Forest Ecology and Management 261 (2011) 154–161

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Improving landscape connectivity in forest districts: A two-stage process for prioritizing agricultural patches for reforestation

C. García-Feced[a,*], S. Saura[b], R. Elena-Rosselló[a]

[a] ECOGESFOR-UPM, Research Group of Ecology and Sustainable Forest Management, Universidad Politécnica de Madrid. E.U.I.T. Forestal, Ciudad Universitaria s/n, 28040 Madrid, Spain

[b] Department of Forest Management and Economics. E.T.S.I. Montes, Universidad Politécnica de Madrid, Ciudad Universitaria s/n, 28040 Madrid, Spain

ARTICLE INFO

Article history:
Received 13 July 2010
Received in revised form
27 September 2010
Accepted 28 September 2010

Keywords:
Forest landscape planning
Potential functional connectivity
Graph theory
Ecological networks
Wildlife corridor
Stepping stone

ABSTRACT

Connectivity is a key concern in natural resource planning. Many studies have focused on the development of methods, tools and indices for the assessment of both components of connectivity: structural and functional. In particular, approaches based on graph theory principles have been recently proposed and are being increasingly applied to guide landscape connectivity conservation. However, forest planners and managers still need effective and operational methodologies to detect those landscapes where connectivity should be treated as a particularly critical conservation concern. In addition, in the Mediterranean, as in other parts of the world, socioeconomic changes in the last decades have driven the abandonment of many formerly cultivated lands. This poses both a challenge and an opportunity for managers intending to restore ecological connectivity in forested areas. In this context, setting adequate priorities for the reforestation of agricultural lands is of outmost importance. Here we show how a two-stage hierarchical methodology based on network analysis can be used to meet these needs. In particular, we apply a graph metric based on the measurement of habitat availability at the landscape scale (the Integral Index of Connectivity) to two Mediterranean forest districts in Spain with different management objectives and environmental heterogeneity. First, we identify those landscapes where efforts to improve forest connectivity should be concentrated. In a second stage, we prioritize within those landscapes the individual patches of agricultural lands that, being available for a potential reforestation program, would contribute most to uphold connectivity and ecological flows at wide spatial scales. We show how the extent of the agricultural patches is not strictly related to the contribution to connectivity they would provide if reforested, and how the results of such analysis vary with species traits (dispersal capabilities). We discuss the suitability of the proposed approach for forest landscape planning purposes and conclude that it can provide a useful diagnosis and helpful guidelines for the development of efficient reforestation programs that might be applied in a variety of situations for improving the ecological coherence of forest landscapes.

1. Introduction

Landscape connectivity can be defined as the degree to which the landscape facilitates movement across its existing resources (Taylor et al., 1993). It can be considered as an emergent property of the landscape that results from the interaction between landscape structure and landscape function (Leitao et al., 2006). As Crooks and Sanjayan (2006) explained, we can identify two primary components of connectivity: structural, i.e., the spatial arrangement of different types of habitat or other elements in the landscape, and functional, that refers to the behavioural response of individuals, species, or ecological processes to the physical structure of the landscape, which is ignored by structural connectivity approaches (Taylor et al., 2006; Tischendorf and Fahrig, 2000). According to Calabrese and Fagan (2004), we can distinguish two classes of functional connectivity: potential and actual. Potential functional connectivity combines the physical attributes of the landscape with limited information about dispersal ability to predict how connected a given landscape or patch will be for a particular species. Actual functional connectivity relates to the empirical observation of individuals moving into or out of focal patches, or through a landscape, and thus provides a concrete estimate of the linkages between landscape elements or habitat patches. Potential functional connectivity analysis is more usual in forest management

* Corresponding author at: Departamento de Ecología Forestal, Escuela Universitaria de Ingeniería Técnica Forestal, Universidad Politécnica de Madrid, Ciudad Universitaria s/n, 28040 Madrid, Spain. Tel.: +34 913367660.
E-mail address: celia.garcia.feced@upm.es (C. García-Feced).

0378-1127/$ – see front matter
doi:10.1016/j.foreco.2010.09.047

due to the intensive and often unavailable data that are required for actual functional connectivity approaches.

Connectivity is a key topic in ecological research due to its potential for mitigating the effects of habitat fragmentation (Anderson and Jenkins, 2006; Bailey, 2007). In particular, the study of connectivity is essential in forest resource management since one of the main goals of many plans is the conservation of certain endangered or focal species (keystone, umbrella or flagship species) whose persistence is in many cases dependant on the degree of habitat connectivity. Landscape structure and connectivity can be easily characterized by means of a large set of indices and widespread programmes specifically designed for that purpose. However, the challenging point is to transfer this information into forest planning effectively, going beyond a descriptive analysis to one oriented towards the decision making process.

Once the forest connectivity within a landscape has been evaluated, and the need for its improvement has been established, one of the major solutions is the reforestation of abandoned agricultural lands. In this sense, 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" (adopted in November, 2008) recommends promoting reforestation activities that contribute to the improvement and restoration of ecological connectivity. This is particularly important in Mediterranean regions that have suffered large-scale agricultural abandonment processes (Vogiatzakis et al., 2008). Particularly, in Spain the abandonment of agricultural lands mainly started in the 1960s during the rural population exodus to cities and has continued ever since. In the 1990s, the European Union established a series of policies (Common Agricultural Policy, CAP) determined to promote reforestation activities in abandoned agricultural lands. This program has been particularly successful in Spain, where 685,000 ha of agricultural lands have been converted to forests in the period from 1994 to 2006 (Sociedad Española de Ciencias Forestales, 2009). Indeed, in 2009 the annual reforestation rate of Spain was the highest of Europe (2.19% in comparison to the European mean of 0.51%). However, in some cases these reforestations have been performed with no or little consideration to the spatial coherence of the forest pattern and to the role of forested lands in a wider landscape context. There is still a strong need of tools and methodological approaches aimed to optimize this activity. Land planners and managers require effective methodologies to perform ecologically-based detection of critical landscapes and to select priority agricultural patches for the potential enhancement of forest connectivity.

In order to achieve this goal, we adopted a methodology based on graph theory principles and tools. A graph represents a landscape as a set of nodes (habitat patches) functionally connected to some degree by links that join pairs of nodes (Urban and Keitt, 2001). This is a potential functional connectivity approach where the graph is simply a means of summarizing the spatial relationships between landscape elements in a concise but spatially explicit way. The use of graph theory for the study of ecological connectivity and the setting of related metrics has been widely reviewed (Cantwell and Forman, 1993; Dale and Fortin, 2010; Fall et al., 2007; Urban et al., 2009; Zetterberg et al., 2010). Bunn et al. (2000) remarked that the simplicity and flexibility of graph-theoretic approaches to landscape connectivity offers much to land practitioners, such as making decisions based on which patches are most critical to uphold landscape connectivity (Calabrese and Fagan, 2004), allowing to increase the scope and effectiveness of resource management. Pascual-Hortal and Saura (2006) analysed the properties and behaviour of a wide set of graph metrics for prioritizing the key habitat patches for connectivity. The best performance in this respect was found for those metrics that were based on the concept of measuring habitat availability (reachability) at the landscape scale, as further developed by Saura and Pascual-Hortal (2007) and Saura and Rubio (2010).

Nevertheless, the importance of connectivity for actual resource management varies among the different landscapes and conservation contexts. In particular, due to the complexities and uncertainties involved in the measurement and analysis of connectivity, the question that arises is if focusing on the amount of habitat (independently of the spatial arrangement of forest habitat in the landscape network) might be an easier and more effective conservation strategy (Hodgson et al., 2009; Saura and Rubio, 2010). Thus, it is desirable to be able to identify in advance those landscapes where connectivity should be really treated as a critical concern for the conservation goals and to avoid overweighting (or under-representing) connectivity considerations in the final management plans.

For these reasons, we here propose a methodology that, based on a graph-theoretical and habitat availability approach, aims to answer two specific questions: (1) which are the critical landscapes of a forested region in terms of connectivity? And, with that knowledge, (2) which agricultural lands would be more effective as connectivity providers if they were reforested? This study adopts a different analytical approach and provides new insights compared to previous applications based in the same type of tools and metrics because (a) it does not focus on habitat loss processes (e.g. Estrada-Peña, 2003; Jordán et al., 2003; Rothley and Rae, 2005) but on the improvement of connectivity through reforestation and on the benefits of a habitat network perspective to enhance the ecosystem services provided by the new forested lands and (b) it refines and more directly relates the connectivity analysis to the manager needs by identifying those units where connectivity should be really treated as a major management concern, unlike previous studies that have evaluated large forested areas without considering the variable conservation and planning contexts at the finer spatial scales where the actual management is implemented (e.g. Pascual-Hortal and Saura, 2008; Saura and Pascual-Hortal, 2007). We applied this methodology by evaluating the potential functional connectivity of different management units in two Spanish Mediterranean forest districts, although the same approach might be adopted in other study areas and management situations. In Spain, the demand for forest plans designed at the district scale has recently arisen and rapidly increased after the Spanish Forest law (passed in 2003) established PORFs (*Planes de Ordenación de los Recursos Forestales*) as the basic planning instrument to broaden the traditional management scale focused on the individual forest, but similar initiatives, needs and trends apply as well to many other countries (Lafortezza et al., 2008). Instead of orienting the design of the wildlife linkages to a single focal or indicator species, which might be controversial to select and uncertain as a reliable measure of conservation management success (Lindenmayer et al., 2000), we performed the potential functional connectivity assessment adopting a multi-species point of view (i.e. setting different dispersal distances), as recommended, among others, by Beier et al. (2008) and Rothley and Rae (2005).

2. Materials and methods

The proposed methodological process can be structured in two stages (Fig. 1). The first one consists in identifying the critical landscapes in terms of forest connectivity. Within these, the second stage should accomplish the selection of those individual agricultural patches that potentially would contribute more to the improvement of connectivity. Finally, once the ecological diagnosis is completed after these two stages, its results should be combined and integrated with other possible constraints and planning considerations such us ownership or administrative regulations.

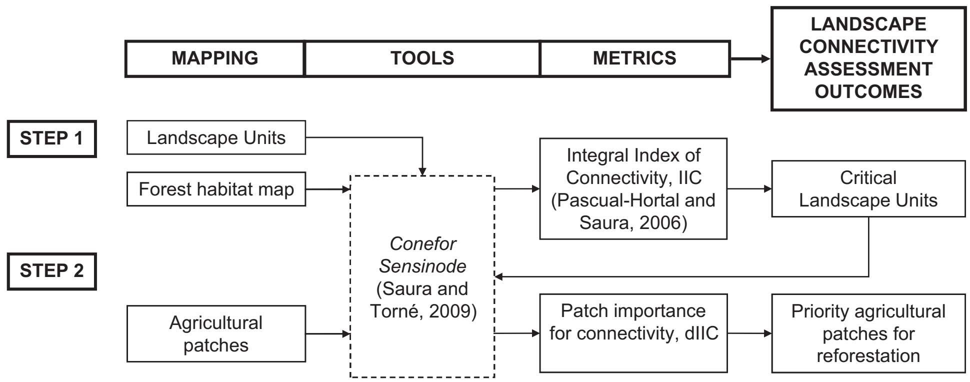

Fig. 1. Scheme of the proposed methodology. The data of both steps were processed through the Conefor Sensinode 2.2 software package (Saura and Torné, 2009).

However, these latter considerations are out of the scope of this study, and we will focus on the first two steps of the process, those specifically related to the ecological connectivity assessment.

2.1. Study areas

The methodology was applied in two Spanish Mediterranean forest districts known as Pinares (UTM coordinates zone 30N: X_{min} = 480,000, Y_{min} = 4,613,000) and Alto Tajo (X_{min} = 540,000, Y_{min} = 4,491,000), with an extent of 127,956 ha and 104,561 ha respectively. Both are located within the most continental part of the Mediterranean region and their average altitudes are 1266 m in Pinares and 1150 m in Alto Tajo. The population density of Pinares district is 12 inhabitants/km^2 while in Alto Tajo is much lower (1.7 inhabitants/km^2). Their forest management intensities are different as well. Management in Pinares district is oriented towards timber production, whereas in Alto Tajo it is focused on the protection of watersheds, biodiversity and geological heritage.

In a previous study, the landscape types of each district were discriminated and mapped according to abiotic and biotic variables (García-Feced et al., 2008). The landscape typologies were differentiated by integrating a land classification based on altitudinal and lithological factors, and land cover information provided by the Spanish Forest Map (SFM) scale 1:50,000 (Ministerio de Medio Ambiente, 2002), as summarized in Table 1. The landscape types were distributed within the forest districts in several spatially separated landscape units, in which different management measures are implemented. In total, 13 landscape units were found in Pinares and 15 in Alto Tajo, corresponding to 7 and 6 different landscape types respectively (Table 1). The forest connectivity analysis was developed in each of these landscape units.

2.2. Forest connectivity analysis

Since we focused on forest planning and forest dwelling species, we considered as nodes the habitat patches with a forested land cover according to the SFM (which included coniferous and deciduous woodlands, *matorral* areas and meadows), as well as those agricultural patches that were available for reforestation (see below).

In order to perform the quantitative connectivity analysis, we used the software Conefor Sensinode 2.2 (Saura and Torné, 2009), available at http://www.conefor.org. This is a powerful tool for analyzing landscape network connectivity that has been applied in numerous studies (Fu et al., 2010; Laita et al., 2010; Neel, 2008; Pascual-Hortal and Saura, 2008; Perotto-Baldivieso et al., 2009; Saura and Pascual-Hortal, 2007). Among all the connectivity indices that can be calculated by this software, we selected the Integral Index of Connectivity (IIC) given the good properties that this index presents for the purposes of this study (Pascual-Hortal and Saura, 2006). IIC is a binary index (each pair of patches is regarded as either connected or not connected) that ranges from 0 to 1 and increases with improved connectivity. It takes into account the connected area existing within the habitat patches (intrapatch connectivity), the estimated dispersal flux between different habitat patches in the landscape, and the contribution of patches as stepping stones or connecting elements that uphold the connectivity between other habitat areas. We selected IIC instead of the conceptually-related Probability of Connectivity (PC) index (Saura and Pascual-Hortal, 2007) because IIC is less data demanding and easier to parameterize, therefore better matching with the amount of ecological information that is usually available or that can be afforded to acquire within the scope of a forest management plan. In addition, IIC is more sensitive to the presence of connecting elements and stepping stones in the forest landscape than PC (Bodin and Saura, 2010) and has been shown to better relate to genetic diversity statistics than PC (Neel, 2008). However, the same methodological process could be applied through PC if considered more appropriate for a particular application.

All the IIC calculations described below were performed for different dispersal distances (200 m, 1000 m, 5000 m, 25,000 m), as related to two types of dispersal processes: natal dispersal (movement between the birthplace and the first breeding site) and breeding dispersal (movement between successive breeding sites) (Cadahía et al., 2010). These four distance values intend to cover the dispersal capabilities of a variety of species present in the forest districts that widely differ in their movement ranges (McComb, 2007), and were selected as representative of diverse animal groups according to several studies (Bowman et al., 2002; Sutherland et al., 2000). For instance, several mammal species present in the study areas have dispersal distances close to the selected ones: surveillance data from wildcats (*Felis sylvestris*) show that the mean length of diary movements is 5.2 km for female adults (Stahl et al., 1988); the mean natal dispersal distance of wolves (*Canis lupus*) has been estimated at 32 km (Blanco and Cortes, 2007); median dispersal distance of male red deers (*Cervus elaphus*) has been reported to be about 21 km (Loe et al., 2009); for small mammals such as the European wild rabbit (*Oryctolagus cuniculus*) the distance from the natal burrow is often lower than 1 km during their first three months (Kuenkele and Von Holst, 1996).

A link between two nodes in the graph was assigned if the edge-to-edge Euclidean distance between them was lower than the selected dispersal distance. We did not consider effective or cost-weighted distance (i.e., modified by landscape resistance/friction (Adriaensen et al., 2003; Theobald, 2006)) because: (1) the latter approach is more data-demanding and the Euclidean distance is

Table 1
Characterization of the landscape types within both forest districts (modified from García-Feced et al., 2010). The list of landscape units corresponding to each landscape type is also reported.

Landscape type	Landscape units included	Relative extent (%)	Average altitude (m)	Lithological type	Dominant land covers	Brief description
Pinares district						
1	P11, P12, P13	4.87	1837	Conglomerates and quartz sands	Grasslands (37%) and *Pinus sylvestris* (34%)	High mountain grasslands
2	P21	26.88	1467	Conglomerates and quartz sands	*Pinus sylvestris* (86%)	*Pinus sylvestris* forest
3	P31, P32	15.3	1160	Quartz sands	*Pinus sylvestris* (59%) and *Quercus pyrenaica* (21%)	Mixed forest (*Pinus sylvestris* and *Quercus pyrenaica*)
4	P41, P42	11.93	1130	Limestones	*Juniperus thuriphera* (58%) and *Pinus nigra* (25%)	*Juniperus thuriphera* forest
5	P51	8.8	1154	Conglomerates	*Pinus sylvestris* (47%), *Quercus pyrenaica* (23%) and crops (17%)	Mixed landscape of forests and crops
6	P61, P62	26.44	1154	Conglomerates	*Pinus sylvestris* (47%) and *Pinus pinaster* (23%)	Pine forest (*Pinus sylvestris* and *Pinus pinaster*)
7	P71, P72	5.77	1103	Sands	Crops (38%) and *Pinus sylvestris* (16%)	Agricultural belt
Alto Tajo district						
1	A11, A12	17.01	1372	Dolomites	*Pinus sylvestris* (46%) and *Pinus nigra* (34%)	Pine forest (*Pinus sylvestris* and *Pinus nigra*)
2	A21, A22, A23	19.6	1191	Dolomites	*Pinus nigra* (86%)	*Pinus nigra* forest
3	A31, A32, A33	19.96	1189	Dolomites	*Juniperus thuriphera* (40%) and *Pinus nigra* (28%)	Mixed forest (*Juniperus thuriphera* and *Pinus nigra*)
4	A41, A42, A43	19.23	1157	Marls and conglomerates	*Pinus nigra* (28%), *Juniperus thuriphera* (21%), shrubs (14%) and crops (14%)	Mixed landscape of conifers, shrublands and crops
5	A51, A52	12.1	904	Conglomerates	*Quercus ilex* (53%) and *Pinus nigra* (23%)	*Quercus ilex* forest
6	A61, A62	12.11	949	Dolomites and marls	*Pinus nigra* (49%)	Tajo river canyons

easier to implement for operational planning purposes, (2) friction coefficients for different land covers strongly vary depending on the analysed species, therefore not being so suitable for the multi-species and more generic perspective here adopted, and (3) friction coefficient assignment is subjected to multiple uncertainties and frequently lacks of enough empirical support. The use of more complex models with a larger number of parameters when there is a lack of validated information to feed them may provide less reliable results than simpler models such as the Euclidean distance. However, we acknowledge the value of the least cost approaches that account for the heterogeneity of the landscape matrix; these could be easily incorporated in the same methodological approach here adopted for those cases oriented to particular focal species with well documented biological and dispersal behaviour. In particular, Rayfield et al. (2010) proposed the delineation of probable movement zones as the combination of multiple low-cost paths between pairs of habitat patches, as a good way to cope with the uncertainity in the friction coefficients that provides more robust outcomes for guiding related conservation management decisions.

In the first stage we calculated the IIC values for the network of forest habitat patches within each landscape unit in order to quantify their degree of forest connectivity ($\mathrm{IIC}_{\mathrm{initial}}$). This allowed identifying which landscape units presented connectivity deficits, as indicated by low IIC values. These would correspond to the priority landscape units in which the efforts for enhancing forest connectivity through the reforestation of abandoned agricultural land should be first promoted.

In the second stage of the analysis, we aimed to measure, within each landscape unit, the contribution of individual patches (nodes) of agricultural lands for enhancing connectivity within that unit if they were converted to forest. This was evaluated through the dIIC values for each agricultural patch (from a total of 275 and 318 agricultural patches in Pinares and Alto Tajo respectively) according to the following expression:

$$\mathrm{dIIC} = 100\,\frac{\mathrm{IIC}_{\mathrm{reforested}} - \mathrm{IIC}_{\mathrm{initial}}}{\mathrm{IIC}_{\mathrm{initial}}}$$

where $\mathrm{IIC}_{\mathrm{initial}}$ is the value of the IIC index in the initial habitat network (before any land is reforested) in each landscape unit (this value had been already calculated in the first stage of the methodology) and $\mathrm{IIC}_{\mathrm{reforested}}$ is the value of the same index that would result in the same landscape unit after the reforestation of a particular agricultural patch. Although the dIIC calculations would be of most relevance in those critical landscape units as identified in the previous stage of the analysis, we calculated dIIC for all agricultural patches in all landscape units. This allowed testing, and eventually demonstrating, that the first stage of the analysis is effective in discriminating those landscape units where connectivity can be effectively improved to a sufficient extent by the reforestation of available lands. In that case, it would be possible to reduce the considerable costs of data gathering, analysis and processing in the rest (potentially most) of the analysed study area. The importance of the agricultural patches for improving connectivity was classified into five categories that were determined by natural breaks of the whole district dIIC values. The class breaks were determined statistically by finding adjacent feature pairs between which there is a relative difference in data value (ESRI, 2005).

Finally, we compared the dIIC values with the area of each current agricultural patch in order to assess whether this methodology provides distinctive results compared to the alternative of prioritizing patches for reforestation just on the basis of the habitat area they would provide.

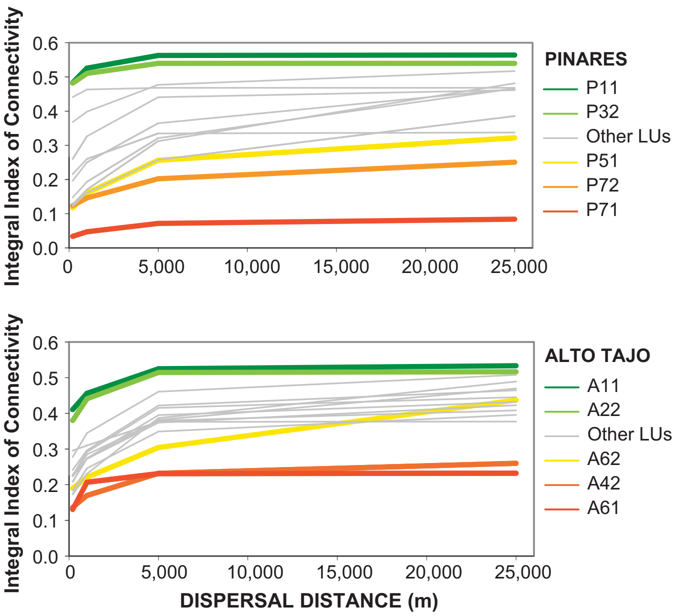

Fig. 2. Degree of forest connectivity (as measured by IIC) for certain landscape units (LUs) and the four selected dispersal distances. Due to the large number of landscape units, the graphs only show in colour those whose IIC values are within the highest two or the lowest three in the majority of dispersal distances considered (out of a total of 13 units in Pinares ("P") and 15 units in Alto Tajo ("A"), as summarized in Table 1).

3. Results

3.1. Landscapes where forest planning should have connectivity as a major concern

The most connected landscape units were P11 (high mountain grasslands) and P32 (mixed forest) in Pinares and landscapes A11 and A22 (both pine forests) in Alto Tajo (Fig. 2). On the contrary, the landscape units with the lower IIC values corresponded to the agricultural belt (P72 and especially P71) in Pinares and to A42 (mixed landscape) and A61 (river canyon) in Alto Tajo; it is in these units where the measures for enhancing the currently limited connectivity would have a more beneficial effect to promote the ecological flows and exchange of forest dwelling organisms. In general, all curves followed a fairly similar pattern (Fig. 2). In spite of that, in some cases the ranking of the landscape units in terms of connectivity (IIC value) varied with the considered dispersal distance. This can be appreciated for example in the values for the Tajo river canyons (A62), which presented one of the largest ranges of variation of connectivity as a function of the dispersal distance (difference between the IIC values at 25,000 m and 200 m of dispersal distance).

3.2. Priority patches for reforestation

In both districts, dIIC values ranged from approximately 1% up to the observed maximum values (Table 2), and presented a highly skewed distribution with strong variations between landscapes

Table 2
Maximum dIIC values (%) found for all the agricultural patches to be potentially reforested in the two forest districts as a function of the dispersal distance considered.

Dispersal distance (m)	200	1000	5000	25,000
Pinares district	222.96	176.57	132.91	120.17
Alto Tajo district	181.72	96.28	82.57	82.20

(Fig. 3). The top patches for connectivity enhancement remained the same regardless of the dispersal distance, but the maximum dIIC values decreased for species with larger movement abilities (Table 2). In Pinares, these maximum dIIC values were reached for all dispersal distances at a patch contained in the agricultural belt (P71) while in Alto Tajo they were associated to a patch located at the river canyon (A61).

In Pinares, 78% of the agricultural patches had a very low importance for connectivity (Fig. 3), while only seven patches (2.54%) had a high importance and only one was classified in the top importance category for a dispersal distance of 1000 m. In the same case in Alto Tajo, 92% of the agricultural patches made a very low contribution for forest connectivity enhancement, and only two and one patch were respectively classified in the first and second categories with the highest dIIC values. These patterns remained similar for all the dispersal distances here considered.

All the important patches as connectivity providers (defined as those with a high or very high importance as shown in Fig. 3) were located in both districts and for all dispersal distances within the landscape units that previously had been identified as critical according to their low IIC values (IIC lower than 0.35 even for the largest dispersal distance here considered). These landscape units where the key agricultural patches for reforestation occurred were P72, P71, P51 and P62 in Pinares and A61 and A42 in Alto Tajo.

There was not a strict relation between the importance for connectivity of a particular patch and its area (Fig. 4). In Pinares the three priority patches for a dispersal distance of 1000 m present quite different extents (485 ha, 196 ha and 119 ha in decreasing importance order, with dIIC values ranging from 32.70% to 176.57%), while the largest patch (702 ha) has a relatively low importance for connectivity (dIIC = 28.67%). In Alto Tajo, the most important patch (dIIC = 96.28%) is not the largest either (163 ha). In fact, there are six larger patches (maximum extent of 503 ha) that have a significantly lower importance. Similar results were found for all the dispersal distances considered.

4. Discussion

Our results clearly show strong differences between the landscape units in terms of forest connectivity, and allow focusing related management decisions first in those landscapes that present evident connectivity deficits. This is accomplished through recent developments in spatial graphs and habitat availability metrics, which are able to combine a pragmatic approach that is operational and meaningful to inform forest management decisions with a functional perspective on landscape connectivity. Therefore, unlike purely structural metrics, which have been shown to be of limited value as a guide for planning (Corry and Nassauer, 2005), the applied methodology is sensitive to the dispersal capabilities of the analysed species. It allows assessing the degree to which different species or taxonomical groups may be affected by the management decisions and the conservation status of forest landscapes. For instance, while we found many landscapes in which most of forest terrestrial vertebrates could disperse freely in absence of other constraints, other species with low dispersal capabilities may find considerable difficulties in traversing the forest habitat existing in certain management units. Indeed, as noted by Saura and Rubio (2010), when the dispersal abilities are large enough, species can move directly from one patch to another without depending on intermediate stepping stones or connecting elements that facilitate this dispersal. Therefore the reforestation of a particular agricultural patch would have a modest contribution to the mobility of these species, which was already largely guaranteed in the initial landscape. This is the case of big and vagile mammal species present in the study areas, such as the wildcat (*F. sylvestris*), the

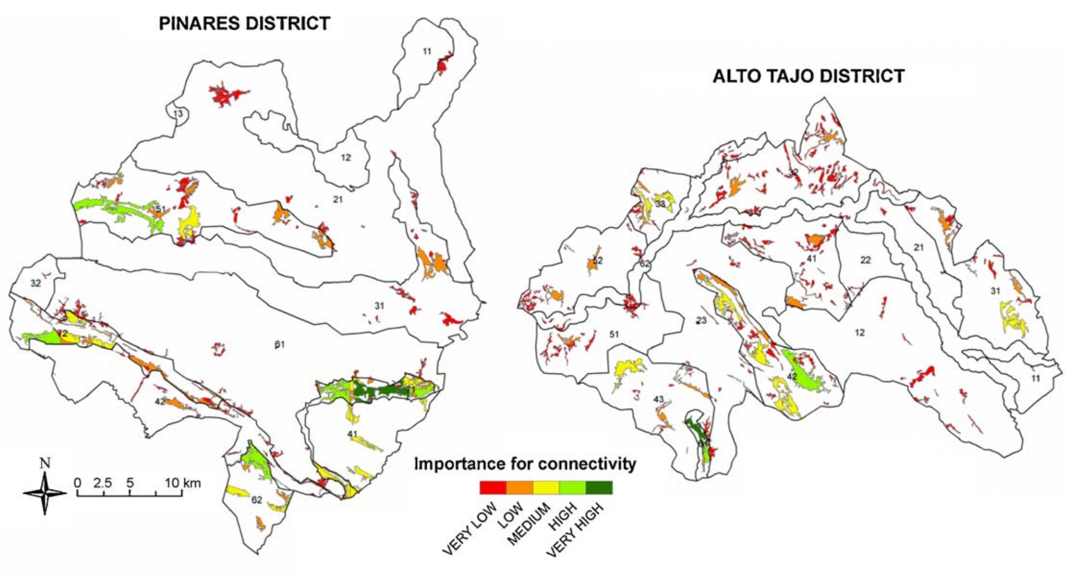

Fig. 3. Categorization of Pinares and Alto Tajo agricultural patches in relation to the contribution to forest connectivity improvement they would provide if reforested (relative increase in the Integral Index of Connectivity, dIIC). The maps represent the resulting dIIC values classified by natural breaks (different values depending on the district) and considering a dispersal distance of 1000 m. The numbers in the maps refer to the existing landscape units according to García-Feced et al. (2008), as summarized in Table 1.

red fox (*Vulpes vulpes*), the European roe deer (*Capreolus capreolus*), the red deer (*C. elaphus*) or the wolf (*C. lupus*) (Blanco and Cortes, 2007; Loe et al., 2009; Stahl et al., 1988), as indicated by the low dIIC values for large dispersal distances in Table 2. When on the contrary dispersal is limited to some degree (but not fully inhibited), the organisms cannot move directly to every other forest patch in the landscape but can more easily disperse to a few other nearer forest patches. These latter patches can serve as stepping stones that allow species to disperse further and reach to a higher amount of available habitat after several steps from one patch to the other (Keitt et al., 1997; Saura and Rubio, 2010). It is in this case when the positive contribution of an individual reforested patch can be more critical and cause a significant raise in the ability of a species to successfully reach to other high-quality or large forest habitat patches in the landscape. In our study, the species that were more dependent on the contribution of selected reforested patches (and more likely to be particularly benefited by them) were those with dispersal distances below 1 km, for which higher dIIC values were obtained (Table 2). This is the case of small vertebrates present in both districts, such as the European wild rabbit (*O. cuniculus*) and the Iberian hare (*Lepus granatensis*) (e.g. Kuenkele and Von Holst, 1996), which are essential elements in the food chain, and may particularly increase their possibilities of successful natal dispersal with improved forest connectivity. This seems to be the case as well for the dispersal of other small rodents, reptiles, amphibians, passerine birds and plant seeds by wind, which rarely goes well beyond 1 km (Sutherland et al., 2000; Tackenberg et al., 2003; Smith and Green, 2005; Vittoz and Engler, 2007). This highlights the importance of adopting a multi-species approach when planning and designing forest landscapes, as recommended as well by Beier et al. (2008) and Rothley and Rae (2005).

Our results also show that the contribution to connectivity (as measured by dIIC) is fairly unevenly distributed among the different individual agricultural patches. The great majority of these patches had very low dIIC values and their reforestation may not significantly uphold forest species population persistence and dynamics. A few agricultural patches concentrated most of the importance for connectivity, with only eight and three patches in the high and very high importance categories in Pinares and Alto Tajo respectively. This highlights the need for an adequate identification and consideration of these critical patches in the reforestation plans and in the management of the spatial arrangement of forest habitats.

It should be noted however that in some cases the advantages of investing in connectivity improvement are uncertain in comparison with the simpler and more classical alternative of increasing the amount of habitat (Hodgson et al., 2009). Although most managers would not be satisfied with forest plans that are just based in a spatially-blind assessment, they face at the same time the problem of deciding the weight that connectivity considerations should have in the final management, given that these may be conflictive or not fully coincident with other management objectives and constrains. This problem is alleviated however by the use of metrics that, like IIC, do not only measure the connectivity between different patches but also account for the connected area existing within the forest patches (intrapatch connectivity) and the habitat characteristics therein (Pascual-Hortal and Saura, 2006). In this way the role of habitat amount is not set aside but integrated in a single currency and enriched metric that is able to provide the adequate weight to both alternatives in the final prioritization of landscape elements (Saura and Rubio, 2010). This includes the possibility of prioritizing the habitat patches just basing on the area they provide if this might be more appropriate in a particular situation. However, in our study we found that the individual area of the agricultural

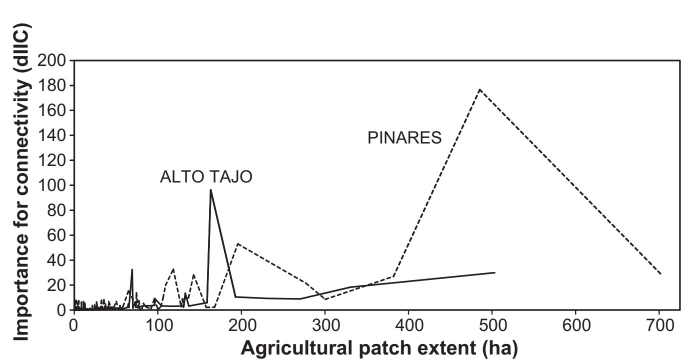

Fig. 4. Relationship between the area of each agricultural patch and the contribution to uphold forest connectivity in case it was reforested, as measured by the relative increase (%) in the Integral Index of Connectivity (dIIC). The figure represents the resulting values for a dispersal distance of 1000 m.

patches is not strictly related to their importance for connectivity. Therefore, the largest patches are not necessarily the most convenient for reforestation. For example, in the district of Pinares, the reforestation of a particular 119 ha patch would enhance forest habitat connectivity and availability more than a much larger patch with an area of 702 ha (Fig. 4).

In addition, it is important to note that the approach here proposed also contributes to significantly diminish the uncertainties of connectivity benefits by previously detecting the critical landscapes where forest planning should have connectivity as a major concern, instead of indiscriminately trying to incorporate connectivity considerations in the management of all the forest units irrespectively of the actual benefits that this might provide in each of them. Indeed, the proposed two-stage methodological process proved effective by being able to discard, in the first stage, many landscape units from further connectivity analyses that were later confirmed to contain only low importance agricultural patches. We found that all the key connectivity providers (as indentified in the second stage) were patches located within the landscape units that had been previously identified as critical. Therefore, this would allow that in a particular application connectivity is integrated with a sound basis but largely reducing the efforts of data gathering and analyses, and of integrating many (and potentially conflictive) criteria in the final planning. This is an important practical aspect given the time and budget constraints that usually affect many management plans, particularly in Mediterranean forests with low economic productivity.

Our results suggest that the methodology here adopted could be extended to a higher number of planning scales and hierarchical levels. Although the first stage was here applied at the level of forest districts, the first analysis step could as well comprise a broader scale (regional or even continental) strategic planning, as in Laita et al. (2010), Pascual-Hortal and Saura (2008), or Vergara et al. (2010). This has the advantage that, compared to the forest districts or landscape units here considered, the ecological flows and connectivity relationships are not constrained by the boundaries of the administrative or management units, which may not always correspond to actual barriers or limitations in the dispersal of species, particularly when broad spatial and temporal scales are considered. In turn, the results of such regional analysis are too coarse and generally do not match with the scale and spatial units at which forest management is really designed and implemented in practice. However, both approaches are not conflictive but complementary, with the regional planning being able to indicate in which (potentially only a few) forest districts connectivity should be incorporated as a relevant management objective, in which additional stages (the two here applied for the two Spanish districts) would be applied to refine the analysis and take it closer to the forest manager needs. In this respect, our results suggest that the variable importance of connectivity considerations for the forest management plans is not shown only at the level of individual landscape units, but also at the level of entire forest districts. In particular, the connectivity considerations (and limitations) seem to be more important in Pinares than in Alto Tajo, given the higher dIIC values obtained in the first district (Table 2). This is also reflected in the lower number of important or very important patches in Alto Tajo compared to Pinares (Fig. 3).

5. Conclusions and other possibilities

As mentioned before, the application of graph theoretical tools and improved habitat availability metrics is an innovative and suitable approach for reforestation planning. Although here we focused on a generic and multi-species forest connectivity assessment with two Mediterranean forest districts in Spain as case studies, the same approach could be applied to other areas or to particular forest species or habitat types. In addition, other habitat availability metrics like PC (Saura and Pascual-Hortal, 2007) could be used instead of IIC. The links in the forest network may also be characterized through effective distances that account for the matrix heterogeneity and resistance to species movements (Adriaensen et al., 2003; Drielsma et al., 2007; McRae et al., 2008; Pinto and Keitt, 2009; Theobald, 2006).

Additionally, we recognize that given the mutifunctionality of forest resources, the planning cannot just rely on the results of the methodology here proposed, which does not fully embrace all the relevant aspects that need to be considered in the decision making process. The integration of the outcomes of the connectivity analysis with those that inform about ownership, social and legal constraints and other management objectives is essential in order to achieve an ecologically beneficial and economically feasible selection as required for the sustainable use of forests. Moreover, the involvement of stakeholders is necessary during the exploration of alternatives and to guide the final decision making.

Transferring ecological connectivity concepts and methods into forest planning is advisable. Previous works (Bunn et al., 2000; Calabrese and Fagan, 2004; Ferrari et al., 2007) have remarked the potentiality of the graph theory for ecologically-based resource management. In this paper we have shown how the graph-theoretic approach can be useful for the selection of critical landscapes within forest districts in terms of connectivity. Likewise, the proposed methodology offers a sequential process for the efficient selection of patches to be reforested within planning units, matching with the scale at which the management is really applied. Land assignment in wildlife resource planning and landscape design with special attention to corridors and stepping stones can be favored by this methodology as well.

Acknowledgements

This work has been funded by the *Ministerio de Ciencia e Innovación* (Spain) through the research projects AGL2005-06648 ("Ecological bases at landscape level for designing PORFs: Pilot study on two Spanish representative forest districts", ECOPORF) and AGL2009-07140 ("Development of improved methods an quantitative tools for the analysis of landscape connectivity and their application to forest planning and monitoring", DECOFOR). Two anonymous reviewers made helpful comments on a previous version of this article.

References

Adriaensen, F., Chardon, J.P., De Blust, G., Swinnen, E., Villalba, S., Gulinck, H., Matthysen, E., 2003. The application of 'least-cost' modelling as a functional landscape model. Landscape and Urban Planning 64, 233–247.
Anderson, B., Jenkins, C.N., 2006. Applying Nature's Design: Corridors as a Strategy For Biodiversity Conservation. Columbia University Press, New York USA, pp. 231.
Bailey, S., 2007. Increasing connectivity in fragmented landscapes: an investigation of evidence for biodiversity gain in woodlands. Forest Ecology and Management 238, 7–23.
Beier, P., Majka, D.R., Spencer, W.D., 2008. Forks in the road: choices in procedures for designing wildland linkages. Conservation Biology 22, 836–851.
Blanco, J.C., Cortes, Y., 2007. Dispersal patterns, social structure and mortality of wolves living in agricultural habitats in Spain. Journal of Zoology 273, 114–124.
Bodin, Ö., Saura, S., 2010. Ranking individual habitat patches as connectivity providers: integrating network analysis and patch removal experiments. Ecological Modelling 221, 2393–2405.
Bowman, J., Jaeger, J.A.G., Fahrig, L., 2002. Dispersal distance of mammals is proportional to home range size. Ecology 83 (7), 2049–2055.
Bunn, A.G., Urban, D.L., Keitt, T.H., 2000. Landscape connectivity: a conservation application of graph theory. Journal of Environmental Management 59, 265–278.
Cadahía, L., López-López, P., Urios, V., Negro, J.J., 2010. Satellite telemetry reveals individual variation in juvenile Bonelli's eagle dispersal areas. European Journal of Wildlife Research (in press). doi:10.1007/s10344-010-0391-z d.

Calabrese, J.M., Fagan, W.F., 2004. A comparison-shopper's guide to connectivity metrics. Frontiers in Ecology and the Environment 2, 529–536.
Cantwell, M.D., Forman, R.T.T., 1993. Landscape graphs – Ecological modelling with graph-theory to detect configurations common to diverse landscapes. Landscape Ecology 8, 239–255.
Corry, R.C., Nassauer, J.I., 2005. Limitations of using landscape pattern indices to evaluate the ecological consequences of alternative plans and designs. Landscape and Urban Planning 72, 265–280.
Crooks, K.R., Sanjayan, M., 2006. Connectivity conservation: maintaining connections for nature. In: Crooks, K.R., Sanjayan, M. (Eds.), Connectivity Conservation. Cambridge University Press, New York, pp. 1–20.
Dale, M.R.T., Fortin, M.J., 2010. From graphs to spatial graphs. Annual Review of Ecology Evolution and Sytematics 41, 21–38.
Drielsma, M., Manion, G., Ferrier, S., 2007. The spatial links tool: automated mapping of habitat linkages in variegated landscapes. Ecological Modelling 200, 403–411.
ESRI, 2005. ArcGIS 9.1. Redwoods, California, USA.
Estrada-Peña, A., 2003. The relationships between habitat topology, critical scales of connectivity and tick abundance Ixodes ricinus in a heterogeneous landscape in northern Spain. Ecography 26, 661–671.
Fall, A., Fortin, M.J., Manseau, M., O'Brien, D., 2007. Spatial graphs: principles and applications for habitat connectivity. Ecosystems 10, 448–461.
Ferrari, J.R., Lookingbill, T.R., Neel, M.C., 2007. Two measures of landscape-graph connectivity: assessment across gradients in area and configuration. Landscape Ecology 22, 1315–1323.
Fu, W., Liu, S., Degloria, S.D., Dong, S., Beazley, R., 2010. Characterizing the "fragmentation-barrier" effect of road networks on landscape connectivity: a case study in Xishuangbanna, Southwest China. Landscape and Urban Planning 95, 122–129.
García-Feced, C., González-Ávila, S., Elena-Rosselló, R., 2008. Metodología para la tipificación y caracterización estructural de paisajes en comarcas forestales españolas. Forest Systems (Formerly Investigación Agraria-Sistemas y Recursos Forestales) 17, 130–142.
García-Feced, C., Saura, S., Elena-Rosselló, R., 2010. Assessing the effect of scale on the ability of landscape structure metrics to discriminate landscape types in Mediterranean forest districts. Forest Systems 19, 129–140.
Hodgson, J.A., Thomas, C.D., Wintle, B.A., Moilanen, A., 2009. Climate change, connectivity and conservation decision making: back to basics. Journal of Applied Ecology 46, 964–969.
Jordán, F., Baldi, A., Orci, K.M., Racz, I., Varga, Z., 2003. Characterizing the importance of habitat patches and corridors in maintaining the landscape connectivity of a Pholidoptera transsylvanica (Orthoptera) metapopulation. Landscape Ecology 18, 83–92.
Keitt, T.H., Urban, D.L., Milne, B.T., 1997. Detecting critical scales in fragmented landscapes. Conservation Ecology, 1.
Kuenkele, J., Von Holst, D., 1996. Natal dispersal in the European wild rabbit. Animal Behaviour 51, 1047–1059.
Lafortezza, R., Chen, J., Sanesi, G., Crow, T.R. (Eds.), 2008. Patterns and Processes in Forest Landscapes: Multiple Use and Sustainable Management,. Springer.
Laita, A., Monkkonen, M., Kotiaho, J.S., 2010. Woodland key habitats evaluated as part of a functional reserve network. Biological Conservation 143, 1212–1227.
Leitao, A.B., Miller, J., Ahern, J., Mcgarigal, K., 2006. Measuring landscapes. In: A Planner's Handbook. Island Press, Washington USA, pp. 248.
Lindenmayer, D.B., Margules, C.R., Botkin, D.B., 2000. Indicators of biodiversity for ecologically sustainable forest management. Conservation Biology 14, 941–950.
Loe, L.E., Mysterud, A., Veiberg, V., Langvatn, R., 2009. Negative density-dependent emigration of males in an increasing red deer population. Proceedings of the Royal Society B-Biological Sciences 276, 2581–2587.
McComb, B.C., 2007. Wildlife Habitat Management: Concepts And Applications In Forestry. CRC Press, Taylor & Francis Group, Boca Raton, Florida, USA, pp. 319.
McRae, B.H., Dickson, B.G., Keitt, T.H., Shah, V.B., 2008. Using circuit theory to model connectivity in ecology and conservation. Ecology 10, 2712–2724.
Ministerio de Medio Ambiente, 2002. Mapa forestal de España escala 1:50000. Madrid, España.
Neel, M.C., 2008. Patch connectivity and genetic diversity conservation in the federally endangered and narrowly endemnoc plant species Astragalms albens (Fabaceae). Biological Conservation 141, 938–955.
Pascual-Hortal, L., Saura, S., 2006. Comparison and development of new graph-based landscape connectivity indices: towards the priorization of habitat patches and corridors for conservation. Landscape Ecology 21, 959–967.
Pascual-Hortal, L., Saura, S., 2008. Integrating landscape connectivity in broad-scale forest planning through a new graph-based habitat availability methodology: application to capercaillie (*Tetrao urogallus*) in Catalonia (NE Spain). European Journal of Forest Research 127, 23–31.
Perotto-Baldivieso, H.L., Melendez-Ackerman, E., Garcia, M.A., Leimgruber, P., Cooper, S.M., Martinez, A., Calle, P., Ramos Gonzales, O.M., Quinones, M., Christen, C.A., Pons, G., 2009. Spatial distribution, connectivity, and the influence of scale: habitat availability for the endangered Mona Island rock iguana. Biodiversity and Conservation 18, 905–917.
Pinto, N., Keitt, T.H., 2009. Beyond the least-cost path: evaluating corridor redundancy using a graph-theoretic approach. Landscape Ecology 24, 253–266.
Rayfield, B., Fortin, M.-J., Fall, A., 2010. The sensitivity of least-cost habitat graphs to relative cost surface values. Landscape Ecology 25, 519–532.
Rothley, K.D., Rae, C., 2005. Working backwards to move forwards: graph-based connectivity metrics for reserve network selection. Environmental Modelling and Assessment 10, 107–113.
Saura, S., Pascual-Hortal, L., 2007. A new habitat availability index to integrate connectivity in landscape conservation planning: comparison with existing indices and application to a case study. Landscape and Urban Planning 83, 91–103.
Saura, S., Rubio, L., 2010. A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. Ecography 33, 523–537.
Saura, S., Torné, J., 2009. Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. Environmental Modelling and Software 24, 135–139.
Smith, M.A., Green, D.M., 2005. Dispersal and the metapopulation paradigm in amphibian ecology and conservation: are all amphibian populations metapopulations? Ecography 28, 110–128.
Sociedad Española de Ciencias Forestales, 2009. Avance del informe sobre la "Situación de los bosques y del Sector forestal en España". V Congreso Forestal de España. Ávila.
Stahl, P., Artois, M., Aubert, M.F.A., 1988. Organization spatiale et déplacements des chats forestriers adultes (*Felis silvestris* Schreber 1777) en lorraine. Revue d'Ecologie (Terre et Vie) 43, 113–132.
Sutherland, G.D., Harestad, A.S., Price, K., Lertzman, K.P., 2000. Scaling of natal dispersal distances in terrestrial birds and mammals. Conservation Ecology 4.
Tackenberg, O., Poschlod, P., Bonn, S., 2003. Assessment of wind dispersal potential in plant species. Ecological Monographs 73, 191–205.
Taylor, P.D., Fahrig, L., Henein, K., Merriam, G., 1993. Connectivity is a vital element of landscape structure. Oikos 68, 571–573.
Taylor, P.D., Fahrig, L., With, K.A., 2006. Landscape connectivity: a return to the basics. In: Crooks, K.R., Sanjayan, M. (Eds.), Connectivity Conservation. Cambridge University Press, New York, pp. 29–43.
Theobald, D.M., 2006. Exploring the functional connectivity of landscapes using landscape networks. In: Crooks, K.R., Sanjayan, M. (Eds.), Connectivity Conservation,. Cambridge University Press, New York, pp. 416–444.
Tischendorf, L., Fahrig, L., 2000. On the usage and measurement of landscape connectivity. Oikos 90, 7–19.
Urban, D., Keitt, T., 2001. Landscape connectivity: a graph-theoretic perspective. Ecology 82, 1205–1218.
Urban, D.L., Minor, E.S., Treml, E.A., Schick, R.S., 2009. Graph models of habitat mosaics. Ecology Letters 12, 260–273.
Vergara, P.M., Hahn, I.J., Zeballos, H., Armesto, J.J., 2010. The importance of forest patch networks for the conservation of the Thorn-tailed Rayaditos in central Chile. Ecological Research 25, 683–690.
Vittoz, P., Engler, R., 2007. Seed dispersal distances: a typology based on dispersal modes and plant traits. Botanica Helvetica 117, 109–124.
Vogiatzakis, I.N., Pungetti, G., Mannion, A.M. (Eds.), 2008. Landscape Series Vol. 9. Springer Publishing.
Zetterberg, A., Mörtberg, U.M., Balfors, B., 2010. Making graph theory operational for landscape ecological assessments, planning, and design. Landscape and Urban Planning 95, 181–191.