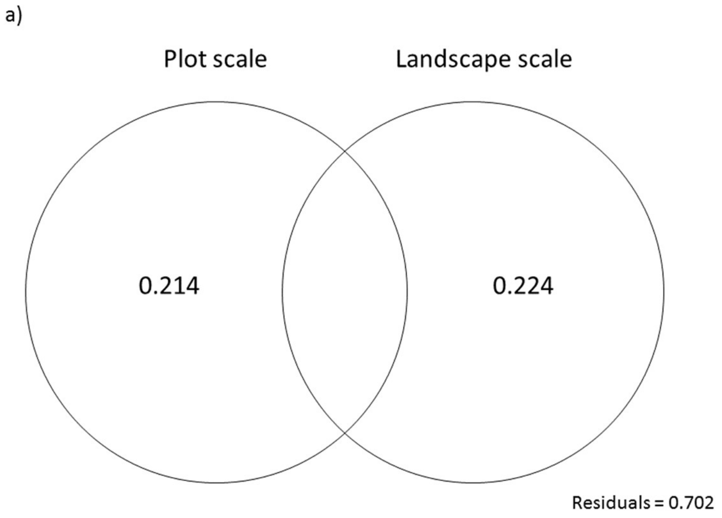


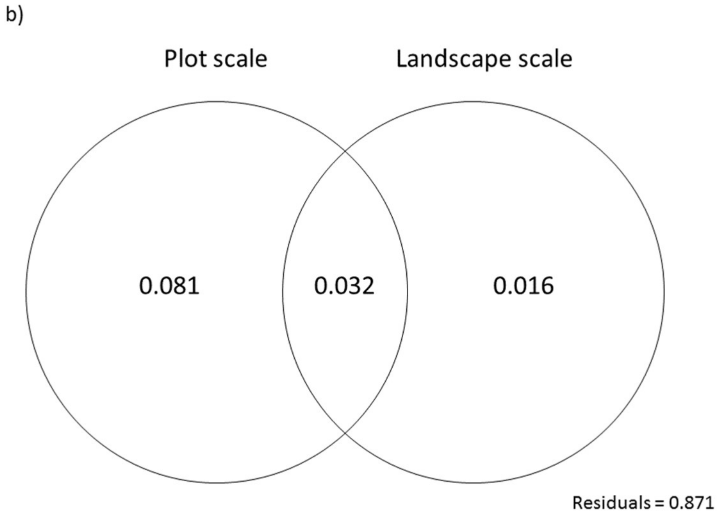
Figure 3

Model variance decomposition analyses showing proportions of variance explained by plot-scale and landscape-scale variables (values<0 not shown) in the European Badger (a) and the Hazel Dormouse (b) presence best models.

European Badger presence



Hazel Dormouse presence



5.4 Discussion

In agricultural landscapes, hedgerows play an important ecological role for several mammal species, both as corridors and as additional reproductive habitats (Červinka et al. 2013; Gelling et al. 2007; Laurance and Laurance 1999; Šálek et al. 2009; Silva and Prince 2008). However, the effectiveness of hedgerows as ecological corridors or reproductive habitats depends on their internal features, such as floristic and structural characteristics (Šálek et al. 2009; Silva and Prince 2008). The influence of the internal characteristics of hedgerows in determining their use is strongly species-specific and, thus, the application of a multi-species approach could allow obtaining more comprehensive information. In this study, we analyzed which hedgerows characteristics facilitate or discourage their use by the European Badger and the Hazel Dormouse.

According to our analyses, width is the most important structural characteristic that makes a hedgerow suitable for the European Badger. The same result was obtained in other studies regarding both wildlife in general (Hilty et al. 2006), and small and medium-sized carnivores in particular (Hilty and Merenlender 2004). Indeed, wider corridors may have several diverse microhabitat structures, fulfilling more species-specific ecological requirements (Hilty and Merenlender 2004). For the European Badger, hedgerows are important in providing shelter (O'Brien et al. 2016) and food (Gelling et al. 2007; Thomas and Marshall 1999) and, obviously, a wider hedgerow is safer and richer of trophic resources than a narrow one. Our results also suggest that the European Badger avoids hedgerows with a high percentage of grass cover. This finding could be linked to the avoidance of hedgerows completely lacking a shrub cover, and, thus, of hedgerows characterized by high percentages of grass cover, which do not ensure shelter for the species. Indeed, even though shrub cover was not found to be significant in our analyses, probably because an excessive shrub development has negative effect on the European Badger, a certain degree of shrub cover is a very important factor in determining the use of hedgerows as corridors by carnivores (Mangas et al. 2008). Moreover, the analyses pointed out that the hedgerows importance for the European Badger have a positive gradient from East to West. This result can be explained considering that hedgerows become more important habitats for the European Badger where the forest cover decreases and the agricultural matrix progressively prevails (O'Brien et al. 2016). In our study area, forest cover decreases from East to West, moving away from the continuous forests of the Ticino Natural Park (Fig. 1).

As regards the Hazel Dormouse, the presence model revealed a positive effect of the cover of continuous hedgerows within a 250-m buffer on presence probability of the species. This finding highlighted the importance of physical continuity in making hedgerows suitable for the Hazel Dormouse. Indeed, discontinuous hedgerows have been proved to be detrimental for several species (Gelling et al. 2007). Specifically, radio-tracking studies carried out in UK by Bright (1998) showed a significant difference in the crossing frequency of different size gaps within hedgerows by the Hazel Dormouse. In particular, one meter gaps were crossed in 55% of the cases, three meter gaps only in 6%, while gaps over six meters were never crossed during the study period. Both the presence and abundance models clearly highlighted the strong importance of shrub cover in determining the suitability of a hedgerow for the species. This probably depend on the fact that a high percentage of shrub cover could play an important role in satisfying the ecological requirements of the Hazel Dormouse (Bright et al. 1994; Juškaitis and Šiožinytê 2008; Ramakers et al. 2014). Indeed, this species needs safe shelters where nests can be built away from predators (Bright 1998; Bright and Morris 1991), vegetal materials for nest construction (Berg and Berg 1998; Wolton 2009) and food resources, such as berries, seeds and insects (Bright and Morris 1992, 1991, 1990). These requirements can only be found in a hedgerow with a well-developed shrub layer, and the higher the percentage of shrub cover is, the larger is the Hazel Dormouse population that the hedgerow can support. Moreover, the analyses showed that the presence of oaks within a hedgerow negatively affects both the probability of presence and the abundance of the Hazel Dormouse. Indeed, both the Pedunculate Oak (*Quercus robur*) and the Sessile Oak (*Quercus petraea*) are tree species with an average height ranging between 30 and 35 meters and a wide canopy and hedgerows hosting these species are often characterized by a poorly developed shrub layer due to the excessive shading provided by trees. The abundance model clearly highlighted the importance of hedgerows floristic characteristics on the Hazel Dormouse population size. First, the Hazel Dormouse seems to prefer poplars rather than oaks in the tree layer, indeed poplars are typical of new-planted hedgerows where trees are often placed at a certain distance from each other. This leads to a not excessive canopy closure allowing the development of the shrub layer. However, the model also showed the negative effect of poplars when they characterize the shrub layer. The presence of poplars probably determines a too simple structure of the shrub layer preventing the hedgerows of supporting high Hazel Dormouse densities. Indeed, higher population abundance of this species are generally associated with a more complex structure of the shrub layer (Sozio et al. 2014), that is provided by the presence of other species. The abundance model, for instance, highlighted the importance of common alders in the shrub layer in determine higher Hazel Dormouse densities. The common alder is a native tree species that typically grows in wet areas and that is often used in reforestations in order to provide a complex physical structure of both the shrub and tree layers.

5.5 Management implications

In this study, we evaluated which hedgerow characteristics facilitate or discourage their use by two mammal species particularly sensitive to fragmentation, which respond to this phenomenon at two different spatial scales: the Hazel Dormouse and the European Badger. In order to identify proper management measures aimed at making hedgerows suitable for a larger part of the entire community, it is fundamental to consider species responding to different spatial scales. In particular, we identified some management practices that should be applied in order to make a hedgerow an effective ecological corridor for the European Badger, the Hazel Dormouse, and any other species with similar ecological requirements and that responds to forest fragmentation at similar spatial scales. The fact that, overall, only eight out of 18 and 21 hedgerows occupied by the European Badger and Hazel Dormouse, respectively, were simultaneously occupied by both species, suggests that these two mammals have different ecological requirements and that the current management of hedgerows in the study area is only partially adequate to make them suitable for both species.

Based on our results, we suggest that, in lowland areas characterized by a high degree of forest fragmentation, hedgerows should be kept as wide as possible. This is consistent with the findings of Hilty and Merendlener (2004), who suggested that, in agricultural landscapes, the maintenance of wider hedgerows is crucial in order to protect more wildlife species with different ecological requirements. It is interesting to point out that if we had used only the Hazel Dormouse as a target species, this result would not have emerged. On the other hand, considering the Hazel Dormouse allowed us to highlight another crucial factor, i.e. hedgerow continuity, and the importance to fill hedgerow gaps with trees and shrubs. Hedgerows should also be managed in order to increase the shrub layer cover. A well developed shrub layer can be maintained avoiding an excessive tree canopy closure, that has a negative effect on the Hazel Dormouse. For this reason, we suggest to control the development of the tree cover in new hedgerows by planting large tree species not too close one to each other. Also a direct management of the shrub layer would be advisable. Particularly, it is important preventing the spread of species that provide a simple structure of the shrub layer, such as poplars, in favor of native species characterized by a more complex physical structure, such as common alders.

From our knowledge, this is the first study that investigated which hedgerows characteristics are selected by the European Badger and Hazel Dormouse in a highly intensive agricultural landscape in northern Italy. The management guidelines we provided could be particularly important in order to conserve wildlife in highly modified European landscapes. Indeed, the population viability of many animal species traditionally present in European agricultural landscapes depends upon the availability of hedgerows with internal characteristics suitable to provide shelter, breeding sites, refuge and foraging habitat (Bennett 2003). Finally, hedgerows characteristics that resulted particularly important for wildlife conservation in this study (e.g. hedgerow width and continuity), would also have a key role in providing ecological functions that are crucial for agriculture services, such as preventing soil erosion, flooding and wind damage to crops (Baudry and Burel 1984).

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Chapter 6

Priority areas to plant new hedgerows to enhance functional connectivity for the Hazel Dormouse in fragmented landscapes

Abstract and links to other chapters

The study presented in this chapter was carried out in collaboration with the Ecogesfor (Ecologia y gestión forestal sostenible) group of the Department of Forest and Environmental Engineering and Management of the Polytechnic University of Madrid.

As the study in Chapter 3 showed the key role of hedgerows in ecological networks for the Hazel Dormouse, in this chapter different analytical strategies to identify priority sites where new hedgerows should be located in order to enhance the landscape connectivity for this species were compared. The study was carried out in a 140 km^2 study area in Lombardy Region (northern Italy). By applying a graph theory approach, the increase of connectivity reached by using the traditional strategy of habitat restoration along existing least-cost paths (LCPs) (strategy 1) and an alternative strategy, which identifies new strategic areas where hedgerows should be located (strategy 2), were compared. Starting from the information obtained in Chapter 3 and 5, a resistance surface for the Hazel Dormouse was set up and the LCPs between every pair of nodes were identified. Following the strategy 1, the increase of connectivity provided by each LCP if it was restored was calculated and the restoration of the most important LCPs by planting 30 new hedgerows was simulated. Subsequently, following the strategy 2, which only considers the existing nodes in the landscape and disregards the LCPs connecting them, the implementation of every possible connection between every pair of nodes was simulated and the importance of every new connection in enhancing connectivity was calculated. Then, the restoration of the most important new connections by locating 30 new hedgerows was simulated. By comparing the increase of connectivity reached through the two strategies, it resulted that implementing new connections guarantees an increase of connectivity about four times greater than the increase obtained by the traditional restoration of existing LCPs. Starting from the best strategy (strategy 2), 15 priority hedgerows in increasing connectivity were selected adopting two different methods (iteratively estimating the percentage of connectivity variation by removing or adding each individual hedgerow to the original landscape) and the increase of overall landscape connectivity provided by their plantation was calculated. The increase of connectivity obtained with this last simulation resulted twice that obtained restoring 30 existing LCPs, demonstrating that with half of the resources, by creating new connections it can be obtained a result two times better than those achievable by restoring existing LCPs.

6.1 Introduction

In fragmented forests, the movement of dispersing individuals through the modified matrix separating forest remnants is hindered or even completely interrupted (Laita et al. 2010; Nogués and Cabarga-Varona 2014), leading to a progressive isolation of the small populations of forest-dwelling species inhabiting the residual patches (Pascual-Hortal and Saura 2006, 2008). The long-term conservation of these species completely depends on the existence of functional connectivity between the spatially separated populations (Fischer and Lindenmayer 2007; Nogués and Cabarga-Varona 2014). Conservation studies generally focused on promoting or increasing connectivity by conserving or restoring areas that currently sustain animal dispersal flows (e.g. Carranza et al 2012; Moqanaki and Cushman 2016) under the present composition and spatial arrangement of the landscape (McRae et al. 2012). These areas are usually identified by mapping the matrix surrounding the residual patches in the form of a resistance surface where each pixel is associated to a value corresponding to the resistance that the land-cover to which it pertains offers to the species movement. Subsequently, paths of minimum cumulative resistance, which correspond to the routes across the landscape matrix where individuals' movements are likely concentrated, are identified between each pair of residual patches by using different tools (e.g. lest-cost corridor modeling, Adriaensen et al. 2003, Beier et al. 2009; circuit theory, McRae et al. 2008). Conserving existing paths of minimum resistance is a good strategy when the species of conservation interest is characterized by a high or intermediate dispersal ability (Saura and Rubio 2010). Indeed, species with a high dispersal ability can move among forest remnants following paths of low resistance, easily crossing gaps of unsuitable habitat often present even along the minimum resistance paths. When the species of conservation concern is characterized by an intermediate dispersal ability, an effective conservation strategy could be the conservation or the restoration of existing low resistance paths strengthening them by placing few small patches of optimal or sub-optimal habitat, playing the role of stepping stones along the paths (Saura and Rubio 2010). On the other hand, species with a very low dispersal ability, typically characterized by a high degree of forest specialization, hardly move among forest remnant even along minimum resistance paths, as they cannot cross unsuitable habitat gaps longer than their dispersal ability in the matrix, that is typically very small. For this reason, some authors suggested that the conservation of these species would be better promoted by increasing the amount and quality of habitats inside residual patches, rather than focusing on promoting dispersal processes (e.g. Gurrutxaga et al. 2011). However, the long-term survival of low dispersal forest-dwelling species in highly fragmented landscapes cannot disregard the maintenance of functional connectivity among residual patches. Indeed, in these landscapes it is arduous that they could persist in residual patches able to maintain populations large enough to avoid detrimental fluctuations of population size, due to genetic (inbreeding and genetic drift) and stochastic demographic factors, that likely lead populations to local extinction. The survival of low dispersal species in highly fragmented landscapes can be thus guaranteed through conservation measures that allow the maintenance of the gene flow among residual patches. In this case, the only effective conservation strategy is the almost complete restoration of the minimum resistance paths using optimal or permeable habitat. Considering that we should locate new habitat to virtually cover the entire paths extent, is it still convenient restoring existing minimum resistance paths or could it be more effective, in terms of connectivity increase, locating new habitat in other strategic areas within the landscape?

Starting from this issue, we carried out a comparative study in a real landscape of northern Italy, focusing on a small forest-dwelling species of conservation concern because of its sensitivity to fragmentation: the Hazel Dormouse *Muscardinus avellanarius* (Mortelliti et al. 2014). In Italy, the optimum habitats for this species are represented by broadleaved forests characterized by a well-developed and diversified shrub layer, that can offer route-ways through branches, suitable sites for building nests and shelter from predators (Panchetti et al. 2007). However, in the Italian highly modified lowland areas, broadleaved forests are fragmented in small residual patches scattered within the agricultural matrix. In these landscapes, the survival of the Hazel Dormouse depends, other than on forest remnants, on semi-natural elements perceived by the species as optimal (hedgerows) or sub-optimal (poplar cultivations, plantations for biomass production and reforestations) habitats (Chapter 3). Different agricultural management plans include funding measures addressed to the plantation of hedgerows, poplar cultivations, plantations for biomass production and reforestations. The last three arboreal plantations generally replace traditional crops for periods ranging from 12 to 20 years. Conversely, hedgerows are permanent continuous or closely spaced lines of shrubs and trees located along crop field borders. Several authors highlighted the crucial role of hedgerows for wildlife conservation in lowland agro-ecosystems (Hinsley and Bellamy 2000; Hilty and Merenlender 2004; Wolton 2009). The importance for wildlife conservation, the existence of an economic support for their plantation, the small surface subtracted to cultivable areas and their characteristics of optimal habitat, make hedgerows ideal elements for management actions designed to increase connectivity for the Hazel Dormouse.

The aim of this study was to evaluate if functional connectivity for the Hazel Dormouse would greater increase by placing new hedgerows (i) along existing paths of minimum resistance or (ii) in other strategic areas not involved in current dispersal flows (i.e. creating new dispersal paths). For this purpose, we adopted a graph-based approach, that is an effective and flexible tool to analyse changes in landscape connectivity incorporating species functional attributes in spatial analyses (Urban and Keitt 2001; Urban et al. 2009; Carranza et al. 2012). Particularly, we realized two maps simulating the plantation of 30 new hedgerows within the investigated landscape (a plausible management effort compared to the extent of the considered study area) first along those existing minimum resistance paths that would confer a greater increase in connectivity if they were restored (management strategy 1) and, second, along those new paths that would confer a greater increase in connectivity if they were created (management strategy 2). Subsequently, we compared the connectivity increase that would be obtained if we implement these two conservation strategies respect to the current value of landscape connectivity (see Fig. 2).

As financial resources for environmental management actions are very often limited, it is crucial to develop and to test methods to identify management priorities among all the possible interventions (Bottrill et al. 2008; Chazdon 2008; Menz et al. 2013). Thus, starting from the most effective management strategy identified through our comparison, we finally compared two different analytical methods, based on the graph theory, in order to identify a number of priority hedgerows corresponding to half of the initial management effort.

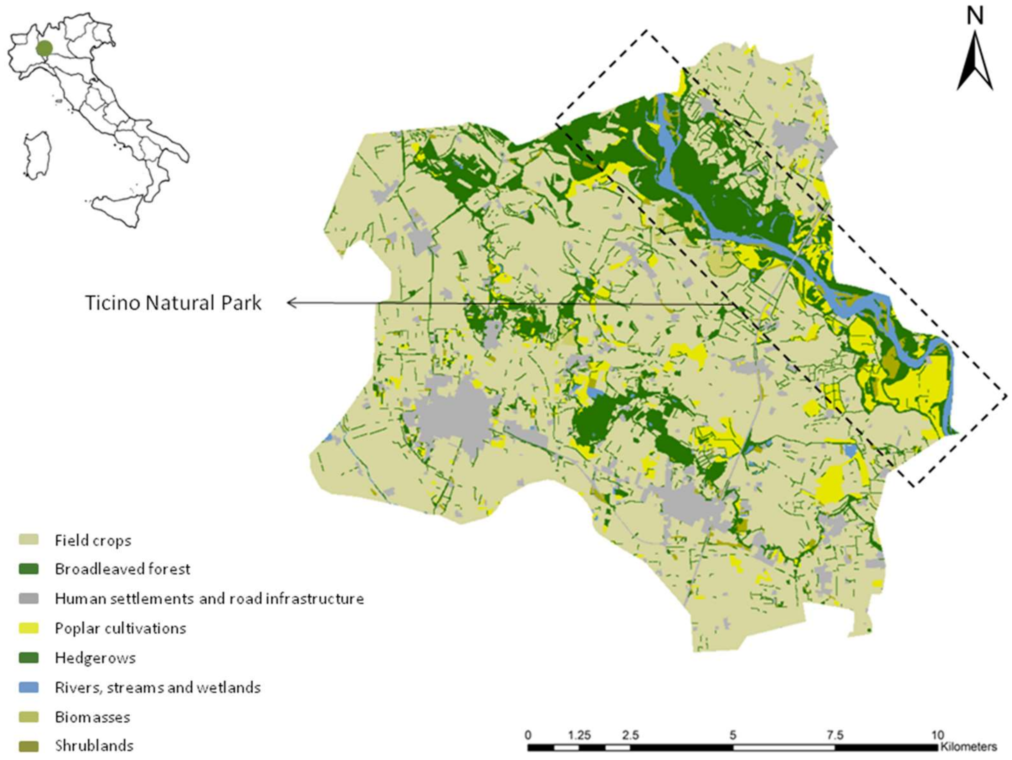
6.2 Methods

6.2.1 Study area

The investigated landscape is an area of about 140 km^2 located in the central part of the lowland area of Lombardy region (northern Italy, 45°25’ N 8°98’ E). It is a typical agro-ecosystem dominated by intensive field crops (63%). The remaining surface of the study area is composed of broadleaved forests (14%), human settlements and road infrastructures (8%), poplar cultivations (6%), hedgerows (4%), rivers, streams and wetlands (3%), plantations for biomass production (1%) and shrublands (1%). Most of the broadleaved forests within the study area follow the Ticino River and fall within the boundaries of the protected area pertaining to the Ticino Natural Park (Fig. 1). The remaining forest surface is reduced in patches of different size, connected to each other to some degree by hedgerows.

Figure 1

Study area in northern Italy (45°25’ N 8°98’ E).



6.2.2 Nodes, landscape resistance and least-cost paths identification

In the investigated landscape, the elements that can play the role of nodes for the Hazel Dormouse are represented by patches of broadleaved forest and by continuous hedgerows (Chapter 3 and Chapter 5), making a total of 1035 nodes. Consequently, the nodes were represented in a map corresponding to the spatial configuration of broadleaved forests and continuous hedgerows within the study area. The raster resistance surface was built up using ArcGis 10.0 (ESRI 2011), starting from the most recent digital land-use cartography available for the study area (DUSAF 4, ERSAF 2014) (Fig. 2b). We adopted a small pixel size (20 m) adequate respect to the size and movement ability of the target species. The pixel values were assigned with an expert-based approach considering the Hazel Dormouse’s perception of the different land-covers constituting the investigated landscape. Lowest resistance values (1) were assigned to nodes (i.e. broadleaved forests and continuous hedgerows) and largest values (1000) to human settlements, road infrastructures and rivers. Such high values were assigned to avoid that least-cost paths (LCPs) would cross human settlements, road infrastructures and rivers (i.e. impermeable barriers for the target species) unless no other possibility of movement existed (Nogués and Cabarga-Varona 2014). Within this range, we assigned lower values (8) to discontinuous hedgerows (Chapter 5) and to poplar cultivations and plantations for biomass production (Chapter 3), an intermediate value (50) to shrublands (Bright et al. 1994; Juškaitis and Šiožinytê 2008; Ramakers et al. 2014), and higher values (500) to field crops (Mortelliti et al. 2013). Starting from the map of nodes and the raster resistance surface, we then identified the LCPs connecting every pair of nodes (Beier et al. 2008). Specifically, we calculated the cost-weighted distance of all pixels to a node, generating a raster of cost-weighted distance values, by using the ArcGis extension Linkage Mapper 1.0.2 (McRae and Kavanagh 2011). Subsequently, adding together the cost-weighted distance rasters from every pair of adjacent nodes, we identified the paths associated to the lowest cumulative movement cost between each pair of nodes (McRae et al. 2012). To identify adjacent nodes, ArcGis creates raster files that associate each pixel to the nearest node in terms of cost-weighted distance creating an allocation zone for each node. If a path from one node to another must pass through the allocation zone of a third node, the two nodes are considered non-adjacent (McRae and Kavanagh 2011). This way, we identified all the LCPs existing between every pair of adjacent nodes and the cumulative resistance encountered moving along each LCP corresponds to the ecological distance among nodes.

6.2.3 Management strategies comparison

6.2.3.1 Identifying priority areas for restoration (strategy 1) or new paths' implementation (strategy 2)

Once defined the nodes and resistance maps and identified the LCPs among every pair of nodes within the investigated landscape, we wanted to evaluate if functional connectivity for the Hazel Dormouse would greater increase by placing hedgerows (i) along existing paths of minimum resistance (strategy 1) or (ii) in other strategic areas not involved in current dispersal flows (strategy 2).

To identify which minimum resistance paths would confer a greater increase of connectivity for the Hazel Dormouse if they were restored (strategy 1), we integrated the least-cost approach and the graph-based theory. Specifically, to prioritize all the identified LCPs according to the increase of connectivity that they would confer if they were restored, we used the command line version 1.0.21 of the software Conefor 2.7.1 (Saura and Torné 2009; Torné and Saura 2013) adopting the Probability of Connectivity (*PC*) index, which is a robust index to detect changes in landscape connectivity (Saura and Pascual-Hortal 2007) (Fig.2c). *PC* is a habitat availability metric, which considers nodes as spaces where connectivity occurs and combines nodes' attribute (such as patch size, quality-weighted patch size, habitat suitability) and their connections (links) in a single index (Pascual-Hortal and Saura 2006). Basing on this index, and using the ecological distance as distance measure, the dispersal probability of an individual moving from one node to another is affected by both the attributes of the source and destination nodes (in our case the size of the source and destination patches) and on the ecological distance of the LCP connecting the two nodes (i.e. the links of the graph) (Carranza et al 2012). Specifically, *PC* is given by:

$$PC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} a_i \cdot a_j \cdot p_{ij}^{*}}{A_L^2}$$

Where a_i and a_j, in our case, correspond to the area of nodes *i* and *j*, respectively; *n* is the total number of nodes in the investigated landscape; and A_L is the total landscape area (considering both nodes and matrix) (for more detail see Saura and Pascual-Hortal 2007). p_{ij} is the probability of dispersal between each pair of adjacent nodes, calculated using a negative exponential function of the ecological distance between them. p_{ij}* corresponds to the maximum product probability of all possible combination of LCPs that can be crossed to reach the node *j* starting from the node *i*, passing through other nodes if necessary (where the product probability of a combination of LCPs is obtained by multiplying the *pij* values associated to each single LCP pertaining to the considered combination). In this study, we defined a dispersal probability of 0.5 for a threshold distance of 700 m, considered as the median distance crossed by the Hazel Dormouse through suitable habitats and calculated starting from a mean home range size (Bowman et al. 2002) of 1 ha (Mortelliti et al. 2013).

To identify the LCPs that would greater increase landscape connectivity if they were restored, we calculated the percentage of variation in the *PC* index (*dPC*) caused by the restoration of each LCP, according to the following expression:

$$dPC = \frac{PC_{after\ restoration} - PC_{initial}}{PC_{initial}}$$

Where $PC_{initial}$ is the value of the *PC* index in the original landscape (before any restoration) and $PC_{after\ restoration}$ is the value of the *PC* index that would result in the same landscape after the restoration of each LCP. Specifically, we systematically recalculated the index after changing the ecological distance of each LCP to the value that would correspond to the new restored scenario affecting that particular LCP, when the ecological distance of all the other LCPs remain unchanged. We simulated a complete restoration of each LCP connecting two nodes, by reducing its cumulative movement cost to the distance in meters of the LCP (i.e. simulating that it only cross habitats with resistance equal to 1). The distance of a restored LCP will be shorter than the original ecological distance or equal for those areas already associated to the minimum resistance value. The *dPC* values for this analysis were calculated through the *link importance* functionality using the *link change* modality in the Conefor software. The analysis returned the list of all the LCPs associated to the *dPC* value obtained after their restoration. This way, we could sort the LCPs according to the increase of connectivity they would confer when restored. To test a realistic management action, we simulated the restoration of the LCPs associated to the highest values of *dPC* through the plantation of 30 hedgerows using the software ArcGis. We designed the new hedgerows following the LCPs and maintaining a width comparable to that of existing hedgerows represented in digital format within the DUSAF 4. Moreover, to propose feasible interventions, we designed the new hedgerows along field boundaries avoiding unrealistic crossings of roads, human settlements or rivers by using the most recent digital orthophotos of the investigated area (available at http://www.geoportale.regione.lombardia.it).

The second management strategy proposed in this study consists in identifying which new paths within the landscape would confer a greater increase of connectivity for the Hazel Dormouse if they were newly implemented, i.e. which currently not connected nodes would greatly increase connectivity if they were connected by hedgerows, regardless where the LCP connecting the two nodes passes and which is its ecological distance (strategy 2, Fig. 2e). To reach this objective, we adopted a different habitat availability index: the integral index of connectivity (*IIC*). *IIC* is based on a simple binary connection model in which two habitat patches either are or are not connected (Saura and Rubio 2010) if they lie, respectively, within or beyond a threshold distance defined by the user. Specifically, *IIC* is given by:

$$IIC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i \cdot a_j}{1 + nl_{ij}}}{A_L^2}$$

Where a_i and a_j, in our case, are the areas of nodes *i* and *j*, respectively; and $nl_{ij}$is the number of links in the shortest path (topological distance) between nodes *i* and *j*.

For this study, we set the *IIC* threshold distance to 200 m, which corresponds to the maximum dispersal ability of the Hazel Dormouse in unsuitable habitats (considering that nodes are separated by inhospitable matrix). We calculated this value by averaging different values of Hazel Dormouse dispersal capability in unsuitable habitats available from literature (100 m, Bright 1998; 250 m, Büchner 2008; 100-300 m, Mortelliti et al. 2013). To identify which new paths would greater increase landscape connectivity, we calculated the percentage of variation in the *IIC* index (*dIIC*) caused by the implementation of each new path, according to the following expression:

$$dIIC = \frac{IIC_{after\ new\ path\ implementation} - IIC_{initial}}{IIC_{initial}}$$

Where $IIC_{initial}$ is the value of the *IIC* index in the original landscape (before the implementation of any new path) and $IIC_{after\ new\ path\ implementation}$ is the value of the *IIC* index that would result in the same landscape after the implementation of any new path. Particularly, we added one at a time a path connecting each pair of nodes not directly linked in the initial landscape as they lie beyond the threshold distance, and systematically recalculated the *IIC* value after that addition. The variation of the *IIC* value before and after the addition of each path (*dIIC*) was calculated through the *link importance* functionality using the *link improvement* modality and specifying a maximum distance of 2000 m. The calculations were performed only for the pairs of nodes not farther than that distance, as it is unrealistic hypothesizing the plantation of so long hedgerows. The analysis returned the list of every possible new path associated to its *dIIC* value and we simulated the implementation of the new paths with the highest values of *dIIC*. Since one path could be obtained by planting more than one hedgerow (when the path connects two not adjacent nodes), we simulated the creation of a feasible number of paths through the plantation of 30 hedgerows, following the same criteria used for the plantation of 30 hedgerows along the most important LCPs described in the previous section.

6.2.3.2 Evaluating how landscape connectivity changes after management strategies implementation

To compare the two simulated management strategies, we created four new maps, i.e. two maps of nodes (adding to the original nodes the 30 new hedgerows designed within the strategy 1 and 2, respectively), and two resistance surfaces (setting to 1 the resistance value associated to the pixels pertaining to the 30 new hedgerows designed within the strategy 1 and 2, respectively).

Starting from the new maps, we used the UNICOR software (Landguth et al. 2012) to evaluate how the areas sustaining animal dispersal flows would change after the adoption of the two strategies (see Mateo-Sánchez et al. 2014, 2015). Particularly, we re-identified and mapped the LCPs between each pair of nodes and calculated the total number of LCPs crossing each pixel of the investigated landscape in order to represent the areas where species movements are concentrated both in the original landscape and after the implementation of the two management strategies proposed. As UNICOR requires a threshold distance corresponding to the maximum dispersal ability of the target species in suitable habitat, we set this threshold to 4000 m, following the relationship between home range size and dispersal ability proposed by Bowman et al. (2002).

Subsequently, by using the software Conefor, we calculated the *PC* index for both the maps simulating the two management strategies, in order to obtain comparable values. Even for these analyses, the *PC* index was obtained by calculating the probability of dispersal between each pair of adjacent nodes using a negative exponential function of the ecological distance (calculated on the new resistance surfaces through Linkage Mapper) and setting a dispersal probability of 0.5 for a threshold distance of 700 m. Particularly, we quantified the increase of connectivity we would obtain with the two strategies, calculating the Equivalent Connectivity index (*EC(PC)*, Saura et al. 2011) of the maps representing the implementation of the two strategies and comparing them with the *EC(PC)* index calculated for the original landscape. The *EC(PC)* index corresponds to the size of a single node of suitable habitat (within which we have the highest degree of connectivity) that would provide the same value of the *PC* metric as the actual nodes pattern in the landscape, which depend on the nodes size and on how they are connected to each other. We selected this index, as the relative variation in *EC(PC)* after landscape changes can be easily and directly compared to the variation in the total area of nodes respect to the original landscape after these changes (Saura et al. 2011).

6.2.4 Identifying management priorities

Once defined the most effective strategy in enhancing connectivity for the Hazel Dormouse in the investigated landscape, starting from the map of nodes and the resistance surface of the landscape that simulate this management strategy, we tested two different methods to identify 15 priority hedgerows, out of the 30 designed, by using the *PC* index (with a dispersal probability of 0.5 for 700 m) and by considering hedgerows as nodes rather than links. Through the first method, we assessed the contribution of each new hedgerow out of the 30 designed to the overall landscape connectivity, by calculating the decrease in connectivity caused by its lost. Specifically, we iteratively estimated the percentage of the variation in *PC* (*dPC*) caused by the removal of each individual hedgerow from the landscape when all the other 29 new hedgerows are maintained unchanged (Urban and Keitt 2001, Saura and Pascual-Hortal 2007). The *dPC* values for this analysis were calculated through the *node importance* functionality provided by the software Conefor. Conversely, through the second method, we assessed the contribution of each new hedgerow to the overall landscape connectivity by calculating the increase in connectivity caused by its addition. In this case, we iteratively estimated the percentage of the variation in *PC* (*dPC*) caused by the addition of each individual hedgerow to the original landscape (i.e. to the map in which all the other 29 new hedgerows were not represented) by using the *node to add* functionality in Conefor. Finally, we calculated and compared the *EC(PC)* index for the two new landscapes simulating the implementation of the 15 priority hedgerows identified with the two abovementioned methods (Fig. 4).

6.3 Results

6.3.1 Management strategies comparison

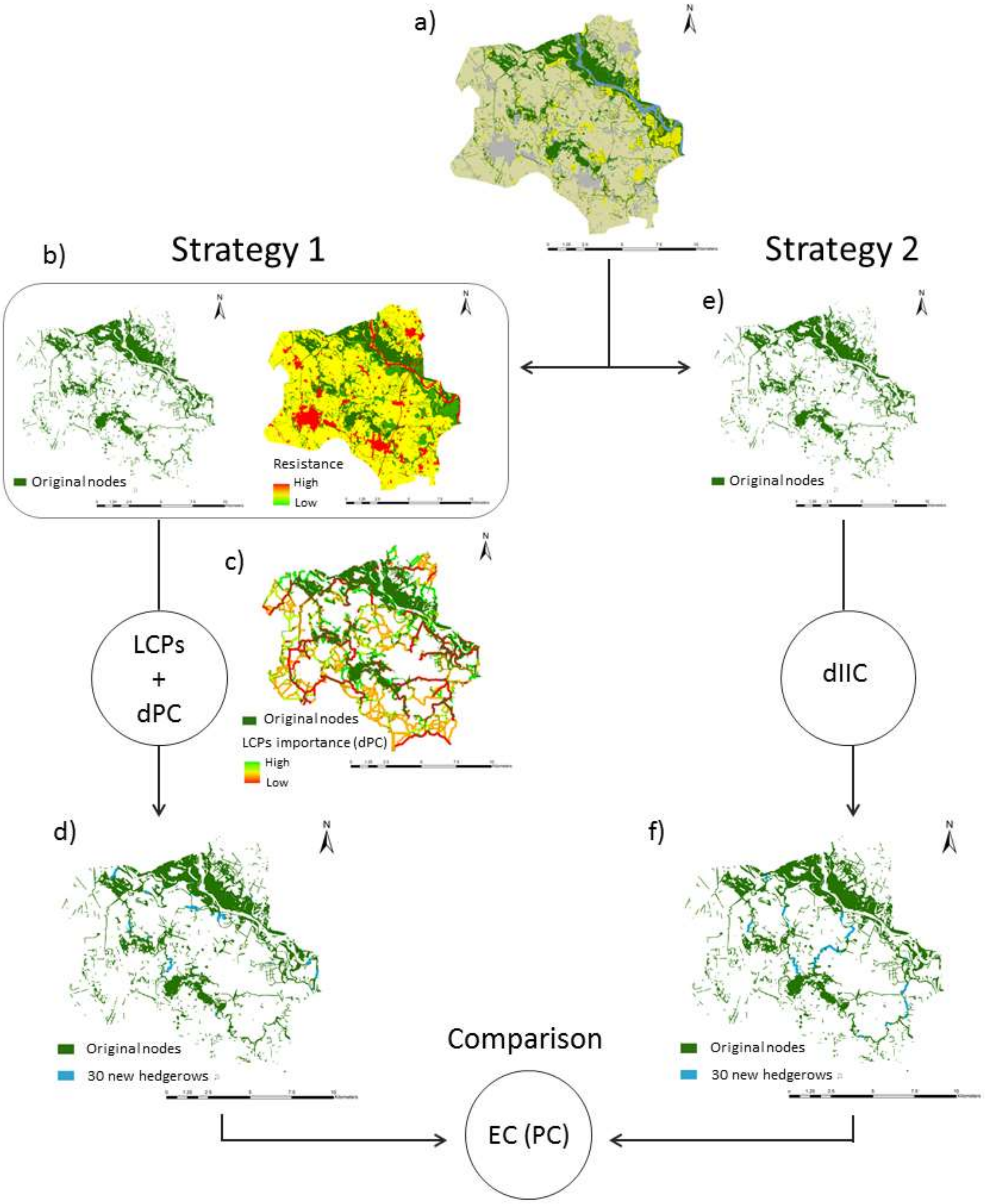
6.3.1.1 Identifying priority areas for restoration (strategy 1) or new paths' implementation (strategy 2)

Starting from the map of original nodes and the resistance surface of the original landscape (Fig. 2b) we identified 3158 LPCs between adjacent nodes. The computation of *dPC* associated to each LCP through the *link change* modality (i.e. *dPC* associated to the LCP permeabilization) allowed to sort all the LPCs according to the increase of connectivity they would provide if restored (strategy 1, Fig. 2c). The absolute value of *dPC* associated to each LCP ranged between 65.3 and 0. The simulation of the restoration of the most important LCPs until 30 hedgerows were designed, led to a total addition of 15.7 ha of new habitat respect to the original landscape. The map representing the 30 new hedgerows, designed to simulate the restoration of the most important LCPs, is shown in Figure 2d.

Starting from the map of original nodes only (Fig. 2e), the computation of *dIIC* through the *link improvement* modality (i.e. to make all the paths between two nodes totally plausible [p=1]) allowed to order all the possible new paths between nodes no more distant than 2000 m, according to the increase of connectivity they would provide if we placed new habitat along each of them (strategy 2). In this case, the absolute value of *dIIC* associated to each new path ranged between 4.3 and 0. The simulation of the creation of the most important new paths until 30 hedgerows were designed, led to a total addition of 20.2 ha of new habitat respect to the original landscape. The map representing the 30 new hedgerows designed to simulate the creation of the most important new paths is shown in Figure 2f.

Figure 2

Flowchart synthesizing the procedure to increase the connectivity for the Hazel Dormouse by simulating two alternative management strategies. Strategy 1: starting from the map of nodes and the resistance surface (b) derived from the original landscape (a), the least-cost paths (LCPs) between adjacent nodes were identified and sorted by importance calculating the *dPC* values (c). Finally, the restoration of the most important LCPs was simulated by designing 30 new hedgerows (d). Strategy 2: starting from the map of nodes (e) the *dIIC* value for every potential link connecting nodes (new potential paths) was calculated. Finally, the implementation of the most important new paths was simulated by designing 30 new hedgerows (f).

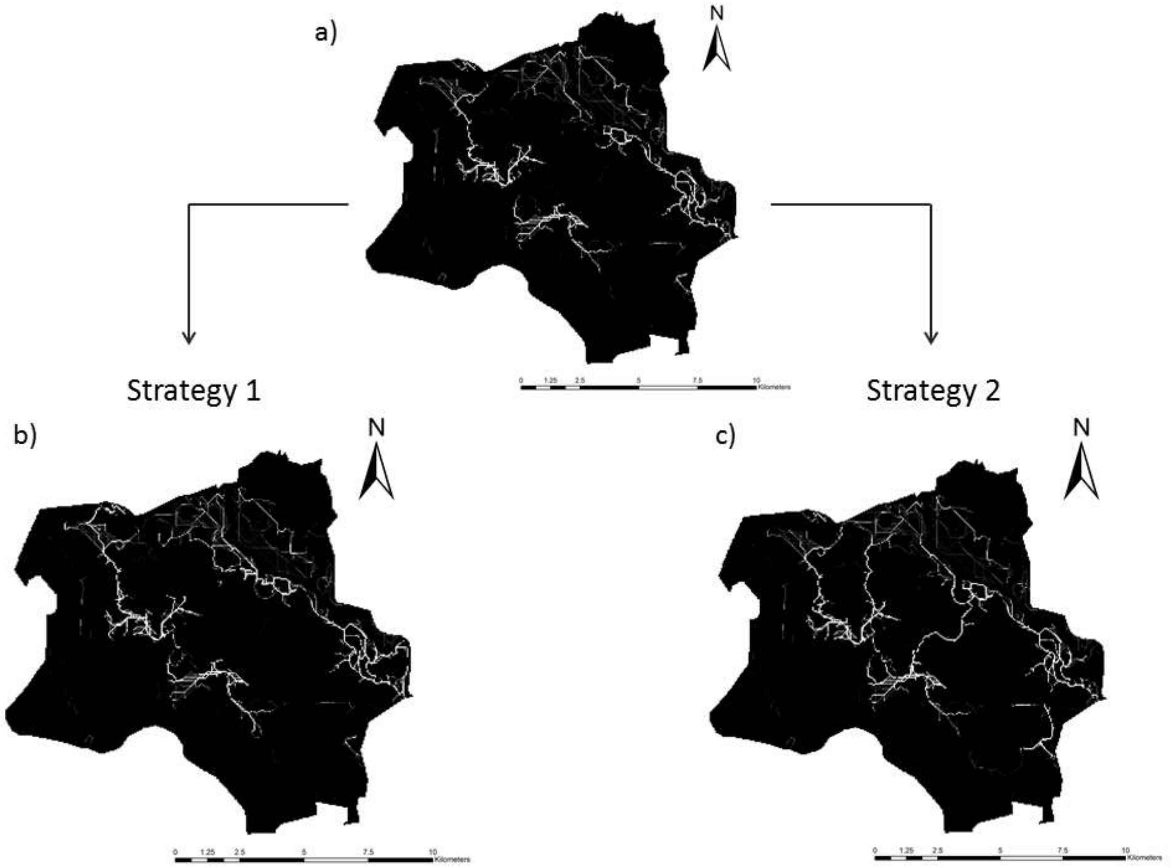


6.3.1.2 Evaluating how landscape connectivity changes after management strategies implementation

The number of LCPs crossing each pixel of the original landscape, of the landscape simulating the location of 30 new hedgerows along the original LCPs (strategy 1), and of the landscape simulating the location of 30 new hedgerows along new paths (strategy 2) ranged between 0 and 301, 516 and 417, respectively. The representation of the LCPs density for each pixel allowed to identify the areas where individuals' movements are concentrated, both in the original landscape and after the implementation of the two management strategies (Fig. 3).

Figure 3

LCPs density for each pixel of the study area in the original landscape (a) and after the implementation of the two management strategies to increase connectivity for the Hazel Dormouse: restoring existing least-cost paths (b) or implementing new paths (c).



The computation of global connectivity for the landscape (i.e. the *EC(PC)* index) showed that, despite the habitat increase is very similar (strategy 1: 0.7%, strategy 2: 0.9%), restoring original LCPs led to an increase of the total connected area equal to 11%, while implementing new paths led to an increase of the connected area equal to 38% (Table 1).

To exclude that the difference of the percentage increase of the area covered by nodes between the two strategies, although small, may have had a significant influence on the increment of connected area when applying the strategy 2, we repeated the analyses maintaining unchanged the final area between the two strategies and, thus, excluding five hedgerows located along the less important new paths among the 30 hedgerows identified through the strategy 2. The computation of the *EC(PC)* index showed that the increase of the total connected area locating 25 new hedgerows along the most important new paths (37%) is almost identical to that obtained locating 30 hedgerows through the same strategy (38%) (Table 1).

Table 1

Comparison of two different strategies to increase landscape connectivity for the Hazel Dormouse in an agro-ecosystem in northern Italy. Absolute and percentage increase of the total and of the connected area covered by nodes after simulating the location of 30 new hedgerows along existing LCPs (strategy 1) or after simulating the location of 30 and 25 new hedgerows along new paths (strategy 2).

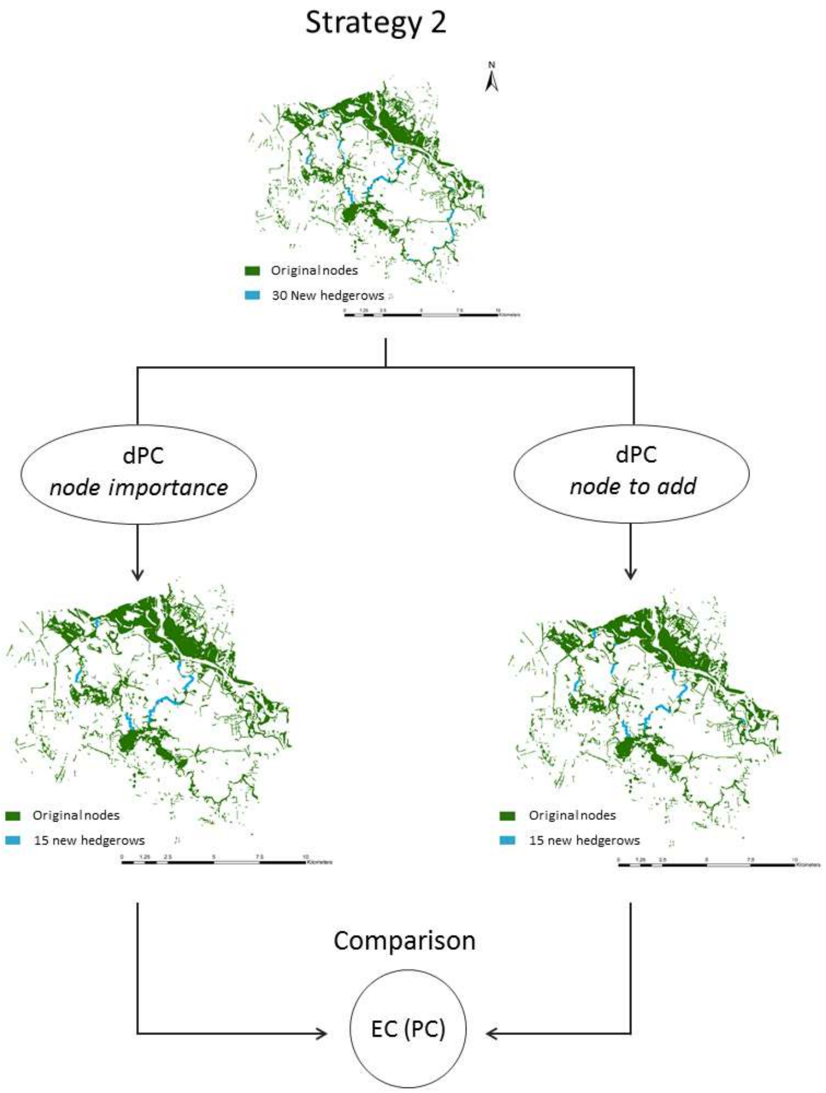
Management strategy	Number of new hedgerows	Initial area (ha)	Initial connected area (ha)	Final area (ha)	Final connected area (ha)	Area increment (%)	Connected area increment (%)
Strategy 1: Restoring existing LCPs	30	2329	791	2344	878	0.7%	11%
Strategy 2: Creating new paths	30	2329	791	2350	1091	0.9%	38%
Strategy 2: Creating new paths	25	2329	791	2344	1082	0.7%	37%

6.3.2 Identifying management priorities

Starting from the map simulating the strategy that would ensure a greater increase of connectivity for the Hazel Dormouse (i.e. the strategy 2), we selected the 15 priority hedgerows through two different methods (i.e. using the functionality *node importance* and *node to add*, respectively). Within the set of 15 hedgerows selected through each of the two methods, 12 were the same, while three differed between the two methods. The maps representing the 15 new hedgerows designed through the two methods are shown in Figure 4.

Figure 4

Flowchart synthesizing the procedure to identify 15 priority hedgerows out of the 30 designed through the management strategy 2.



Starting from an increment of the total area equal to 0.6% for both the simulations, the percentage increment of the *EC(PC)* index (i.e. of the final connected area) resulted 20% for both the two new landscapes simulating the implementation of the 15 priority hedgerows identified with the two methods (Table 2).

Table 2

Comparison of different methods to identify priority hedgerows for the Hazel Dormouse in an agro-ecosystem in northern Italy. Absolute and percentage increase of the total and of the connected area covered by nodes after the simulated location of 15 new hedgerows through assessing the contribution of each new hedgerow by calculating the decrease in connectivity caused by its lost (i.e. using the functionality *node importance*) and through assessing the contribution of each new hedgerow by calculating the increase in connectivity caused by its addition (i.e. using the functionality *node to add*).

Hedgerows priorization strategy	Number of new hedgerows	Initial area (ha)	Initial connected area (ha)	Final area (ha)	Final connected area (ha)	Area increment (%)	Connected area increment (%)
Node importance	15	2328.6	791.3	2342.6	952.4	0.6%	20%
Node to add	15	2328.6	791.3	2343.2	946.8	0.6%	20%

6.3 Discussion

Conservationists and practitioners urgently need reliable methods to identify priority areas where developing habitat conservation and/or restoration to guarantee the long-term maintenance of functional connectivity in fragmented landscapes. Indeed, managing for conservation is arduous without the support of maps to guide interventions (Beier et al. 2011). Mapping the degree of landscape permeability (or resistance) to animal dispersal flows in order to maintain and/or strengthen connectivity by conserving and/or restoring existing paths of minimum resistance is a good strategy when the species of conservation interest are characterized by high or intermediate dispersal ability (Saura and Rubio 2010). Species with a high dispersal ability can easily move among patches following paths of low resistance, otherwise, for the conservation of connectivity for species with an intermediate dispersal ability, it is sufficient placing stepping stones along the low resistance paths. In these cases, the protection of existing low resistance paths, or their restoration through few and small stepping stones, respect to restore habitats elsewhere, is also convenient from an economic point of view. Conversely, Saura and Rubio (2010), by simulating the response to landscape changes of species with different dispersal abilities, demonstrated that species characterized by a low dispersal ability cannot simply cross minimum resistance paths, not even using stepping stones. To ensure the long-term conservation of functional connectivity for these species, which are also the most sensitive to fragmentation because of their biological and ecological traits (Mortelliti et al. 2009), decreasing the resistance of existing paths of low resistance along almost their entire extension is normally proposed as a major option.

Based on this assumption, this study was aimed to assess whether, in the case of low-dispersal species, it is still more convenient to locate management actions along existing low resistance paths or in new strategic areas to implement new dispersal pathways in the landscape. To this aim, we simulated the implementation of management actions to increase connectivity for the Hazel Dormouse by planting new hedgerows in an agro-ecosystem in northern Italy.

From our analyses, it resulted that locating hedgerows in new strategic areas produces a definitely higher increment in connectivity, respect to locating them along low resistance paths existing in the original landscape. It happens because, through the restoration of LCPs identified in the original landscape (strategy 1), we are only strengthening already existing corridors, i.e. the areas where animal dispersal flows were concentrated in the original landscape. This phenomenon can be observed in Figure 3, where lighter pixels representing the areas of dispersal flows concentration in the simulated landscape corresponding to the strategy 1 (Fig. 3b), even if associated with higher absolute values of LCPs density, basically fallow the original corridors showed in Figure 3a. Conversely, by locating new hedgerows in other strategic areas (strategy 2), we are creating new corridors, i.e. new areas carrying additional animal dispersal flows. The new dispersal pathways characterizing the simulated landscape obtained by implementing the strategy 2 are evident in the central and northern part of the map represented in Figure 2c. The increase of LCPs density in correspondence of pixels that already supported dispersal flows in the original landscape when we apply the management strategy 1, is also evident considering the maximum absolute value of LCPs density reached within this scenario

(516), respect to that reached in the scenario simulating the management strategy 2 (417), where LCPs are concentrated in new areas respect to the original corridors.

The percentage increment respect to the original landscape of the *EC(PC)* index quantified the differences among the two strategies in terms of connectivity increase. Restoring existing LCPs by adopting the strategy 1 produced an increase in landscape connectivity equal to 11%, while implementing new paths by adopting the strategy 2 led to an increase equal to 38%. Thus, generating new corridors by implementing new paths led to an increase of connectivity almost four times greater than strengthening existent corridors by restoring existing paths. The importance to create new corridors in high resistance areas, rather than strengthening existing permeable corridors, was already highlighted by McRae et al. (2012). By creating new dispersal pathways, we increased the overall connected area within the landscape and simultaneously created alternative routes along which dispersal may occur among nodes. This is crucial because individuals seldom follow a single optimal rout (Pinto and Keitt 2008), and because redundant corridors ensure the maintenance of connectivity even in the face of future landscape changes (McRae et al. 2008), strengthening the effectiveness of the ecological network.

Hence, to increase the functional connectivity for the Hazel Dormouse, planting new hedgerows generating new dispersal pathways is a much more effective strategy than strengthening existing corridors. From our analyses, it resulted that new corridors are generated only when we applied the strategy 2, i.e. when we did not consider the already existing LCPs to identify the nodes that would lead to a greater increase in connectivity if they were connected. However, it was not obvious that new corridors could be identified with the strategy 2 and not with the strategy 1. Indeed, through the strategy 1, we evaluated how much every LCP (even the LCPs not belonging to the original areas of high LCPs density), connecting every pairs of adjacent nodes, would increase the landscape connectivity if it was restored. Thus, theoretically, the strategy 1 could have had selected the LPCs that, if restored, originated new corridors. However, the most important LCPs to be restored identify with the strategy 1, does not correspond to the most important pairs of nodes to be connected in order to create new corridors, identified with the strategy 2. This result was probably simultaneously due to a methodological and an ecological issue. The methodological issue relies on the procedure to identify LCPs, which only considers adjacent nodes. Indeed, if the connection of two non-adjacent nodes would lead to a consistent increase of connectivity, they will be surely associated to a high value of *dIIC* through the strategy

2 (if they fall within the 2000-m buffer), but not necessary to a high value of *dPC* through the strategy 1. It occurs because the LCPs among adjacent nodes, which, if joined, would connect the two considered non-adjacent nodes, not necessary will be associated to high values of *dPC* when considered alone. The ecological issue for which important pairs of nodes identified through the strategy 2 are disregarded when adopting the strategy 1, relies on the simulation of the complete restoration of a LCP by setting the new distance between the pair of nodes connected by that LCP equal to its distance in meters. LCPs have a much more tortuous pattern compared to hypothetical direct links between two nodes, indeed more permeable areas, surrounded by high resistance ones, often do not follow the shorter paths between two nodes. It is thus clear that the distance in meters of the LCPs will be in some cases much greater than the Euclidean distances between nodes. Considering a species with a very low mobility, the probability that it moves from one node to another crossing this long and tortuous path is very low. Consequently, LCPs with these characteristics will not be associated to a high *dPC* value and will not be selected as priority LCPs to be restored through the strategy 1. The LCPs associated to high *dPC* values are probably those characterized by distances in meters not so long when restored and belonging to areas of high LCPs density in the original landscape. Conversely, the pairs of nodes which maximize the increase of connectivity if they were connected are identified by the strategy 2 because this strategy does not consider the movement probability of the species among nodes, but it simply simulates that two nodes are connected, whatever it is their distance. Starting from the results of the strategy 2 we are then able to design hedgerows following the most feasible direct way between two nodes, disregarding the longer and more tortuous LCP.

This strategy is applicable in the proposed case study as the effort of the management actions is equal in the two strategies, as we are simulating a complete restoration of both the LPCs or alternative links by using 30 new hedgerows. If we would increment the functional connectivity for species with a low dispersal ability, but larger in size respect to the Hazel Dormouse, we should consider more conspicuous management actions to simulating the creation of new paths. In this case, when comparing the two management strategies, we should also consider the feasibility and the cost of the simulated actions. Certainly, in these cases, increasing habitat quality along paths already characterized by a low resistance is more realistic and cheaper than locating new habitat in new strategic areas (Hobbs et al. 2009; Pardini et al. 2010). However, if restoring existing low resistance paths have a much smaller effect in increasing connectivity respect to creating new paths, even the conversion in suitable habitat of small and few strategic areas in the landscape could have a greater effect respect to restore existing low resistance routes. Thus, it is necessary to have appropriate tools able to identify the priority actions, not only considering in which strategic areas of the landscape they should be implemented, but managing them in the form of concrete patches, characterized by a specific size and position respect to the other habitat patches in the landscape.

To this aim, starting from the 30 new hedgerows designed along the new strategic paths identified with the strategy 2, we decided to select the 15 most important in increasing connectivity and to evaluate if they provide a smaller, equal or greater increase of connectivity respect to planting 30 hedgerows according to the traditional strategy of LCPs restoration (Fig. 4). The ideal procedure to identify the 15 most important hedgerows would be to consider the increase in connectivity provided by every possible combination of 15 hedgerows out of the original 30, following a similar rationale to that proposed in Rubio et al. (2014). However, when the number of considered patches is high, this method become computationally prohibitive (McRae et al. 2012), particularly when the procedure includes the re-calculation of the LCPs after every simulation. Thus, we decided to adopt two simpler methods provided by the software Conefor: assessing the contribution of each new hedgerow by calculating the decrease in connectivity caused by its lost (when all the other 29 new hedgerows are unchanged) and assessing the contribution of each new hedgerow by calculating the increase in connectivity caused by its addition (when all the other 29 new hedgerows are not considered). The sets of 15 priority hedgerows identified with these two methods only differed for three hedgerows (see Results) and both led to an increase of the connected area respect to the original landscape equal to 20%. The similarity among the two sets of priority hedgerows, and the increase of connectivity they provided, suggests that the two methods, even if imperfect as they only consider one hedgerow at time, are sufficiently efficient in identifying priority elements for connectivity. Moreover, this result suggested that even halving the number of new hedgerows, the strategy 2 led to an increase in connectivity (+20%) almost two times greater than that obtained with the strategy 1 by simulating the plantation of twice the new hedgerows (+11%). Thus, in our example, with half of the effort (in this case both in terms of new habitat amount and costs) we would obtain an increase in landscape connectivity two times greater if we create new strategic paths, rather than if we restore existing LCPs.

One of the strengths of this study was to have considered a real landscape and to have concretely simulated alternative management strategies by proposing reliable actions and by evaluating how they would affect landscape connectivity. Indeed, as management actions, we simulated the plantation of hedgerows, which represent suitable habitats for the Hazel Dormouse and would subtract a very small surface to cultivable areas. Moreover, hedgerows plantation is supported by financial contributions, due to their importance both for wildlife (Bennett 2003) and environment (Baudry and Burel 1984) conservation. Finally, the maps representing the simulated management actions were designed realistically, as regards both to the number of new hedgerows and to their size and position within the landscape. This way, our results allowed to highlighting that, to greatly enhance the functional connectivity (+38%) for the Hazel Dormouse in an agro-ecosystem, it is sufficient locating few new hedgerows in new strategic areas, respect to existing corridors, adding a negligible amount of new suitable habitat respect to the original landscape (+0.9%).

In conclusion, the information provided by this study about the effectiveness of different management strategies could have important implications concerning the Hazel Dormouse conservation in highly modified agro-ecosystems. The simplicity of the procedure concerning the identification of strategic areas and priority patches, and the use of realistic actions, make the proposed method of management simulations easily to be integrated in landscape management plans for the conservation of the Hazel Dormouse and of other small mammals with similar ecological requirements (e.g. red squirrels *Sciurus vulgaris*, Celada et al. 1994; Mortelliti et. al. 2009). Moreover, the proposal of management interventions in the form of maps representing the position, size and shape of the hedgerows that should be planted to maximize the landscape connectivity increase is easily understandable and exploitable either by financial authorities, practitioners and stakeholders.

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Chapter 7

General conclusions

7.1 Conclusions

This research aimed at providing practical suggestions to design an effective ecological network for forest-dwelling species in a highly fragmented landscape in northern Italy. In this context, specific objectives of the study were: (i) to define how to correctly manage residual forest fragments, i.e. primary nodes; (ii) to define which landscape elements (among the typical semi-natural elements characterizing European agricultural landscapes) can be used as secondary nodes and connectivity elements in an ecological network designed for forest-dwelling species; (iii) to identify priority areas where secondary nodes and connectivity elements should be located to enhance the current landscape connectivity provided by the ecological network.

As the effect of forest fragmentation is strictly species-specific, and acts at very different spatial scales, an ecological network designed for a single target species hardly ensures the long-term conservation of the entire animal community, or at least of a great part of it. For this reason, this research followed a multi-species and multi-scale approach by focusing on multiple bird and mammal target species with different habitat requirements, forest specialization degrees and dispersal abilities. This way, suggestions for (i) the correct management of the internal characteristics of forest remnants, (ii) the correct choice of landscape elements which could play the role of nodes or connectivity elements, and (iii) deciding where to locate them within the landscape, were provided considering, from time to time, the most suitable set of target species in terms of their sensitivity to the phenomenon analyzed and their effectiveness in playing the role of umbrella species.

In Chapter 2, the independent and the combined effect of forest structure and floristic composition on three bird species with a high degree of forest specialization and on a fourth more generalist species were evaluated by developing an innovative method. This study showed the importance of tree species richness for all the considered target species, highlighting the importance of maintaining this features in forest remnants. Moreover, a relatively high number of large trees and a high variety of stem diameters resulted crucial structural forest features for the conservation of the three more specialist target species. This study also demonstrated that models which consider the combined effect of forest structure and floristic composition are characterized by a better performance in modeling bird data and are more effective in providing information useful for forest management with respect to models considering independently the effect of forest structure and floristic composition. Starting from the results obtained by models that consider the combined effect of these two forest features, quantitative forestry guidelines for singular plant species and diameter classes were provided for woodlands characterized by different exploitation degree starting from the ecological requirements of the different bird indicator species considered. The provided management indications should be applicable in most of the European broadleaved forests, where the four target bird species are present.

Once defined the correct management strategies to maintaining high-quality forest remnants, it is crucial to adopt specific analytical tools to identifying which landscape elements could play the role of secondary nodes or connectivity elements able to ensure the ecological connectivity among nodes.

In Chapter 3, the main limitations of the existing analysis procedures to identify the elements of ecological networks are discussed, and a new approach implemented for this aim is described. This approach consists in a new method to objectively define the role that every land-cover type within a landscape can play in an ecological network designed for a given species by comparing different simulated species landscape perceptions. The application of this new method allowed to highlight that the Hazel Dormouse perceives hedgerows, other than woodlands, as nodes, and poplar cultivations, biomasses and reforestations as connectivity elements, providing practical information on how to use these landscape elements in an ecological network designed for this species.

In Chapter 4, the application of the new proposed method on other two mammal target species allowed to obtain further important information. First, it highlighted that a generalist species with an intermediate vagility (the European Badger) and a species with an extremely higher vagility, but with a higher degree of forest specialization (the Roe Deer) have virtually the same dispersal ability through the anthropogenic matrix. Second, the analyses showed that in an ecological network designed for the European Badger, poplar cultivations and hedgerows, other than woodlands, can be used as nodes, while biomasses as connectivity elements. Conversely, in an ecological network designed for the Roe Deer, poplar cultivations and reforestations, other than woodlands, can play the role of nodes, and biomasses and hedgerows that of connectivity elements.

The results of Chapter 3 and 4 are interesting because these are the first studies that objectively demonstrated the role of hedgerows as nodes, and so as suitable habitats and not only as connectivity elements, for both the Hazel Dormouse and the European Badger, and which defined the perception of poplar cultivations, biomasses and reforestations by the Hazel Dormouse, European Badger and Roe Deer in a typical European agro-ecosystem. The very low amount of the area simultaneously suitable and well connected for the three species (see Chapter 4), confirmed that the design of an ecological network using a single target species is not recommendable, and that it is crucial adopting a multi-species and multi-scale approach in order to ensure the conservation of the largest part of the forest animal community. In general, the method proposed in Chapter 3 and applied in Chapter 4 can virtually be adopted to design ecological networks for every species sensitive to fragmentation, providing an effective tool that can be easily integrated in landscape management plans.

The application of this method in our study area revealed the key role of hedgerows in ecological networks designed for forest-dwelling species in agricultural landscapes, particularly for the Hazel Dormouse and the European Badger. Thus, in Chapter 5 and 6 two different studies on how to manage hedgerows to make them effective elements of an ecological network are presented. As the effectiveness of hedgerows as ecological corridors or secondary nodes depends on their internal characteristics, in Chapter 5 a study aimed at evaluating which structural and floristic features make hedgerows suitable for the Hazel Dormouse and the European Badger was presented. The study suggested that the current management practices of hedgerows in the study area is only partially adequate to make them suitable for both species. To simultaneously increase the suitability for both mammal species, hedgerows should be wide and continuous and they should be managed to allow the growth of native species with a complex physical structure in the shrub layer and to promote shrubs development by preventing an excessive tree canopy closure. The information obtained by the two-species approach provided important suggestions for an effective management of hedgerows, which might guarantee the conservation of virtually any forest-dwelling species with ecological requirements and dispersal ability similar to those of the two considered target species. Once defined how to manage hedgerows' internal features, in Chapter 6 the issue of where to plant new hedgerows to improve the overall landscape connectivity for the Hazel Dormouse was addressed. The comparison of the increase of connectivity reached by planting new hedgerows along existing dispersal paths or by planting hedgerows in new strategic areas creating new corridors showed that this last strategy guarantees an increase of connectivity about four times greater than the first one. The information provided by this study about the effectiveness of different management strategies could have important implications concerning the conservation of species characterized by a very low dispersal ability within the hostile matrix in highly modified agro-ecosystems.

7.2 Future applications and perspectives

One of the innovative results of this study was the identification of secondary nodes and connectivity elements among the semi-natural elements characterizing agro-ecosystems for three mammal target species. To be effective components of an ecological network, secondary nodes and connectivity element should have suitable internal characteristics and should be located in areas with a strategic role for landscape connectivity. Specifically, this research provided management information, regarding both internal features and spatial configuration, to make hedgerows suitable elements in ecological networks. The present study focused on hedgerows, as their role for biodiversity is widely recognized, and because hedgerows would subtract a very small surface to crop areas making them optimal elements for conservation interventions in human-dominated contexts. In addition, from the studies reported in Chapter 3 and 4, it resulted that also poplar cultivations, biomasses and reforestations play an important role in ecological networks designed for the three mammal target species, as connectivity elements or even secondary nodes. Thus, future studies should be addressed to identify the correct management strategies to make these plantations effective elements in an ecological network, focusing on internal structural characteristics for poplar cultivations and biomasses (which are generally monospecific plantations) and on both structural and floristic characteristics for reforestations. Moreover, the most suitable strategy to identify sites where they should be planted to enhance connectivity should be assessed, focusing on evaluating whether, for species with an intermediate dispersal ability comparable to those of European Badger and Roe Deer, it is more effective to strengthen the existing corridors sustaining the current individuals’ flows or to creating new corridors by locating plantations in strategic areas where they could play the role of effective stepping stones.

For each proposed strategy, it would be also interesting to associate the costs required for its realization, in order to translate in economic terms the proposed conservation interventions. In fact, the economic resources available for conservation interventions are often limited, and decisions should be supported by information about how much we could increase landscape connectivity by implementing different interventions achievable with the same available resources. Finally, it would be interesting to insert the proposed management strategies within dynamic temporal scenarios, which take into account landscape connectivity changes, consequent to the cut of plantations scheduled by the crop planning.

Scientific papers and conference proceedings

Scientific papers

Some of the results presented within this thesis (Chapter 2 and Chapter 5) were published in the following papers:

Dondina, O., Orioli, V., Massimino, D., Pinoli, G., Bani, L., 2015. A method to evaluate the combined effect of tree species composition and woodland structure on indicator birds. *Ecological Indicators*. 55: 44–51. DOI: 10.1016/j.ecolind.2015.03.007.

Dondina, O., Kataoka, L., Orioli, V., Bani, L., 2016. How to manage hedgerows as effective ecological corridors for mammals: a two-species approach. *Agriculture, Ecosystems and Environment*. 231: 283–290. DOI: 10.1016/j.agee.2016.07.005.

Other results obtained during the three years of Ph.D research are included in the following papers:

Dondina, O., Orioli, V., D'Occhio, P., Luppi, M., Bani, L., 2016. How does forest species specialization affect the application of the Island Biogeography Theory in fragmented landscapes? *Journal of Biogeography*. DOI: 10.1111/jbi.12827.

Bani, L., Orioli, V., Pisa, G., Fagiani, S., Dondina, O., Fabbri, E., Randi, E., Sozio, G., Mortelliti, A., 2016. Population genetic structure and sex-biased dispersal of the hazel dormouse (Muscardinus avellanarius) in a continuous and in a fragmented landscape in central Italy. *Conservation Genetics*. 1–14. DOI 10.1007/s10592-016-0898-2.

Conference proceedings

Poster: Dondina, O., Orioli, V., Bani, L., 2015. Evaluation of the combined effect of tree species composition and woodland structure on secondary cavity-nesting birds. Student Conferences on Conservation Sciences, Cambridge, UK.

Poster: Scancarello, E., De Pasquale, D., Dondina, O., Crippa, L., Veronese, M., 2015. Habitat selection by the European roe deer (*Capreolus capreolus*) in a lowland area of the northern Italy. Student Conferences on Conservation Sciences, Cambridge, UK.

Poster: Chiatante, G., Dondina, O., Lucchelli, M., Bani, L., Meriggi, A., 2016. Habitat selection of European Badger in a highly fragmented forest landscape: the importance of agroforestry. X Congresso Italiano di Teriologia, Acquapendente, Italy.

Contributed talk: Dondina, O., Orioli, V., Bani, L., 2016. Ecological network identification by means of simulated species landscape perceptions. 101st Ecological Society of America Annual Meeting, Fort Lauderdale,FL.

Poster: Chiatante, G., Dondina, O., Meriggi, F., Bovo, K.S., De Pasquale, D., Ferrario, F., Fusari, S., Lucchelli, M., Procaccio, E.L., Scancarello, E., Meriggi, A., 2016. Range expansion of the Roe Deer (*Capreolus capreolus* L., 1758) in the north-western part of the Po Plain, Northern Italy. III Convegno nazionale sulla fauna problematica, Cesena, Italy.

Poster: Chiatante, G., Dondina, O., Bovo, K.S., De Pasquale, D., Ferrario, Lucchelli, M., Scancarello, E., Meriggi, A., 2016. The Roe Deer (*Capreolus capreolus* L., 1758) in the Ticino Regional Park: from the reintroduction to nowadays. III Convegno nazionale sulla fauna problematica, Cesena, Italy.

Contributed talk: Meriggi, A., Lombardini, M., Dondina, O., Piacentini., I., Nugnes, C., De Pasquale, D., Scancarello, E., 2016. Comparison of four different methods to estimate population density of roe deer *Capreolus capreolus* in a lowland forest in Northern Italy. III Convegno nazionale sulla fauna problematica, Cesena, Italy.