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Ecology of fragmented Natural Systems under Semiarid Conditions

Ecología de Sistemas Naturales fragmentados en Condiciones Semiáridas

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Ecology of fragmented natural systems under semiarid conditions

Ecología de sistemas naturales fragmentados en condiciones semiáridas

Dissertation submitted by **Mr. Víctor Manuel Zapata Pérez**

to obtain the PhD degree in Ecology by the University of Murcia with

international mention.

Director:

Dr. Francisco Robledano Aymerich

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A mis padres, que lo son todo...

Hubo árboles antes de que hubiera libros,

y acaso cuando acaben los libros continúen los árboles.

Y acaso llegue la humanidad a un grado de cultura tal que no necesite ya de libros,

pero siempre necesitarán de árboles,

y entonces abonará los árboles con libros.

Miguel de Unamuno

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GENERAL ABSTRACT

0. General Abstract

Biodiversity has evolved in the Mediterranean basin throughout geological history and along the different stages of human colonization. Different phases of enrichment related to tectonic and climatic phenomena, and subsequent processes of migration and refuge, have resulted in a diverse landscape that qualifies as one of the world's biodiversity hotspots. In recent decades, urban development processes have been promoted continuously throughout all this territory, taking advantage of its privileged climate and the uniqueness of the landscape and cultural heritage.

The interaction between these two types of development, natural and urban, is the origin of one of the main problems currently faced by Mediterranean biodiversity. In this regard, the southeast of the Iberian Peninsula has been no exception. Because of the predominance of a particular semi-arid climate and as a result of its lithological diversity, there are many species that have evolved in this area or have found there a niche in which to settle. Many of these habitats have been occupied directly by human constructions totally incompatible with their persistence. In other cases, what urban development has caused is the fragmentation of natural areas causing changes in local species composition and variations in their regional distribution areas.

If the focus is placed on the main actors of fragmentation of natural areas in the southeast of the Iberian Peninsula, it should not only be directed at urban processes, but needs also to consider other agents, such as agriculture. The consequences for ecosystems are very different depending on the type of fragmentation. Similarly, the consequences will also vary depending on whether the fragmented ecosystem is restored with species or communities well within their potential growing range or, on the contrary, man is trying to use species or life forms in conditions near the edge of their niche potential.

On this basis, the current thesis presents advances in the understanding of the biological communities that inhabit these fragmented areas, showing the effects of the fragmentation types on the biodiversity of natural patches depending on the dominant ecosystem type, studies the performance of some biological interactions in these systems as well as their possible use as passive restoration techniques, and finally, proposes management measures for the natural development and recovery of these areas.

In the current work, birds and woody vegetation have been chosen as bioindicators of the communities that develop in these forest fragments. In the case of birds, a great experience is needed for identification and recording, but at the same time, the information provided by them is very important because most species of this group are in the upper links of the food webs. In the case of vegetation, sampling is seasier, and its importance lies in being one of the basic structural

pillars on which other communities are organized, as well as one of the most important components of the landscape.

The studies that compose this thesis have been developed in forest patches of diverse nature that are distributed throughout the municipality of Murcia, located in the south east of the Iberian Peninsula (West Mediterranean). The dominant climate is semiarid Mediterranean with mean temperature of 18.6 °C and annual rainfall of about 300 mm, the latter with uneven distribution although tending to be concentrated in autumn-winter.

In this area, the reforestation policies implemented in the middle XXth century plagued the mountain with aggressive techniques like monospecific plantations of Aleppo pine (*Pinus halepensis* Miller). Semi-arid climatic conditions may be limiting for the development of this species, so the result are often underdeveloped forested areas with little representation from other plant species. This simplification of the ecosystem proves to be problematic at the ecological level, but also at the social level due the low landscape quality that confers to the restored areas, and their low appreciation by the public.

The current thesis is structured in 5 chapters that address the problem of fragmentation from different scales. Chapter 1 and 2 look for patterns and responses at local scale considering networks of forest patches. In this way, these chapters explore the influence of connectivity or isolation between patches and species responses. Chapter 3 extends the focus to the territorial scale studying group of patches of different physiognomy spread around the whole municipality, to determine the response of biodiversity regarding their intrinsic and extrinsic characteristics. Finally, chapters 4 and 5 provide a final, more detailed focus to the study of the dispersal process as a biological interaction, at the patch level. Additionally, these chapters seek to identify measures of potentially direct application.

The results of **chapter 1** demonstrate that there is no correlation between the legal status of a patch and its conservation value. In this way, greater values of conservation indexes were obtained in parches without or with a lower protection category than in other patches with regional or higher protection. Also, a negative correlation between floristic value and pine density was found, which suggests that the selective extraction of pines could be a measure to reach a self-sustaining forest formation. However, this negative relationship was not found for the birds, so a complementarity between indexes is evident, and should be regarded when adopting management or protection decisions.

Chapter 2 works at the same scale, and its main results show different effects of urban fragmentation depending on the biological group studied. In reference to this, vegetation appears as more vulnerable and sensitive to fragmentation. For birds, changes in community composition and diversity occur as a result of the edge effect created by urbanization. The effect will be different depending on the intrinsic characteristics of each patch. For that reason, at landscape level it is necessary to identify management measures that buffer or mitigate urban effects. Also, these guidelines should be considered in the creation of new urban areas that fragment Mediterranean landscapes.

At territorial scale, **chapter 3** displays greater conservation value for flora in Mediterranean shrubland patches than in forest ones. For birds, the results are different depending on the geographical or administrative scale of the index involved. For example, patches with steppic physiognomy or scrublands have greater conservation values for European indexes due to the scarcity of such landscapes in the Community territory and the consequent value of species associated to them. The opposite happens with national conservation indexes where steppe landscapes are more common, and forest species have greater value. On the basis of all this, management measures applicable to Mediterranean forest patches have been proposed. It is recommended to reduce the number of pine trees in those patches where their density is high (and usually unsuitable for the prevailing climate conditions). Also it is necessary to promote ecological interactions as passive restoration techniques and as a measure to connect nearby isolated forest patches. Besides, protection of non forested areas (open shrublands or scrub-woodlands) should be implemented to complete the landscape diversity protected under local and regional regulations.

Chapters 4 and 5 focus at a more detailed level, in search of deepening the knowledge about the dispersal of seeds of shrubs by birds in forest patches, and on the use of this interaction to restore degraded areas or to improve underdeveloped natural areas.

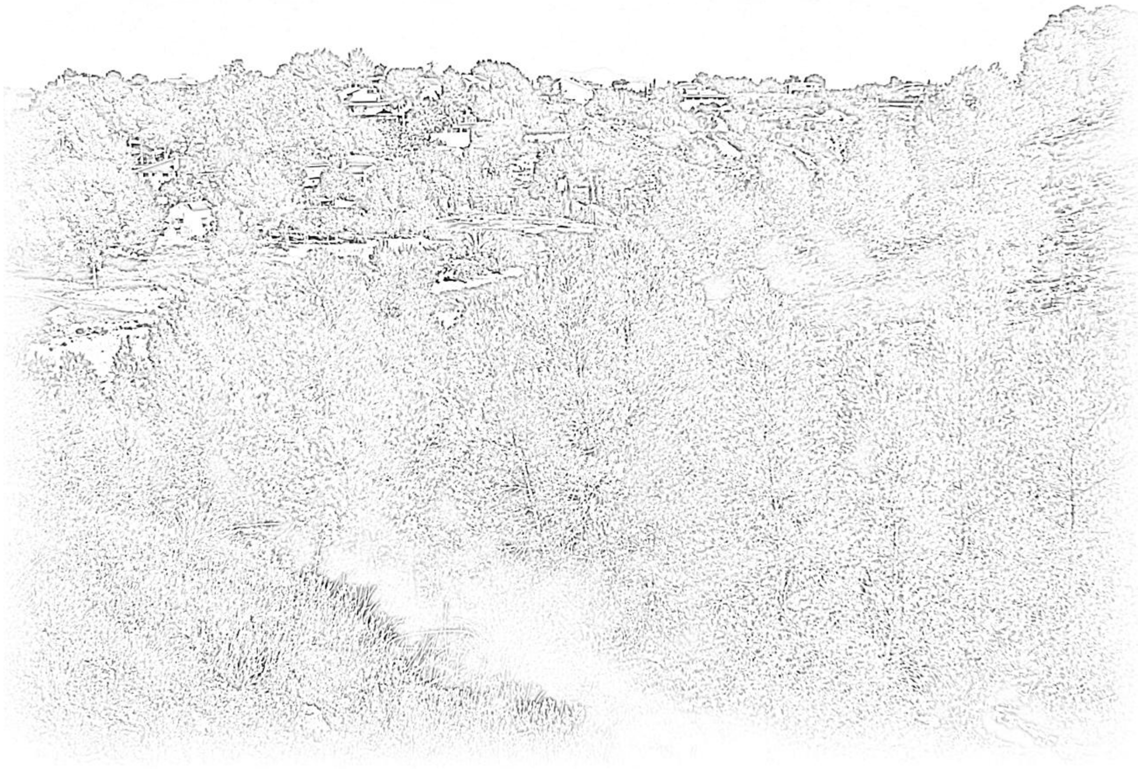
Chapter 4 focuses on the functioning of dispersion within the patch and in the role of *P. halepensis* trees as perches and later nurse plants of Mediterranean fleshy-fruited shrubs (*Rhamnus lycioides, Pistacia lentiscus, Asparagus albus*). Greater consumption was observed in *A. albus*. However, dispersion directed to the pines was higher for *R. lycioides* seeds. On the other hand, more than half of adult shrubs of the patch were under the influence of canopy cover (and much more when considering also the dead trees or stumps), despite canopy cover is very sparse. Hence, if conditions in these microhabitats are favorable for seedling and sapling growth, reducing pine density is supported again, but preferably eliminating individuals (among those weakened or decaying) with a shrub growing under their influence.

Finally, **chapter 5** pursues learning more about the direction of seed dispersal by birds, and about the performance of frugivory during the periods when seed production is limited. Higher production and seed consumption were observed in the interior of the studied patch, and preferent dispersion to specific places within it was found. Two alternative dispersion models in fragmented forest Mediterranean patches were proposed.

The first model proposes important interior consumption by adult and territorial individuals who have no need to move significantly (either outside the patch or within its limits). In this way, consumption and dispersal are concentrated in the interior of the patch. Only less dominant individuals who would be displaced to the peripheral areas could contribute to seed transport at larger distances (outside the patch).

The second model proposes a similar behavior in the interior of the patch, but in this case with higher influence of urban areas as attractors (i.e. sources of water and food resources). Directed dispersal of seeds of patch shrubs towards garden areas (that would act as sinks) is in this case ecologically irrelevant, while the risk of propagules of exotic species being transported into the patch is greatly increased. Again, a reduction of pine density in the patch is proposed as a way to contribute to a more homogeneous dispersal within it. It is also proposed to install artificial perches attractive to birds in peripheral zones, to favor their passive restoration.

GENERAL INTRODUCTION



1.1. General Introduction

Tropical and boreal forest cover approaches 30% of the world surface (Bonan, 2008) providing ecological, economic, social, and aesthetic services to natural systems and humankind (Hassan *et al.*, 2005). Although world forest distribution is skewed towards humid and temperate climates, dryer areas host some forest ecosystems with higher diversity of tree species (Scarascia-Mugnozza *et al.*, 2000), well adapted to high temperatures, low water availability and different edaphic conditions. In this context, a focus in the Mediterranean realm is pertinent. Mediterranean forests are not only highly diverse (Thompson, 1999; Myers *et al.*, 2000), but also particularly sensitive and exposed to climate change in the drier sections of the biome, in their transition towards desert climates (Esteve-Selma *et al.*, 2010).

Strong differences in forest coverage have been found between the north and south sides of the Mediterranean Sea. In the former, forest cover varies around 20-30% of the land, which is lowered to 5-10% in southern Mediterranean countries (Scarascia-Mugnozza, *et al.*, 2000). Land use and restoration policies are in part responsible for that difference. Focusing in the south of Europe, and especially in the south east of the Iberian Peninsula, forest coverage has experienced important increases due both to land abandonment and to intensive reforestation policies implemented in the middle of the XXth century (Esteve *et al.*, 2003; Maestre and Cortina, 2004). The ecological importance of such increases will be discussed later.

The value of forests for mankind is based on the exploitation of their wood productivity and other economic functions. The benefits derived from these products and services vary depending on the forest biome. For example, Bernard *et al.* (2009) described the economic importance of tropical forest in Costa Rica distributing human benefits among water supply, recreation and tourism, and maintenance of biodiversity. In the subtropical forests of Wulingyuan Natural Reserve (China), the value of forest services was quantified in terms water conservation, CO_2 sequestration, O_2 release, soil fertility conservation, environmental purification and forest recreation (Rong, 2004). In Swedish boreal forests, Zanchi *et al.* (2014) incorporated to the study of the economic value of forest ecosystems aspects such as productivity, carbon storage, wood production and water quality. Whatever the ecosystem type under consideration, research about ecosystem services attribute them to four general fields: carbon sequestration, watershed protection, biodiversity benefits and landscape beauty (Landell-Mills and Porras, 2002; Grieg-Gran *et al.*, 2005). Due to the specific climatic, lithological and biological characteristics of Mediterranean areas, and their intrinsic variability (Mazzoleni *et al.*, 1992), the focus of research must be brought on the share between economic, biodiversity and amenity values of Mediterranean forests in different areas.

Besides productive functions, people and the forestry sector are connected with forests trough tourism, recreation services or nature conservation (Kangas *et al.*, 2008), but the reality is that current studies about the problems faced by Mediterranean forests focus mainly on internal forest characteristics and on the functioning on which forest services depend. The main areas of research are forest biodiversity conservation, forest sustainability, forestation, regional planning, forestry industry, and risk and uncertainty (Díaz-Baltero and Romero, 2008). In a broader context, a set of methods to solve forest management problems, grouped under a Multiple Criteria Decision Making (MCDM) system (Maroto *et al*., 2013), has been applied in many non-Mediterranean areas (Ananda and Herarth, 2003; Kazana *et al*., 2003; Nordström *et al*., 2009). In the case of Mediterranean areas, the criteria, consequences and action plans have been defined by Maroto *et al*. (2013) (Figure 1.1). As it will be seen later, not all these action plans are applicable to all types of Mediterranean forests, especially when fragmented and poorly developed forest areas are considered.

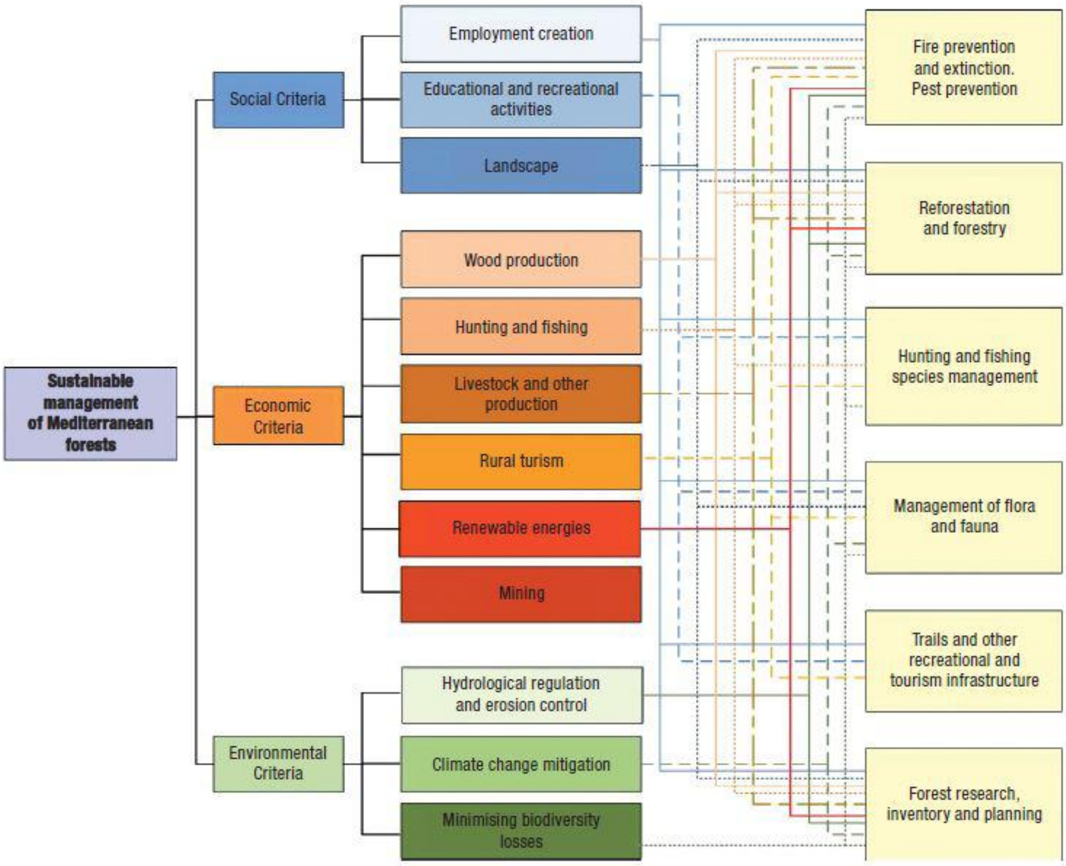


Figure 1.1: Scheme of criteria, consequences and action plans for Mediterranean forests (Maroto *et al*., 2013)

Championing the cause of the prominent values and importance of forest areas, international reforestation policies, whose financial investment has been immense, have been developed with varying success. In Asian countries like China, Laos, Vietnam or Indonesia, similar forest land policies have been implemented including settlement programmes, land classifications, devolution of forest management and reforestations schemes (Clement & Amezaga, 2008). Chazdon (2013) defined four types of forest regeneration, ranging from strict natural succession to artificial plantations. Although defined for tropical areas, they could be properly applied to any forest policy worldwide, being related with: a) spontaneous natural regeneration; b) assisted natural regeneration; c) agroforestry;

d) commercial tree plantations. To these could be added another type, defined as intensive non-commercial tree plantations. This fifth class refers to some reforestations, which have been quite usual in semiarid areas during the XXth century, especially in the south east of the Iberian Peninsula. Such reforestations, driven by policies aimed at promoting rural employment and preventing soil degradation, monospecific plantations (in this case with *Pinus halepensis* Mill.) have been a common practice in rural landscapes (Zapata and Robledano, 2014). As a result, in these zones *P. halepensis* is now more widespread than its ecological niche would suggest (Esteve *et al*., 2003; Chirino *et al*., 2006) with many potential effects on biodiversity at local and landscape levels (Díaz *et al*., 1998; Reino *et al*., 2010; Shochat *et al*., 2001).

In Mediterranean areas reforested conifer forests have been considered low-quality habitats when compared to open areas (Vance *et al*., 2007; Mateos *et al*., 2011). Monospecific pine plantations cause decreases in diversity and conservation value of some groups like terrestrial Gastropoda (Torre *et al*., 2014), spiders (Zapata *et al*., 2012) vegetation (Zapata and Robledano, 2014) or vertebrates (Moreira and Russo, 2007). Also, effects on internal soil processes like the amount of organic matter, mineralization rates or C sequestration have been found (Carreira *et al*., 1996; Martínez-Mena *et al*., 2002; Rashid and Ryan, 2004; Almagro and Martínez-Mena, 2012).

Another element to be considered, when dealing with forest formations in the south east of the Iberian Peninsula, is global change, and particularly climate change. Its effects are expected to be enhanced through two key features affecting Mediterranean ecosystems: limited water supply and unpredictable rainfall (Peñuelas *et al*., 2004). Even if summarizing all recent research about the effects of global change on Mediterreanean biodiversity is an impossible task to take, some examples of effects on forest species can be cited. Trees are particularly affected by climate change due to their longevity, for example, genetic change to adapt to the new conditions is not possible (Kramer *et al*, 2000). In other groups such as birds, changes are expected in their distribution and geographical ranges, phenology, or demographic factors (Crick, 2004).

Under such scenario, the effects of climate change on under-developed trees and weakened forest formations, make it even more difficult to maintain these monospecific forest formations. In semiarid *P. halepensis* reforestations, reduced water availability will place environment conditions definitely out of the species' ecological niche, especially when these conditions were already close to the edge of it.

Therefore, there is a need to manage these monospecific reforestations in order to: i) Improve biodiversity at all scales and levels of organization; ii) Change the present monotonous landscape to a heterogeneous, visually more attractive, Mediterranean semiarid scenery; iii) Raise the social appreciation for these areas. As superimposed on figure 1, under the current state of the ecosystem, of the three criteria described by Maroto *et al*. (2013), the economic one is difficult to improve, and therefore, research and management should focus on improving the values and services represented by the other two criteria (Figure 1.2).

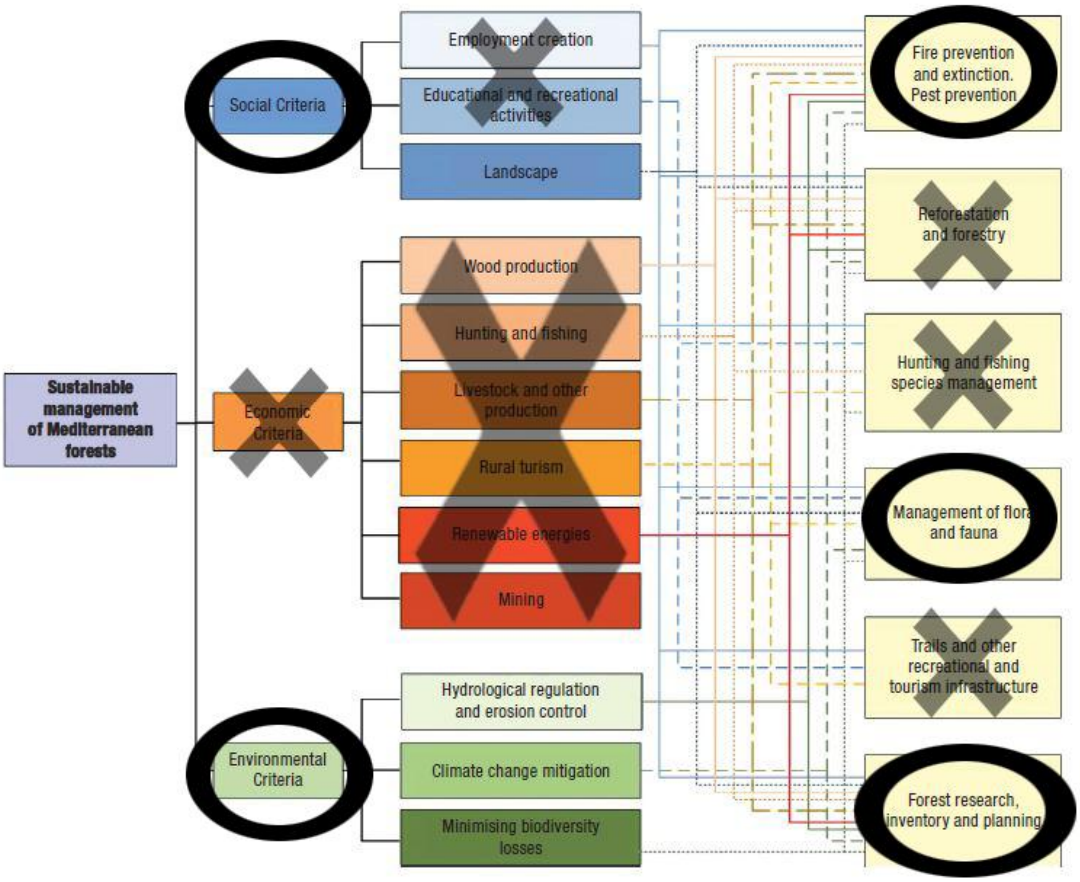


Figure 1.2:Priority fields of action according to the current state of monospecific semiarid forest patches. Modified from Maroto *et al* (2013).

From a social perspective, it is difficult to create new employment or to try to improve the educational or recreational activities in these areas nowadays. Therefore, management should focus at re-directing the landscapes towards a diverse Mediterranean woodland or shrubland. This change is made feasible by the actions derived from environmental criteria.

Normally, these forests are small patches without important soil degradation or hydrological problems associated, but the pthe hysiognomy resulting from old reforestations causes poorly developed shrub and herbaceous strata (due to biotic interactions), with the risk of localized erosion processes (physical effects). Besides, the effects of climate change can be considerable due to the already restrictive environmental conditions for actual species, which predicts a decline of biodiversity in these habitats in the mid-term. Therefore, the utility of ecological approaches lies in their contribution to restructuring these forest habitats and to developing management techniques that will add compositional value to them.

The current thesis deals about fragmented forest systems located in the semiarid Mediterranean domain of the Iberian Peninsula. The recognition of this sector of the Mediterranean basin as one of the world hotspots of biodiversity relies on its high species richness and endemicity of several animal and plant groups like vascular plants (Groombridge, 1992; Lobo *et al.*, 2001; Mota *et al.*, 2004), water beetles and general freshwater biodiversity (Abellán *et al.*, 2005; Abellán *et al.*, 2007), spiders (Murienne and Giribet, 2009), amphibians and reptiles (Sillero *et al.*, 2009), as a result of a complex interplay of geological, biogeographical, historical and human factors (Blondel *et al.*, 2010).

Semiarid Mediterranean areas are considered as frontiers of change between European and North African climates, and have acted as such historically as paleoclimatic studies show (Vaks *et al.*, 2006). Influences from the north and south make semiarid Mediterranean areas of the south east of the Iberian Peninsula biogeographical crossovers, either as glacial refuge of species (Hewitt, 2001; Tzedakis *et* al., 2002; Carrion *et al.*, 2003; Hampe *et al.*, 2003) or as a corridor during the Messinian salinity crisis (Hsü *et al.*, 1977), allowing local taxon diversification and species exchange between Europe and Africa. Climate and ecological niche similarity between North Africa and the south east of the Iberian Peninsula allowed the establishment of some of these African species in the semiarid Spanish Mediterranean contributing to its present biodiversity, further enriched by human intervention in more recent times (Dobson, 1998).

In the same way that Mediterranean biological richness and ecological diversity is high, the occupation of the landscape by man throughout history has also been continuous and intense (Blondel, 2006) thereby increasing human pressures (Hewitt, 2001). The problem is that while traditional land-use systems have contributed to preserve and evolve even higher levels of biological and ecological diversity (Blondel and Aronson, 1999; Blondel *et al.*, 2010) current models of human growth and development associated with residential, commercial and infrastructural uses, have drastically altered the landscape and habitat setting (Zapata *et al.*, 2012). Models predict that in the XXIth century, land use changes will be one of the most important drivers of change at global scale with intermediate effects in Mediterranean areas (Sala *et al.*, 2008).

Focusing on urbanization as one of the current agents of land use change, many negative effects to biodiversity can be identified. The main one results from the net reduction of natural surface area and its isolation by built areas, which can be considered a case of extreme fragmentation since the boundaries are very narrow and there is a steep gradient in environmental characteristics between the urban matrix and the natural remnants (Crooks *et al.*, 2004). The edge area between urban and natural habitats is expected to be the most sensitive to negative impacts originated in the former. Some studies restrict such effects to a band of 40-50 m of urban influence (e.g. Chen *et al.*, 1993, 1995), but other research shows different distances of effect stabilization depending on the variable under study. For example, soil temperature stabilization occurs at shorter distances (Davies-Colley *et al.*, 2000) while some microclimatic variables like humidity or wind speed need more than 200 m for urbanization effects to disappear (Chen *et al.*, 1995). The edge effects can also be noticed in the forest structure since alterations in canopy, light penetration, distribution of foliage and vertical stratification have been observed (Heithecker and Halpen, 2007).

Another impact of urban fragmentation is a structural simplification and biotic homogenization, leading to a similar species composition in all or most the natural patches embedded in a urban matrix (McKinney, 2006; Schwartz *et al*., 2006; Olden, 2006; Wang *et al*., 2014). This can respond, for example, to the removal of shrubs and dead wood and the consequent increase in grasses and herbs (Marzluff and Ewing, 2001), which has a negative effect on birds and other animal taxa (Savard *et al*., 2000).

In addition to the general effects of urbanization, there are specific ones that occur when fragmentation is caused or intensified by the construction of roads. Direct effects on vegetation result from the physical occupation of soil by an impermeable layer which is totally unsuitable for plant establishment, although side effects are also expected both in the form of microenvironmental changes, and the spread of exotics. However, road impacts (direct mortality and barrier effects) influence mainly the fauna, from small invertebrates to large animals (Smith-Patten and Patten, 2008; Carvalho and Mira, 2011) even causing local extinctions in small mammals (Rico *et al*., 2007). Regarding birds, forest patches fragmented by roads experienced a reduction in their occurrence, forest species being the most affected (Brotons and Herrando, 2001).

Since human transition from foraging and hunting to farming and herding, agricultural expansion in Mediterranean areas has had a considerable effect on biodiversity (Zeder, 2008). Given the historical pace of the development of this transformation, some species have adapted to agriculture, leading to biodiversity increases in some cases. In more recent times, and especially in the north of the Mediterranean basin, an inverse process is occurring which causes a polarization of agricultural landscapes between extreme scenarios of abandonment and intensification (Nainggolan *et al.*, 2012). This will affect biodiversity in different ways depending on the taxocenoses under consideration (Moreira and Russo, 2007; Robledano *et al.*, 2014).

In an enumeration of threats to Mediterranean biodiversity, exotic and invasive species would be the next. Due to continuous human presence with high population density in the Mediterranean Basin, agriculture and forest exploitation have been important activities (Gritti *et al*., 2006), leading to frequent introductions of exotic species (National Research Council, 2002). The introduction of exotic species is also associated with gardening and pet ownership, linked with urbanization activities. The most invasive animal exotic species of the Mediterranean Basin are related with that origin, like the Common Slider (*Trachemys scripta elegans* (Schoepff)) or the Rose-ringed Parakeet (*Psittacula krameri* (Scopoli)) (DAISIE, 2009). Regarding plants, garden species represent 58% of total invasive plant species in Europe (Pyšek *et al*., 2009) among them the Tree of Heaven (*Alianthus altissima* (Mill.)) or the Pampas Grass (*Cortaderia selloana* (Schult. & Schult.f.)) (DAISIE, 2009)

In the strict sense fire cannot be considered an impact in Mediterranean areas, although at least since the early Holocene (Ali *et al.*, 2003) it has been a crucial process controlling ecosystem structure and dynamics (Retana *et al.*, 2002; Baeza *et al.*, 2007). But human activity has increased fire recurrence in the Mediterranean. The alteration of drought events caused by climate change (Pal *et al.*, 2004), has increased the risk of wildfires in Mediterranean Europe (Lloret and Marí, 2001; Mouillot *et al.*, 2002).The influence of fire is greater in Mediterranean areas than elsewhere, due to the fact that Mediterranean-type vegetation represents one of the world's most fire-prone biomes (Bond *et al.*, 2005). Measures to prevent fires must be imposed through forest and land use policies, and their control or suppression should be based on fire-fighting plans out of the scope of this work. Although the present thesis does not address fire prevention or management directly, it is felt that its focus on achieving diverse and resilient forest formations points in that direction.

Most impacts and threats listed previously for Mediterranean ecosystems are equally applicable to their semiarid areas. Loss of natural cover due to the expansion of intensive agriculture and urbanization, fragmentation from infrastructure building, overexploitation and mismanagement of water resources, and soil degradation, are presently widespread in semiarid areas. With the aggravating circumstance of the slow resilience of these ecosystems subjected to greater environmental constraints. Any human action that threatens the weak ecological balance of semiarid ecosystems, will increase the impacts previously described.

Nowadays, science and society have pointed out these impacts in semiarid landscapes and many research and restoration actions have been performed to tackle some of these problems, in cases with considerable financial support. In this thesis neither the success of these actions, nor the way they have been performed, will be discussed. Instead, we stress that studies on the ecological interactions to assist ecosystem restoration, and that the use of ecosystem services to reduce its cost, should have a bigger role in future applied research projects. Within these, dispersion and facilitation appear as the most important interactions that can help ecological restoration in the Mediterranean.

Among plant-plant interactions, facilitation has widely studied, as the main process with direct application in restoration actions. In fact, the survival and performance of plants in Mediterranean-type ecosystems usually improve when associated with neighbors (Maestre *et al.*, 2001, Gómez-Aparicio *et al.*, 2004), as shown by the positive association of seedlings and saplings of tree species and shrubs (Gómez *et al.*, 2003, Gómez-Aparicio *et al.*, 2004) and *vice versa* (Zapata *et al.*, 2014). Under the influence of nurse (facilitator) species, conditions for establishment and survival are better than in bare ground due to improved soil fertility (Moro *et al.*, 1997), microclimatic amelioration (Franco and Nobel, 1989) and greater water availability (Raffaele and Veblen, 1998). For these reasons, this is an important phenomenon in semiarid regions. Besides tree-shrub or tree-tree facilitation processes, grass-shrub interactions seem to play an important role for some Mediterranean fleshy-fruited shrub species (Maestre *et al*., 2003).

For facilitation to occur, a propagule must arrive to a favorable microsite (shrub or tree canopy, in most cases). This makes necessary to introduce the concept of seed dispersal effectiveness which can be expressed as the "number of seeds dispersed by a dispersal agent multiplied by the probability that a dispersed seed produces a new adult" (Schupp *et al* 2010). The second part of the definition refers to the quality of the microsite, and the first one to the activity of the dispersal agent. Related with it, in Mediterranean areas bird frugivores play a key role as dispersal agents for plant species that benefit from a scattered mode of seed deposition (Zapata *et al.* 2014). The use of this ecosystem service in semiarid Mediterranean areas lacks the amount of ecological knowledge that supports its application in other environmental contexts (García *et al*., 2010). The current thesis makes an effort to clarify the role of this ecosystem service in order to promote its integration in the future management measures for fragmented semiarid forest areas.

To summarize, human development in Mediterranean areas has led to the reduction and fragmentation of natural areas. Initiatives to counteract such impacts have paradoxically led to arboreal reforestations extending to semiarid landscapes with limiting environmental conditions for tree growth. Further fragmentation and the degradation of the land matrix threatens the survival of species by decreasing suitable habitat, impeding genetic exchange, or increasing direct mortality. Currently, management measures are needed to increase the ecological value of these secondary landscapes, through the structural and functional improvement of the natural remnants integrated in them, the maintenance of functional connectivity and the enhancement of the matrix permeability. These measures should be as based as much as possible in natural processes, taking advantage of the passive restoration forces that the environment provides, in the form of ecosystem services. In urban-dominated landscapes, such services are linked to the concept of green infrastructure, defined as "*Spatially and functionally integrated systems and networks of protected landscapes supported with protected, artificial and hybrid infrastructures of built landscapes that provide multiple, complementary ecosystem and landscape functions to the public, in support of sustainability*" (Ahern, 2011). For that purpose, the general aim of this thesis is to provide ecological basis for improving the biodiversity of fragmented forest patches in Mediterranean semiarid areas that are currently subject to anthropogenic pressures like urbanization or agriculture. In particular, the specific objectives that develop this overall goal, can be organized in an ascending range of scales:

At a local scale, urbanization effects represent the most immediate impacts to fragmented natural areas. Thus, the local (**patch**) level research and management objectives of the study are:

- To study the biodiversity (birds and woody flora) of reforested natural patches under semiarid Mediterranean conditions from a compositional and structural perspective.

- Similarly, to investigate the effects of the type of fragmentation (total or partial) from urban development.
- To evaluate the conservation importance of these fragmented areas through multispecific ornithological and floristic value indexes, and to determine the best structural conditions (dominance of shrubs *vs* trees) which maximize biological value depending on local patch conditions
- To apply research results to the internal design of new urban developments in order to minimize their negative effects of green spaces or neighbouring natural areas.

At the scale of **patch networks**, the objectives are:

- To study the response of biodiversity in relation to the proximity between patches, their size and connectivity, in order to evaluate the importance of these factors and its potential application to future urban development policies.
- To assess the fragmentation role of different human activities surrounding forest patches, with respect to biodiversity conservation, focusing on the filtering capacity of urbanization and the differences between urban and rural (agricultural) activities.

At the **landscape** and **territorial** scale, the objectives are:

- To assess the importance of ecosystem type (shrublands, scrub woodlands or forest areas) in regeneration of all kinds of patches and nearby ones, and landscape influence on woody flora and birds.
- To propose a list of territorial measures which should guide urban ordinance enhancing the value of the semiarid Mediterranean landscapes, and minimizing negative impacts of urban developments in the fragile stability of Mediterranean ecosystems.

Finally, in relation with the **ecosystem services**, the objectives are:

- To improve the knowledge about the functioning of plant regeneration processes in natural patches of fragmented semiarid Mediterranean areas, focusing on fleshy-fruited Mediterranean shrubs dispersed by frugivore birds and the factors limiting.
- To apply this knowledge to the restoration and improvement of biodiversity of isolated patches, coupling landscape-scale measures with other management proposals identified, at different scales, in other sections of this work

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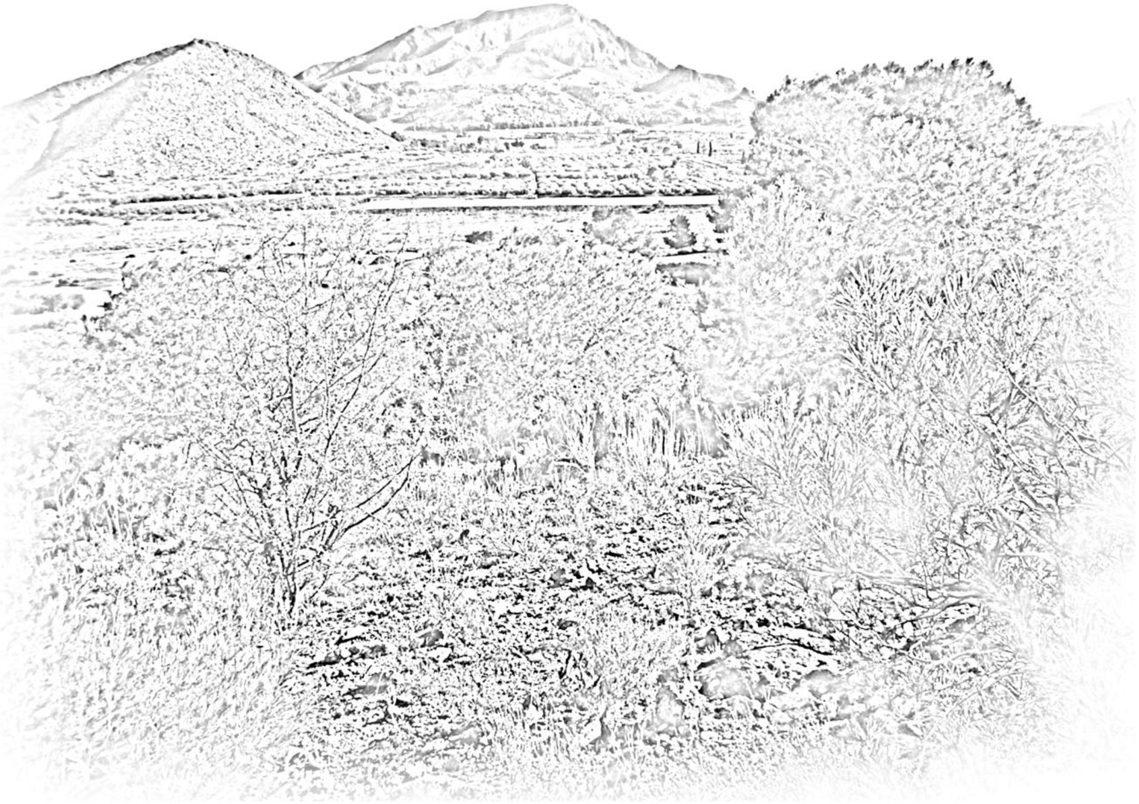
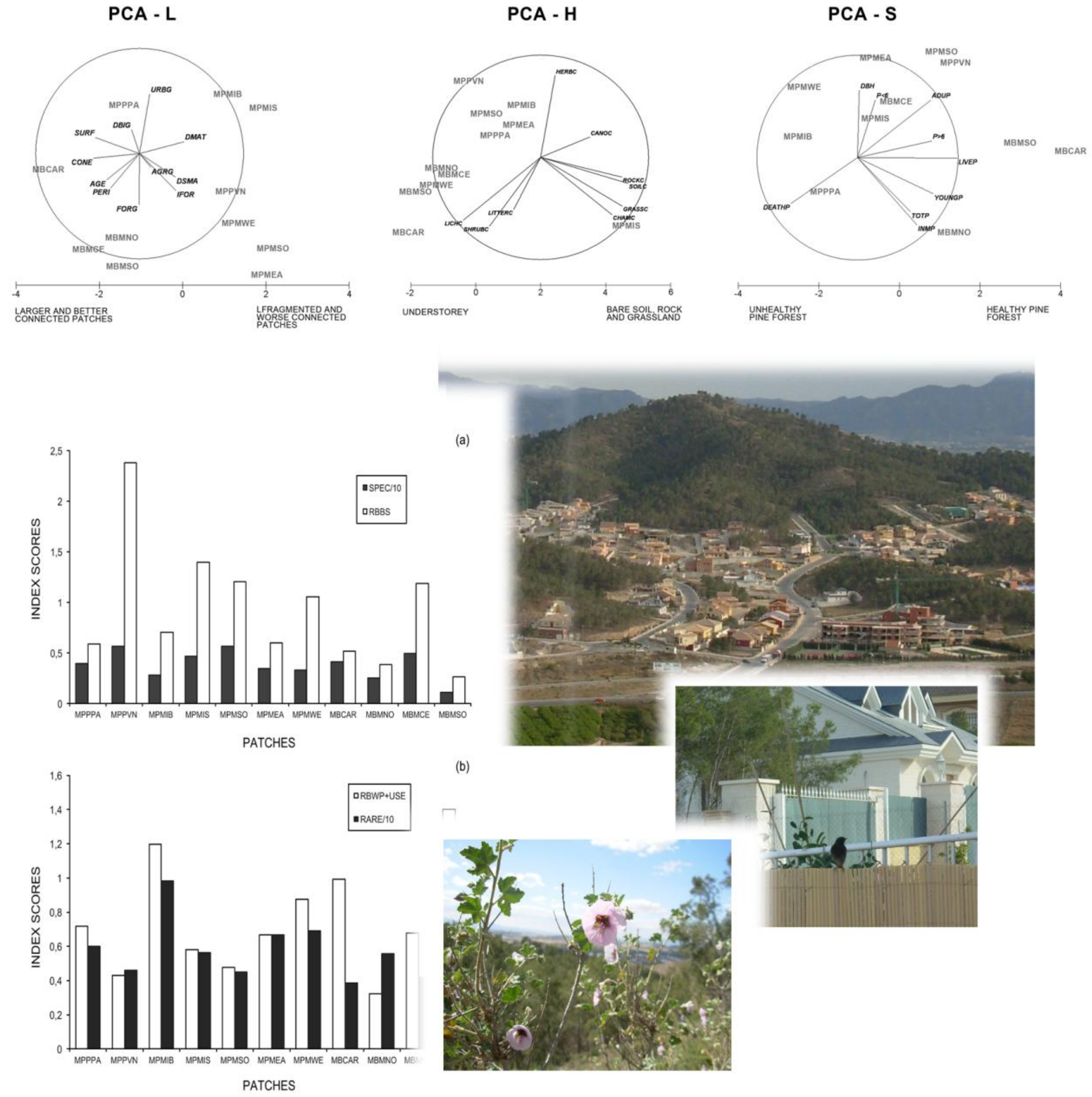
CHAPTER 1



Assessing biodiversity and conservation value of forest patches secondarily fragmented by urbanization in semiarid southeastern Spain

Graphical Abstract



Abstract

Semiarid Mediterranean areas patchily forested with *Pinus halepensis* Mill. are often affected by urbanization plans, which influence the conservation of biodiversity in the remaining fragments. We assessed forest biodiversity in eleven *P. halepensis* patches of the municipality of Murcia (SE Spain), on the basis of the abundance, richness and conservation value of birds and woody flora. Under the hypothesis that biodiversity is conditioned by pine density, modulated by other patch features (age, physical substrate, surrounding land uses), mixed regression models were used to relate biodiversity indicators with environmental gradients, detected through principal component analyses (PCAs) performed on three sets of variables: landscape (L), habitat structure (H), and stand development (S). PCA-L related patch size and connectivity to the abundance of typical woodland birds and to floral and bird species richness. PCA-H opposed shrub cover associated with *Sylvia melanocephala* Gmelin., to rock and dry grassland cover, a feature of moderately disturbed areas preferred by *Lanius senator* L. PCA-S associated taller trees and better physical conditions with canopy dwelling and soil probing bird species, respectively. The bird or plant conservation value of patches did not generally match their protection status, which was confirmed by the analysis of a larger sample of 36 natural patches. A negative relationship between floristic value and pine density suggests that reducing the density of reforestations can lead to more diverse and self-sustaining wooded formations. In contrast, conservation value for birds did not decrease with reforestation, stressing the need to use ornithological and floristic indexes as complementary assessment tools. The results seem useful for building a municipal network of reserves connecting higher status protected areas.

2.1. Introduction

The distribution patterns of species are determined by the environmental conditions prevailing in their respective geographical areas (Andrén, 1994; Santos et al., 2002). These conditions can be altered by natural disturbances that cause important changes in the habitats of species. The result is a heterogeneous landscape driven by natural processes (Hansson et al., 1995), where biodiversity is the expression of the suitability of the landscape mosaic for species of different ecological requirements. The situation is complicated in Mediterranean areas, whose landscape pattern is strongly influenced by human activities (clearing, grazing…), equivalent to some extent to natural disturbances (fire, wild herbivores). Traditional agroforestry mosaics can sustain highly diverse biological communities (Sitzia et al., 2010), but their abandonment or intensification can lead to the loss or degradation of valuable habitats (Underwood et al., 2009). In addition, Mediterranean habitats and species are exposed to a combination of two adverse factors: limited water supply and unpredictable rainfall, both expected to be exacerbated by climate change (Peñuelas et al., 2004).

In south-eastern Spain, a representative region of Western Mediterranean semiarid areas, reforestation with *Pinus halepensis* Mill. has been a common practice in rural landscapes. *P. halepensis* is now more widespread than its ecological niche would suggest (Esteve et al., 2003; Chirino et al., 2006), with many potential effects on biodiversity at local and landscape levels (Díaz et al., 1998; Reino et al., 2010; Shochat et al., 2001). These effects have been little studied in semiarid Mediterranean Spain (López and Moro, 1997; Maestre and Cortina, 2004).

Where artificially expanded *P. halepensis* plantations meet limiting environmental conditions, they can show a naturally patchy configuration and reduced (shrub-like) growth (Bonet, 2004), both enhanced by traditional pastoral and agricultural activities. The secondary fragmentation of these patchy forests due to urban development can be considered a case of extreme fragmentation since the boundaries between urban and forest areas are very narrow and there is a steep gradient in environmental conditions (Crooks et al., 2004). However, the responses of the different habitat variables will vary greatly depending on the characteristics of the forest-urban interfaces.

For many plants and animals, native species richness decreases with increasing building density along a rural-urban gradient (Hansen et al., 2005). Critical thresholds are reached when the intensity of urbanization reduces the minimal area required by some species, or prevents the recolonization of patches by those with restricted dispersal. Moreover, the relationship between species abundance and urbanization is often not linear since many species peak at intermediate levels of development (Hansen et al., 2005), which further complicates our understanding of landscapes influenced by urban developments and the management of forest remnants within them.

Vegetation and birds have often been selected as biodiversity response indicators in forest ecosystems. Vegetation is an easily observable and recordable component that provides information on the structure and composition of the forest and is closely related to the faunal component (Kati et al., 2004). Bird sampling requires significant prior expertise, but once acquired, it delivers information on the structural preferences and differential responses of species to several human stressors affecting forest patches (direct disturbance, fragmentation, land use changes in the surrounding matrix), This paper seeks to interpret the current status of biodiversity in forest fragments located along urban-rural gradients, capturing and synthesizing a complex range of conditions with gradient analysis techniques (McDonnell and Hahs 2008). The specific objectives are: (i) to assess the biodiversity value of these urban-fragmented forests through the characterization and evaluation of discrete spatial units (patches) from a structural and compositional point of view; (ii) to use the urban gradient approach (*sensu* McDonnell and Hahs 2008) to relate biodiversity indicators (birds and woody flora) to patch characteristics (structure, development status and landscape context). Our hypothesis is that pine density, modulated by other patch characteristics (age, physical substrate, surrounding land uses) conditions forest biodiversity, expecting higher biological richness and conservation value in areas with intermediate density (e.g. self-thinning) or moderate disturbance. These objectives seem relevant for nature conservation in municipalities whose protected area network is primarily the result of compensations for urban developments in rural areas, where decisions on which areas should be protected (site selection), and how to manage them (tree density), should be based on sound biodiversity assessments.

2.2. Material and methods

2.2.1. Study area

The study was conducted in two sections of the municipality of Murcia, the main city of the Autonomous Community of Murcia in the south-east of Spain (Figure 2.1). Both sections (Montepinar and Majal Blanco) are mid- to low altitude landscapes, traditionally devoted to agriculture or agro-forestry, and undergoing a process of gradual urbanization during the last three decades (1980's to 2010's). The fragments studied are *P. halepensis* plantations, representative of recent historical attempts (late XIXth-early XXth century) to restore arboreal cover in semiarid south-eastern Spain (Esteve et al., 2003). Although they can host remnants of native pine cover, in most cases density and site preparation techniques make them deviate from natural conditions. They can be considered test areas on which to conduct research that can be extrapolated, in the future, to a number of similar forest areas which are being donated to the Municipality to establish new nature reserves. They were selected in order to include six protected forest patches and five satellite fragments with different regimes of use and management (hereafter referred to as 'patches'). Although part of the protected patches lie within areas of higher protection status (Regional Parks and sites of Community Importance under Habitats Directive EEC 92/43), the majority are designated at most as Municipality Forest Parks.

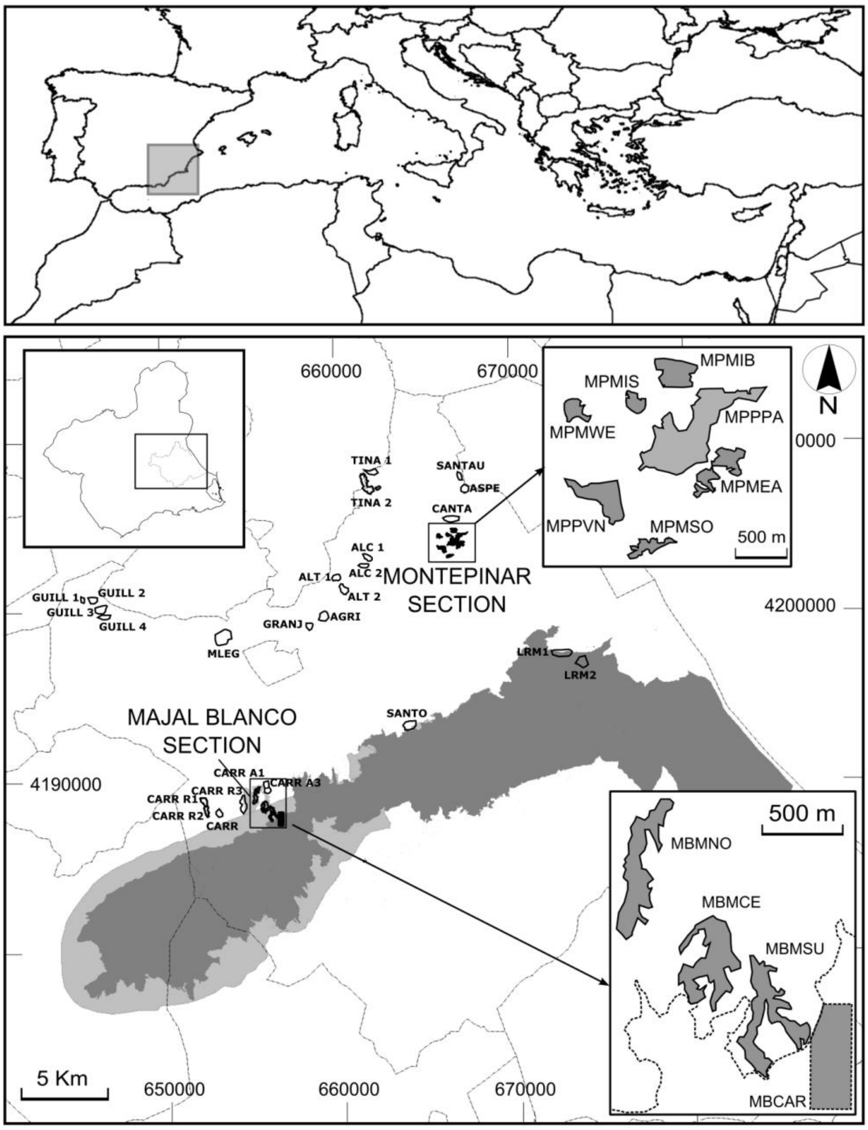


Figure 2.3: Location of the two sections studied (Forest Parks and associated forest fragments) in the Municipality of Murcia, also located within the Region of Murcia and the Iberian Peninsula/Mediterranean in the left and uppermost boxes. The patches studied (solid polygons, with acronyms as in Table 3) are enlarged in the zoomed windows to show their relative size and spatial configuration (except for MBCAR which is only partially represented). Other polygons correspond to forest patches proposed as future Forest Parks. Also shown are the main Natural Protected areas included in the Municipality: Site of Community Importance (dark grey core area) and Regional Park (surrounding light grey area, plus the surface overlapping the former). Coordinates refer to the Universal Transverse Mercator (UTM) grid.

In the Majal Blanco section none of the patches studied was totally isolated. There, the configuration of the landscape mosaic together with the persistence of corridors between the fragments and the continuous forest habitat, superimpose a gradient of connectivity on that of urban influence. In Montepinar there are cases of total isolation caused by urban development, and other in which the patches are embedded in rural habitats (agricultural, pasture or scrubland). Consequently, several types of patch isolation are found, depending on the main fragmentation cause, which can be strictly urban, agricultural, or even natural (a lithological, edaphic or topographic limitations to forest growth

2.2.2. Sampling methodology

Vegetation was sampled using circular plots of ca. 100 m^2 (set out in the field with a 5.64 m rope attached to a nail), in which the presence of all woody species was recorded in order to determine their frequency of occurrence. Annuals and biennials were not sampled as well as gramineae, to overcome identification problems which would have impeded a rapid survey, but were quantified as cover (see later). Plots were distributed systematically in a grid or transect, with a separation of 50 m on either direction. The exact number was coarsely proportional to the size of the patch, modulated by shape as well as by soil and topographic heterogeneity. Patches larger than 10 ha generally had 9 or more plots, and those smaller usually 5-7. An exception was MPMIB (12 plots), due to its greater topographic complexity. A significantly greater sampling effort (36 plots) was devoted to MBCAR due to its high soil and topographic variety. We counted all Aleppo pines located within each plot to calculate its mean density, and determined the height (in m) of each individual, its age and the presence of cones, from which we derived different stand descriptors. We also recorded the presence of dead trees and stumps. Finally, we sampled the linear coverage of all physical and vegetation layers (rock, bare ground, scrubs, chamaephytes, perennial grasses, litter, mosses and lichens, and annuals) as their projection on a tape extended along four 10 m transects oriented towards the cardinal points. The values were expressed as mean percentage.

Bird sampling was conducted through 5 min point counts (Bibby et al., 1992) in predetermined locations, scoring all the visual or auditive contacts with birds, with no limit of detection (Blondel et al., 1981). Plot centres were located 150 m apart from each other and at least 100 m from the edge of the patch. Since in practice the detection of most forest species took place within 50 m, this reduced the possibility of overlap among areas covered by adjacent points. Coreau and Martin (2007), in a study using the same method to sample songbirds in structurally comparable habitats, state that birds are typically recorded at less than 100 m and most within 50 m, and discard the occurrence of spatial autocorrelation in species richness even with sampling points closer than100 m and on average 140 m apart. In our closest sampling plots, in the events of birds flying between them, we discounted these individuals from the subsequent sample.

Like vegetation plots, the number of bird sampling points was approximately proportional to the size of the patch, in combination with topographical heterogeneity (slope, exposures). Larger patches (> 10 ha) generally contained 3-4 points, the exact number depending on internal topographic heterogeneity (lowest in MBMNO, with only 2 points), and the smaller ones 1-2 points. Sampling was carried out during the first four hours of daylight. Two visits were made to each patch during the breeding period (the first one between April 15th and May 15th, and the second between May 15th and June 15th). For each patch, the mean number of birds recorded per plot (averaged among the two sampling dates) was used as an index of abundance. According to Rost et al., (2010) detectability was expected to correlate with raw abundance, so we are aware that species present in low numbers could have passed undetected in some plots, rendering richness values also indicative. Sampling was carried out by both authors in all cases. To overcome the potential limitations arising from the restricted sample of patches, we made an extensive survey of other forest areas of the Municipality, with a similar sampling procedure, recording the density of *P. halepensis* and the variables used to generate bird and woody plant conservation indexes. A total of 25 additional patches or equivalent parts of continuous forest landscapes were surveyed in 5 additional sections, resulting in a total sample of 36 patches in 7 sections (Figure 2.1; Appendix 1). 'Sections' are loosely defined landscape complexes whose forest patches share general ecogeographical characteristics (same physiographic unit, substrate, land use pattern…), distinctive with regard to patches belonging to other complexes. Since the study focuses on the influence of human uses on fragmented forest landscapes, only sections external or marginal to natural areas with a high level of protection (Regional Parks, NATURA 2000 sites) have been considered.

2.2.3. Assesment of conservation value

The ornithological and floristic interest of each fragment was assessed by calculating an index of conservation value obtained as the summatory of the average abundance of each species recorded in a patch multiplied by a numerical value corresponding to its category of protection (adapted from Pons et al., 2003). For more detail on its calculation see http://www.conama10.es/conama10/download/files/CT%202010/1000000201.pdf. Bird species were ranked according to two conservation assessments (Table 2.1): i) SPEC categories (Species of European Concern) as reported in the 'Birds in Europe' assessment (Birdlife International, 2004); ii) IUCN threat status from the Red Data Book of the Birds of Spain (Madroño et al., 2004). On the basis of these ranks, two indexes were constructed (SPEC and RBBS), their categories and corresponding numerical values being presented in Table 2.1. In the case of flora, the conservation ranks considered were the categories of threat proposed in the Red Data Book of Protected Wild Plants of Murcia Region (Sánchez-Gómez et al., 2002) and the rarity of the species according to Sánchez-Gómez and Guerra (2007), to generate three indexes (Table 2.1): RBWP (based on the threat status), RBWP+USE (value of RBWP index plus one point, in the case of species whose exploitation may be subject to management measures), and RARE (based on the degree of regional rarity). We could not generate an index based on any national or European ranking for plants because none of the species sampled was included in conservation assessments at these scales. The species were given importance values depending on their position in each of the rankings above described (Table 2.1).

	BIRDS		FLORA	
VALUE	**SPEC**	**RBBS**	**RBWP**	**RARE**
0	-	NOT EVALUATED	-	VC
1	NON-SPEC	NT, LC, DD	SMS	C
2	-	VU	SI	X
4	SPEC-3	-	-	R
8	SPEC-2	-	-	VR

Table 2.1. Score assigned to each category in the ranks of conservation importance for birds and flora, used in the calculation of conservation value indexes. NT: Near threatened, LC: Least concern, DD: Deficient data, VU: Vulnerable, SMS: Species whose exploitation may be subject to special management measures, SI: Special interest, VR: Very rare, R: Rare, X: Uncommon, C: Common, VC: Very common.

2.2.4 Statistical analyses

Principal Component Analysis (PCA) was used to reduce the main intrinsic habitat characteristics of the fragments and the variables that characterize their landscape relationships into a few interpretable gradients (Soliveres et al., 2011). The PCA was performed using the correlation matrix. PCA is a commonly used technique to describe the response of species or other biodiversity features to complex environmental gradients (e.g. Blair, 2004; Fernández-Juricic, 2004; Coreau and Martin, 2007; Mangas et al., 2008). Three subsets of variables were summarized through separate PCAs (Coreau and Martin, 2007). The variables included in the analyses, presented in Table 2.2, were grouped into: (1) general landscape characteristics, including patch age; (2) structural variables describing patch vegetation characteristics; and (3) stand development variables (tree mortality, regeneration). Since the variables were in different units of measurement, a correlation-based PCA analysis was used, performed with PRIMER v6 (Clarke and Gorley, 2006).

CODE	Description/units	PCA groups
SURF	Surface (km^2)	1 (PCA-L)
PERI	Perimeter	(Landscape scale variables)
IFOR	Shape Index (Perimeter/Area)	
DSMA	Distance to patches < 30 ha (km)	
DBIG	Distance to patches >30 < 100 ha (km)	
DMAT	Distance to patches >100 ha (km)	
CONE	Connectivity with the continuous forest habitat. (Index of increasing connectivity, with range 1-3)	
URBG	Degree of urbanization in the patch surroundings (% of the patch perimeter bordered by urban areas)	
AGRG	Agricultural surface in the patch surroundings (% of the patch perimeter bordered by cultivated land)	
FORG	Non-arboreal forest surface in the patch surroundings (% of the patch perimeter bordered by cultivated land)	
AGE	Age of forest (Index of increased time since plantation or existence, with range 1-3)	
SOILC	Bare ground cover (%)	2 (PCA-H)
ROCKC	Rock cover (%)	(Habitat structural variables)
LITTERC	Litter cover (%)	
GRASSC	Grass cover (%)	
CHAMC	Chamaephyte cover (%)	
SHRUBC	Shrub cover (%)	
LICHC	Moss and lichen cover (%)	
HERBC	Annuals cover (%)	
CANOC	Crown cover (%)	
TOTP	Max (live+dead) pine tree density (number/ha)	3 (PCA-S)
LIVEP	Live pine tree density (n/ha)	(Stand development variables)
DEATHP	Dead pine tree density (n/ha)	
YOUNGP	Young pine tree density (n/ha)	
INMP	Immature pine tree density (n/ha)	
ADUP	Adult pine tree density (n/ha)	
P<6	Density of pines < 6 m high (n/ha)	
P>6	Density of pines > 6 m high (n/ha)	

Table 2.2: Description and grouping of the variables included in the PCA analyses. Connectivity (CONE) scaled as 1: Patch completely disconnected from other forest area, 2: Patch in contact with another forest area in at least 50% of its perimeter. 3: Patch in contact with another forest area through wooded connections along more than 50% of its perimeter; Time since plantation or existence (age of forest) scaled as 1: Recent reforestation (< 50 years), 2: Old reforestation (>50 years), 3: Natural forest or ancient reforestation (Source: http://cartomur.imida.es/visorcartoteca).

Besides summarizing environmental variables, PCA was used to produce predictors (explanatory variables) that were statistically independent. Following Blair (2004) we explored the variation of biodiversity indicators along these PCA gradients through regression analyses. In these analyses, PCA axes were used as independent variables (predictors), and biotic variables or indexes as dependent (response) ones. Mixed model analyses (LMMs) were performed with the *lme* function of the *nlme* package (Pinheiro *et al.,* 2013) of the freely distributed “R” statistical software (R Development Core Team, 2009). Acknowledging the nested spatial structure of the study area (patches within sections), and to minimise spatial autocorrelation problems (Betts et al., 2006), the highest level of spatial aggregation (‘section’) was included as a random factor in the models and the biotic variables of interest as fixed factors (Vergnes et al., 2012). The latter included abundance values (density or frequency) of the different species of birds and woody flora, bird species richness, woody plant species richness (total and within particular life forms), and the conservation value indexes. LMMs have proven useful for studying the univariate response of species to vegetation gradients in Mediterranean forest habitats (Pons et al., 2012). Table 2.3 shows the main characteristics of each patch studied, summarized through the descriptive statistics of some relevant compositional and structural variables.

Code	Woody plant sps richness	Plant sampling plots	Bird species richness	Bird sampling points	Surface (ha)	Max pine density (n/ha)	Live pine density (n/ha)	Bare ground (%)	Chamae-phyte cover (%)	Scrub cover (%)	Crown cover (%)
MPPPA	27	25	22	4	30.72	624.00	236.00	35.76	18.53	2.78	13.81
MPPVN	17	7	22	2	8.70	485.71	485.71	39.41	10.50	1.00	46.70
MPMIB	22	12	10	1	6.77	500.00	241.67	33.91	22.81	0.00	19.81
MPMIS	22	6	11	1	2.23	616.67	416.67	36.40	19.69	1.00	24.42
MPMSO	15	6	12	1	2.9	550.00	533.33	39.96	8.46	0.52	41.79
MPMEA	19	6	13	1	6.4	383.33	366.67	35.29	20.42	0.08	36.13
MPMWE	23	5	13	1	2.7	380.00	300.00	23.15	19.80	5.50	29.70
MBCAR	36	36	18	3	28.00	494.44	472.22	27.54	29.11	7.82	34.00
MBMNO	26	9	14	2	15.36	555.56	555.56	55.11	11.30	0.69	38.34
MBMCE	24	9	24	3	15.49	400.00	400.00	27.27	7.06	1.83	45.96
MBMSO	37	9	17	3	14.27	742.86	728.57	29.05	17.34	8.21	43.71
MEAN ± SE	24.36 ± 6.99	-	16 ± 4.90	-	-	521.14 ± 112.64	430.58 ± 146.82	34.8 ± 8.62	16.82 ± 6.75	2.68 ± 3.07	34.03 ± 10.96

Table 2.3: Main descriptive characteristics concerning biological features and habitat structure of the patches studied. Maximum pine density refers to the standing plus dead individuals (i.e. all the individuals once planted or growing in the patch). Live pine density refers to those surviving at the time of sampling. Patch acronyms as in Figure 1, with the first two letters identifying them as belonging to Montepinar (MP) or Majal Blanco (MB) section, combined with an abbreviation of some distinctive feature of their location or visual characteristics (only for map labelling purposes).

A larger dataset of 36 forest patches was analyzed through linear regression models (LMs), using live pine tree density as explanatory variable and conservation value indexes as response ones. Also within this larger sample, Kruskal-Wallis non-parametric tests were used to compare the values of floristic and ornithological conservation indexes, using two characteristics of patches as grouping factors: protection status, with 3 levels (none, municipal, regional or higher) and reforestation, with two (reforested with *P. halepensis* or not). Visual examination of box-plots and the post-hoc test '*kruskalmc*' (from the '*pgirmess*' R package; http://giraudoux.pagesperso-orange.fr/) were used to determine the sign of paired differences.

2.3. Results

2.3.1. Environmental gradients and their associated ecological and landscape features

The principal components (Figure 2.2) can be considered as new variables accounting for much of the variance in a multidimensional data set. The most important components -usually the first two ones- can be interpreted as correlations of other underlying factors (in ecological studies, a physical or environmental gradient). In our case, the first components of PCA-1, 2 and 3 -hereafter PCA-L, PCA-H and PCA-S (for 'Landscape' 'Habitat Structure' and 'Stand development', respectively)- were selected (Soliveres et al., 2011), not only because they accounted for more than one third of the total variance of each PCA (PCA-L: 37.7%; PCA-H: 36.2%; PCA-S: 41.5%), but also because of their clear ecological interpretation (see below).

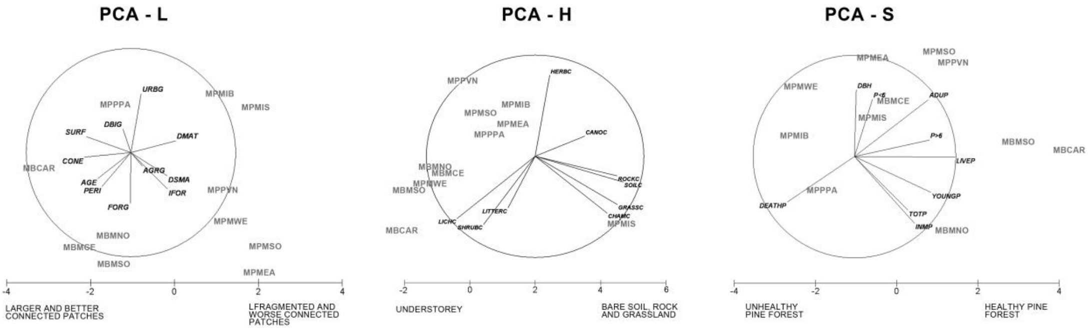


Figure 2.2: Representation of the three main gradients represented by the first PC (horizontal axis of each biplot) of the Principal Component Analyisis summarizing each set of patch variables ('L' for 'Landscape scale', 'H' for 'Habitat structure' and 'S' for 'Stand development'). Patch acronyms as in Figure 2.1 and Table 2.3, and codes of variables as in Table 2.2.

The first axis of the PCA performed on landscape variables (PCA-L) displays a gradient of size and connectivity with larger patches on the negative side as well as those more tightly connected with the forest matrix. Towards the opposite end, increases the distance to continuous forest or to other patches (Figure 2.2). In fact, this reflects that smaller patches tend to lie farther and to become isolated from larger ones with higher probability.

The first axis of the PCA performed on habitat structural variables (PCA-H) summarizes the cover of physical and vegetation layers, and depicts a gradient of understorey *vs* rock, bare soil and grassland (on the opposite side). Lichen and shrub coverage load on the negative side (Figure 2.2). This gradient may also reflect the types of soils that remain as forest habitat, either as protected areas (forest parks) or as green spaces within urbanizations, since more 'natural' patches tend to lie on the negative side while more 'urban' ones do on the opposite one.

With higher positive loadings on the first axis of PCA-S (Figure 2.2), a number of variables -Live pine tree density, Adult pine tree density, Young pine tree density, Pines over 6 m high- can be interpreted as measures of stand development (in terms of vitality and age structure). On the positive side of this axis we find two patches belonging to Majal Blanco section, the older but more dynamic of those studied, as well as the two Montepinar patches with best edaphic conditions. Considering the variables that load on the positive side, these patches represent a more healthy pine forest with internal regeneration. Among the Montepinar patches, all of them located on the negative side, we can differentiate MPPPA, MPMIG and MPMIP (patches of various size totally isolated by urban development or bordered by it in a high proportion). Dead pine tree density load on this side reinforcing the interpretation of this axis as a gradient of pine forest vitality. These patches represent pinewoods outside their ecological optimum. The remainder of Montepinar section corresponds to 'satellite' patches more indirectly impacted by urbanization and located in an intermediate position along the urban-agroforestry gradient.

	PCA-L			PCA-H			PCA-S		
Variable	p	(Sign) Coef.	% DE	p	(Sign) Coef.	% DE	p	(Sign) Coef.	% DE
Aegithalos caudatus (Long-tailed Tit)							***	(+) 0.330	29.31
Columba palumbus (Woodpigeon)	*	(-) 0.510	61.31						
Carduelis chloris (Greenfinch)	*	(+) 0.206	24.68	*	(+) 0.164	18.12			
Lanius senator (Woodchat Shrike)	*	(+) 0.350	23.21	***	(+) 0.352	42.03			
Motacilla alba (White Wagtail)				*	(+) 0.447	39.11			
Parus cristatus (Crested Tit)	*	(-)1.509	0						
Passer domesticus (House Sparrow)				*	(+) 0.182	14.65	**	(-) 0.365	45.62
Sturnus unicolor (Spotless Starling)	*	(-) 0.285	23.91						
Sylvia melanocephala (Sardinian Warbler)				*	(-) 0.284	48.59			
Sylvia undata (Dartford Warbler)	*	(-) 0.766	52.88						
Upupa epops (Hoopoe)							*	(+) 0.479	40.74
Bird species richness	*	(-) 1.827	19.33						
Woody plant species richness	***	(-) 0.118	51.53						
Nanophanerophyte species richness	***	(-) 0.470	44.43	**	(-) 0.638	16.68			
Shrub species richness	*	(-) 0.314	65.41						

Table 2.4: Gradients identified by the first axis of each PCA and summary of linear mixed models explaining their effect on biodiversity indicator variables (birds and woody flora). Only significative responses (p < 0.05 = '*'; p < 0.01 = '**'; p < 0.001 ='***') are shown, together with the percent deviance explained (%DE). 'Sign' denotes positive or negative relationships of variables with PCA gradients.

Regarding the mixed model analyses, table 2.4 shows the biotic variables related to the gradients summarized in PCs. Among the 26 forest bird species recorded, the abundances of 11 (see Table 2.4 for common names) responded to at least one of the gradients, but none of the composite indexes of conservation value did so. Given the relatively small number of patches and sampling plots, however, caution is needed when interpreting the responses of the less abundant species, e. g. *Columba palumbus* L., *Motacilla alba* L., *Sylvia undata* Boddaert. and *Upupa epops* L., all with less than

2 individuals/10 ha on average. Species of conservation concern are not necessarily rare, in any case (e. g. the SPEC-2 *Lanius senator* L. had a mean abundance of 8.5 individuals/10 ha). Bird and woody plant species richness were negatively related with PCA-L, as did the abundance of *S. undata*, *Parus cristatus* L. and *C. palumbus* (Table 2.4). By contrast, species like *L. senator* or *Carduelis chloris* L., positively related with this axis, seem to be favoured by heterogeneous mosaic patches (smaller and more isolated by other landscape types). Both species also correlated positively with PCA-H, favoring open, rocky or herbaceous areas with sparse trees (Table 2.4). The same response was shown by more antropophilous generalists like *M. alba* and *Passer domesticus* L. By contrast, the abundance of *Sylvia melanocephala* Gmelin. was negatively related to PCA-H. PCA-S was associated with three species: on the positive side with *U. epops* and *Aegithalos caudatus* L., both favoured by more dynamic pine patches displaying a better tree condition and age structure; and on the opposite side only with the generalist *P. domesticus* (Table 2.4).

2.3.2 Assesment of biodiversity conservation value

Although the aggregated calculation of conservation indexes (a single value per patch) prevents a statistical validation based on the reduced sample, there are obvious differences, both between and within sections, depending on the taxa on which the assessment is based (Figure 2.3).

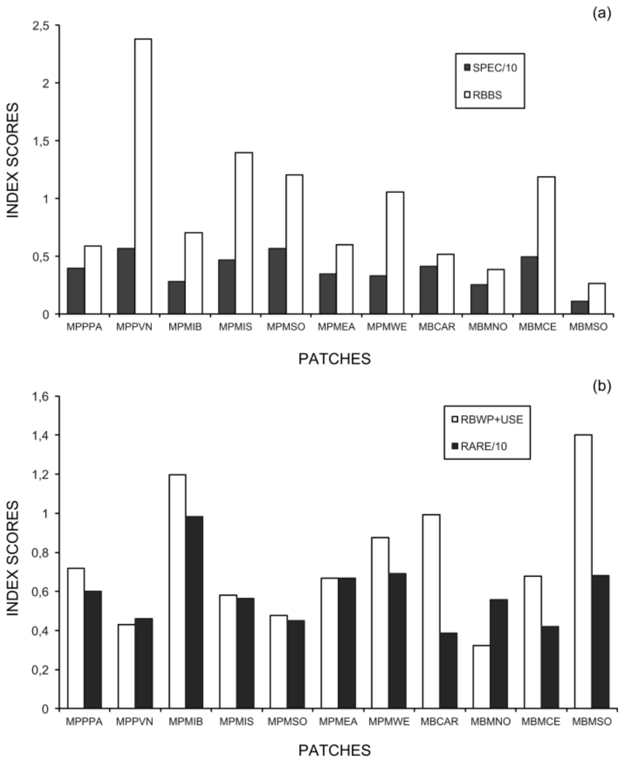


Figure 2.3: Graphical comparison of the conservation value of patches for the two taxocenoses considered. a) Bird-based indexes: Birds in Europe categories (SPEC) and Spanish Red Book (RBBS); (b) Indexes based on flora: Murcia Region Red Book (RBWP+USE) and Regional Rarity (RARE). The index with higher scores within each pair has been divided by 10 to facilitate comparisons. Patch acronyms as in Figure 2.1 and Table 2.3.

The conservation value of a patch for birds (Figure 2.3a) does not match its level of protection (Appendix 1). Isolated patches like MPPVN, MBMCE or MPMSO, with a low level of protection (or none), reached higher values of the SPEC index than MBCAR, which has the strongest protection (Site of Community Importance, Regional Park and Municipality Forest Park). For vegetation-based indexes (Figure 2.3b), despite the evident differences from bird-based ones, the pattern looks alike: patches with low protection levels (MPMIB, MBMSO) exceed the conservation value of those with higher levels of protection. Examples for the RBWP+USE index are MPMIB (immersed in an urban matrix), or MBMSO (connected with the core area of Majal Blanco), both with greater value than the forest matrix itself. Regarding the RARE index, many patches with low protection status surpass the conservation value of the core area (MBCAR). Species in category X (uncommon) which includes those not fitting any of the other categories (Sánchez-Gómez and Guerra, 2007), could bias the results, although the concordance between the two floristic indexes suggests that no such bias exists. When ornithological and floristic value indexes are compared across the larger sample of patches, using protection status as grouping factor, the above patterns are confirmed. There are not significant differences in any of the indexes either for flora or birds (Kruskal-Wallis test, $p>0.05$), except in the case of the RBWP index (Regional > Municipal, None; Kruskal-Wallis $H=8.08$, $p=0.017$). Thus, for all but one of the five indexes, patches with weak or absent protection status had a comparable value to sites afforded regional or higher protection status. In contrast, when pine reforestation was the grouping factor, significant differences were found between patches reforested with *P. halepensis* for two floristic indexes, RARE (Kruskal-Wallis $H=5.19$, $p=0.022$) and RBWP+USE (Kruskal-Wallis $H=8.83$, $p=0.002$), with Non reforested > Reforested patches, and also for the two ornithological ones, SPEC (Kruskal-Wallis $H=8.43$, $p=0.003$) and RBBS (Kruskal-Wallis $H=4.74$, $p=0.02$) but with opposite sign (Reforested > Non reforested patches).

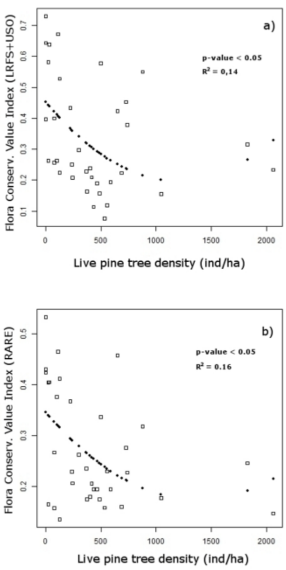


Figure 2.4: Scatterplot and fitted regression line of the relationship between the indexes of conservation value for flora and the density of live pine trees in the 35 patches surveyed in the Municipality of Murcia.

The above results point to a negative relationship between the botanical conservation value and the density of pine trees, confirmed through the regression analysis performed on all the 36 patches surveyed (Figure 2.4), with both indexes decreasing significantly with increasing density of live pines (p < 0,05). This negative relationship is also evident in the subset of patches of Montepinar (Figure 2.5), where live pine tree density explains a high proportion of the variation in the conservation value of flora.

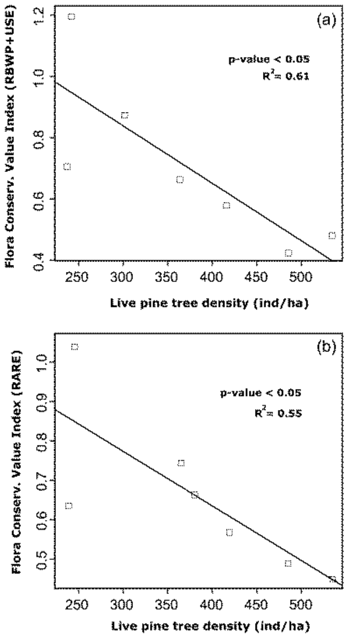


Figure 2.5: Scatterplot and fitted regression line of the relationship between the indexes of conservation value for flora and the density of live pine trees (Montepinar section).

Regarding birds, their relationship with reforestation is more complex, since the tree stratum can have some purely structural value for them. Pine formations have often some shrub-like physiognomy, when growing under limiting conditions (Bonet, 2004) or in young stages of reforestations (Reino et al., 2010). Such formations give also some chance to fleshy-fruited shrubs to regenerate under the tree canopy, providing food resources for birds in addition to protective cover. The two sets of indexes have in any case to be regarded as complementary (Appendix 1). When considering the larger sample of patches (N=36), significant correlations among them are always negative, while redundancy (positive correlation) is more frequent among indexes based on the same taxa (Table 2.5).

	Floristic indexes			Ornithological indexes	
	RBWP+USE	RBWP	RARE	SPEC	RBBS
RBWP+USE		0.52**	0.80***	-0.58***	-0.62***
RBWP			0.41*	-0.32 MS	NS
RARE				-0.53***	-0.64***
SPEC					0.73***
RBBS					

Table 2.5: Spearman rank correlations between floristic and ornithological indexes. Values of the correlation coefficient (rho) are shown together with significance codes (p-value: ‘*’ < 0.05; ‘**’ < 0.01; ‘***’ < 0.001; ‘MS’ <0.1; NS=non significant).

2.4. Discussion and conclusions

The influence of urbanization and associated fragmentation on diversity has been established for many animal and plant groups (e.g. Germaine and Wakeling, 2001; Hansen et al., 2005). However, the fragments of a formerly continuous habitat cannot be attributed to a poor conservation status without a detailed survey and multi-taxa assessment (Barbaro et al., 2005). To be useful in nature management, evaluations should take into account the main factors explaining the distribution of species. Key factors influencing biodiversity are patch size and connectedness with the forest matrix, as found in studies about fragmentation of urban forests (Fernandez-Juricic, 2004), oak woodland (Santos et al., 2002) and pinewoods (Díaz et al., 1998). Extreme fragmentation can eliminate core habitat, whose persistence increases bird species richness, allowing the persistence of patch interior species (Bianconi et al., 2003), in our case *C. palumbus, P. cristatus* and *S. undata* (positively associated with PCA-L.). *Sturnus unicolor* Temminck. is a ubiquitous species that apparently favours larger patches simply due to its dispersion from nearby urbanized areas. The negative relationship of *S. undata* with PCA-L suggests a stronger effect of urban fragmentation on this species, apparently unable to stay as breeder in pinewoods <30 ha.

Plant species richness (particularly that of nanophanerophytes and smaller shrubs), is also negatively associated with PCA-L, showing that larger fragments retain species that are lost in smaller ones. But, besides size and connectedness, it is important to consider the vegetation structure in terms of density and coverage of different strata (Chirino et al., 2006). Intra- and inter-patch heterogeneity favour species richness both at local and landscape scale. Scrubland can play an important role in semiarid ecosystems (Mangas et al., 2008). The negative relationship of *S. melanocephala* with PCA-H illustrates its preference for areas with greater nanophanerophyte species richness, indicative of protective or nesting cover (López and Moro, 1997), and of food resources (fruit) at the end of the breeding season (Jordano, 1985).

Regarding PCA-S, the association of *A. caudatus* with well developed pinewoods (taller and healthier trees) was expectable for a canopy-dwelling species. Healthy pine plantations have also better soil physical properties and their management (e.g. clearing of understorey) can provide enough bare or short grass areas to allow species like *U. epops* to feed efficiently (Barbaro et al., 2008). Our regression models also identified species with preferences for moderately disturbed habitats (Crooks et al, 2004) or heterogeneous mosaic landscapes (Coreau and Martin, 2007), a characteristic feature of Mediterranean biodiversity (Blondel and Aronson, 1999). The species favoring rural mosaics tend to have higher conservation value than those of forest-dominated areas (Pino et al., 2000; Pons et al., 2003). A typical example is *L. senator*, positively associated with increasing fragmentation, bare soil and dry grassland cover (Isenmann and Fradet, 1998). Since more antropophilous species (*M. alba, P. domesticus*) respond in the same way, the footprint of urban encroachment on bird communities is also evident, warning about the danger of increased human disturbance.

If the objective is to create a biologically diverse and self-sustainable forest formation, it is doubtful that it can be met through dense *P. halepensis* plantations (Osem et al., 2009; Cortina et al., 2011). Limiting semiarid conditions compromise the survival of individual trees and the self-regeneration of the stand. Botanical indexes did not respond to any of the composite gradients, but their negative direct relationship with standing live pine density supports a competing effect of trees on other vegetation layers and ultimately on floristic richness. Under less favorable ecological conditions, the natural reduction of pine tree density can have positive effects on understorey development (Chirino et al., 2006). Thus, a wooded formation best adapted to semiarid Mediterranean climate (e.g., a sparse woodland) seems the best reafforestation target. To maximize local biodiversity, however, managers should not just focus on the tree stratum, but try to match the variety of soil and topographic conditions with the range of structural components available (shrubs, chamaephytes, perennial grasses).

Plant and bird species restricted to small, isolated and poorly structured patches raise the biodiversity value of physically stressed areas, usually with minimum tree cover. When dealing with such areas, however, the possibility of an “extinction debt” (*sensu* Tilman et al., 1994) should not be ruled out. On the other hand, enhanced climatic stress might increase the suitability of these patches for species like the Ibero-African shrub *Periploca angustifolia* Labill. (recorded in MPMIB but not represented in samples), whose optimum is predicted to come closer to the study area (Chacón, 2011). These isolated patches can also attain comparatively higher ornithological value (Appendix 1), as found by Pino et al., (2000), who recorded greater species richness and greater conservation value in unprotected territories located between natural parks.

As regards vegetation, the highest-valued patch (MPMIB) is a fragment apparently poorly preserved and embedded in the urban matrix, in which severe pine thinning (from natural mortality or preventive logging of damaged trees) has had a positive effect on the richness of chamaephytes (Figure 2.5). The Mediterranean area stands out for its diversity of endemic and native chamaephytes, which evolved once the biome's climate was already established (Herrera, 1992), making places like MPMIB priority candidates for the establishment of botanical microrreserves.

As a management recommendation, reducing the density of pine trees where environmental conditions limit tree growth, will improve the structure of the understory and raise the biodiversity value of the studied patches. At a larger scale, the relationship of biodiversity with pine density also supports such management strategy (Figure 2.4).

On a landscape perspective, a network of small patches of different sizes spread around the urban-rural matrix and isolated from core forest areas (usually in large mountain ranges) can result in a high accumulated species richness with no need to include areas already given higher protection level. This is particularly useful if the aim is to promote biodiversity in the more intensely developed areas of the municipality (Miller and Hobbs, 2002). However, as we have seen, some large patches (>30 ha) are essential for retaining Mediterranean bird species of high conservation value like *S. undata* (Pons et al., 2012).

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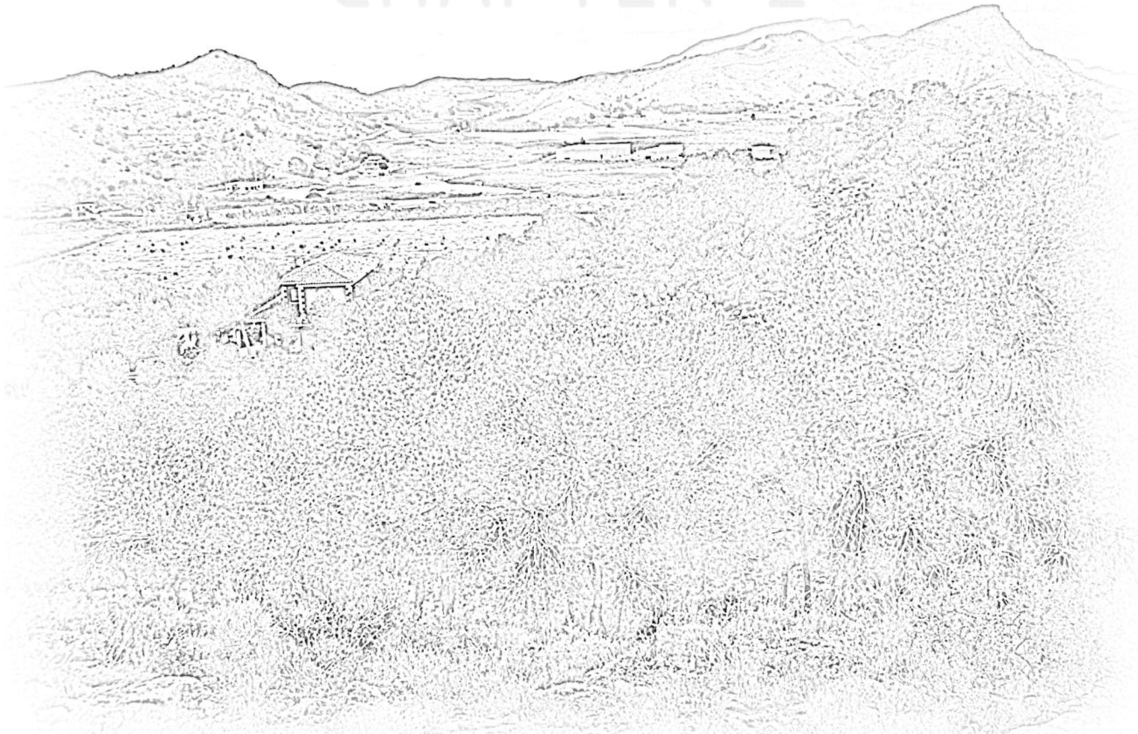
Appendix

APPENDIX 1. Main characteristics and conservation index values of additional patches surveyed in the Municipality of Murcia.

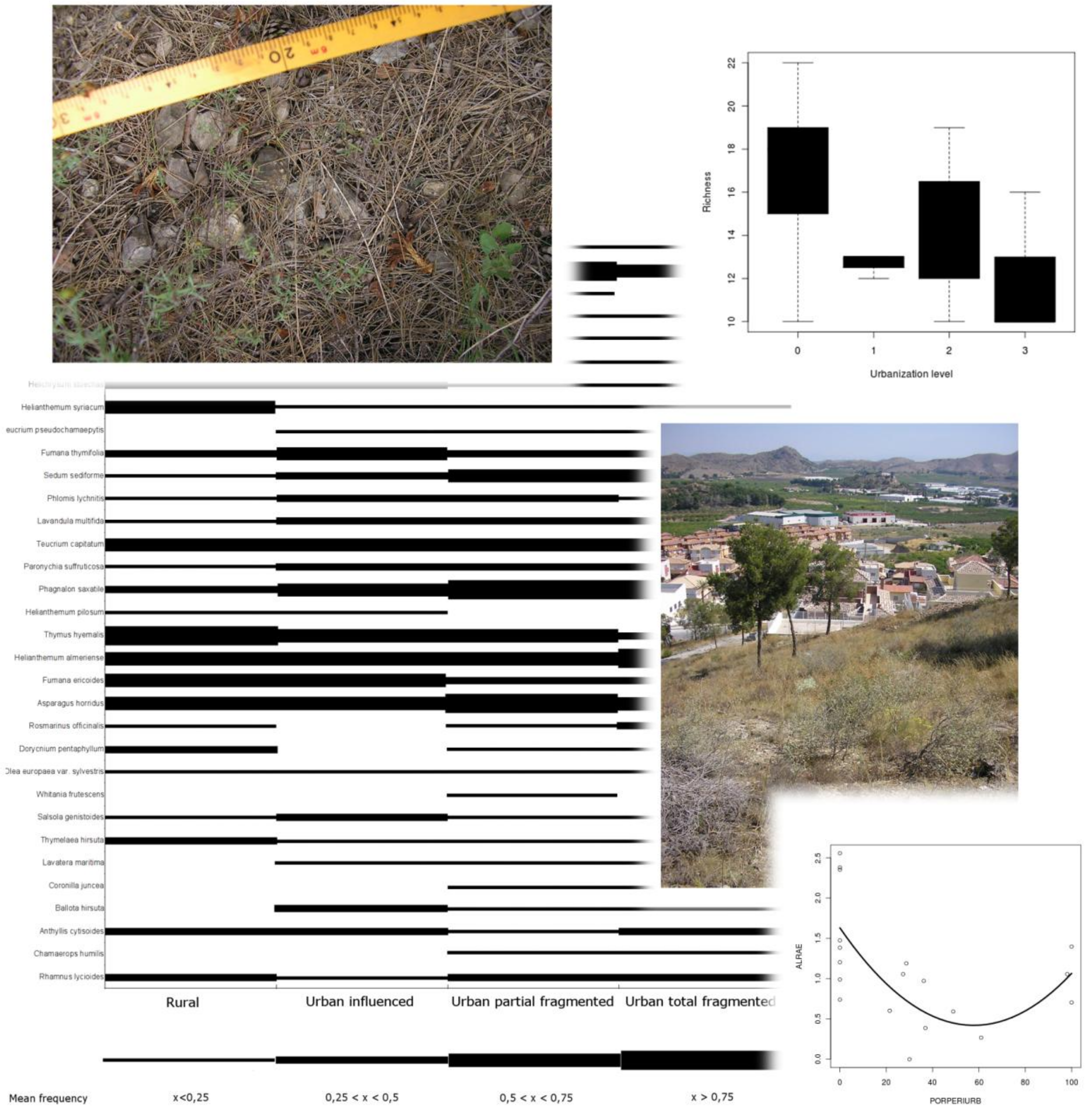
PATCH	SECTION	PROTL	PINREFOR	LIVPD	DEAPD	MAXPD	RBWP+USE	RARE	RBWP	SPEC	RBBS
ALTI	ALT	NON	Y	466.67	22.22	488.89	0.21	0.21	0	5.49	1.4
ALTII	ALT	NON	Y	1,044.44	66.67	1,111.11	0.17	0.19	0	6.83	1.69
ALCA1	ALT	NON	N	33.3	0	33.3	0.89	0.5	0	2.07	0.24
ALCA2	ALT	NON	N	22.2	0	22.2	0.79	0.5	0	1.37	0.13
SANTO	CSV	REG	Y	737.5	125	862.5	0.46	0.26	0.23	1.73	1.17
LRM2	CSV	REG	Y	650	37.5	687.5	0.53	0.58	0.1	1.86	0
LRM1	CSV	REG	N	0	0	0	1.07	0.7	0.64	3.3	0.47
GRANJ	ESP	NON	Y	433.33	44.44	477.77	0.12	0.21	0	4.24	1.63
AGRI	ESP	NON	N	0	0	0	0.49	0.53	0	3.03	0.22
MLEG	ESP	NON	N	222.73	22.73	245.46	0.54	0.44	0	2.62	1.04
GUILIII	GUI	NON	Y	688.89	44.44	733.33	0.25	0.17	0.04	7.15	3.25
GUILI	GUI	NON	Y	2,062.5	25	2,087.5	0.26	0.16	0	6.02	2.8
GUILL2	GUI	NON	Y	1,828.57	100	1,928.57	0.37	0.28	0	3.29	0.94
GUILL4	GUI	NON	Y	877.78	22.22	900	0.73	0.37	0.17	3.83	0.72
CARR	CMB	REG	N	587.5	62.5	650	0.21	0.22	0	5.31	1.78
CARR_A1	CMB	NON	N	75	50	125	0.29	0.17	0	3.6	0.81
CARR_A3	CMB	NON	N	100	0	100	0.3	0.46	0	7.86	1.92
CARR_R1	CMB	MUN	N	125	0	125	0.25	0.15	0	3.11	1.68
CARR_R2	CMB	MUN	N	75	37.5	112.5	0.49	0.31	0.2	2.98	1.66
CARR_R3	CMB	MUN	N	22.2	0	22.2	0.3	0.18	0	3.74	0.91
CANTA	MP	NON	Y	371.43	200	571.43	0.18	0.19	0	3.14	1.29
TINA1	SAN	MUN	N	0	0	0	0.9	0.54	0.18	3.06	1.49
TINA2	SAN	MUN	N	125	12.5	137.5	0.69	0.51	0	3.21	0.73
ASPE	SAN	NON	N	112.5	0	112.5	0.96	0.59	0.1	2.4	0.57
SANTAU	SAN	NON	N	500	28.57	528.57	0.78	0.4	0	1.27	0.28

CHAPTER 2



Urbanization and biodiversity in semiarid landscapes under strong environmental and anthropogenic stress

Graphical Abstract



Abstract

The Mediterranean Basin (MB) is a representative test region for research on the effects of urbanization on biodiversity, being one of the world's most impacted areas as a result of its early human occupation. Semiarid areas within the MB are key places for the study of such impacts, since they experience strong anthropogenic pressures on a naturally stressed environment. Southeastern Spain, a predominantly semiarid Mediterranean area, has undergone a process of recent urban sprawl with increases over 60% of land devoted to urbanization with respect to previously built area. In its interaction with natural -mainly forest- habitat, urbanization is superimposed on natural climatic fragmentation and on cultural fragmentation caused by agriculture and livestock. This has led to a pattern of relatively large core natural areas within an ample transformed matrix with sparse forest islets, usually Aleppo Pine (*Pinus halepensis*) planting or remnants, which have not only intrinsic biological value but an important functional role (as a part of the connectivity network). Their closeness to urban areas also confers them a relevant social role as places favouring man's contact with Nature.

In the demanding and dynamic environmental context of the semiarid MB, there is scarce research focused at studying the response of biodiversity to urban development, and to devising management strategies for its preservation under a global change scenario. To fill this gap, between 2006-2010 a survey was made of 19 remnant forest patches embedded in the anthropogenic matrix (urban and agricultural) inside the 885,9 km^2 of Murcia municipality (SE Spain). Biodiversity descriptors and indicators chosen were woody species (trees, shrubs and chamaephytes) among plants, birds among vertebrates and ground spiders among invertebrates. The urbanization process was characterized by two variables related with urbanization, indicative of habitat fragmentation and direct urban influence. Through this multispecific and multiscale approach, we have evaluated the effects of urbanization on the selected taxocenoses, emphasizing the influence of edge effects on invertebrates, and the landscape and habitat influences on birds and flora.

The results of ongoing studies suggest different effects of urbanization on fragmented forest landscapes, depending on the biodiversity indicator considered. Vegetation appears as the most vulnerable component due to its higher sensitivity to fragmentation (low mobility and barriers to dispersal). In the case of birds, changes in community composition and diversity occur as a result of edge effects and numerous intrinsic characteristics of each islet (patch size, urban influence…). A landscape-patch-microhabitat approach allows the identification of management measures to mitigate the impact of urbanization, and the establishment of guidelines to preserve biodiversity when new urban areas are created.

Urban planning and the management of "urban forest" remnants should take into account these guidelines to allow natural islets to reach their full social and ecological potential.

3.1. Introduction

3.1.1. Urbanization and biodiversity. A Mediterranean perspective.

Habitat fragmentation is a process through which a continuous habitat is partially or completely removed, thereby altering its original configuration (Villard *et al.*, 1999). The main effects are a reduction in habitat size through an increase in the number of fragments, a greater distance between fragments, and an increased vulnerability of the remaining habitat due to its exposure to a higher number of disturbance factors. Such a process results in an alteration of habitat conditions and in changes in the patterns of distribution and interaction among species (Brandt *et al.*, 2009; Andrén, 1994). A special case is when fragmentation is due to human activities. In the Mediterranean Basin (MB), the situation is complicated because landscape disturbances and the spatial pattern of the landscape mosaic are driven mainly by current and past human activities. While traditional land-use systems have contributed to preserve and even evolve higher levels of biological and ecological diversity (Blondel and Aronson, 1999; Blondel, 2006), the superimposition on these of new land uses –namely urbanization, with its associated commercial and infrastructural developments-, is drastically altering the landscape and habitat setting.

The effects of the urbanization at any level of organization are not completely understood since traditional ecological studies have failed to incorporate the human being as an agent of ecosystem function (McDonnell and Pickett, 1990). Although urban ecology is a discipline which already accumulates many years of study, many researchers believe that these studies were initially conducted in areas not subjected to high degrees of human pressure. This view changed during the 90's when researchers began to recognize the human domination on terrestrial ecosystems (Vitousek *et al*., 1997; Blair, 2004;). Remaining biodiversity spots in highly humanized landscapes (e.g., isolated forest or wetland patches) are faced with growing and increasingly diverse pressures.

Habitat fragmentation due to urban development could be considered as a case of extreme fragmentation since the boundaries between urban and forest areas are very narrow and there is a steep gradient in environmental characteristics (Crooks *et al*., 2004). The main impacts associated with it are an increase in soil imperviousness and an alteration of the structure of the remaining vegetation (Beissinger and Osborne, 1982; Arnold and Gibbons, 1996; Germaine *et al*. 1998; Melles *et al*., 2003). However, the responses of the different habitat variables will vary greatly depending on the structure and composition of the natural-urban interfaces, the most obvious consequence of fragmentation being an increased susceptibility of fragments to the negative effects of edges (Tallmon and Mills, 2004). In these fragments, although the land retains natural characteristics, the composition and ecological functions of the habitat are compromised by human intervention (Debinski and Holt, 2000).

It should be remarked that some species (e.g. small or medium sized predators) benefit from the human structures and constructions as these can provide shelter, or even increase the food supply. This can pose problems when these species are non-native and may have a substantial impact on native ones (Churcher and Lawton, 1987; Coleman and Temple, 1993). Therefore, although the severity of the damage caused by urban expansion may be similar to that caused by habitat clearing for other purposes, fragmentation due to urbanization is much more persistent and the areas capable of reversing to the initial conditions are less (McKinney, 2002).

The MB is a representative test region for research on the effects of urbanization on biodiversity, being one of the world's most impacted areas as a result of its early human occupation. Semiarid areas within the MB are key places for the study of such impacts, since they experience strong anthropogenic pressures on a naturally stressed environment. Southeastern Spain, a predominantly semiarid Mediterranean area, has undergone a process of recent urban sprawl with increases over 60% of land devoted to urbanization with respect to previously built area (Robledano and Zapata, 2011). In its interaction with natural -mainly forest- habitat, urbanization is superimposed on natural climatic fragmentation and on cultural fragmentation caused by agriculture and livestock. This has led to a pattern of relatively large core natural areas within an ample transformed matrix with sparse forest islets, usually Aleppo Pine (*Pinus halepensis*) planting or remnants, which have not only intrinsic biological value but an important functional role (as a part of the connectivity network). Their closeness to urban areas also confers them a relevant social role as places favouring man's contact with Nature.

3.2. Material and methods

3.2.1 Study area

In the demanding and dynamic environmental context of the semiarid MB, there is scarce research focused at studying the response of biodiversity to urban development, and to devising management strategies for its preservation under a global change scenario. Only recently some of the expected changes brought in by climate change have been subject of monitoring and modelling studies (Martínez-Fernández *et al.*, 2009; Esteve *et al.*, 2010). To fill this gap, between 2006-2010, a survey was made of 19 representative forest patches embedded in the anthropogenic matrix (urban and agricultural) inside the 885,9 km^2 of Murcia municipality (SE Spain). During the study period, these forest remnants of different origin are being used as outdoor laboratories for testing biodiversity responses to various landscape and management processes, upon which the effect of climate change is superimposed.

The municipality of Murcia (UTM30 X=666,800; Y= 4,200,000; Datum ETRS89) is located in the South-east of the Iberian Peninsula, in the central-eastern part of the Murcia Region Autonomous Community. Due to its geographical location, the municipality has a great diversity of natural and semi-natural habitats with high ecological value, currently threatened by human activities (intensive agriculture, urban reclamation, etc.) Therefore, much of the municipal territory has seen a net reduction in the proportion of their remaining natural ecosystems, with the remaining surface split into large tracts of protected areas plus many small fragments of remnant vegetation that have not yet been eliminated from the anthropogenic matrix.

The forest patches selected (Figure 3.1.; Table 3.1.) are discrete spatial units totally isolated from the main forest matrix, or still connected with it through corridors (Gurrutxaga and Lozano, 2008), but surrounded along most of its perimeter by non forest-habitats. This spatial pattern is the final stage of the process of forest fragmentation in our study areas. Continuous forest surfaces associate with the main mountain ranges and hillslopes, while lowland areas represent a first fragmentation scenario, given their traditional suitability for cultivation which has prompted their early reclamation. In the less suitable places for tree growth, due to limiting topographic, soil or microclimate conditions, natural processes add up to the fragmentation of forest landscapes The impact of traditional agriculture on the connectivity of natural systems has been in part mitigated by different elements of the natural and cultural infrastructure of the landscape (watercourses, hedges, cattle ways…), as well as by its fine-grained mosaic pattern. In more recent times, however, fragmentation is due mainly to other processes, namely the growth of artificial uses -urban and industrial- and infrastructures -transport, water and energy distribution- (Martínez Fernández *et al.*, 2009).

		PENPR	**SUPH**	**PERI**	**PERIURB**	**PORPERIURB**
	Patch	Mean slope	Surface (has)	Perimeter	Urban perimeter	% Urban perimeter
1	CANTA	18.41	12.79	1.75	1.71	98.11
2	MPMIG	12.62	5.48	1.35	1.35	100.00
3	MPMIP	7.45	2.43	0.61	0.61	100.00
4	MPMWE	15.51	3.26	0.92	0.25	27.28
5	MPPVN	6.99	9.00	1.40	0.00	0.00
6	MPMSU	13.49	2.98	1.19	0.00	0.00
7	MPMES	17.78	6.16	1.80	0.39	21.50
8	MPPPA	11.69	34.35	3.55	1.74	48.89
9	ALTI	12.38	13.18	2.38	0.00	0.00
10	ALTII	4.85	6.11	0.98	0.00	0.00
11	GRANJ	6.47	23.45	2.38	0.00	0.00
12	GUILIII	9.37	7.52	1.34	0.00	0.00
13	GUILI	6.76	2.00	0.66	0.00	0.00
14	CARR	11.75	5.80	1.25	0.00	0.00
15	MBMNO	14.70	20.65	3.09	1.14	36.88
16	MBMCE	10.88	17.29	3.78	1.08	28.66
17	MBMSU	12.56	18.90	4.06	2.47	60.93
18	SANTO	19.57	8.57	1.55	0.56	36.19
19	LRMII	54.30	26.63	2.85	0.86	30.05

Table3.1: Forest fragments studied and their main characteristics related to habitat fragmentation and urban development.

As a result of the most recent spread and intensification of residential development (Robledano and Zapata, 2011), extant forest habitat patches embedded in the urban-farmland matrix, can undergo a second process of fragmentation leading to archipelagos of smaller fragments or to a constellation of satellite spots around a main patch. Another type of fragmented landscape occurs where satellite spots are totally or partially separated from the border of a continuous forest section (matrix). Summarizing, we find a high diversity of forest patches isolated by various anthropogenic processes.

The forest patches studied are valued as remnant “green” spots, amenable to protection for landscape and recreational uses. Some patches have in fact been declared as Municipality Forest Parks, and incorporated to the public heritage as urban concessions. In many cases, these lands are adjacent to urban developments, and hence embedded to a variable degree in a predominantly urban matrix. On the other hand, current development policies allow residential growth to come closer to large tracts of continuous forest habitat, and concessions for natural protection are usually concentrated in such areas. As a result, residential developments are tightly associated with forest patches or forest edges, while land protection policies usually neglect natural fragments in other landscape situations. In doing so, the Municipality not only fails to complement, but can be weakening land conservation at the regional level. Depending on various factors, isolated forest patches may contain significant biodiversity and high conservation value and serve as connection areas between large protected natural zones. This has already been established by other studies in Mediterranean forest landscapes (Pino *et al*., 2000). This chapter uses a representative test area to assess the effects of urbanization on forest patches within an increasingly stressed semiarid Mediterranean landscape.

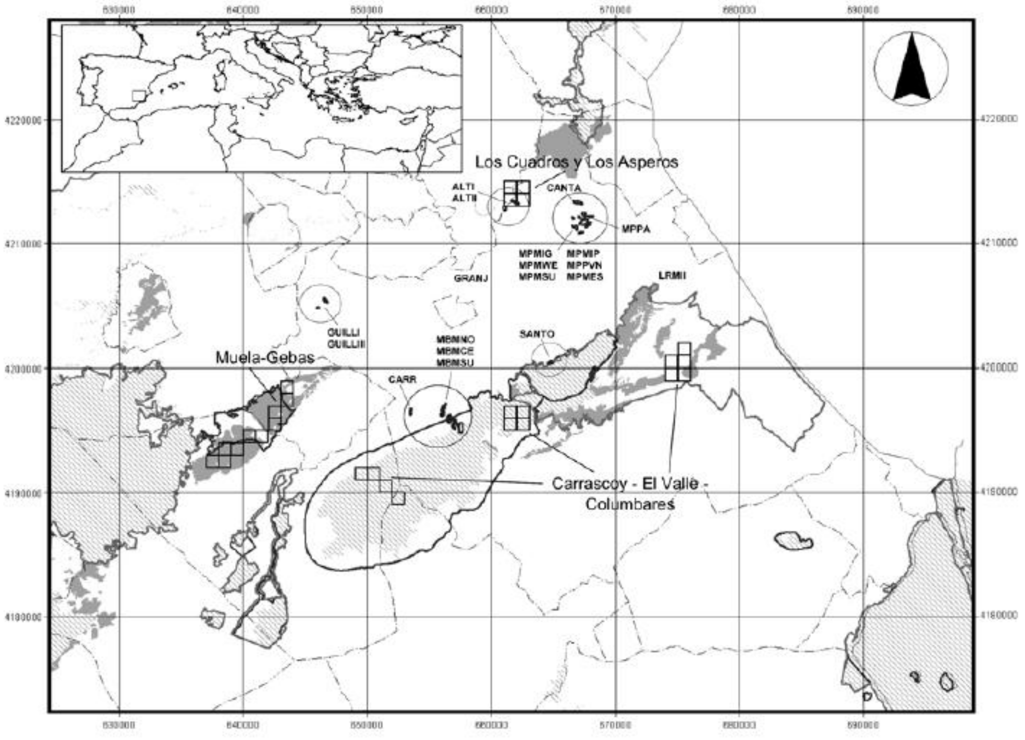


Figure 3.1: Location of the forest fragments studied in the municipality of Murcia (SE Spain). Circles enclose the main groups (archipelagos or constellations) of forest patches. Squares represent reference quadrats (not included in this study) located within three continuous reference areas (Los Cuadros y Los Asperos, Muela-Gebas and Carrascoy-El Valle-Columbares). Also represented are the main protected areas, with regional (shaded grey and black lines) and European status (hatched and grey lines).

3.2.2 Selecting biodiversity indicators

Biodiversity descriptors and indicators chosen were woody species (trees, shrubs and chamaephytes) among plants, birds among vertebrates and spiders among invertebrates. For the first two groups, the scale of work is the landscape level, while in the case of spiders, a zoom is made to the local scale due to the small size of these organisms and their high compositional diversity. The urbanization process was characterized by landscape variables related with urbanization, indicative of habitat fragmentation and direct urban influence. Through this multispecific and multiscale approach, we have evaluated the effects of urbanization on the selected taxocenoses, emphasizing the influence of edge effects on invertebrates, and the landscape and habitat influences on birds and flora.

3.2.2.1 Flora and vegetation

Vegetation sampling was carried out with systematically distributed 100 m^2 circular plots (spaced 50 m), in number proportional to the size of each patch. Vegetation variables recorded inside each plot were the occurrence of all woody species and the density of trees (number of individual Aleppo pine *Pinus halepensis*). From each individual tree the height and DBH (Diameter at Breast Height) was recorded, as well as its general condition, age, the presence of cones and other relevant observations. The presence of stumps and standing dead tree specimens of Aleppo pine was also annotated. Finally, the linear coverage of all physical or vegetation layers (rock, bare ground, scrubs, chamaephytes, grass –*Brachypodium*-, litter, mosses and lichens, and annuals) was recorded along four 10 m transects oriented towards the four cardinal points, and expressed as percentage of the total transect length.

3.2.2.2 Birds

Breeding birds were sampled through point counts (Jansen and Robertson, 2001) of 5 minutes of duration, during which all the sightings and sound contacts were recorded. Data were used to generate specific or aggregate (community) indexes of abundance, i.e. individuals per point count, upon which diversity metrics (species richness and the Shannon index) were calculated. More details on bird sampling can be found in Robledano and Zapata (2011).

3.2.2.3 Conservation value indexes

The conservation value of each fragment was assessed through several indexes of conservation value adapted from the procedure used by Pons *et al.* (2003), and calculated for woody flora and birds. This method of evaluation has been widely used in similar contexts (see e.g. Robledano and Zapata, 2011, for a more detailed explanation of the background and application of such indexes).

3.2.3 Urbanization assessment and statistical analyses

Regression analysis was used to explore the response of birds and vegetation community metrics and indexes to the two measures related with urban development: PORPERIURB (percentage of urbanized perimeter around the patch) and SUPH (surface of the patch in ha), a measure of fragmentation. These two variables are weakly correlated, although the length of urbanized perimeter increases with total patch perimeter (a variable in turn positively related with patch size).

For the same purpose, we compared the conservation value of flora and birds among categories of urbanization using box-plots and ANOVA, which were implemented with the freely distributed "R" statistical package (The R Foundation for Statistical Computing, 2005; http://www.r-project.org).

3.3. Results

3.3.1 Flora and vegetation

We expect vegetation to be the component most directly affected by historical changes (Guirado *et al.*, 2008). These changes can lead to the disappearance of some species, and because of their restricted movement, the dispersal and re-colonization of native species can be prevented. Due to this fact, it is hard to find a plant species with a specific response in each kind of forest patch. In such a situation the forest edge will have an important function and the frequency of occurrence of each plant species will be determined by the strength and nature of the physical gradients (Matlack, 1994).

For this reason, we assessed changes in the frequency of occurrence (as a measure of abundance) of each plant species among forest patches with different degrees of urban development. For such assessment we have classified the forest patches into four groups. We considered patches as *rural* (GUILLI, GUILLIII, CARR, ALTII) when the distance to an urban area is greater than 1 km; *Urban influenced* (MPPVN, MPMSU, ALTI, GRAN) are patches whose distance to an urban area is less than 1 km and there is no contact with the urban zone. Patches are considered *urban partially fragmented* (MPMWE, MPMES, MPPPA, MBMNO, MBMCE, SANTO, LRMII) when their perimeter in contact with urbanization is less than 50% of the total perimeter length, and *urban totally fragmented* (CANTA, MPMIG, MPMIP, MBMSU) when the contact with urbanization occurs along more than 50% of the total perimeter of the patch.

Figure 3.2 shows the mean frequency of occurrence of the main woody plant species recorded in the patches. It is evident that there are no abrupt changes among the categories established on the basis of urban influence. Frequencies increase or decrease gradually in both directions along the gradient, in a way such that some species tend to increase towards rural areas, for example *Helichrysum stoechas, Thymus hyemalis* or *Fumana ericoides*, while other belonging to the same life form (chamaephytes) follow the opposite way, like *Sedum sediforme, Asparagus albus* or *Phagnalon saxatile*. Also, some species do not show either of these trends and appear with the same frequency across all the different patch types.

There are several plants adapted to edge conditions, which could increase plant richness in the patches with higher perimeter length (López de Casenave *et al*., 1995). We observed an intermediate maximum on plant richness when the value of PORPERIURB was close to 50-60%, i.e. patches characterized by edge variety with agro-forest and urban-forest gradients. The microclimate conditions of each type of edge can influence plant communities (Gehlhausen *et al*., 2000), creating a different species assemblage from that of the patch interior (Harris, 1984; Noss and Cooperrider, 1994), and leading to increased plant richness (Figure 3.3). A similar response is found when plant richness is compared directly among urbanization levels (Figure 3.4). Besides, the strip of land that can act as an edge may be broadened by factors related to human behavior and lifestyle (Boyle, 2004), which would further increase microclimatic heterogeneity and hence plant richness.

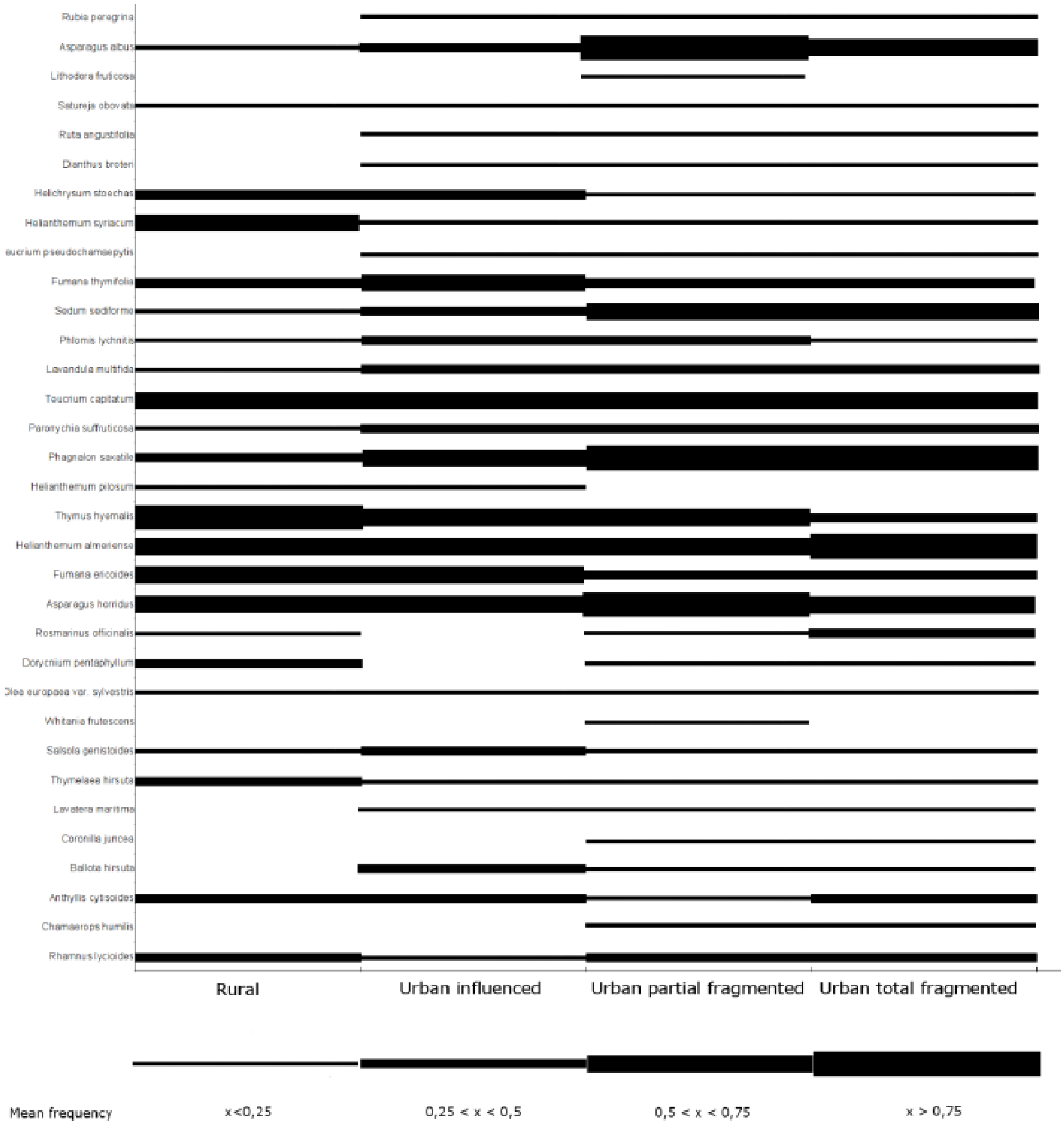


Figure 3.2: Mean frequency of occurrence of the main woody plant species recorded in the forest patches studied, according to the degree of urban influence (categories defined in the text).

The indexes of plant conservation value (see Robledano and Zapata, 2011) LRFSUSO ($R^2 = 0.34$; $p < 0.01$) and RARE ($R^2=0.24$; $p < 0.05$) –based respectively on protection status and rarity-, respond similarly than does plant richness when related with PORPERIURB (Figure 3). They reach maximum values with intermediate values of the variable used to describe urban influence across the patch edge. In fact, Figure 2 shows how some plant species contributing to such indexes (e.g. *Asparagus albus, Fumana thymifolia, Teucrium capitatum, Phagnalon saxatile, Thymus hyemalis, Helianthemum almeriense, Asparagus horridus*) occur with high frequency in intermediate classes (Urban influenced and Urban partially fragmented) which attain higher plant conservation value.

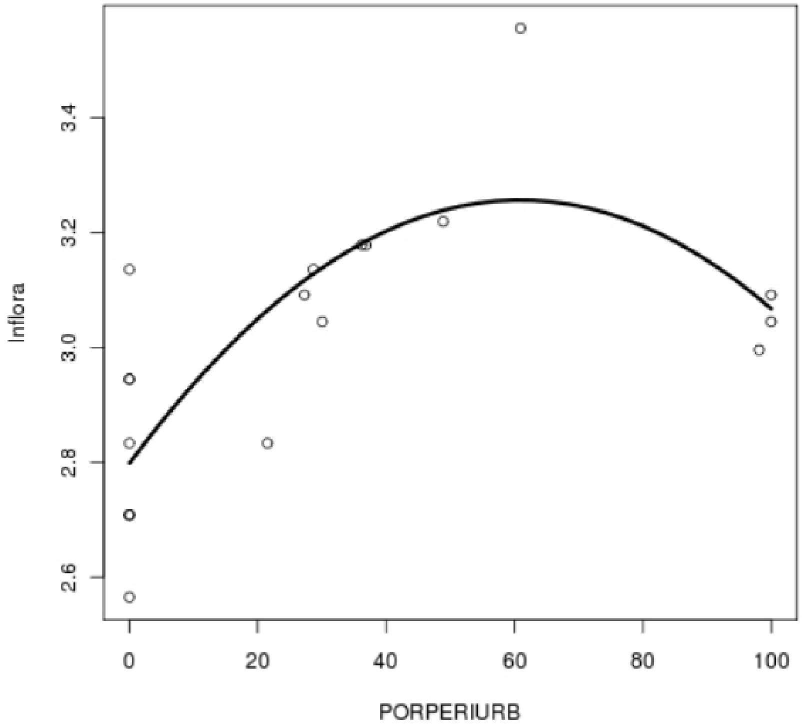


Figure 3.3: Linear regression model relating woody plant richness with the proportion of patch perimeter in contact with urban areas (quadratic response of log species richness vs PORPERIURB).

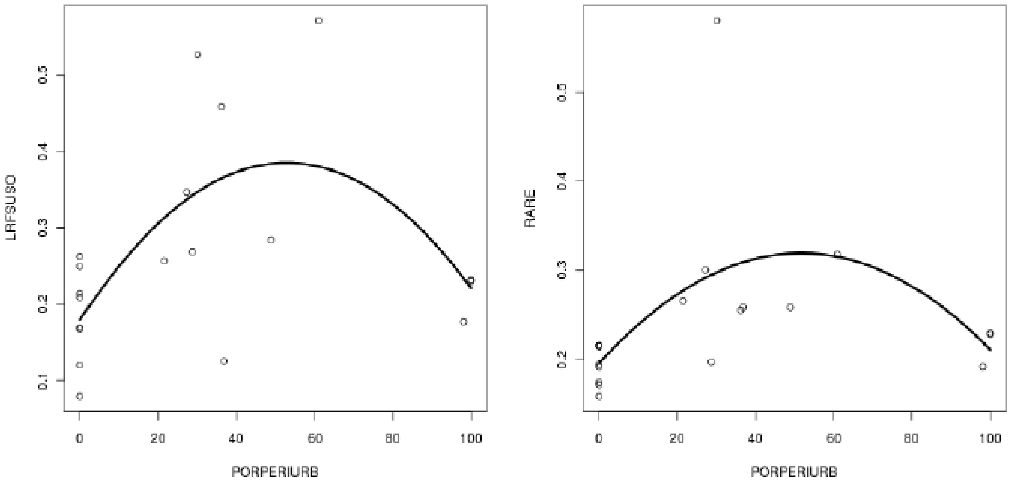


Figure 3.4: Linear regression models relating the indexes of conservation value for flora with the proportion of patch perimeter in contact with urban areas: quadratic responses of LRFSUSO (left graph), and RARE (right graph) vs PORPERIURB.

3.3.2 Birds

Differential and quick responses of birds species to changes in the ecosystems, make them one of the most used bioindicator taxocenoses in forest remnants within urbanizing landscapes (Chace and Walsh, 2006; McKinney, 2008). Changes in the bird community are highly related to urbanized area (Chan-Ryul *et al.*, 2005).

Following the same method used for screening the frequency of occurrence of plants, in Figure 3.5 the mean abundance of bird species is compared among the four classes of urban influence established. We can classify most bird species according to their tolerance to urban alterations, with four main types of responses: predominantly urban, rural (intolerant to urban changes), intermediate level of tolerance, and indifferent. Regarding this last group, birds could show indifference in two main ways. First, when the abundance of the species is high under all levels of urban influence, for example *Turdus merula* or *Serinus serinus.* On the other hand, there are species with similarly low abundance in all categories. We can adscribe to this group *Sylvia undata, Picus viridis, Streptopelia turtur* and *Sturnus unicolor*. The former three are species that can be considered rare in these extremely fragmented landscapes, while *Sturnus unicolor* is a typically urban species.

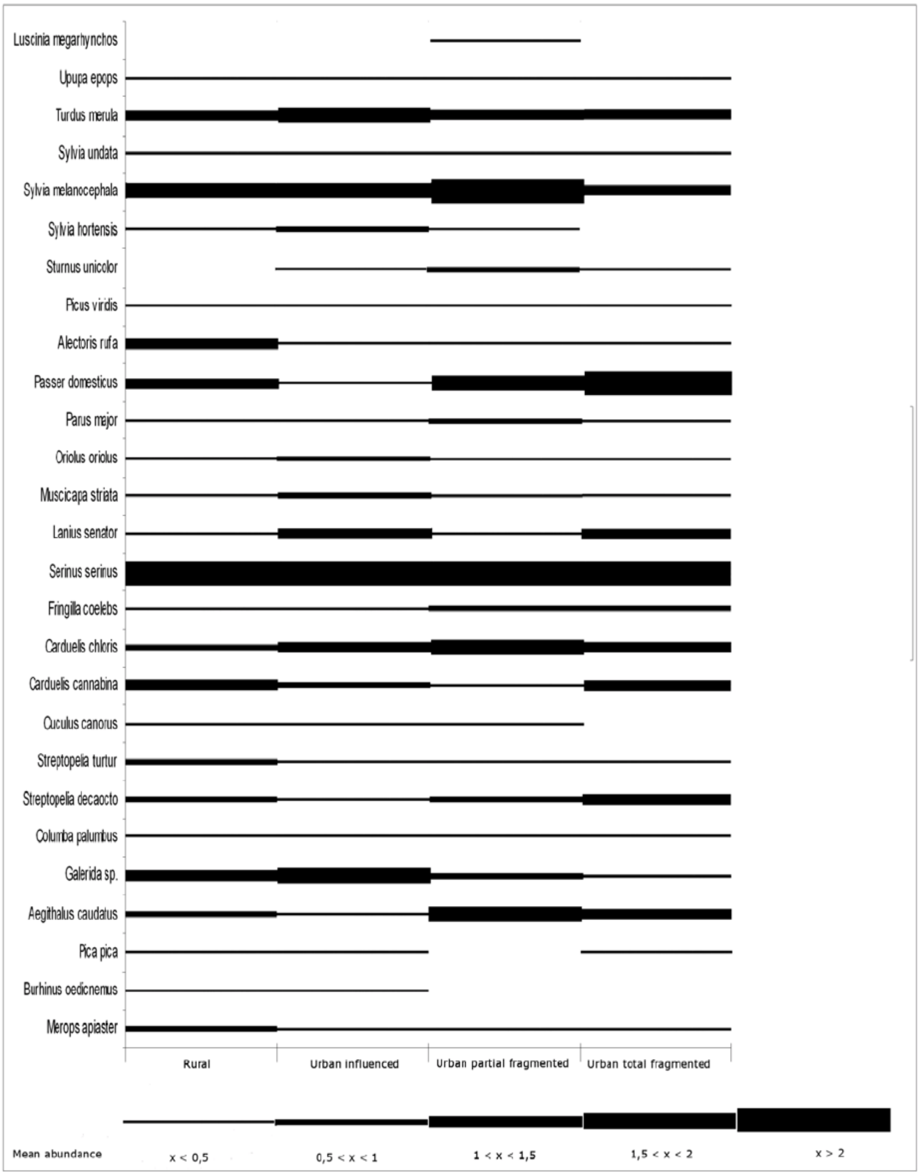


Figure 3.5: Mean abundance of the main bird species recorded in the forest patches studied, according to the degree of urban influence (categories defined in the text).

As for plants, there are some species that reach maximum abundance in patches with intermediate levels of disturbance (Crooks *et al.,* 2004). Usually this is the case of species favoured by a high availability of open areas (scrub and grasslands), or other productive habitats like urban gardens, combined with forest patches providing refuge and nesting places. It can be interpreted as a strategy aiming to maximize the exploitation of resources, which is a characteristic feature of Mediterranean biodiversity (Blondel and Aronson, 1999). Within this group of species we can mention *Sylvia melanocephala, Carduelis chloris* or *Aegithalus caudatus*.

A second group is the one formed by rural species. It comprises birds with low tolerance to habitat changes caused by urban expansion. Within this group, we highlight the abundance of the representatives of the family *Alaudidae* (*Galerida sp.*) within which there are species of conservation concern at both national and European level (Tucker, 1997; Burfield, 2005). Similarly, *Merops apiaster*, *Cuculus canorus*, *Burhinus oedicnemus* and *Alectoris rufa* -species that prefer rural landscapes, composed by agroforestry habitats- seem to be negatively influenced by urban land.

Finally, there are some species for which urban development generate increased, easily available resources. Urban areas provide a continuous supply of food for birds, which is exported towards related areas (Marzluff, 1997). *Passer domesticus* and *Streptopelia decaocto* are the main species that display greater abundance in forest land influenced by urban development.

Two special cases are *Lanius senator* and *Carduelis cannabina* which show high abundance in urban total fragmented patches. Both are species of conservation concern at the European scale (Birdlife International, 2004), which is not consistent with its presence in highly urbanized habitats. Their preferences in the patches studied are interpreted as a transient exploitation of the early successional stages following the disturbance caused by the spread of urbanization towards rural landscapes, which can also explain the shape of the response curve relating bird conservation indexes to urbanization shown in Figure 3.6. These open stages can represent good feeding places both for insectivores and granivores. It is predictable that when these disturbed areas become more stable, pioneer species of high conservation value will be displaced by the more competitive typical urban species, as well as by habitat changes.

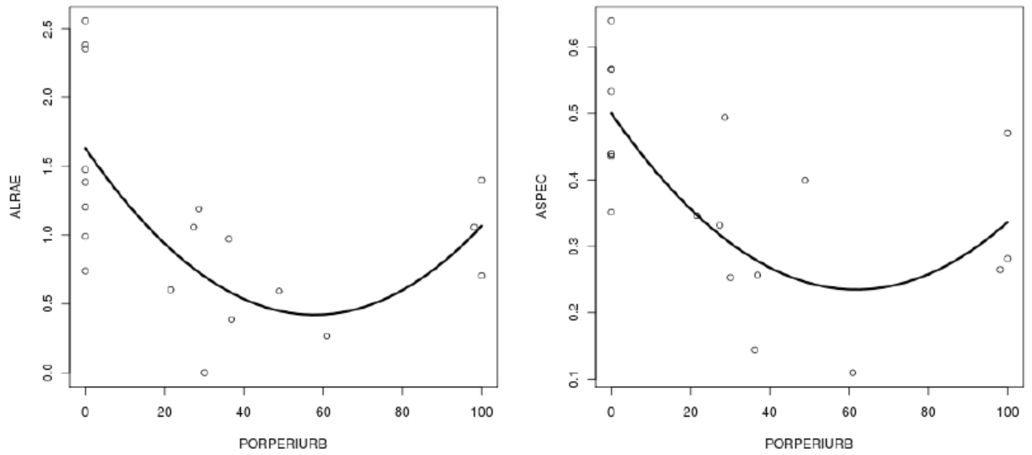


Figure 3.6: Linear regression models relating the indexes of conservation value for birds with the proportion of patch perimeter in contact with urban areas: quadratic responses of LRAE (left graph), and SPEC (right graph) vs PORPERIURB.

The response of bird richness and conservation value indexes (see Robledano and Zapata, 2011) shows the negative effects of urbanization on avifauna. Figure 3.7 also displays a marked decrease of bird richness as the level of urbanization rises in the vicinity of the patch, with statistically significant differences (ANOVA, $p < 0.05$) among categories of urban influence. Such a relationship has been documented by other authors (Bolger *et al.*, 1997; Germaine *et al.*, 1998; Chace and Walsh., 2006; McKinney, 2008). SPEC (R^2=0.46; $p < 0,01$) and LRAE (R^2=0,4; $p < 0,01$) indexes have also a significant negative response to the proportion of the patch perimeter occupied by urbanization (PORPERIURB; Figure 3.6). The inverse relationship shown by both indexes (Figure 3.5) suggest that in general, the species which are lost when urbanization approaches a patch have greater conservation value than the species positively associated with urban development, with the exceptions previously mentioned.

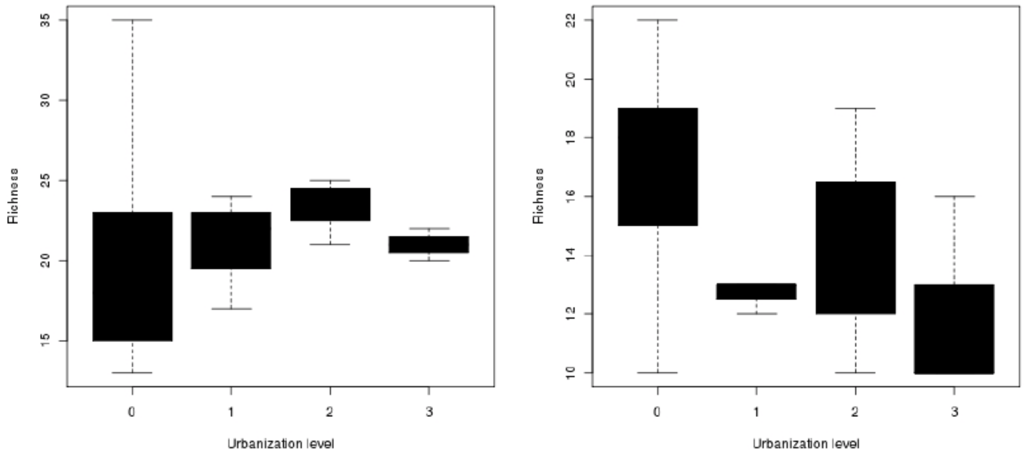


Figure 3.7: Comparison of woody plant and bird richness among categories of urban influence. Although for plants, the box-plot representation suggests a similar pattern to that shown by figure 3, differences are not statistically significant (ANOVA, p=0.38). In the case of birds, differences among categories are significant (p<0.05), with rural patches attaining higher species richness.

3.4. Discussion and conclusions

3.4.1 Lessons for urban planning and natural areas management

Learning from history to avoid making the same mistakes is a point applicable to all disciplines. In the municipality of Murcia, these mistakes have occurred repeatedly when planning the arrangement of urban and forest areas. The priority has always been to build urban resorts in lands with the greatest potential for forest development, while allocating to forest restoration places where soil and climatic conditions often hinder the proper development of a tree stratum. At the same time sparse building development around the main cities has progressively approached forest areas, both natural and planted (often in suboptimal locations). As a result landscapes converge in a mosaic where urban and forest habitats coexist in close vicinity, whether in the edge of large forests or around isolated wooded patches. Natural areas devoid of trees (scrubland and pasture) have been historically the target of re-afforestantion initiatives, and systematically neglected by current nature conservation policies. Therefore, to maximize heterogeneity at the local scale, efforts should be made to adscribe to forest use the greatest diversity of soil and topographic conditions. This would lead forest areas towards different structures and may help to achieve the maximum diversity of species at a landscape scale.

Fragmented areas cannot be attributed a poor conservation status until their biodiversity is characterized and explained through detailed studies. These should take into account the number of factors that determine the presence or absence of different taxa. The closeness of urban areas to forests modifies the communities, having undesirable effects on biodiversity (Theobald *et al*., 1997). By changing the physical characteristics and the environmental conditions of the area, some patches can undergo a process of substitution of species, with the new ones being able to exploit the increased resources occurring in the area (Blair, 2004). Historical and current management of an area will also influence the composition of the biodiversity and the ecological interactions.

The vegetation will be the biological component most affected by historical changes. These changes can lead to the disappearance of some species, and because of their restricted movement, dispersal and recolonization by native species can be prevented. The response of plant species to disturbances like fragmentation and urban encroachment often peaks at intermediate levels of these variables, which is a characteristic of Mediterranean biodiversity (Blondel and Aronson, 1999). It seems that although forest fragmentation can lead to complete isolation, an appropriate patch size allows the maintenance of species with functional significance for the ecosystem (e.g. *Rhamnus lycioides* and *Chamaerops humilis*).

Chamaephytes, a common life form in semiarid Southeastern Spain, do not seem particularly affected by urban fragmentation (Figure 3.2), but are sensitive to the availability of habitat resources for which they compete with the tree layer. Therefore, reducing the density of pines in areas with limiting conditions for tree growth, will improve the conditions for this important plant group in the Mediterranean area due to the high number of endemic species it includes (Herrera, 1992).

Besides the size, it is important to consider the structure of vegetation, in terms of density and coverage of the various strata represented (López and Moro, 1997). In the case of man-made forest systems, the capacity for self-regeneration will be affected by environmental conditions. Thus, when restoring semiarid areas, choosing structures with low tree density (scrub-woodland, dry pasture or chamaephyte-dominated) can make the ecosystem more self-sustaining and resilient facing global change.

The easy movement of birds can prevent extinctions at the landscape scale through metapopulation dynamics (provided enough habitat of sufficient quality persists), but also allow species more adapted to new conditions of urban and agricultural influences to colonize the forest patches and invade the former assemblages. Normally, as time from the start of urban development increases, the bird community becomes more homogeneous with other urban areas and become distinct from the native populations (Chace and Walsh, 2006), although the removal of native vegetation will also cause major changes in bird faunas (Tweit and Tweit, 1986). Maintaining heterogeneous patch constellations seems to be a good choice for enhancing local ornithological diversity. Thus, integration of forest habitat within urban planning should be addressed at the landscape level (Jokimäki, 1999). As noted with indexes, high conservation value for birds may occur in marginal or peripheral forest areas with low-medium urban pressure, and outside protected areas. This has also been observed in other Mediterranean forest landscapes (Pino *et al.*, 2000). It is essential that urban pressure does not increase around these patches, because of the expected loss of conservation value in these areas with high rates of urbanization.

Regarding urban design, the presence of smaller patches located in intermediate positions may have also critical importance, as corridors between natural areas or reservoirs of biodiversity.

3.4.2 Conclusions

Our results suggest that the urbanization process is an objective threat to many species of remnant forest patches, but also that adequate planning and design provide a good opportunity to increase biodiversity, especially in Mediterranean environments where many species can benefit from a certain level of disturbance. In addition, these species tend to have greater conservation value than those favouring predominantly forested areas. In this sense, there are studies supporting the non-linear relationship between the abundance of species and the degree of urbanization (Blair, 2004; Hanssen *et al.*, 2005).

Ongoing studies show different effects of urbanization on fragmented forest landscapes, depending on the biodiversity indicator considered. Vegetation appears as the most vulnerable component due to its higher sensitivity to fragmentation (low mobility and barriers to dispersal). In the case of birds, changes in community composition and diversity occur as a result of edge effects and numerous intrinsic characteristics of each islet (patch size, urban influence…). A landscape-patch-microhabitat approach allows the identification of management measures to mitigate the impact of urbanization, and the establishment of guidelines to preserve biodiversity when new urban areas are created.

It is important to maintain natural patches of various sizes. Large patches (>30 has) can host by themselves a sufficient extension and diversity of habitats for the presence of species adapted to each of these habitats, as well as those sensitive to the size of the habitat. In this regard, the often negative effect of edges should be remarked, since a too close urban development can reduce the effective size of patches. Urban developments, particularly residential resorts, often seek the proximity of the forest for landscape and amenity reasons. To soften their edge effect, it is advisable that the urban front gets not too close to the forest patches and, whenever possible, buffer rural areas are established. Herbaceous systems (strips of pasture or grasslands) are also valuable as bird habitat, providing foraging areas for many open-land species. Finally, they can protect the urban space against the risks of excessive proximity to forest (e.g. forest fires), particularly if controlled grazing is allowed to control forest biomass.

At the landscape scale it is also advisable to seek for the greatest variety of ecological conditions (size, substrate, soil, vegetation) within a constellation or network of patches. This will provide habitat for a greater number of species as well as potential refuges or elements favouring connectivity, particularly in a dynamic context.

It seems also crucial to take into account the positive effect that rural landscapes exert on forest patches, which if surrounded solely by urban areas will have impoverished or trivial biotas. Urban planning and the management of “urban forest” remnants should take into account the former guidelines to allow these natural islets to reach their maximum ecological potential. Therefore, we advocate for an urban-agroforestry integration as a way to preserve their full range of ecological and social functions and values.

3.5. Acknowledgements

The studies on which this chapter is based are the result of ongoing collaborative fieldwork undertaken since 2005 with the participation of many Environmental Sciences´ undergraduate students from the Faculty of Biology of the University of Murcia (see Robledano and Zapata, 2011). We also thank the Caja de Ahorros del Mediterráneo CEMACAM’s Directors and staff, for allowing the use of their facilities during the fieldwork in the Majal Blanco Forest Park, and Francisco Carpe Ristol and the staff of the Murcia City Council´s Environmental Protection Service for authorizing and facilitating research on Forest Parks and municipally owned protected estates.

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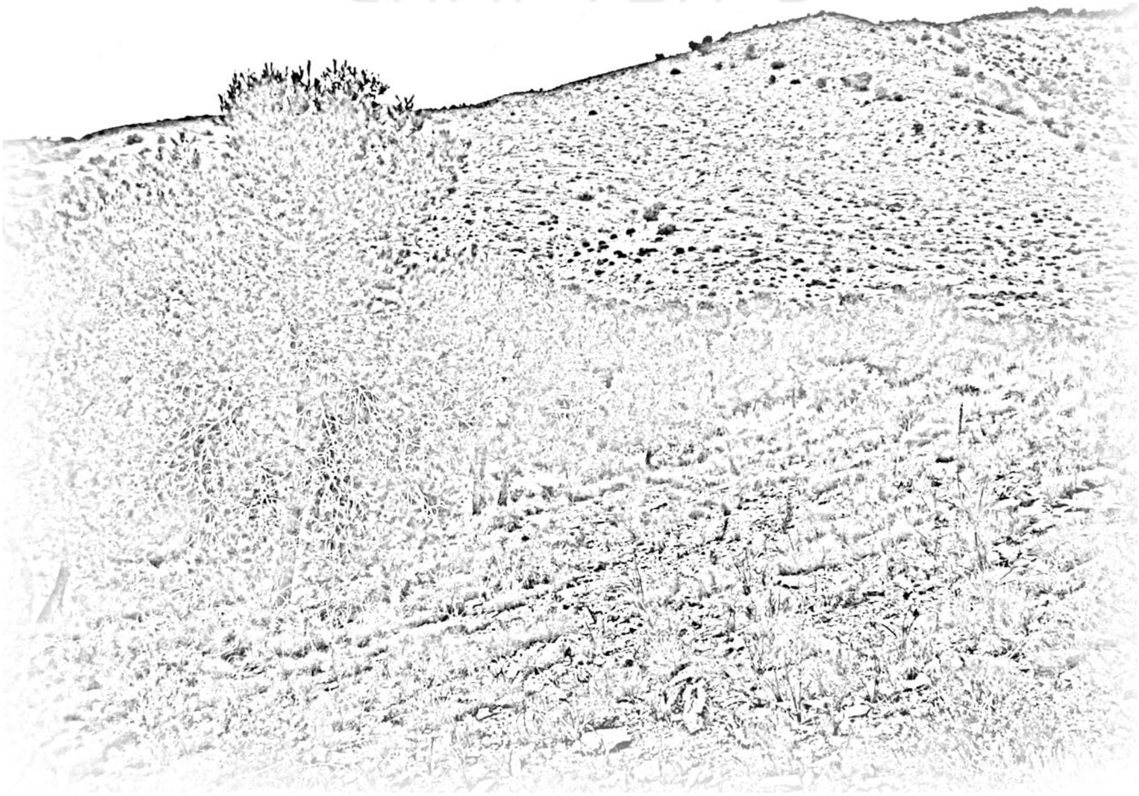
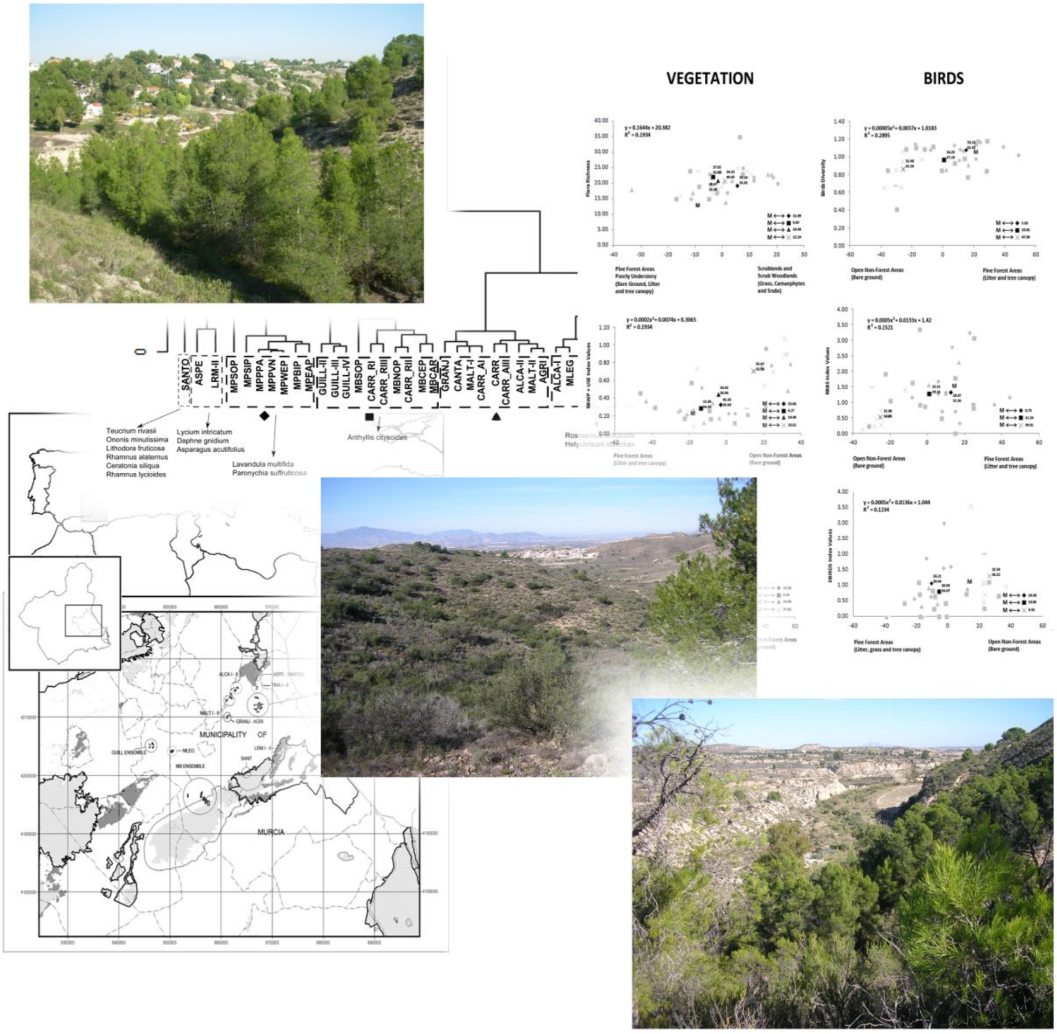
CHAPTER 3



Woody vegetation and bird response to fragmented landscapes in the semiarid Mediterranean. How can biodiversity be managed from inside and outside the remaining forest patches?

Graphical Abstract



Abstract

Mediterranean areas have experienced human exploitation along centuries which has resulted in fragmented mosaic landscapes and subsequent effects on flora and fauna. Land management inside and outside the remaining natural patches affect their diversity both in the composition and structure of taxocenoses, and in their mutual relationships.

Vegetation and birds have been chosen to indicate how fragmentation context affects the biodiversity of natural patches (forest, shrublands and recent reforestations) in semiarid Mediterranean areas. 37 patches were surveyed for vegetation and 34 for birds. Vegetation (presence of woody species and percent cover of different strata) was sampled in 100 m^2 plots, and birds through point counts. Conservation value indexes were calculated for birds and vegetation. Statistical analyses were based on classification techniques and on the search of indicator species (IndVal). Relationships between response variables (richness, diversity and conservation value indexes) and predictors (physical, cover and structural variables) were sought through Partial Least Square Regressions (PLSR) using each patch as a single unit.

Classification and IndVal analyses organized shrublands and forest areas in separate groups representing different levels of urban affection (as fragmentation agent). Significant PLSR results were obtained for cover variables, with higher values for plant conservation value indexes in shrublands than in forest areas. For birds, results varied with the scope of the conservation index, open and shrubland areas scoring higher in the European index but lower in national ones. Higher bird diversity was found in shrubland and forest patches far from urban or agricultural developments.

From these results, management guidelines for each type of patch have been proposed, including recommendations like clearing areas with high pine densities, enhancing seed dispersal and other natural interactions in isolated patches, or protecting well structured forest and shrub patches.

4.1. Introduction

Human development has influenced natural ecosystems, modifying terrestrial landscapes for several purposes (logging, fuelwood extraction, grazing with domestic herbivores, and agricultural conversion), which have caused a generalized retreat of natural vegetation since the Neolithic (Blondel and Aronson, 1995; Blondel *et al.*, 2010). Changes occurring throughout the XXth century in the more populated areas (coastal areas and lowlands), like industrial development, urban growth, and the expansion of modern agriculture, have locally fragmented the remaining natural areas, increasing also the direct impacts through abrupt landscape boundaries. The growth and expansion of these sectors affect natural ecosystems in different ways, depending on their intensity and on the buffering capacity of the receiving environment, as well as on the varying permeability of the landscape matrix (Fischer and Lindenmayer, 2007)

Urban growth close to natural areas causes strong isolation of patches and severe changes of their internal conditions. Besides strict fragmentation effects, urban developments close to natural areas result in an increase of human presence (which brings with it direct damage, disturbance, waste disposal) and the risk of invasions of allochthonous biodiversity through gardening and other man-driven biotic exchanges (e. g. ornamental plants, pets: Marco *et al.,* 2010; Balogh *et al.* 2011). Generally, agricultural growth causes different effects in natural ecosystems since the suitability of the agricultural matrix is greater than the built one (Vandermeer and Perfecto, 2006). Effects depend mainly on the type of agriculture (extensive *vs* intensive). Although the reduction of natural remnants (in area and habitat complexity) is undeniable, in many cases the fauna can use cultivated fields and take advantage of their resources, while much biodiversity can survive in field margins and other landscape elements of extensive agricultural landscapes (Duelli and Obrist, 2003; Burel *et al.*, 2013).

Since the boundaries between natural and urban areas are usually very narrow, a steep gradient is created (Crooks et al., 2004). So, effects on biodiversity are usually stronger than those resulting from agricultural fragmentation, except in the case of intensive (e.g. greenhouse) cultivation. In fact, urban fragmentation is considered as a major threat to biodiversity and the cause of important local extinctions (Czech et al., 2000; McKinney, 2002). Several studies concluded that urban expansion promotes biodiversity homogenization, leading to a similar composition of species in fragmented natural patches (McKinney, 2006; Schwartz *et al.*, 2006; Olden, 2006; Wang *et al.*, 2014). Such effects could be increased in Mediterranean semiarid areas where dry conditions and high temperatures slows biodiversity recovery, while intensive land uses can be speeded by mild climate and human subsidies (particularly imported water resources; Martínez Fernández and Esteve Selma, 2004).

In the case of agricultural expansion, the effects can be similar. Agriculture is the most important land use in Europe (Reidsma *et al.*, 1996) resulting in a pressure to biodiversity and this is likely to continue (Tillman *et al*., 2001). Extinctions of some species have been found after the spread of greenhouses in the Southeast of Spain (Mota *et al*., 1996). Furthermore, agriculture can be a vector of exotic species which could replace natural vegetation (Brockerhoff *et al*., 2008). However, traditional Mediterranean agriculture is known to contribute greatly to the biological and cultural richness of landscapes, either compositionally, structurally or functionally (Reidsma *et al*., 2006; Blondel *et al*., 2010; Peco *et al*., 2012, Robledano *et al*., 2014)

Whatever the type of fragmentation considered, in all cases and edge effect is created. The consequences are similar to those already discussed. In this case, the effects occur in the natural strip closest to the interface increasing the risk of extinction in these zones (Fahrig, 2001; Kupfer *et al*., 2006).

Besides the agent of fragmentation involved, the type of habitat of the patch (its structure and composition) is also expected to affect the response of biodiversity to fragmentation. In semiarid Mediterranean areas, shrublands or sparse forests are the typical forest formations when restrictive climatic conditions dominate (namely where yearly rainfall is below 300-400 mm). Despite this fact, in the XXth century pine plantations (almost exclusively of *Pinus halepensis* Miller) extended massively in the Southeast of the Iberian Peninsula, occupying areas with favorable conditions for the growth of this species, but also large surfaces below the above indicated threshold. Nowadays, most of these plantations are no longer exploited, becoming either dense unmanaged forests (Andrés and Ojeda, 2002), or decaying, low-stature (shrub-like) sparse woodlands (Bonet, 2004). Studies of the response of biodiversity to these monospecific plantations have found negative effects for local fauna (Díaz *et al*., 1998; Romero-Alcaraz and Ávila, 2002) and flora (Mazurek and Romane, 1986; Chiarucci and De Dominicis, 1995). Greater tree canopy results in more floor litter (Rosetti *et* al., 2015) and the allelophaty of pine litter also contributes negatively to biodiversity values (Fernandez *et al*., 2013). Increasing human growth close to reforested areas has often caused further fragmentation of these man-made ecosystems, with cumulative effects on their biodiversity. Fragmentation has also occurred in shrublands, but with fewer studies having assessed the response of biodiversity in these areas. The importance of these seniarid ecosystems relies mainly in its plant component which is characterized by high species richness and level of endemism (particularly edaphic endemisms), this latter characteristic being strongly associated with the shrub habit (Ojeda *et al*., 2001). Moreover, the value of shrubby habitats for birds and other vertebrates in Mediterranean mosaic landscapes is well established (López and Moro, 1997; Mangas et al., 2008; Pita et al., 2009).

Management measures are necessary to improve the biodiversity inside the natural patches (whatever forest or shrub), and to take advantage of the opportunities outside the patch, like land abandonment, for the same aim. To determine these measures, vegetation and birds have been used in this research as biodiversity indicators, with the advantage of having already been used in previous studies in the same territories (Zapata and Robledano, 2014; Zapata et al., 2014). Knowledge about vegetation structure and composition is easy to obtain, and provides important information on intrinsic floristic value and habitat physiognomy, being also related with the faunal component (Kati et al., 2004). Bird sampling requires significant prior expertise, but once acquired, delivers valuable information on the structural preferences and differential responses of species to several human stressors affecting forest patches (Zapata and Robledano, 2014).

This paper seeks to define, for these isolated patches, management guidelines to be included in reforestation and ecological restoration policies, that allow them to fulfill their ecological, economic and social functions, as pursued by the new European Union's Forest Strategy (http://ec.europa.eu/agriculture/forest/index_en.htm). The specific aims of the study are: i) to assess biodiversity and conservation value of different ecosystem types represented in the forest patches, including pine and shrub reforestations, and natural shrublands, in different landscape contexts (natural, agricultural, urban); ii) to classify patches according to their intrinsic ecological features and to search for relationships between these and physical or human characteristics defined by such context; and iii) to propose management measures to be implemented inside and outside the forest remnants, in order to improve biodiversity in the existing patches and to buffer the fragmentation-related effects of future developments.

Our working hypothesis is that the density of the tree layer (of *Pinus halepensis*) is the key internal factor for ecosystem development when human reclamation has already acted on the forest landscape. For that reason, the management measures should be different depending on the original type of ecosystem that experiences further fragmentation, as shall the recommendations for the creation of new patches of forest habitat.

4.2. Material and methods

4.2.1 Study area

Research was conducted in the Municipality of Murcia, the capital city of Murcia Region's Autonomous Community, located in the South East of the Iberian Peninsula (Figure 4.1) and with a total surface area of 881.83 km^2 (https://www.murcia.es/medio-ambiente/medio-ambiente/estado/poblacion.asp). The prevailing climate is Semiarid Mediterranean, mean annual rainfall not exceeding 300 mm, but with great irregularity in its distribution, both between and within years.

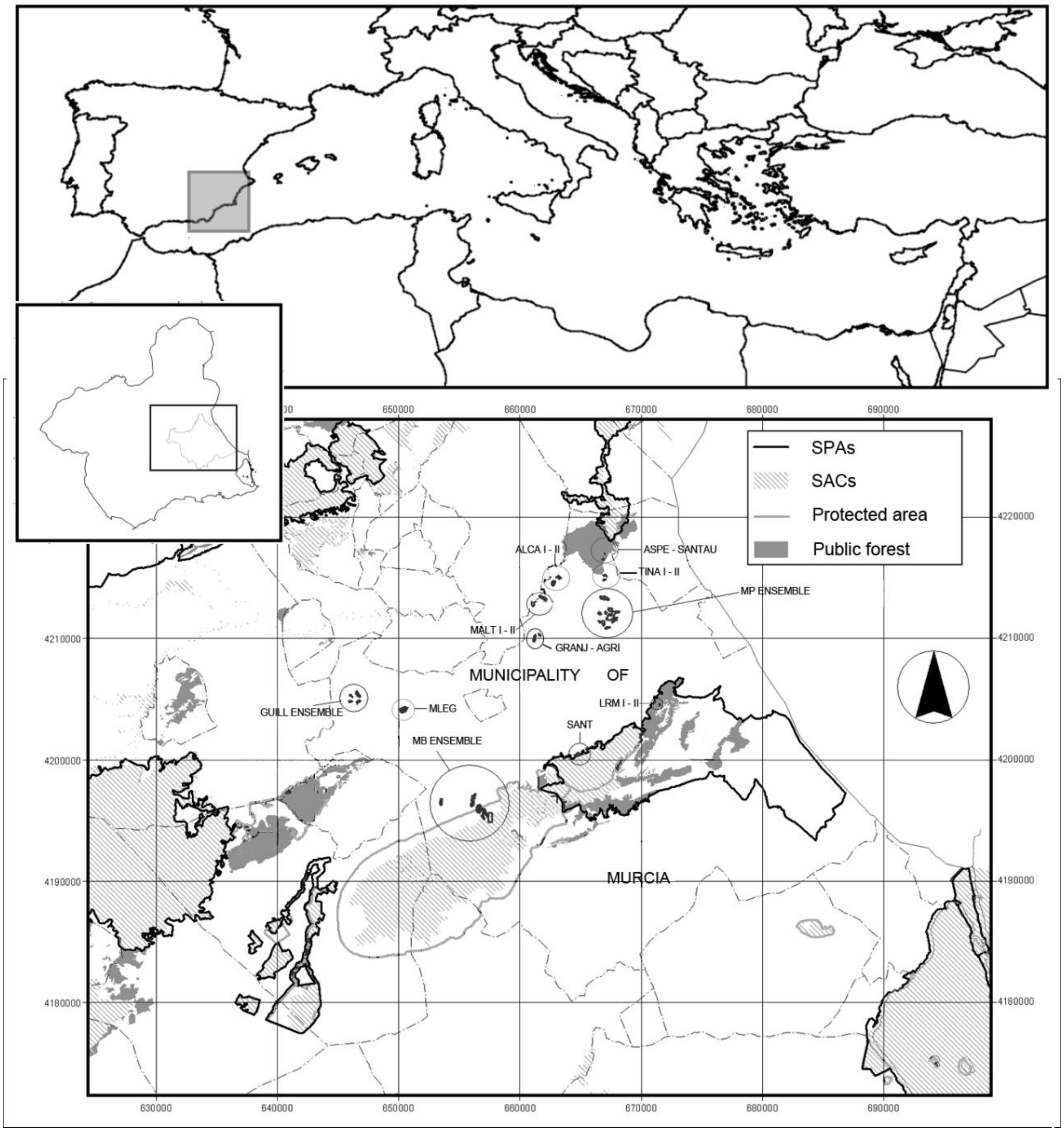


Figure 4.4: Studied patches or ensembles distributed in Murcia´s Municipality. Also figure shows protected areas, public forest and European protected areas.

Historically, various development drivers (mostly agricultural and urban demands for land) have fragmented natural landscapes in Mediterranean areas. Consequently, numerous natural patches of different size and isolation can be found in man-modified landscapes, particularly in lowlands between mountain areas. Specifically, in Murcia Region, the landscapes originated from earlier forest reclamation for fuelwood, agriculture and pasture, have been secondarily modified through monospecific *Pinus halepensis* (Aleppo Pine) reforestations carried out mostly in the 50-60´s of the XXth century. Commonly, this tree was planted beyond its rainfall ecological limit (Esteve *et al.*, 2003), where it naturally grows in low density, and with a sparse or patchy distribution. Pine plantations were made both over large public surfaces and in smaller private estates, many of which have been secondarily fragmented by later agricultural developments (irrigated agriculture), urbanizations, and other land use changes approaching forest limits. In the less favourable locations for pine growth, i.e. poor soils with scarce rainfall (tipically <300 mm) in sun-facing slopes, forest areas have a poorly developed pine canopy, and an understory whose development can be strongly conditioned by pine density (Zapata and Robledano, 2014; Zapata *et al.*, 2014).

Ensemble	**Patch**	**Surface (has)**	**Isolation Agent**	**Age of Reforestation**	**Live Pine Density (id/has)**
MP	CANTA	12.74	Agricultural-Urbanization	>50 years	371.43
	MPBIP	5.48	Urbanization	>50 years	241.67
	MPSIP	2.43	Urbanization	>50 years	416.67
	MPWEP	3.26	Urbanization	>50 years	300.00
	MPPVN	9.00	Agricultural	>50 years	485.71
	MPSOP	2.98	Agricultural-Urbanization	>50 years	533.33
	MPEAP	6.16	Urbanization	>50 years	366.67
	MPPPA	34.35	Agricultural-Urbanization	>50 years	236.00
MB	MBNOP	20.64	Urbanization	>100 years	555.56
	MBCEP	17.28	Urbanization	>100 years	400.00
	MBSOP	18.90	Urbanization	>100 years	728.57
	CARR	5.80	Agricultural	>100 years	587.50
	CARR_AI	5.55	Agricultural	-	75.00
	CARR_AIII	6.29	Agricultural	-	100.00
	CARR_RI	10.50	Agricultural-Urbanization	<50 years	100.00
	CARR_RII	3.30	Agricultural-Urbanization	<50 years	0.01
	CARR_RIII	8.82	Agricultural-Urbanization	<50 years	22.20
GUILL	GUILL-I	2.00	Agricultural	<50 years	2062.50
	GUILL-II	3.96	Agricultural	>50 years	1828.57
	GUILL-III	7.52	Agricultural	>50 years	688.89
	GUILL-IV	5.40	Agricultural	>50 years	877.78
-	MALT-I	13.18	Agricultural	<50 years	466.67
-	MALT-II	6.11	Agricultural	<50 years	1044.44
-	GRANJ	23.46	Agricultural	>50 years	433.33
-	SANTO	8.57	Urbanization	>100 years	737.50
-	LRM-I	3.64	Agricultural	-	0.00
-	LRM-II	26.63	Urbanization	>50 years	650.00
-	ALCA – I	2.96	Agricultural	-	33.30
-	ALCA – II	5.02	Agricultural	-	22.20
-	TINA – I	1.45	Agricultural	-	0.00
-	TINA – II	6.83	Agricultural	-	125.00
-	ASPE	5.71	-	<50 years	112.50
-	SANTAU	9.22	-	<50 years	500.00
-	MLEG	20.99	Urbanization	-	222.72

Table 4.1: Main patches characteristics and their pertinence to ensembles or not.

The number of patches sampled was 36 for vegetation and 34 for birds. There was a variety in size, surrounding landscape, internal vegetation physiognomy, protection status and connection with other natural continuous areas. (Table 4.1). Some patches were located close together and can be considered functionally related. Such local networks of patches are called ‘ensembles’ hereafter. The three ensembles defined were Montepinar (MP), Majal Blanco (MB) and Guillermos (GUILL). Although the age since reforestation was in most cases greater than 50 years, three patches reforested only some 20 years ago (CARR_R), and two patches of 30 years old abandoned crops, naturally reforested without human intervention (CARR_A), have been sampled.

4.2.2 Sampling methodology

Vegetation was sampled using circular plots of 100 m2 (set out in the field with a 5.64 m rope attached to a nail), in which the presence of all woody species was recorded in order to determine their frequency of occurrence. Annuals and biennials were not sampled as well as grasses, to overcome identification problems which would have impeded a rapid survey. Systematic sampling was performed with plots distributed in a grid or transect, with a separation of 50 m on either direction. The exact number was coarsely proportional to the size of the patch, modulated by shape as well as by soil and topographic heterogeneity. Patches larger than 10 ha generally had 9 or more plots, and those smaller usually 5-7. Finally, the linear coverage of all physical and vegetation layers (rock, bare ground, scrubs, chamaephytes, perennial grasses, litter, mosses and lichens, and annuals) was calculated, from their projection on a tape extended along four 10 m transects oriented towards the cardinal points. The values were expressed as mean percentage.

Bird sampling was conducted through 5 min point counts (Bibby et al., 1992) in predetermined locations, scoring all the visual or auditory contacts, with no limit of detection (Blondel et al., 1981). Plot centres were located 150 m apart from each other and at least 100 m from the edge of the patch. Like vegetation plots, the number of bird sampling points was approximately proportional to the size of the patch, in combination with topographical heterogeneity (slope, exposures). Larger patches (> 10 ha) generally contained 3-4 points and the smaller ones 1-2 points. Sampling was carried out during the first four hours of daylight. Two visits were made to each patch during the breeding period (the first one between April 15th and May 15th, and the second between May 15th and June 15th). For each patch, the mean number of birds recorded per plot (averaged among the two sampling dates) was used as an index of abundance.

4.2.3 Bird and flora indexes

Conservation indexes for birds and woody flora have been calculated. The indexes are based in the original expression of Pons et al. (2003) (Equation1), adapted by Paquet et al. (2006) to be used with frequencies. They combine the status of protection or threat level (SPEC in this case), with the relative abundance (frequency, density) of each species (A_i).

$$\sum_{i=1}^{k} \left[\log(A_i + 1) \times SPEC\ value_i\right]$$

The same equation has been used to obtain conservation value indexes based on the European Directive 2009/147/CE (Birds Directive), the Red Data Book of the Birds of Spain (Madroño *et al.*, 2004), and the Murcia Region's Red Data Book of Wild Flora (Sanchez-Gómez et al. 2002)

4.2.4 Statistical analyses

From the composition of woody flora and avifauna, two classification analyses were performed. Then, Indicator Value Analyses (IndVal) were used to extract indicator species that could help interpretation of the groups obtained by the classification.

Conservation value indexes, woody plant richness and bird diversity were used as response variables (biodiversity indicators), and physical, cover and pine structure variables as predictors (environmental variables). These variables defined the habitat type and management of each patch, which was also characterized by its landscape setting (defined by the surrounding development). Their relationships were analyzed by means of Partial Least Squares Regression (hereafter PLSR) (Swold *et al.*, 2001; Tobias 2003; Mevik and Wehrens, 2007) using each patch as a simple unit. This technique has been used in several ecological studies of various types of communities (Zhang *et al.*, 1998, Zhang *et al.*, 2007; Moya-Laraño and Corcobado, 2008), including forest habitats (Rey-Benayas *et al.*, 2010; Sánchez-Oliver *et al.*, 2014). It is particularly recommended when sample size is low and in cases of severe multicollinearity (Carrascal *et al*, 2009). Analyses were performed with the free statistical package R, using MASS, ade4, labdsv and pls libraries (R development Core Team, 2009).

Three groups of PLSR analyses have been performed: the first one with physical variables (Altitude, Distance to patches smaller than 30 hectares, Distance to patches between 30-100 ha, Distance to patches greater than 30 ha, Shape index, Minimum slope, Average slope, Perimeter and Surface area); the second one with cover variables (Bare ground, Rock, Litter, Grass, Perennial grasses, Chamaephytes, Shrubs, Lichens and Mosses, Herbaceous and Tree Cover); and the third one with pine stand physiognomy and development, including the density of different age/condition classes (seedling, sapling, live pines, dead pines, pines up to 6 meters height), and mean Diameter at Breast Height (DBH).

The significance of each PLSR analysis was tested, and only significant PLSR results have been interpreted. For each response variable, only analyses with significant relative contribution of predictors in both positive and negative sides were taken into account, to facilitate the interpretation of axes. The relative contribution of each predictor was calculated by means of the square of predictor weights (Rey-Benayas *et al.*, 2010). Those analyses that meet the requirement were plotted indicating the group of the classification to which each patch belongs. Quadratic or linear fit was calculated and the minimum or maximum point (critical point) was represented also in the plot. The centroid of the data cloud for each group was represented, as well as the standard deviations in both directions to determine the degree of dispersion of the data within it. The distance between the centroid of each group and the critical point was also indicated in the plot. Finally, the fitting equation (linear or quadratic) was presented in the same figure.

4.3. Results

4.3.1 Classification and indicator species

Vegetation

The best plant grouping produced six clusters (Figure 4.2), but two of them included only one or two areas and were not taken into account.

Group ♦ consists of all patches of the Montepinar ensemble. This local network of patches has achieved a similar patch plant development, even though the reforestation of these areas occurred after the first agricultural fragmentation. During the last 20 years the urbanization process has caused a secondary fragmentation process that has isolated some patches. In that short term, urban development has not changed the plant community. Two indicator species appear for this group, *Lavandula multifida* and *Paronychia suffruticosa.*

Group ■ is formed by forest patches with a good representation of some typical forest shrubs and chamaephytes. The case of CARR_R patches indicates that the reforestation of these areas was made according to the surrounding plant communities. The difference with natural shrublands is that the growth of these reforestations has been slow, and thus their physiognomic appearance differs. *Anthyllis citysoides* appears as the only indicator species, which indicates that species composition differs considerably among the constituent patches, their grouping being explained by the similarity in structure of the resulting vegetation.

Group ▲ is a mixture of forest areas and scrublands, for which it's hard to find a unifying characterization. In the same way, no indicator species have been found.

Group × is composed by well developed and extensive scrublands. GUILL-I is an exception since it is the area with the highest pine density, but trees are underdeveloped (two m of maximum heigth) and confer it a shrub-like physiognomy. It is noteworthy that in these areas human influence is usually localized or buffered by linear landscape elements (forest strips, hedgerows or dry riverbeds).

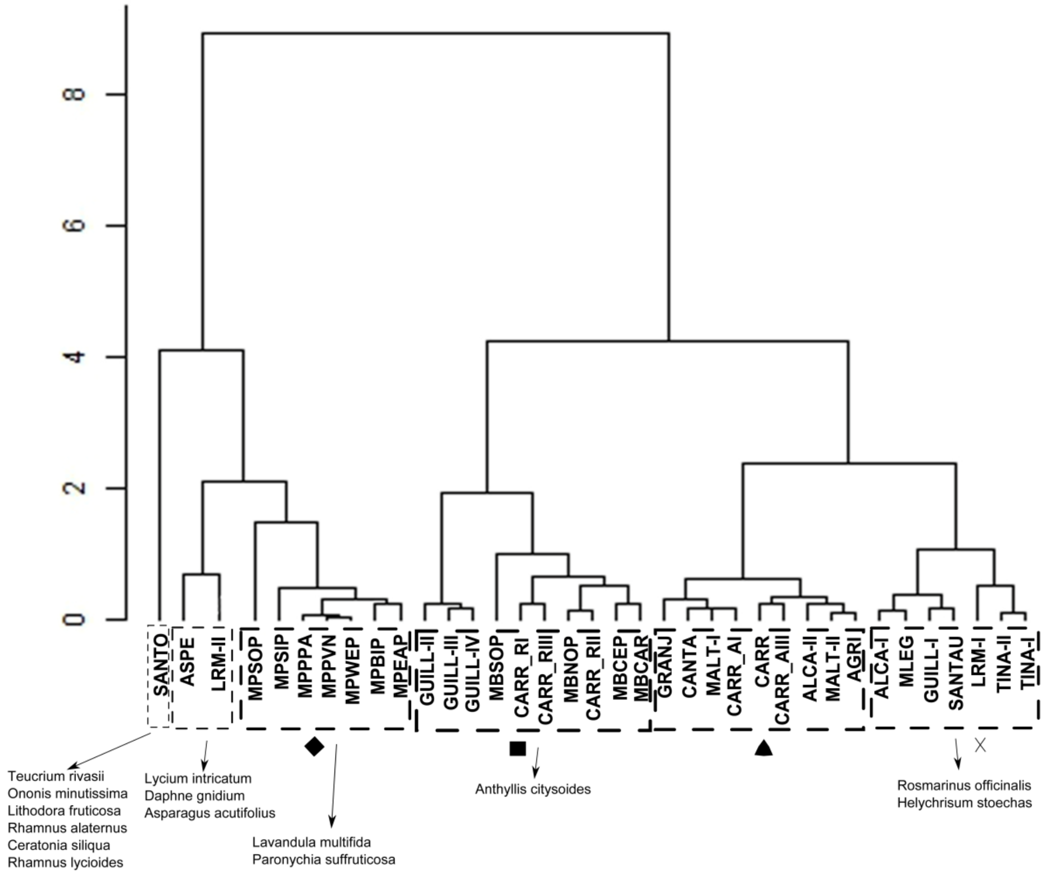


Figure 4.5: Classification of patches by their flora composition and indicator species of each group. Additionally, symbols that represent each group have been added.

Birds

The classification results in that case in five groups (Figure 4.3), two of them again formed by one or two patches and not taken into account.

Group ♦ is formed by the best structured forest areas (except GUILL-I). In this group, surrounding landscapes become less important, while the inner forest patch formation seems to gain importance. In fact, the only indicator species is *Aegithalos caudatus*, a typical forest-dwelling species. Most of the patches of this group are included in ♦ and ■ vegetation groups.

Group ■ mixes the worst structured forest patches with open forest-shrubland areas, including reforested and naturally recolonized abandoned agricultural fields. These heterogeneous mixes of ecosystems doesn´t result in any indicator species. The patches included in this group are shared with ■ and ▲ flora groups

Group × is formed by shrub patches in a landscape context with low influence or total absence of forest. The predominant physiognomy is a steppic one (dry pasture or dwarf shrubland). Although, there is no indicator species, hopefully different bird community occupy them due to the type of ecosystem that they represent. Similar characteristics are found in patches of the × vegetation group.

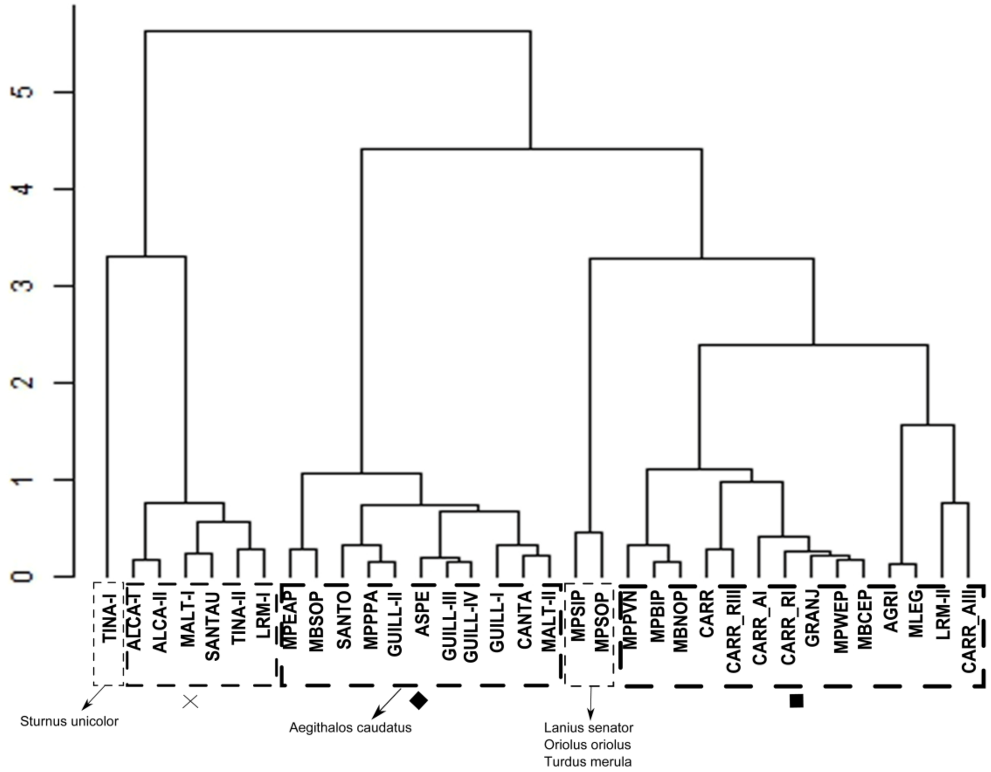


Figure 4.3: Classification of patches by their birds composition and indicator species of each group. Additionally, symbols that represent each group have been added.

4.3.2 PLSR Results

Table 4.2 shows all PLSR results. The analyses with significant response variables on both sides (positive and negative) are shown in bold. Only Cover PLSR fulfilled this condition. The other two groups of variables (physical and pine stand variables) were not taken into account.

In general, a forest development axis is found, showing an influence of canopy and pine litter in one side (which can be associated to forested areas), and of bare ground in the other side, which is interpreted as characteristic of open, less forested areas, like shrublands or sparse woodlands (with a scrub layer and pine trees only in favored locations).

Response variables	Predictors	Variance	Scores -	Scores +
RFLOR	**COVER**	**23.25**	**Bare Ground, Litter, Canopy**	**Grass, Chamaephytes, Shrubs**
RFLOR	PHYS	12.22	-	Average Altitude
RFLOR	PINE	4.6	Inmature, Live	-
RBWP+USE	**COVER**	**30.51**	**Litter, Canopy**	**Bare Ground**
RBWP+USE	PHYS	15.02	-	Average Altitude
RBWP+USE	PINE	12.23	Inmature, Live	-
RARE	**COVER**	**43.08**	**Litter, Canopy**	**Bare Ground**
RARE	PHYS	21.34	-	Average Altitude
RARE	PINE	20.24	Inmature, Live	-
SPEC	**COVER**	**5.85**	**-**	**Litter, Canopy**
SPEC	PHYS	6.07	Average Altitude	-
SPEC	PINE	4.12	-	Inmature, Live
RBBS	**COVER**	**10.03**	**Bare Ground**	**Litter, Canopy**
RBBS	PHYS	5.43	Average Altitude	-
DBIRDS	**COVER**	**7.54**	**Litter, Grass, Canopy**	**Bare ground**
DBIRDS	PHYS	2.23	-	Average Altitude
DBIRDS	PINE	5.47	-	Inmature, Live
RBIRDS	**COVER**	**6.27**	**-**	**Litter, Canopy**
RBIRDS	PHYS	2.60	Average Altitude	-
RBIRDS	PINE	4.80	-	Inmature, Live
DIVBIRDS	**COVER**	**22.24**	**Bare Ground**	**Litter, Canopy**
DIVBIRDS	PHYS	4.91	Average Altitude	-
DIVBIRDS	PINE	16.30	-	Inmature, Live

Tabla 4.2: PLSR results. Bold results shows significant results (p-value <0.05) and predictors variables with significant + or – score.

4.3.3 PLSR plot and trends.

Figure 4.4 represents the values of the response variables against the first axis of the PLSR analyses. In bold, the average value of each group of patches and their dispersion degree in both directions and the distance between this central point and the critical point (minimum in Flora PLSR and maximum in Birds PLSR depend on fitting equations)

Plant richness

Although variance explained was 23.25%, minimum richness was at close distance from the central points of patch groups. Even so, maximum richness was found in open forest areas, although the difference with scrublands and human-isolated areas (forest or shrubland) was only 3-4 species on average. The number of species recorded in all patches is similar.

RBWP + USE plant conservation index

Variance explained rises to 30.51%. The scrubland group is the farthest from the minimum. Besides, higher index values are found with low dispersion of data. Casually, the group with higher plat richness appears now close to the minimum and has low dispersion in both axis, so all the patches are located close to the minimum index value.

RARE plant conservation index

This PLSR obtained the highest variance explanation with 43.08%. Results are similar than for the RBWP+USE index. Again, scrublands obtain higher index value and are farther from the minimum than the rest of groups. The forest patches group is found close to the minimum again.

Bird diversity

This analysis attained the highest variance explanation of the birds' PLSRs (22.42%). Forest and scrubland isolated from human developments (agricultural and urban) obtain higher diversity and are close to the maximum. It is noteworthy that scrublands are the group that is found farther from the maximum in the negative sense.

RBBS bird conservation index

The results of this index are similar to those found for diversity. The same group is close to the maximum, and scrublands are the farther group. The variance explained by this PLSR analyses was low (10.03%).

DBIRDS bird conservation index

Opposite results were found, the group with higher index values and close to the maximum being scrublands and open (non forest) areas. This analysis, however, had the lowest percent of variance explained (7.54%).

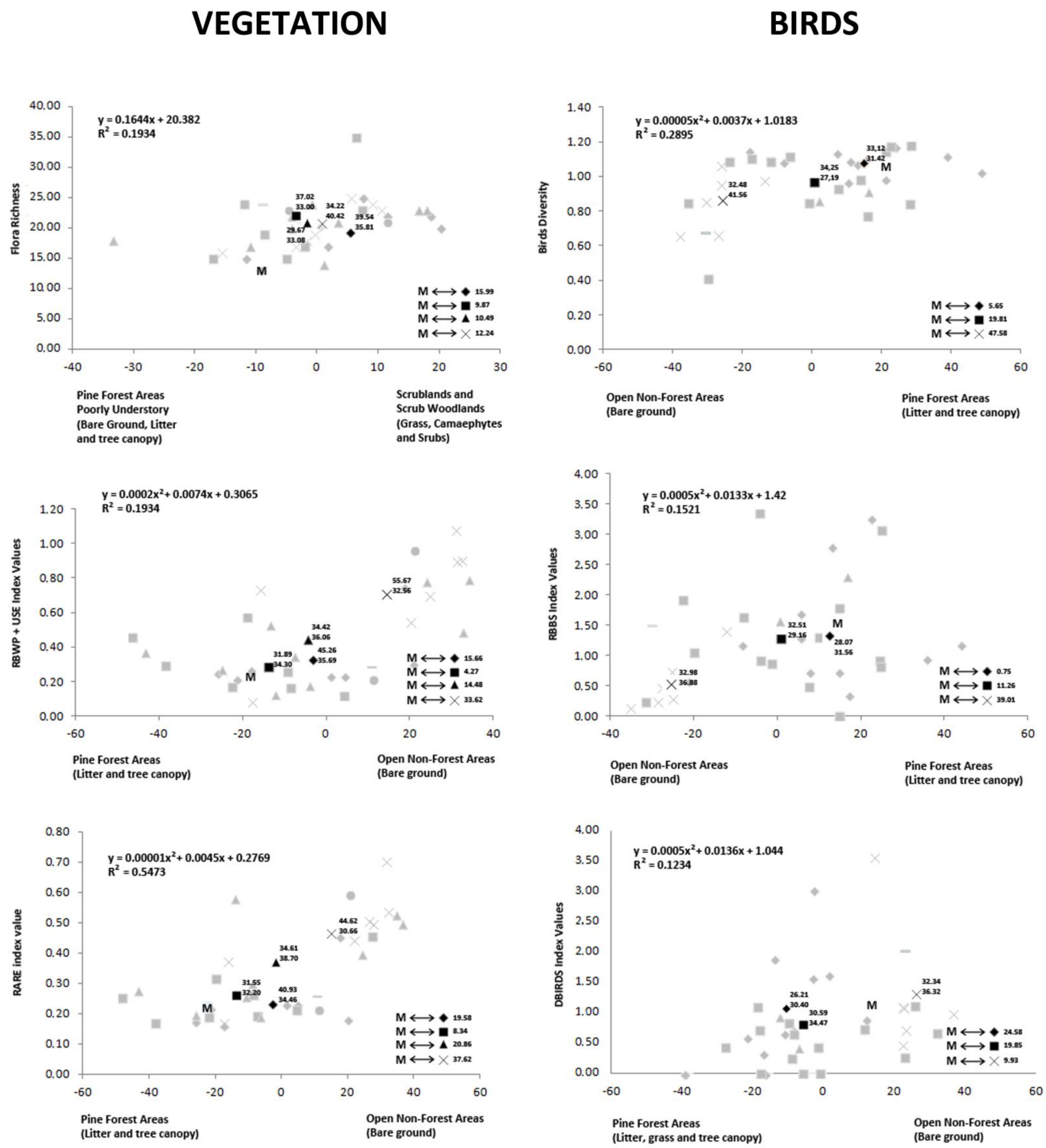


Figure 4.4: Graphical representation of PLSR results with centroid (and their location in the spatial area) of each group and critical point (minimum or maximum point) in bold. Furthermore, lineal or cuadratic adjust of points are showed. Also, distance between centroids and critical points are indicated.

4.4. Discussion and conclusions

4.4.1 Response of semiarid vegetation by patch groupings

Forest patch ensembles

It is noteworthy that classification grouped Montepinar areas in a separate cluster from the rest of patches. Before the second event of human fragmentation these patches formed a larger single unit, and similar species appear in the resulting sub-units. Urbanization has physically fragmented the original area, but time has not been enough to change the plant community. Some studies have that fragmentation can decrease genetic diversity and promote inbreeding (Lande, 1986), increase exposition to wind, warm the air and soil temperature (Heithecker and Halpern, 2007), alter plant-animal interactions (Ferreria and Laurence, 1997), limit seed dispersal (Cordeiro and Howe, 2003), and reduce insect pollinator services (Tomimatsu and Ohara, 2002). So it is expected that plant composition will change in the future, at least in the smaller fragments.

Regarding *L. multifida* as indicator species, it is typical of deforested *P. halepensis* primary forests subjected to overgrazing, habitat fragmentation and overexplotation as main threats (Chograni *et al.*, 2008). According to these authors, *L. multifida* maintains higher genetic diversity after fragmentation than other *Lavandula* species. In the case of *P. suffruticosa*, the other indicator species, as exozoochorous and animal-pollinated species (Melendo *et al.*, 2004), it is expected to be affected by the problems previously listed. In the case of *P. halepensis* (the only arboreal species), it was reforested in the 1950´s and its present development is low. Its dispersal is conditioned by the suitability of the receiving microhabitats.

For patches with high pine densities, but generally unsuitable for tree growth, the reduction of such density seems the better option. By doing so, the development chamaephytes will be favored (including the indicator species) thus increasing the conservation value of these areas (Ojeda *et al.*, 2001; Zapata and Robledano, 2014). In addition, solutions should be sought to reduce the long term effects of fragmentation and land-use change on isolated plant populations. One potential measure, the use of autochthonous species in gardens, has several benefits. It increases permeability of edges since these species increases nectar availability for pollinators (Corbet *et al*, 2001), also enhancing habitat connectivity (Parker *et al.*, 2008) and functional connectivity (Rudd *et al,* 2002). The Montepinar experience (an *outdoor laboratory* for recent research, e.g. Zapata *et al.,* 2012; Zapata *et al.*, 2014; Martínez *et al.*, in prep.) can be useful for the design of future urban developments, since the lack of buffers surrounding forest patches embedded in urban areas is expected to maximize isolation and minimize total habitat area (Goddard *et al.*, 2010). These buffer areas could be green spaces or extensive agricultural ones.

Structured forest areas

Some reforestation techniques, popular along the XXth century, were implemented without ecological basis (high tree densities, intensive preparation techniques, low species diversity…). But in this case, the patches belonging to this group correspond with newly reforested areas (during the last 20 years) as well as areas reforested areas in the beginning of the XXth century.

One of the most frequent reforestation technique in the mid-XXth century was the plantation of high densities of Aleppo pine. Several studies have demonstrated that monospecific pine reforestations in Mediterranean areas have negative effects on the regeneration of other vegetation groups (Andrés and Ojeda, 2002; Pausas *et al*., 2004), and on soil characteristics (Ruiz-Navarro *et al*., 2009). In our case the patches included in this group show an opposite trend, exhibiting a valuable plant representation. Reforestations of the beginning of the XXth century are more developed than new reforestations. Specifically, some reforestations done at the beginning of the last century in the Region of Murcia have been considered as model examples of high quality reforestations at national level (Esteve *et al*., 2003)

The only plant indicator species found, Anthyllis cytisoides is a legume particularly common in disturbed areas (Haase et al., 1997). Normally, it is not used in reforestation but its colonization potential allows it to occupy all forest of reforested areas which were disturbed in the past. This specie appears as early an colonizer and can survive in the landscape several decades after disturbance process (Pugnaire, 2006)

Mediterranean shrublands

Limiting physical environmental conditions (aridity) cause that some interactions as competition and facilitation determine community structure and functionality (Gross *et al.* 2013). But historically human factors have been the main determinant of shrubland developement through activities like grazing, fire and cutting. Also, the absence of human activity has promoted the persistence of shrublands (Calvo *et al*., 2012).

In our case, fragmented shrublands contribute to the local flora with different species than the other groups. Among these, native chamaephytes are usually the most representative species. The species of this life form have global importance since they evolved once the climate of the Mediterranean biome was already established (Herrera, 1992).The absence of indicator species make these areas more interesting since although physiognomy and structure are similar between patches, composition species is different depending on the area. This increases plant diversity at local scale (patch ensembles) and in the whole municipality.

4.4.2 Bird response to fragmentation effects

Importance of well structured and connected forest areas

Restrictive climatic conditions in semiarid Mediterranean make well structured tree forests formations a rare vegetation endpoint. In addition, climate change drives vegetation species substitutions and changes in forest physiognomy (Thomas *et al*., 2004; Svenning and Skow, 2006; Lenihan *et al*., 2008; Esteve-Selma *et al*., 2012).

In this study, the long tailed tit (*Aegithalos caudatos)* was the only bird indicator species. Is a typical forest-dwelling species which responds negatively to the increasing distance between patches and positively to the amount of habitat in 1 km^2 (Jansson and Angelstam, 1999). So, networks of nearby forest patches could minimize problems caused by fragmentation on species with such requirements (i.e. allowing bird communities to keep these forest representatives).

At landscape scale, the patches that are part of this group are medium-sized fragments with human habitat (agricultural or urban) interspersed. Although fragmentation occurs, its low intensity (i.e. closeness between forest remnants, persistence of undeveloped soils) still allows the maintenance of functional networks of forest patches and bird communities representative of them. Therefore, we should advocate for maintaining these forest mosaics without an increase of human fragmentation. In this way, forest bird communities could be kept close to people, either visitors of forest patches or residents in their neighbourhood (social benefits), and contribute to the conservation of bird species at the scale of the municipality (particularly in lowland semiarid areas with a rapid decline in forest surface).

At the intra-patch scale, structural vegetation enrichment combined with the maintenance of woodlands and patches of shrubs without trees, will allow the presence of forest generalists and other bird species typical of open habitats (López and Moro, 1997; Blondel *et al*., 2010). This results in an increase of bird diversity at higher scales, i.e. ensembles and the whole municipality (Zapata and Robledano, 2014)

Birds of shrublands and open habitats

Contrary to the previous group, open and shrubland vegetation patches constitute a separate grouping but with different bird species composition. Although in typical shrubland or steppe ecosystems, the breeding community is formed by a few and characteristic species, in this case, the differences between patches respond mostly to their surrounding land uses (urban or forest). Although these different landscapes determine contribute different species, the community is always dominated by birds of open or shrubland habitats. This explains that no indicator species has been detected.

Generally, steppe communities *sensu stricto* are valued for their rarity (Suarez-Seoane *et al*.,2002) and specifically for their greater ornithological biodiversity singularity (Blondel et al., 2010). Commonly, birds of open habitats *sensu lato* (including dry pastures and low-statured scrublands) are also valuable species since their distribution is restricted to natural or extensive agricultural landscapes in Europe and North Africa, and have persistent negative population trends due to drastic habitat loss and fragmentation (Cramp, 1988; Tella et al., 2005). Moreover, the lack of ornithological assessments in restorations based on different structural models (shrub formations), gives importance to the knowledge of the behavior of birds in steppe and scrublands. These can be taken as objectives to be achieved in future shrubland and agricultural field restorations.

Flora and birds responses in relation to habitat

Maximum plant richness appears in ensembles of forest patches. Although plant richness increases with patch size (Jacquemyn *et al.,* 2001), in our case patch connection seems the key factor to preserve plant richness. However, the conservation importance of vegetation does not directly relate to this factor. Plant conservation index values are higher in areas with shrub species as the main vegetation component (Centroid point farthest from critical minimum). Historically, protection has focused in mature forest and, since social preferences favor landscapes with continuous tree canopies, scrublands have been considered of minor conservation relevance. However these areas are important for different animal groups as Mediterranean carnivores (Lozano *et al.,* 2003; Mangas *et al*., 2008), birds (Herrera, 1984; Hulme, 1997), ants (Azcarate and Peco, 2003) or bees (Petanidou and Semts, 1995). The Mediterranean area and their characteristic shrublands stand out for their diversity of endemic and native chamaephytes (Zapata and Robledano, 2014), which evolved under typical Mediterranean conditions, unlike other inherited lineages of past climates (Herrera, 1992). For birds, these habitats display the lowest diversity and national index value, while for the European conservation index they reach the highest value. Contrary to this, the highest bird diversity and national index values occur in connected patches.

4.4.3 Final synthesis and management proposals

After having analyzed all the cases and management options, a list of management proposals is necessary for each group of natural forest patches.

Techniques like thinning to reduce the number of trees are required in high density forest patches under semiarid conditions. Rainfall usually does not the development of trees except in favorable sites where compensation factors (shade, runoff concentration…) allow individuals or small woodlots to reach full growth. For that reason, clearings that leave isolated trees or small groups can improve general vegetation structure at the patch level (Torras and Saura, 2008). These non forested, man-made areas should be distributed heterogeneously around the patch. Such a mixture of cleared with forested areas will allow forest-dwelling birds to persist, particularly if some patches larger than 30 ha are secured (Zapata and Robledano, 2014). On the other side, clearings and surrounding open areas could host other bird species typical of shrub-forest mosaics. Besides, maintaining isolated trees in the cleared areas can speed shrubland recovery since these structural elements are essential for fruit dispersion as bird perches (Pausas *et al.*, 2006). Furthermore, in areas climatically unsuitable for tree establishment (except in favorable microsites protected from abiotic stress), a facilitation process can occur between pine trees and the seedlings of shrubs (Zapata *et al.*, 2014).

Urban fragmentation affects birds and vegetation in a different way. The mobility of birds allows some species to obtain benefits from anthropogenic habitats (Zapata *et al.*, in preparation). But while moderate disturbance can enhance bird richness, plant density and diversity could be diminished. In that case, it is advisable to apply measures that promote ecological dynamics (e.g. enhancing dispersal), and to reinforce vegetation communities with localized direct plantations of shrubs and chamaephytes.

Some patches present an adequate vegetation structure with optimal tree densities and a good representation of shrub and chamaephyte layers. There, management measures should be limited to monitoring and preserving the area from further human impact. There are gaps in the knowledge of biodiversity patterns and ecological relationships in this type of forest habitats. Few studies have addressed these topics in Mediterranean semiarid natural patches (Maestre and Cortina, 2005; Zapata *et al.*, 2012; Sánchez-Oliver *et* al., 2014; Zapata and Robledano, 2014; Zapata *et al.*, 2014). Consequently, some of them should be taken as outdoor laboratories in which to study the response of different groups of biodiversity indicators (besides birds and flora), as well as the processes of dispersal, facilitation, predation and mutualism, all them expected to be strongly conditioned by the restrictive semiarid environment. Improved knowledge of all these responses and processes is essential to balance the application of active and passive restoration in areas (e.g. oldfields, degraded grasslands) candidate for forest restoration.

Scrubland patches are the third type of natural patches characteristic of the studied landscapes. The results regarding birds and flora reveal some relevant values at European level. These patches often appear embedded in human-forest mosaics, to which they add a shrub habitat were species with Mediterranean origin can coexist with other with more steppic character and greater affinity for semiarid desert landscapes (Blondel *et al*., 2010; Ambarli and Bilgin, 2014; Robledano *et al*., 2014).

In these areas management measures must focus in maintaining their physiognomy and improving shrubland quality. To this aim, shrub reforestations and passive techniques will be needed. Since some Mediterranean birds are legitimate frugivores (Herrera, 2004), passive restoration techniques must be directed at favoring bird movement along natural patches. The use of artificial perches has proven effective in other ecosystems (McClanaham and Wolfe, 1993; Aide and Cavelier, 1994; Sarmiento, 1997; Holl, 1998; Shiels and Walker, 2003; Zanini and Ganade, 2005; Heeleman *et al.,* 2012;

Graham and Page, 2012), but it has still to be assessed in semiarid Mediterranean areas. More studies about this technique are necessary for its future application in these areas and elsewhere. In addition to managing from inside the patch, measures can be applied outside it. Urban pressure causes numerous negative effects on the biodiversity of a patch. Although biodiversity loss is not an immediate net effect of urbanization, as new habitats are created (Grimm *et* al., 2008; Shochat *et al*., 2010), over time this loss is evident and a reduction of the conservation value results from the emergence of generalist species (Zapata and Robledano, 2014). Furthermore, the availability of resources is usually higher in urban gardens than in forest patches, so flying fauna (birds, bats and invertebrates) tends to use the former to get resources (food and water), with the risk of introducing exotic plant propagules into natural patches (precedent of garden plant species) and of causing the loss of ecosystem services in the natural area (dispersal, polinization, etc.). Thus, *outside-patch* management measures in forest-urban interfaces of already built areas should be directed to changing plant composition of private gardens. Educational measures, and even incentives for the use of autochthonous vegetation in private gardens close to natural patches can be highly cost-effective, creating a wildlife-friendly buffer space that can mitigate the effects of urbanization and allow urban people to approach biodiversity.

Future urban development should integrate the set of measures described in this paper. In the Iberian Southeast, built areas have occupied valleys and ascended upslope until construction or mobility costs made it economically unfeasible. This model of sparse development has produced a landscape of urban resorts isolating natural areas in mountain borders and lowland hills. Future planning should address the maintenance of ecological corridors between isolated patches and the natural matrix, as well as the preservation of shrub or extensive agricultural buffers whose protection will incorporate new habitats to the mosaic and increase biodiversity.

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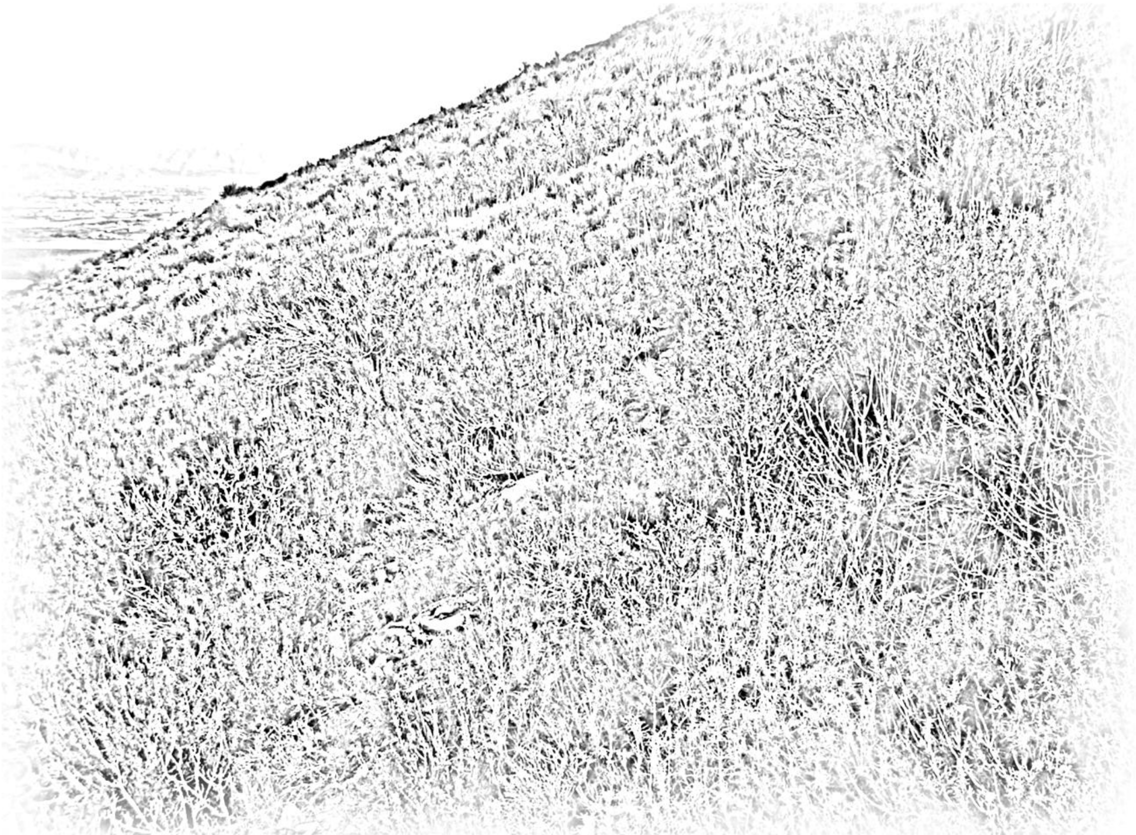
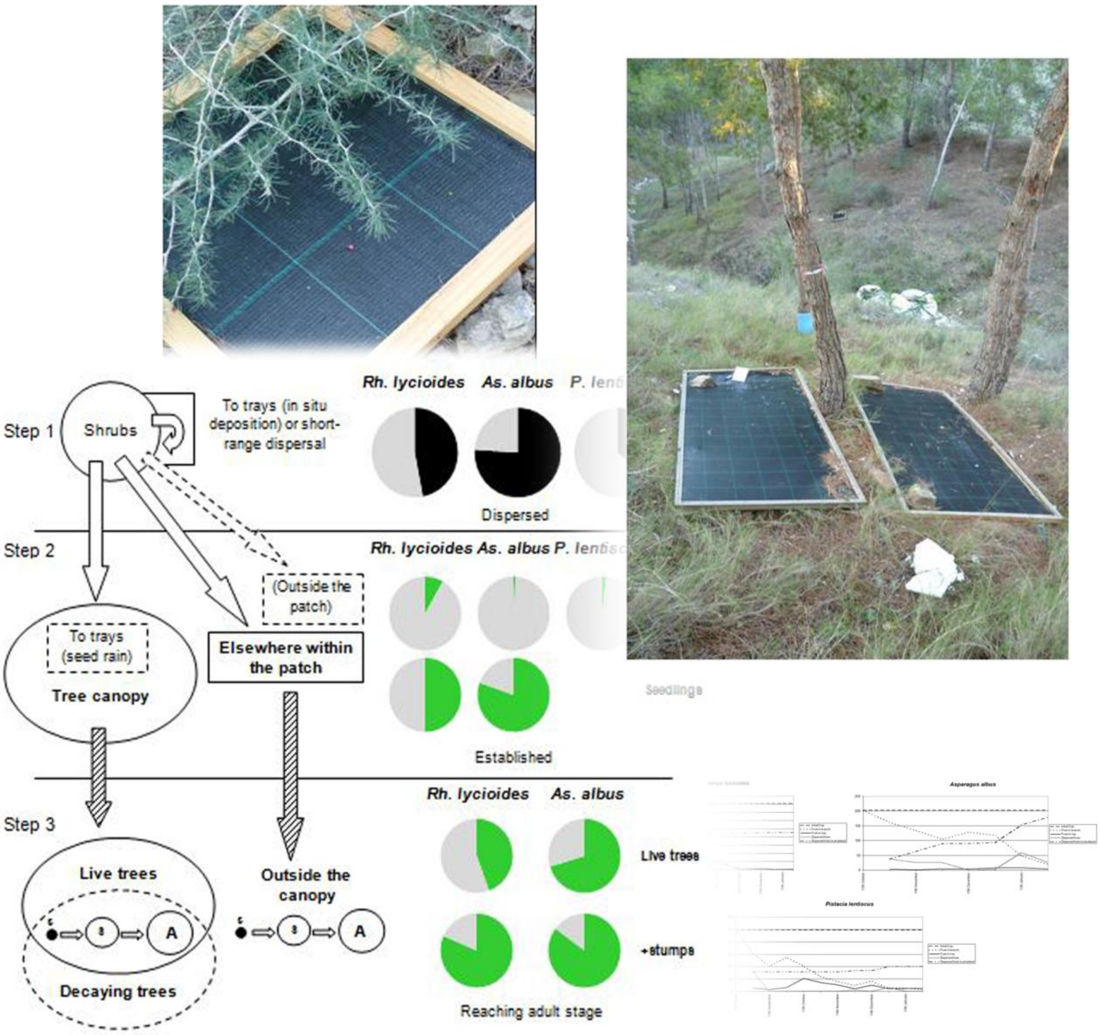
CHAPTER 4



Bird-mediated seed dispersal of fleshy fruits of Mediterranean shrubs in semiarid forest patches. The role of *Pinus halepensis* Miller trees as seed receptors

Graphical Abstract



Abstract

In Mediterranean semiarid areas, limited water supply confers to woody non-arboreal formations (scrublands) a capital importance. Fleshy-fruiting shrubs provide food resources for vertebrates giving rise to well-known mutualisms, poorly studied in marginal Mediterranean areas. We assessed seed dispersal in three shrub species (*Rhamnus lycioides* spp. *lycioides* Brot., *Asparagus albus* L., *Pistacia lentiscus* L.) considering birds as the main vector and pine trees (*Pinus halepensis* Miller.) as key structural elements in seed deposition and establishment. The study area is a small forest fragment with an artificially high standing density of pine trees. The initial hypothesis is that birds direct dispersal towards the tree stratum, thus conditioning shrub regeneration. Collector trays, placed under shrubs and pine trees (potential perches), were used to estimate fruit consumption and seed rain. Shrub density and regeneration were assessed through vegetation sampling differentiating three age classes (seedlings, saplings and adults). The highest estimated consumption by birds was recorded on *A. albus* (76% of the fruits available). *R. lycioides* and *P. lentiscus* were less consumed (47 and 36%) but achieved a relatively important dispersal towards perches (0.88% of the total fruit crop, and 1.89% of the fruits consumed in *Rhamnus*, and up to 2.94 and 8.18% in *Pistacia*, compared to only 0.09 and 0.13% in *Asparagus*). More than 80% of *A. albus* seedlings grew under the tree canopy, also the microhabitat preferred by *A. albus* and *R. lycioides* saplings. Approximately half of the extant adult shrubs were under the tree layer (more than 80% considering dead trees), although canopy cover was only 13.81%. Only a small fraction of the seeds were deposited under trees, but their canopy seemed to provide a favourable regeneration niche, at least during the germination and establishment phases. The management of tree density seems thus crucial for fostering the transition of these reforestations towards healthier and more resilient woody formations.

5.1. Introduction

Mediterranean ecosystems are subjected to a high natural variability and anthropogenic influence that often lead them close or directly under limiting environmental conditions (Di Castri *et al*., 1981; Mazzoleni *et al*., 1992; Esteve-Selma *et al*., 2012). Regarding forest vegetation, the restrictive climate conditions of semiarid areas confer to woody non-arboreal formations a capital importance. Such importance is based both on the significance of scrublands in the functionality of semiarid ecosystems (Mangas *et al*., 2008), and on the vast amount of territory they cover. Furthermore, farmland abandonment and frequent forest fires have expanded woody non-tree habitats (Di Castri *et al*., 1981; Navarro and Blanco, 2006). Surprisingly, despite their limiting conditions for tree growth, in semiarid areas reforestation policies have often prioritized the use of trees over shrubs (Cortina *et al.,* 2011).

The fleshy fruit crops of Mediterranean shrubs represent productivity 'islands' within a 'sea' where food resources are scarce. The dispersal of the seeds of these fruits by vertebrates (mainly birds) has a key role in the dynamics of natural habitats and in vegetation recovery in altered landscapes (Wenny, 2001; Cousens *et al*., 2008). Natural regeneration of shrublands depends on a central concept in dispersal ecology, seed disperser effectiveness (Schupp, 1993), reformulated by Schupp *et al*., (2010) as Seed Dispersal Effectiveness (SDE). Originally presented as the 'number of new adults produced by the dispersal activities of a disperser' (Schupp, 1993), under its new formulation it can be expressed as the 'number of seeds dispersed by a dispersal agent multiplied by the probability that a dispersed seed produces a new adult' (Schupp et al., 2010), the latter involving both quantitative (activity of the dispersal agent) and qualitative (probability of survival across the several stages from removal to deposition) aspects. Within the framework set by Howe (1989), Mediterranean bird frugivores play a key role as dispersal agents for plant species that benefit from a scattered mode of seed deposition. Besides being morphologically and behaviourally suited to the size and characteristics of the fleshy fruits of shrubs, they increase the chances for seeds to escape density-dependent mortality and/or to reach suitable places for germination and survival (Howe and Miriti, 2004).

Although widely studied in typical Mediterranean locations (e.g. Herrera, 1995), the dispersal process, its agents and the resulting patterns of plant density and distribution remain understudied in more marginal areas of the biome. There is a lack of ecological knowledge in which to base the preservation and management of this ecosystem service under semiarid Mediterranean conditions (Garcia *et al*., 2010). These marginal areas represent ecotones between typical Mediterranean and arid subtropical ecosystems (Esteve-Selma *et al*., 2010), and can be expected to display an earlier response to the effects of climate change, a process that can force the transition from tree- to shrub-dominated landscapes (Peñuelas *et al.,* 2004; Valladares, 2007). These shifts have been observed in different compositional and structural features of many other types of ecosystems (Villers-Ruiz and Trejo-Vázquez, 1998; Thuiller, 2004; Calef *et al*., 2005; Wheng and Zhou, 2006; Altorre *et al*., 2008).

To fill this gap, we have studied the dispersal of fruits and seeds from three Mediterranean shrub species within a small (ca. 30 ha) isolated forest fragment of semiarid Southeastern Spain, and the contribution of arboreal structures (reforested trees) to this process. Frugivorous birds are considered to be the main dispersal vector. The study of bird-mediated dispersal and the use of natural or artificial structures (perches) by frugivores as seed-deposition sites has been addressed by many studies (e.g., McClanahan and Wolfe, 1993; Zanini and Ganade, 2005; Heelemann *et al*., 2012; Graham and Page, 2012), but again rarely in regions with semiarid Mediterranean climate.

Semiarid Mediterranean forest fragments are degraded remnants of an original shrubland landscape where the density of trees has been artificially increased through pine plantations. The biodiversity of these fragments is conditioned by the tree layer, largely dependent on human intervention (reforestation episodes). Considering the critical role of remnant trees in the regeneration of shrubs in Mediterranean landscapes (Debussche and Isenmann, 1994; Verdú and García-Fayos, 1996; Rodríguez-Pérez and Traveset, 2007) our main hypothesis is that diaspores of bird-dispersed shrubs will be directed preferentially beneath the tree layer, whose structural role (perch) and biophysical influence (micro-environmental improvement or competition) would condition the regeneration of the shrub layer. Directed dispersal (*sensu* Wenny, 2001) may be common in disturbed landscapes, where it can also be relevant for their restoration. Because the tree layer is more sensitive to environmental change, the study of these processes and interactions is essential to manage the transition towards more resilient and sustainable woody formations (e.g. shrublands or sparse woodlands). Many plants depend on mutualisms with frugivorous animals that must be a priority in the conservation of biodiversity (Tellería *et al*., 2005) due to their key functional role in the preservation of the structure and diversity of plant communities in a global scenario of habitat alteration.

Our specific aims are: i) to analyze the temporal pattern of fruit removal from shrubs and to assess the contribution of frugivorous birds to seed dispersal in the species studied; ii) to study the temporal relationship between fruit dispersal and seed rain under the canopies of Aleppo Pine (*Pinus halepensis* Miller, Pinaceae), indicative of an effective transport by birds to these perching locations, iii) to evaluate the role of such microhabitat as seedling establishment and shrub regeneration places; iv) to discuss the implications of the seed dispersal pattern for the management of fragmented semiarid forests, as a basis for the implementation of measures that enhance the sustainability of these landscapes.

Under the aforementioned hypothesis, if bird-mediated directed dispersal towards the tree stratum occurs in any species, we expect to find: a) higher seed deposition (seed rain) under trees than that expected from their canopy cover (equal to the available perching surface); b) higher seed deposition than that expected from the shrub's relative dispersal potential (fruit abundance and seed removal rate); c) higher survival to adult stage under the tree canopy than in other microhabitats or vegetation strata.

5.2. Material and methods

5.2.1 Study area

The research was carried out in the Municipality Forest Park of Montepinar (38° 2´ 11´´ N, 1° 5´40´´ O), located in the Southeast of the Iberian Peninsula (Figure 5.1). Its total area is 26.86 ha. This is a protected *P. halepensis* forest created as a result of a 50-60 year old plantation (Ramos, 2011). The climatic conditions (290.5 mm mean annual rainfall, distributed mainly in autumn and winter) are limiting for the growth and development of this conifer, which becomes naturally scarce and patchily distributed below 300-400 mm of annual rainfall (Esteve-Selma *et al*., 2003). The study year (2010) was wetter than average with a total precipitation of 414 mm, of which 253 mm were recorded during the study period. The initial reforestation density in Montepinar was too high (624 trees/ha) for such climatic conditions, and the suitability of the area for tree growth seems further limited by restrictive edaphic conditions (lithosols). More than half of the trees planted are dead or have been cleared recently by the Municipality management staff in an attempt to improve the health and survival chance of the remaining specimens.

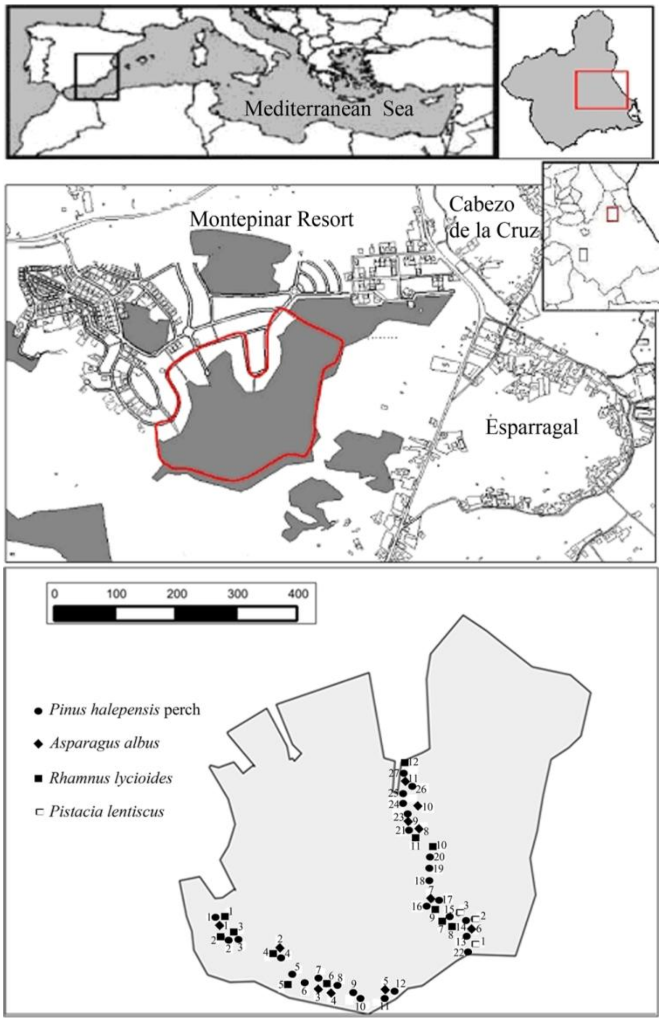


Figure 5.1: Detailed and general geographical setting of the Montepinar Forestry Park in the Municipality of Murcia (location within the Region of Murcia and the Iberian Peninsula/Mediterranean shown in the left and uppermost boxes). The lowermost map shows the distribution of the sampled individuals of shrubs and *Pinus halepensis* (perches) within the Forestry Park.

The understory is dominated by typical Mediterranean shrub species. Three of these species have been chosen for the study: *Asparagus albus* L. (Liliaceae), *Rhamnus lycioides* spp. *lycioides* Brot. (Rhamnaceae) and *Pistacia lentiscus* L. (Anacardiaceae). Table 5.1 shows the phenology of these species, whose density in the study area was different in several orders of magnitude, from the highest value for *A. Albus* (around 2,000 ind/ha), through the intermediate value of *R. lycioides* (around 200 ind/ha) to the lowest for *P. lentiscus* (<1 ind/ha). These densities have been estimated from previous fieldwork in the study area and nearby ones, and refined in this paper through the sampling protocol described later.

The forest bird community comprises a good representation of typical Mediterranean frugivores like *Sylvia melanocephala* Gmelin (Sylviidae), with an average density of 4.60±0.33 individuals/ha in autumn (non breeding season), and of Eurosiberian origin like *Erithacus rubecula* L. (Turdidae), with 3.14±0.28 ind/ha in the same period. The size of these species varies between 12 and 14 cm, and their weight between 10 and 21 g. Although their diet is varied, during fruiting periods these species forage intensively on fleshy fruits. The two species are known to perform an effective dispersal since they are able to consume the pulp and excrete the seeds ready to germinate (legitimate fruit consumers, *sensu* Herrera, 2004), contrary to seed predating species like *Fringilla coelebs* L. and *Serinus serinus* L. (Fringillidae), also present although in lower density (2.70 and 2.35 ind/ha, respectively). Legitimate frugivores account for 42% of the density of the forest bird community and represent 62.2% of all frugivores.

Species	***Flowering***	***Fruit ripening***	**Presence of fruits**
P. lentiscus	March – May	September – January	August – March
R. lycioides	March – June	August – October	August – November
A. albus	August – October	October – January	October – February

Table 5.1: Phenology of the species studied according to Herrera, 1984.

5.2.2 Estimation of seed removal and seed rain

In total, 26 individual shrubs were selected (12 *R. lycioides,* 11 *A. albus* and 3 *P. lentiscus*) located throughout all the topographical range of the area (ranging between 105 and 200 m a.s.l.). The low number of *P. lentiscus* is in relation to its extremely low density in the area. The biovolume of each specimen was determined using the formula proposed by Blanco and Navarro (2003) and assuming a cylindrical morphotype.

Furthermore, 27 *P. halepensis* individuals were selected to study their role as perches. Pine trees were located at distances less than 20 m from the nearest shrub. Two collector trays (2 x 1 m) were placed on both sides of the main trunk of each tree covering an area of 4 m^2 under the tree canopy.

A branch of each selected shrub was chosen, under which a 0.5 x 0.5 m collector tray was placed. Tray dimensions were smaller than those located under trees due to the shrub's lower basal area and the consequent difficulties for their placement. The initial amount of fruits within the vertical projection of the tray was estimated visually by two different researchers. Due to the tendency of these observers to underestimate the number of fruits present in the branches, confirmed in precise test counts, the estimated number was corrected upward by adding a 20% (average underestimate of the observers).

In a similar manner to Sorensen (1981), we estimated the total fruit crop of the area extrapolating the initial average number of fruits per branch to the average biovolume of each species, and multiplying this value by the total number of individuals of each species estimated for the whole study area from their respective densities. Once the mean number of fruits per branch estimated to have been removed by birds was known, the same procedure was used to calculate a whole area bird removal figure and the percent of the total crop it represented.

Between May 2010 and February 2011, according to the species phenology (Table 5.1), 17 regular fortnightly visits were made in which the number of fallen fruits collected in trays were counted, and the number of fruits in the overhanging branches visually estimated.

In the same visits seed rain was recorded by collecting bird droppings and regurgitations found in the trays located below pine perches. Then, the collected material was examined with binocular lens, noting the presence of seeds and capsules of each species. In addition, we collected all fruits, seeds and seed capsules that were found loose in the trays. Due to the absence of another disperser who could leave loose seeds in trays, we assume that all seeds found in trays correspond to bird dispersers.

In both cases, the design of the tray was such that minimised the loss of samples from it (raised edges which retained the seeds and dark material to mimic the colour of fleshy fruits or drops). Even so, the days of heavy rains could cause the loss of part of the sample.

Dispersed (=consumed) fruits were estimated by subtracting from the number of fruits on the branch in the previous visit, the number remaining on the branch in the current visit plus those collected in the tray (Eq. 1).

$$(F_{(dispersed)} = F_{(branch)t-1} - F_{(branch)t} - F_{(tray)t}) \qquad (1)$$

At this stage we do not differentiate between legitimate and illegitimate consumers (*sensu* Herrera 2004), assuming that all fruits removed have a chance to be dispersed to a variable distance from the parent plant. Other potential fruit consumers are neglected.

The estimated mean crop of fruits was calculated for each individual shrub and averaged by species. With this value and the mean biovolume, production per m^3 was also calculated. All means are presented with their Standard Error (±SE). To estimate the disperser effectiveness (Schupp 1993; Schupp *et al*. 2010) we have considered the total fruit production of each species, the percentage of fruit consumption by birds (from Eq. 1), and the number of seeds deposited in relation to the total available in plants (initial crop) and to the total estimated to have been mobilized by birds.

5.2.3 Statistical analyses

Regression models were used to explore the relationship between dispersal variables and perch locations in order to detect potential spatial patterns in seed deposition within the patch. Response variables were the number of seeds deposited, and explanatory ones were altitude (topographic gradient) and distance to patch edge (considering urban and rural edges as the extremes of the main recognizable gradient of external influence). A Generalized Linear Model was implemented, with Poisson error and default (log) link, which is considered adequate for count data. We used the function 'glm()' contained in the *stats* package of the freely-distributed R statistical package (R Development Core Team, 2009), after checking graphically the basic assumptions of the model. Chi-square tests were used to compare the observed frequency of deposition of seeds under the tree stratum and that expected from the available perching area (made equal to the canopy cover within the study area). Canopy cover was available from previous surveys made in the study area and averages 13.8% (Ramos, 2011). We also compared the observed and expected frequency of shrubs of different age (seedlings, saplings and adults) growing under the tree stratum. All comparisons were made considering that there was no substantial export of fruits from the forest patch (i.e. all seeds not deposited under trees fell in open spaces outside the tree canopy within the patch), an assumption confirmed by the virtual absence of seeds in faeces and regurgitations of birds mist-netted in the patch edges (unpublished data). Frequency comparisons and associated Chi-square tests were made on the basis of 17 circular samples of 100 m^2 taken in November 2011 in three transects set along the study area (from the urban edge towards the topographical summit). These transects were located close to the previously investigated shrubs and perches, and ran along both tree-covered and open spaces. In each circular plot, the number of shrubs of different age (seedlings, saplings and adults) and their position in relation to the tree layer (under the canopy vs open spaces) was recorded. Saplings are classed as plants that have acquired vegetative adult characters, generally about 1 year after germination. This age class is considered to last until first reproduction, which in *R. lycioides* and *P. lentiscus* occurs at 3-6 years (Navarro *et al*., 2009). We compared: i) the observed frequency of deposition of seeds (percent recorded in trays of those removed from shrubs) versus the expected frequency of deposition if no selection or avoidance of tree perches occurred (made equal to the percent canopy cover within the study area); ii) the frequencies of germination (seedlings recorded) under and outside the canopy, to the frequencies of deposition used in the previous comparison (taken in this case as expected); iii) the observed and expected frequency of shrubs of different age (saplings and adults) growing under the tree stratum and outside it, in this case considering the proportion of individuals of the precedent age class in the same locations as expected frequencies.

The study area was cleared some years before the start of our work in the area (research was initiated in 2007), to eliminate dead or damaged pines, so some shrubs initially established under the tree canopy could be found in totally exposed sites at the time of sampling (after the trees had been cut). Consequently we considered adult shrubs (> 3 year old) growing close to tree stumps to have settled under a live tree canopy. Since only trunks and branches were removed during clearing operations, the initial density of pines at planting and the individuals cleared were easily recognizable by the presence of stumps.

5.3. Results

5.3.1. Production of fruits

The crop per unit volume (Table 5.2) was 133.30±35.91 fruits/m^3 for *R. lycioides*, 300.25±147.61 fruits/m^3 for *A. albus* and 439.51±41.88 fruits/m^3 for *P. lentiscus*. The total production of fruits of *R.*

lycioides estimated for the study area was 10.2±0.06 millions of fruits. For *A. albus* these value was five times higher (54.3±0.05 millions). *P. lentiscus*, production was 25,185.53 ± 6,091.91 fruits.

Variable	***R. lycioides***	***A. albus***	***P. lentiscus***
Shrub density (individuals/ha)	211.76 ± 65.23	1,876.47 ± 363.55	0.23 ± 0.06
Initial fruits per branch	162.00 ± 6.75	214.00 ± 9.74	496.00 ± 19.21
Dispersed fruits per branch (% of initial)	76.00 (47)	162.00 (76)	178.00 (36)
Mean fruits per individual	1,704.39 ± 716. 61	837.14 ± 331.11	4,076.78 ± 455.10
Fruits/m^3	133.30 ± 35.91	300.25 ± 147.61	439.51 ± 41.88
Fruits/ha	360,921.63 ± 46,744.47	1,570,868.10 ± 113,104.04	937.48 ± 27.31
All study area	9,694,354.98 ± 1,255,556.46	42,193,517.17 ± 3,037,974.51	25,180.71 ± 6,091.91
Consumed by birds	4,808,951.21	41,321,301.39	9,065.06
Deposited under perches (total seeds)	265.00	154.00	2.00
Number of perches	27	27	27
Deposited under perches (seeds/m^2)	2.45	1.43	0.02
Percentage tree cover (%)	13.81	13.81	13.81
Deposition surface (m^2)	37,093.66	37,093.66	37,093.66
Total seed rain under perches	90,879.47	53,043.93	741.86
% of consumed fruits	1.89	0.13	8.18
% of initial crop	0.88	0.09	2.94
Potential perches (236 live pines/has)	6,810.96	6,810.96	6,810.96
Seeds per perch	13.34	7.77	0.11

Table 5.2: Main results of fruit crops, estimated consumption of fruits by birds and seed rain under perches for each shrub species in the study area, and other variables used for their calculation. Dispersed (=consumed) fruits per branch are calculated through equation (1) of section 2.2. Mean fruits per individual are averages of fruit crops of the studied shrubs (calculated after counts of branches extrapolated to the shrub biovolume), and projected to the study area in terms of fruits per biovolume and per surface area. Extrapolations are also made as gross absolute values for the whole study area (total crop and removal) to give an indication of the magnitude of the dispersal process.

5.3.2 Removal of fruits from shrubs

Rhamnus lycioides

The mean number of fruits whose dispersal was attributed to birds is 76 (±0.43) per branch sampled, representing 47% of those initially present (162±6.75). Extrapolating to the whole *R. lycioides* population of the study area (Table 5.2), birds would start the dispersal of 4.8 millions of fruits. The graphical representations of values along the sampling period (Figure 5.2) shows how in 6 months (May to October) nearly all the fruits are removed (about 95% of the total crop), most of them during July and August (91%). After September dispersal activity is scarce.

Asparagus albus

The average dispersal of fruits was 162 (±1.97) per branch, representing 76% of the initial crop (214.00±9.74). Extrapolating to the total population of the study area, 41.2 of the 54.3 millions of fruits produced (Table 5.2) was estimated to be consumed by birds. In this case, fruits were removed from the branches in about four months (October to January) with two periods of greater dispersal (November and January), while the appearance of fruits in the tray was quite regular throughout all the sampling period.

Pistacia lentiscus

The average dispersal was 178 (±103.18) fruits per sampled branch, representing 36% of the initial crop (496.00±19.21). Extrapolating to the entire population, 9,066 fruits were dispersed by birds (Table 5.2). Between August and November, 75% of the fruits available in the branch were lost. The dispersal was concentrated at the beginning of the sampling (August to October).

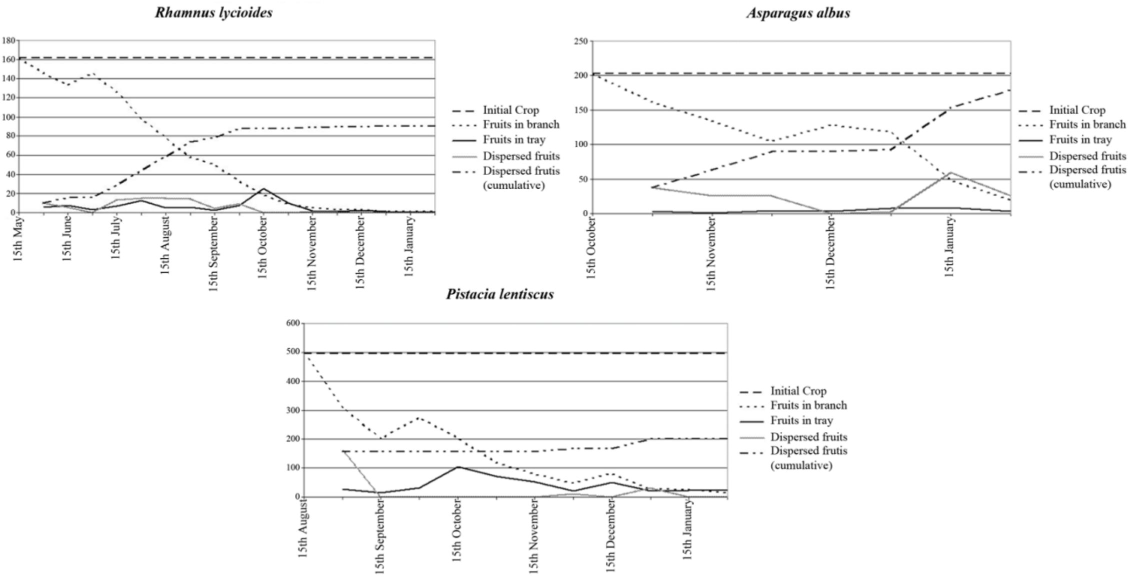


Figure 5.2. Temporal changes in the fate of the fruits of the three species studied (Rhamnus lycioides, Asparagus albus and Pistacia lentiscus). Curves represent the mean value of fruits in the branches sampled and in the trays placed below them, the difference representing the estimated dispersal.

5.3.3 Seed rain under tree perches.

Table 5.3 shows the total number of seeds collected for each species. Seeds of *R. lycioides* were the most abundant (63.9%), followed by *A. albus* (35.6%), and finally *P. lentiscus* (0.5 %). Total seeds collected were 421, the majority not included in droppings (85%). This could have been caused by rain events during the study period (more frequent than average, a characteristic feature of Mediterranean climate) which could wash droppings and separate the seeds from the faecal matter.

For *R. lycioides*, July, August and September were the months with the greatest contributions to the total seed rain. In the case of *A. albus*, seeds appeared in December and January.

Regression models showed that the number of seeds deposited was not significantly related to any of the variables of the perch location (altitude and distance from urban or rural edges; $p > 0.1$ in all cases).

Species	**Total seeds**	**Loose seeds**	**Seeds in droppings**
R. lycioides	265	206	59
A. albus	154	152	2
P. lentiscus	2	1	1
	421	**359**	**62**

Table 5.3: Total seeds collected in trays. Loose seeds were those not included in any dropping.

5.3.4 Temporal concordance between dispersal of fruits and seed rain.

Rhamnus lycioides

The significant volume of fruits consumed resulted in a relatively synchronous seed rain at the perches. However, a gap exists in the months of May and June in which dispersal exists but there are no seeds under the perches. The data from July 31 onwards display a better concordance and a significant relationship (p-value < 0.001; R^2: 0.7032) between the dispersal of fruits and seed rain.

Asparagus albus

For *A. albus* there is not a synchronization between the dispersal of fruits and the seed rain because there are two periods of maximum dispersal of which only one (January) concurs with the seed rain. Apparently, there is a significant amount of dispersed fruits during October and November that does not result in seed rain.

Pistacia lentiscus

In this case there is not a temporal agreement between the two processes studied, mainly because only two seeds of this species were found under perches. Furthermore, the appearance of these seeds does not coincide with the period of maximum dispersal (end of August and early September).

5.3.5 Tree-directed dispersal effectiveness

The estimated total seed rain of *R. lycioides* under *P. halepensis*, extrapolated to the total study area is 90,879.47 seeds. This value represents only 0.88% of the estimated total production of fruits, and 1.89% of the total consumption by birds. For *A. albus*, these values are 53,043.93 seeds, 0.09% and 0.13% respectively. In the case of *P. lentiscus*, the deposited seeds estimated are 741.86, representing 2.94% and 8.18% respectively (Table 5.2).

Of the three species studied, *P. lentiscus* had a higher initial density of fruits (more fruits per branch and per m^3) and hence a larger individual crop, although it was also the species from which a lower percent of fruits were dispersed (36%), and a lower number of seeds found under the perches. *A. albus* had intermediate values of seed density and percent fruits dispersed (76%), but a lower individual crop and the poorest representation under the perches. Finally, despite *R. lycioides* had lower scores on both fruit per branch and density per m^3, and an intermediate position in percentage of fruits dispersed, it was the species best represented in the seed samples collected under the pine trees. *R. lycioides* achieved the highest quantitative effectiveness in its dispersal towards perches, where almost 1% of the total crop and nearly 2% of the fruits consumed by birds were found (Table 5.2).

5.3.6 Seedling establishment and survival to sapling and adult stages

Vegetation sampling provided information on the density and distribution of age classes among cover types (tree canopy *vs* open) only for the two more abundant species, *A. albus* and *R. lycioides* (Table 5.4).

Seed rain		**Under *Pinus* canopy**	
Species:	Deposited (% of those dispersed from shrubs)	YES	NO
Pistacia lentiscus	N= 2 (8.18)	0.0818	0.9182
Rhamnus lycioides	N= 154 (0.13)	0.0189	0.9811
Asparagus albus	N= 265 (1.89)	0.0013	0.9987
	Probability (canopy cover, all cases)	0.1381	0.8619

Seedlings established vs seeds deposited		**Under *Pinus* canopy**	
Species:		YES	NO
Rhamnus lycioides	Seedlings (N= 2)	0.5000	0.5000
	Probability (seeds deposited)	0.0189	0.9811
Asparagus albus	Seedlings (N= 92)	0.8043	0.1957
	Probability (seeds deposited)	0.0013	0.9987

Saplings, adults vs seedlings		**Under *Pinus* canopy**	
Species:		YES	NO
Rhamnus lycioides	Saplings (N= 8)	1.0000	0.0000
	Adults (N= 27)[1]	0.4444	0.5556
	+ adults close to stumps as ‘under canopy’	0.8148	0.1852
	Probability (seedlings established)	0.5000	0.5000
Asparagus albus	Saplings (N= 118)	0.7034	0.2966
	Adults (N= 195)	0.5641	0.4359
	+ adults close to stumps as ‘under canopy’[1]	0.8513	0.1487
	Probability (seedlings established)	0.8043	0.1957

Table 5.4. Comparison of actual seed rain under and outside Pinus canopy (estimated from seeds collected in trays relative to those removed by birds) vs the expected probabilities of deposition among these cover types, assuming an uniform dispersal (expected probabilities derived in this case from the distribution of the precedent age class among cover types (seedlings vs seeds deposited, saplings and adults vs seedlings established). In the last comparison adults recorded close to Pinus stumps are added to those under tree canopy to represent all individuals whose settlement and growth started under the tree stratum. All the comparisons show significant differences (Chi-square tests, d.f.=1, p<0,005) except (1). Pistacia lentiscus was not represented in vegetation samples.

P. lentiscus was not recorded in the samples. Estimated densities were 1,876.47±363.55 ind/ha for *A. albus* and 211.76±65.23 ind/ha for *R. lycioides* (considering only saplings and adults). Since these values are quite close to the previous estimates of shrub density available for the forest patch (section 2.1), they were used to calculate fruit crops for the whole area (Table 5.2). A large proportion of the seedlings of *A. albus* (80.43; N=92) were found under the tree canopy, which contrasts sharply with the low number of seeds estimated to have been deposited there by birds (0.13% of those consumed). Only two seedlings of *R. lycioides* were found in samples (one of which under the tree canopy). Regarding saplings, 70.34% of those of *A. albus* (N=118) and all those of *R. lycioides* (N=8) were also under the tree canopy. The more balanced proportions of adults among cover types (56.41 *vs* 46.59% for *A. albus* and 44.44 *vs* 55.56 % for *R. lycioides*), contrast in the first case with the rate of seedling establishment (strongly biased towards the canopy cover), suggesting that recruitment is a rare event outside the tree canopy (where it needs a high seed input pressure to be successful) and more likely under it (even with a much lower seed input). This points out to a facilitative effect that enhances shrub survival close to a tree. In *Rhamnus*, the low number of seedlings and saplings found makes difficult to draw conclusions about its relationship with trees, although the fact that saplings were always found under trees points to a similar interpretation. Moreover, if all adults close to stumps (of dead or damaged trees having been cleared) are considered to have settled under the tree canopy, then the proportion of shrubs having grown to adult stage in such cover type increases (over 80% in both species).

5.4. Discussion and conclusions

5.4.1 First step: fruit production and removal

As occurs with its individual fruit production compared with other Mediterranean species (Jordano, 1989; Bas *et al*., 2005), the percentage of fruits dispersed in *R. lycioides* (Table 5.2) is relatively low compared with other areas studied in the Iberian Peninsula where birds commonly remove over 90% of the harvest of ripe fruits of this species (Herrera, 2004). Although *a priori* we discarded this possibility, some of the fruits whose consumption is attributed to birds could have been taken by climbing species of small mammals or lizards (Herrera, 1995;Traveset, 1995; Godínez-Álvarez, 2004; Rodríguez-Pérez *et al.,* 2005). Our method to estimate bird consumption could also ignore the potential role of ground-dwelling vertebrate or invertebrate dispersers (Aronne and Wilcock, 1994), although the design of collecting trays aimed at minimising the removal of fallen fruits. Moreover, the impact of harvester ants on shrub species like *P. lentiscus* seems not to be substantial (Barroso *et al.,* 2013).

The dispersal percentage in *A.* albus is closer to the value recorded by Herrera (2004), and suggests that birds concentrate their foraging activity in the most abundant and productive fruiting species. At least for these lower-height shrubs, dispersal by ground-dwelling species (e.g. *Alectoris rufa*, typical game species of these open forest landscapes) can also be significant, as observations of localized dropping accumulations suggests.

The low percentage of fruits removed from *P. lentiscus* compared to other areas of the Iberian Peninsula, where the birds consumed between 80 and 100% of the ripe fruits of the same species (Herrera, 2004), can respond to its extremely low local density making less advantageous for birds to search for its fruits.

Given that the fruit ripening period for *R. lycioides* extends between August and October, if the fruits are consumed before they mature -which raises the question if the birds are able to tolerate the toxic compounds found in this type of fruit (Herrera, 1982; 2002)-, the seeds are not fully formed and could be destroyed when the bird handles the fruit or when passing it through the digestive tract, leading to an ineffective dispersal. During the period of higher consumption (July-August), the birds performing most dispersal correspond to the late breeding assemblage (adults and juveniles of the year), whose feeding requirements are expected to be higher. Moreover, during that period, a higher proportion of the frugivore assemblage is composed by illegitimate species (family *Fringillidae*, personal observations).

5.4.2 Second step: seed rain

The lack of significant relationships between seed deposition and the variables of perch location (section 3.3) suggests that the spatial pattern of dispersal is probably a short-distance local movement of seeds, not influenced by surrounding land uses. This fact could affect more the dispersal of the shrubs towards areas outside the patch than the internal regeneration of the forest.

Interpreting the relationship between the dispersal of fruits and the seed rain under perching trees illustrates the purely structural and rather passive role of the latter, as receptors of part of the seed production of shrubs transported by birds. According to García *et al.,* (2005), spatial patterns of recruitment in bird-dispersed species largely depend on the initial template established by seed dispersal, but are subsequently influenced by successive post-dispersal factors (predation, germination and seedling mortality), leading to variable spatial relationships between seed deposition and plant recruitment.

The shaping of such initial template is influenced by the quantitative and qualitative performance of birds as seed dispersers. For instance, the reduced seed rain of *R. lycioides* in May and June can be related to the composition and activity of the frugivore assemblage during the nesting period. Also, remains of unripe seeds may be undetectable in the material collected under the perches in that period. For *A. albus*, the lack of synchronization between dispersal of fruits and seed rain could not be attributed to the consumption of unripe fruits by the birds as the period of fruit ripening extends in this species from October to January. The absence of seeds under the perches between October and December could be attributed to the fruits being consumed *in situ* by fruit predating birds (*sensu* Herrera 2004) or other vertebrate fauna (favoured by the smaller size of *A. albus* plants) or to exportation outside the patch.

5.4.3 Third step: effective recruitment

Comparing the low number of seeds of *P. lentiscus* found under perches with the high relative fruit crop indicates a general inability to effectively disperse in the study area. This contrasts with the usual appetite of birds for its fruits (Herrera, 2004), and could respond to its extremely low local density, interpreted as the result of overexploitation for firewood and other traditional uses (Diaz-Barradas and Correia, 1999).

A. albus is the species whose dispersal to the perches is less effective in quantitative terms (around 1%; Table 5.2), suggesting a dominance of other dispersal directions different than the perches (open spaces).

For *R. lycioides* the dispersal was intermediate but reasonably effective (22% of seeds found in droppings), which could reflect a preference of legitimate bird disperser for its fruits, in accordance with the temporal adjustment between production, seed dispersal and rain. Fruits of *R. lycioides* are directly consumed by the birds in the perch, and sometimes on the shrub itself without being pre-processed, thus explaining the appearance of a significant percentage of seeds in droppings. When the seeds pass through the digestive tract of birds, the germination cycle is activated by the acid attack of their digestive system (Bas *et al*., 2005). However, it also suggests a short-distance, preferently local dispersal with implications for the internal distribution (within the forest patch) and for the dispersal outside it, that is confirmed by studies on seed transport by birds (Jordano *et al*., 2008; Nathan *et al.,* 2008) and by the authors' unpublished observations in the study area. All these features would be in agreement with a preferent dispersal towards the tree layer, in which legitimate frugivores consuming *R. lycioides* would remain within the forest patch and roost in the trees. Conversely, the superabundant seeds of *Asparagus* would be consumed by all the frugivore assemblage, including seed predators and other generalist species, resulting in a more uniform dispersal pattern within the forest patch.

Of the two species from which we have sufficient data (*R. lycioides* and *A. albus*) none seems to fulfil the two first conditions necessary for directed dispersal to occur (Figure 5.3): disproportionate transport to, and enhanced survival in specific locations (Wenny, 2001). Both species, however, are clearly favoured in their regeneration when growing under the tree canopy. Despite the low proportion of seeds reaching such microhabitat (< 2% in the best case), seedling emergence occurred mainly there, as did sapling growth. Nearly half of the adults sampled of both species were found under the tree canopy, and when those growing by stumps (previously also under the tree canopy) were added, the proportion rose to > 80%. Considering that more than 98% of seeds fall outside the tree layer, direct establishment seems a very rare event far from its protective cover, facilitative effects seemingly outweighting the costs of competition (Maestre *et al*., 2004; Pugnaire *et al.,* 2011).

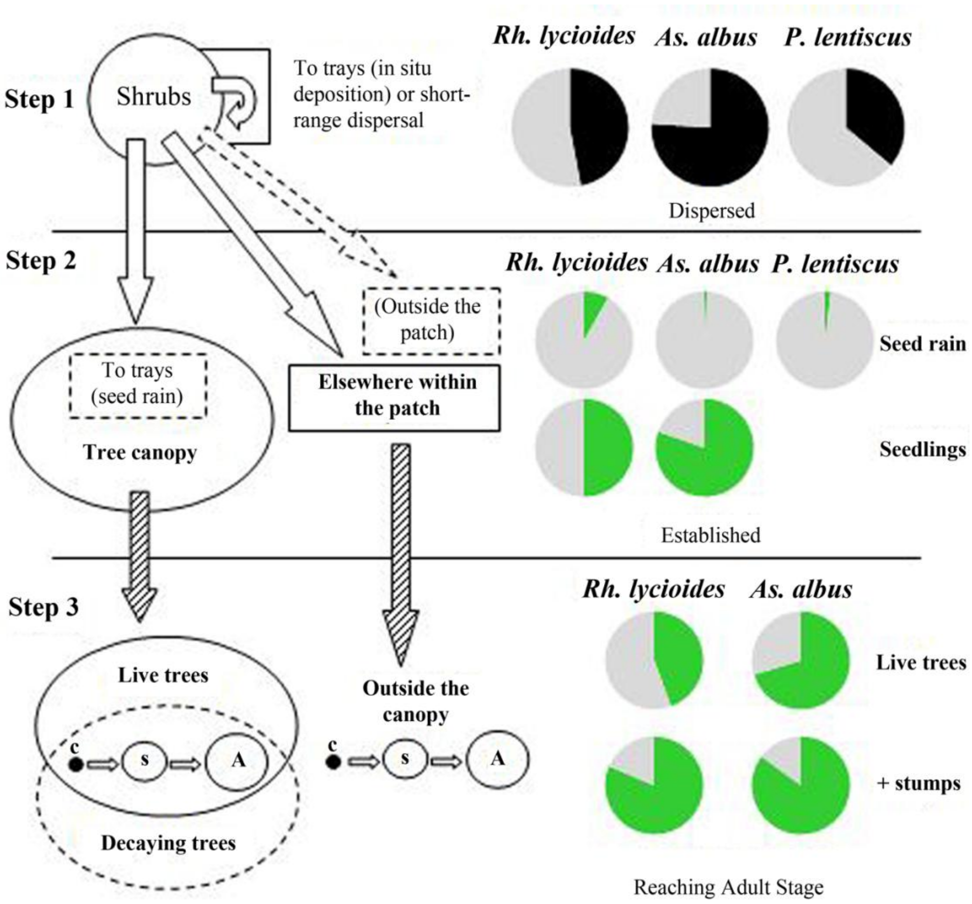


Figure 5.3: Main phases in the dispersal of seeds within the studied forest patch. In step 1, graphs show the proportion of the seed crop dispersed from shrubs (in black). In step 2, the upper graphs represent the seed rain under tree canopy (in % of those dispersed, in green) vs other destinations (in grey), and the lower ones the proportion of seedlings recorded in each of these locations. In step 3, the upper graphs show the occurrence of adult shrubs under the tree canopy (green) vs open spaces, and the lower ones the same distribution including adults growing by tree stumps in the first category.

5.4.4 Future research needs and implications for management

When assessing the role of tree perches, our approach (Figure 5.3) has several shortcomings. First, seed destinations or fates other than deposition, germination and growth (i.e. predation, deposition outside the canopy, death) are subsumed within an indirect estimation in each step. Furthermore, fieldwork had to be restricted to one location (the largest forest unit in the rural landscape studied, and the only one with all the focal species adequately represented) and to a rather narrow recruitment window (a year cycle along which these species exhibited a good performance in all phenological phases). The authors are aware of the difficulties posed by these limitations for the generalization of results, but feel that the understanding of such scenario is a necessary step before interpreting other local or external replicates.

Considering that the reproductive success of plants depends on the dispersal of propagules to germinate in right places (Herrera, 2002), isolated trees and shrubs can play an important role in the colonization of Mediterranean environments because they can change the microclimate and soil conditions, as well as attract bird seed dispersers (Herrera, 2004, Maestre and Cortina, 2005, Pausas *et al.,* 2006). The activity of dispersal agents (birds) interacts with plant reproduction to shape vegetation distribution patterns (Howe and Smallwood, 1982; Nathan and Muller-Landau, 2000; Bas *et al.,* 2006).

Therefore, although the dispersal towards trees is effective, it is limited and there must be a significant proportion of fruit that is not directed towards these microhabitats. However, when considering the survival and development of established shrubs to adult stage, the tree canopy seems to play a critical role, not only as a static component but under a dynamic management approach. Our observations confirm how often the deaths of trees or their removal result in an improved development of shrubs having germinated and grown under the tree canopy, highlighting the reduction of tree density as a desirable measure for biodiversity (Zapata and Robledano, 2014). In the future, research in the area should address in more detail the outcome of the interaction between *P. halepensis* and shrubs (facilitation *vs* competition), and its effect on recruitment (Reid and Holl, 2012). This will improve the knowledge needed to perform a management in favour of the regeneration of the shrub species.

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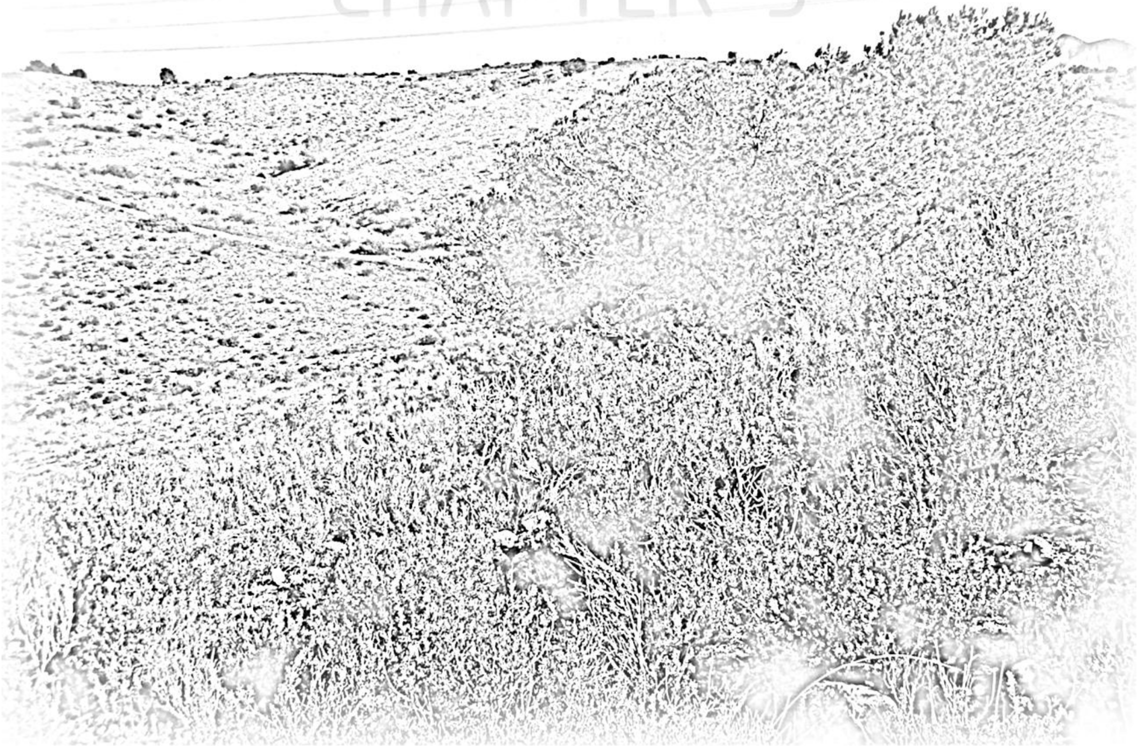
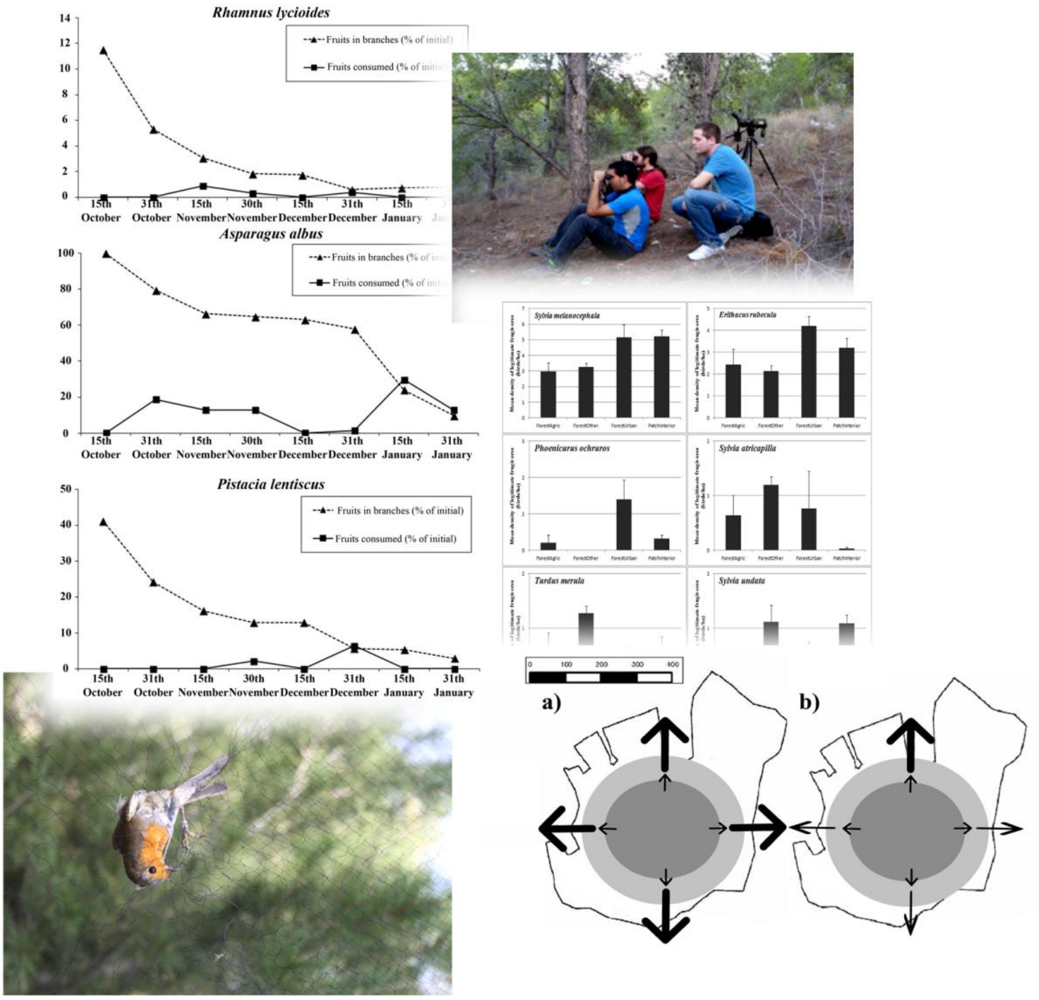
CHAPTER 5



Bird frugivory under limiting environmental conditions. Relating foraging distribution and activity with seed dispersal in semiarid Mediterranean forest patches

Graphical Abstract



Abstract

Assuming that birds are an important vector in the dispersal of Semiarid Mediterranean shrubs, but this critical process can be conditioned by specific limitations imposed by their restrictive environment, this study aims are: (i) to characterize forest fragments bird communities; (ii) to assess the quantitative importance of frugivore-shrub interactions; (iii) to describe and analyze the distribution and activity of frugivorous birds feeding during such periods; and (iv) to propose bird-mediated dispersal models and related guidelines applicable to restoration.

The study was conducted during two years in an isolated *Pinus halepensis* fragment of 26.86 ha, focusing in the period of expectedly higher resource limitation (October-November). Census of forest birds and observations of feeding activity of frugivore species on selected shrubs (*Rhamnus lycioides* and *Pistacia lentiscus*) were done.

Our main findings are that fruit resources seem enough for the birds overwintering in the study area. Although the distribution of fruit production and consumption within the patch favours interior habitats, seed deposition is more important in peripheral ones, suggesting a trend for birds to feed in the former but rest preferently in the latter. Two alternative spatial models are proposed to explain the potential consequences for shrub dispersal and to promote forest regeneration.

6.1. Introduction

The Mediterranean Basin, although far from a pristine natural ecoregion (Garrido *et al.*, 2002; Aparicio, 2005; Aparicio *et al.*, 2008), still represents a hotspot for biological diversity (Médail and Quézel, 1997). This is attributable both to topographical and climatic variability as well as to human influence, which commonly overlaps with, replaces or mimics natural disturbances such as fire or herbivory (Valladares, 2007). Mediterranean biodiversity critically depends on habitat and ecosystem heterogeneity, often expressed as a mosaic landscape (Atauri and DeLucio, 2001; Blondel *et al.*, 2010). Fine-grained landscape mosaics are the result of human activities whose nature and intensity determine the biodiversity value of the remaining natural and low-intensity farmed or grazed habitats (Blondel *et al.*, 2010).

The Mediterranean climate imposes a double limitation to vegetation, in what regards water availability: scarce and irregular supply (Mazzoleni *et al.*, 1992; Gulías and Traveset, 2012). Since these limitations are expected to be intensified by climate change (Peñuelas *et al.*, 2004), in large areas of the biome they can cause the replacement of tree formations by scrublands or cleared forests, with a predominant sub-arboreal component (Mangas *et al.*, 2008). Scrubland expansion is also a consequence of human management of vegetation through fire, cutting and grazing (to provide fuelwood or pasture), at the expense of forests. Similarly, extensive reforestations have taken place in areas with scarce natural potential for tree growth (i.e. average rainfall well below 350 mm; Maestre and Cortina, 2002), delivering unsustainable forest formations (Zapata *et al.*, 2014).

In Mediterranean shrublands, the fleshy fruits of some woody plants represent food patches or 'resource islands' for frugivorous fauna, particularly for birds (Herrera, 1984; Maestre and Cortina, 2005), but also for other vertebrates and invertebrates (Herrera, 1995; Clark *et al.*, 2005; Rodríguez-Pérez *et al.*, 2005). They represent an ideal resource for birds to secure body reserves, and therefore survival, during winter, which makes frugivores the most diverse and abundant components of their wintering bird communities (Tellería *et al.*, 2005). Birds that feed on fruits or seeds drop them at varying distances from the source plants (Jordano, 2000) and contribute to landscape regeneration through secondary succession in reclaimed areas.

The interaction between bird frugivores and fruiting plants has occurred and co-evolved along at least 90 million years (Fleming and Kress, 2011), and plays an essential role in the maintenance of biodiversity (Blendinger *et al.*, 2012).

Seed dispersal by birds is considered a crucial ecosystem function that controls the dynamics of plant communities in natural habitats and vegetation regeneration in landscapes altered by man. The ability of many plant species to respond to fragmentation, for example, is related to the ability of their seeds to be dispersed mostly at long distance (Aparicio *et al.*, 2008). But there are still knowledge gaps for developing conservation and management activities helped by this service (Garcia *et al.*, 2010). In semiarid Mediterranean areas, research about plant-frugivorous interactions and how they affect the regeneration and dispersal of forest plants species is very limited. Most research, like e.g. the extensive work of Herrera (1984), Jordano (1989a), or Traveset (1994), has been conducted in areas that can be considered climatically "typical" of the biome. Besides, the focus is mostly on vegetation and infers the avian service without specific bird studies or through indirect approaches (but see e.g. Bas *et al*., 2005). In natural or reforested woodlands characteristic of the semiarid Iberian Southeast, like those dominated by *Pinus halepensis*, biodiversity assessments are uncommon (López and Moro, 1997; Zapata *et al*., 2012) and the knowledge about bird-mediated dispersal processes is even scarcer. A better understanding of such processes, when operating under limiting semiarid conditions can help to design new strategies to restore degraded ecosystems taking advantage of the frugivorous bird role.

Our focus is on fragmented forest systems of semiarid areas, where previous studies have addressed the biodiversity value and its ecological correlates in forest islets which are the result of the typical fragmentation history of lowland areas (Zapata and Robledano, 2014). In this context, bird-mediated dispersal may be essential for preserving plant populations and for the recolonization and/or restoration of degraded land (Méndez *et al*., 2008). Dispersal can be directed to natural perches like trees or shrubs. The perch canopy can provide transient or permanent microsites that enhance germination and decrease mortality during the first stages of seedling development (APAP, 2003) playing an important role in arid and semiarid climate (Barberá *et al.,* 2006). Ongoing studies (Zapata *et al*., 2014) have quantified the magnitude and directionality of the dispersal process, as well as the patterns of seed deposition and survival. However, there are still a number of questions to respond in relation to the interaction of fruiting plants with frugivorous birds:

a) Are fleshy fruit resources provided by semiarid forest patches enough for bird populations inhabiting them?

b) Is the spatial distribution of frugivory and seed flow related to fruit availability?

c) Can the fruit-feeding and seed transport activity of birds allow an efficient dispersal within and around forest patches?

With the assumption that birds are an important vector in the dispersal of seeds of Mediterranean shrubs in semiarid areas, but that this process can be conditioned by the specific limitations imposed by their restrictive environment, the following objectives have been set out: (i) to characterize the forest bird community, focusing on the frugivore assemblage; (ii) to assess the quantitative importance of shrub-frugivore interactions, in terms of food resources for the birds and seed dispersal, and to identify potentially limiting periods for birds depending on such resources; (iii) to describe and analyze the distribution and behaviour of frugivorous birds in the search and consumption of fleshy fruits of selected Mediterranean shrubs during such periods; and (iv) to propose bird-mediated dispersal models and related guidelines applicable to management and restoration of degraded forest landscapes.

As a general hypothesis, if fruit resources become limiting (i.e. the fruit crop is insufficient to satisfy the energetic requirements of the frugivore assemblage), we expect birds to search for food in nearby sources, which could enhance shrub dispersal (through seed transport) towards other habitats. Since forest patches are often close or surrounded by urbanizations, there is also the risk of losing propagules into unsuitable habitats or that exotic plant species are imported through seed transport. On the opposite, if birds tend to stay inside a forest patch (avoiding surrounding habitats), the opportunities of external dispersal can be reduced.

6.2. Material and methods

6.2.1 Study area

The research has been carried out in the Local Forest Park of Montepinar (38° 2´ 11´´ N, 1° 5´40´´ O), located in the southeast of the Iberian Peninsula (Figure 6.1), with a total area of 26.86 ha. The park is a 50-60 year old *P. halepensis* Miller forest patch planted on a rural area following an intensive degradation through logging and overgrazing. The climatic conditions (290.5 mm of annual rainfall) are limiting for the establishment and growth of tree formations, even of the highly resistant *P. halepensis*. During the main period of study (March to November of 2011), rain concentrated in March-April and November with an intermediate drought period coinciding with the end of spring and the entire summer. Thus, the undisturbed original landscape was most probably a scrubland with sparse trees or small woodlots (growing only in localized favourable places), and the planting scenario a degraded mosaic of dwarf shrubland, dry grassland and bare rock with even lower forest potential.

The initial reforestation density in the park (624 trees/ha) has proven to be too high for the restrictive environmental conditions. In less than half a century, this has obliged managers to remove dead or severely damaged trees accounting for 50% of those initially planted. The understorey of the patch is formed by typical Mediterranean fleshy-fruited shrubs, with three dominant species of which two have been chosen for the study (*Rhamnus lycioides* spp. *lycioides* Brot. and *Pistacia lentiscus* L.). Their density in the study area has been established by Zapata *et al*., (2014), being intermediate for *R. lycioides* (211.76 ± 65.23 individuals/ha) and low for *P. lentiscus* (0.23±0.06 individuals/ha). The third species (*Asparagus albus*) was not included in the study despite its high density (1,876.47 ± 363.55 ind/ha), since it seems much less conditioned by the tree canopy as regards intra-patch dispersal and regeneration. Nine individual shrubs were selected (8 *R. lycioides* and 1 *P. lentiscus*), the number being related with their density in the study area. They were distributed in two transects in different catchments. The biovolume of each shrub was calculated using Blanco and Navarro (2003) equation and considering a cylindrical morphotype.

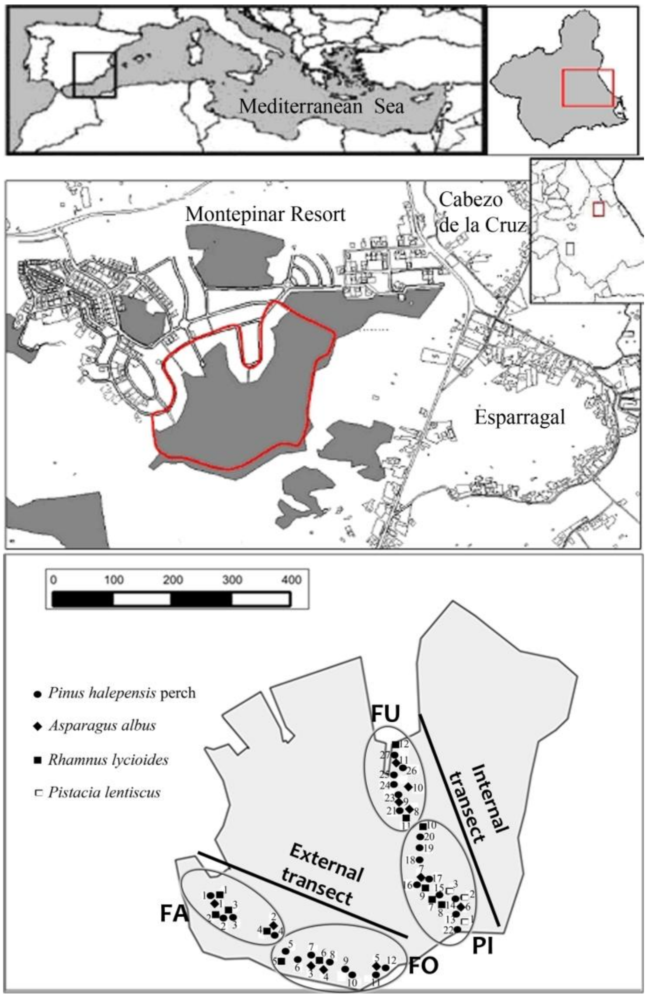


Figure 6.1: Location and limits of the study area (Montepinar Forest Park). The grey shaded polygons are forest patches and the red line the area actually studied, including some plots planned for building (in white). The lower map shows the position of sampled **shrubs and perches along the two main transects, and the four main sections used in the spatial analyses.**

6.2.2 Fruit availability and consumption along the bird wintering period.

In 2010, during a study about seed removal and dispersal (Zapata *et al.*, 2014), we quantified the fruits consumed by birds in a sample of marked individuals of the three dominant shrubs (the two species studied, *R. lycioides* and *P. lentiscus*, plus *A. albus*) distributed in the forest fragment along the two aforementioned transects (Figure 6.1). Fruit dispersal by frugivorous birds was estimated through the examination of faecal and pellet samples collected in trays under tree perches (*P. halepensis*).

On the basis of these results and further sampling, in 2010 and 2011 a spatial and temporal analysis of fruit production, consumption by birds, and seed rain was carried out. To assess spatial differences, the target shrubs and perches were classified as "patch interior" (PI), or among three types of "patch edges": forest-urban (FU), forest-agricultural (FA), forest other (FO). The last class represents forest edges adjacent to natural areas without tree cover (dry grassland and chamaephytes). The inner transect (FU-PI) opposed the shrubs located close to the urbanization, to those independent from its influence, while the external transect (FA-FO) differentiated two groups of shrubs similarly close to the patch edge, but adjacent to different rural landscape types (agricultural *vs* natural). The study area is suited to that differentiation since the four clusters of shrubs falling into the respective spatial classes were separated into relatively independent sections (small sub-catchments or slopes). The temporal analysis explored the occurrence of interannual differences in fruit production and consumption (between 2010 and 2011). To search for differences in seed movement within the forest patch, we also compared seed rain under perches (pine trees and shrubs) in relation to the same spatial divisions used to analyze fruit production and consumption.

6.2.3 Frugivorous bird community characterization

In parallel to the study of fruit availability and consumption, which did not differentiate the contribution of each bird species to crop removal and seed transport, we characterized the ornithological assemblage of the forest patch. First, we made a general characterization of the forest bird community on the basis of unpublished records and surveys made in the area in 2007-2008. Breeding bird censuses were made in may and june 2007, following the scheme of the SACRE monitoring project (Del Moral *et al*., 2008).

In 2011, we characterized in detail the wintering bird community of the forest patch. Bird sampling was carried out by point counts considering a detection radius of 50 m, a method commonly used to characterize bird communities elsewhere (Coreau and Martin, 2007) and in the studied landscape (Robledano and Zapata, 2011; Zapata *et al*., 2012). In our case point count samples were located around selected shrubs, choosing *R. lycioides* as focal species due to its value as feeding resource for birds (Zapata *et al*., 2014), and *P. lentiscus* due to the known energy content of its fruits (Jordano, 2007). We adjusted the time of this intensive sampling to a period *a priori* considered critical for birds, when they depend more on the interaction with these two species as a food resource before they can concentrate on other food, i.e. the later fruiting and superabundant *A. albus*. In fact, whose ripening occurs in full winter. Six sampling visits were made, two of them during the second half of October –one in the afternoon and one in the morning of consecutive days-, and four in November, two in each fortnight, each pair also in consecutive days.

Bird surveys focused on forest avifauna *sensu lato* (i.e. both specialist and generalist forest bird species were included; Gil-Tena *et al*., 2009) with the exception of large birds of prey. Bird species were classified in two categories, regarding the type of frugivory: legitimate and illegitimate frugivores (or seed predators). The first ones consume the whole fruit, digest the pulp and then regurgitate or defecate the seeds. The second ones eat the seed and leave the pulp, or eat the pulp and drop the seed (Herrera, 2004). Among illegitimate frugivores, there are species belonging to the families *Fringillidae* (*Fringilla, Carduelis,…*) and *Paridae* (*Parus*). The legitimate frugivores include families like *Sylviidae* or *Turdidae*.

6.2.4 Observation and assessment of feeding behaviour

In addition to bird sampling, the selected shrubs were watched to record birds perching on the surface or inside the plant canopy, and to determine their feeding attempts and success (fruit consumption). Data were recorded during four fortnightly periods (October and November). Two morning and two afternoon observation periods were made in each fortnight, all with a length of 10 minutes separated by 1 hour (i.e., in total, 160' were dedicated to each shrub). Observations were made with spotting scopes from vantage points allowing full coverage of the shrubs. In case of fruit removal or consumption, the observer recorded the number of fruits taken or ingested.

From the observation we derived two direct estimates of consumption:

- Realized consumption (conservative estimate): Number of fruits that were observed to be effectively ingested by the birds during the period under observation.
- Maximum possible consumption ('optimistic' estimate): Number of fruits effectively consumed plus the number of presumable consumption events (birds seen on the shrub with feeding attitude, but undetermined success, i.e, ingestion not confirmed). Each of these events was made equal to the consumption of a fruit (minimum possible value).

Furthermore, for the period covered by bird sampling and focal observations (October-November), we made an indirect estimation of the consumption of *Rhamnus* fruits based on the trophic requirements of the main consuming species (*Sylvia melanocephala* and *Erithacus rubecula*). The amount of energy (1.03 Kj) contained in a fruit of *R. lycioides* was obtained from 'FRUBASE' (Jordano 2007). The daily amount of energy required by *S. melanocephala* (34.69 Kj/bird) and *E. rubecula* (43.69 Kj/bird) was extrapolated from *Sylvia borin* data (Bairlein, 1987).The number of fruits required per day and bird was calculated by dividing its daily energy requirements by the energy provided by a fruit. Both species' local populations were calculated from their density (derived from the number of individuals sampled within point count radius), and multiplied by the individual fruit requirements to get an estimate of the total number of fruits that each species would consume during these two months. This consumption was estimated for different degrees of frugivory, 100% and the species' specific percent (Izhaki and Safriel, 1989; Bairlein, 1999), because although Iberian frugivorous birds rely largely on fruits, there is variation in their dependence on this resource (Herrera, 2004). Since not all food eaten by birds is metabolized, a quantity of energy is not used. It depends on the efficiency of uptake, which in the case of *Passeriformes* is 73,0 ± 0,6% (Bairlein, 1999). The resulting range of potential consumption of fruits by *S. melanocephala* and *E. rubecula* was compared with the estimation derived from direct feeding observations (also extrapolated to the whole study period).

6.2.5 Collection of faecal samples

Two days of bird trapping were carried out to obtain faecal samples and to determine the direction in which the seeds contained in them were being transported. Two mist-net networks (totalling 120 m length) were placed in transitions from the forest patch to agricultural or urban habitats, in openings coincident with the transects selected for the study. The birds were introduced in cloth bags with a paper cone to facilitate sample collection. After excretion, they were released and the sample deposited in an Eppendorf vial. A total of 20 samples were collected from frugivores, out of a sample of 63 mist-netted birds.

The side of the net in which the birds were caught was recorded in order to determine the directionality of bird movement (and seed flow) between areas with different land uses.

Two more days were dedicated to the search and collection of droppings under perches (shrubs) located in the interior of the forest patch. Perches were selected when there was a significant accumulation of bird droppings. This search yielded 17 faecal samples.

To increase the dataset of fruit deposition and transport, we also included in the analyses data from seed rain studies made the precedent year (2010) in the study area (Zapata *et al*., 2014), considering also the droppings found in collecting trays placed under *P. halepensis* perches. We finally gathered 707 samples, of which 20 correspond to the patch edges (birds caught in mist-nets), 675 to bird faeces/regurgitations collected under perches (trees) of the patch interior, and 17 to faeces collected under interior shrubs (only the 20 first ones could be differentiated by bird species).We used this dataset to determine the relative proportion of seeds of different species dispersed by birds towards the interior and to the edges of the forest patch.

The collected samples were dried and examined for the presence of fruits with a magnifying glass (Optika WF 10x/20), spending 10 minutes per sample (Holt and Warrington, 1996).

6.2.6 Statistical analyses

First, we analyzed the relationship of the variables describing fruit production and consumption (as response variables), with the position of the sampled shrubs with respect to different types of forest edges (urban-agricultural-other) as explanatory ones. The distances to patch edges were measured from satellite ortophotographs, both for shrubs sampled in 2010, and for shrubs on which point counts and observations of feeding behaviour were conducted in 2011. We interpreted these distances as gradients of external disturbance exerted by unfavourable habitats, due to human-related impacts (i.e. deforestation, induced predation, noise). Linear regressions were used to fit models to the relationships between the variables of interest (fruits consumed and number of visits) and the distances to such edges.

We also sought to determine if there was an internal pattern of distribution of *R. lycioides* fruit availability and if it was reflected in the distribution of bird frugivory within the patch (in 2010 and 2011). Generalized Linear Mixed Models (GLMMs), controlling for temporal autocorrelation effects among subsequent sampling dates, tested the effect of shrub location within the patch (using the zonation described in section 2.2) on its fruit production and on the consumption by birds, as separate response variables. We did the same for seed rain (seeds deposited under tree or shrub perches) to search for a pattern of seed movement within the patch (to be complemented with the records of seed transport obtained from mist-netting, see section 2.5).

For 2010 we used estimates of the available fruit crop (still standing in shrubs, i.e. not consumed) in the period of interest. In 2011, we made estimates of the number of fruits in each shrub under observation (to the nearest 50 fruits) at the time of bird sampling. Both estimates were compared in order to detect inter-annual changes in fruit production, characteristic of Mediterranean forest systems (Herrera, 2004). As regards bird frugivory, for 2010 we used data on estimated fruit removal obtained by Zapata *et al.*, (2014), and for 2011 the direct estimates of consumption derived from field observations.

In the analysis of seed rain, we considered the number of seeds of all species deposited (*Rhamnus* plus *Asparagus*) and that of *Rhamnus* alone as response variables. All analyses were performed with the freely-distributed statistical software "R" (R Development Core Team, 2006, packages *lme4* and *nlme*).

We also tried to relate bird frugivory (rates of visit and consumption) with features of the shrubs like fruit density and biovolume. Kruskal-Wallis tests were used to assess the differences in the number of bird feeding visits to shrubs of different fruit density and biovolume. Five types of density were considered (High = >500 fruits; High-Medium = >250 fruits; Medium = >100 fruits; Low-Medium = >50 fruits; Low = <50 fruits). Linear regressions were used to fit models to the relationships between the variables of interest (fruits consumed and number of visits) and the shrub biovolume.

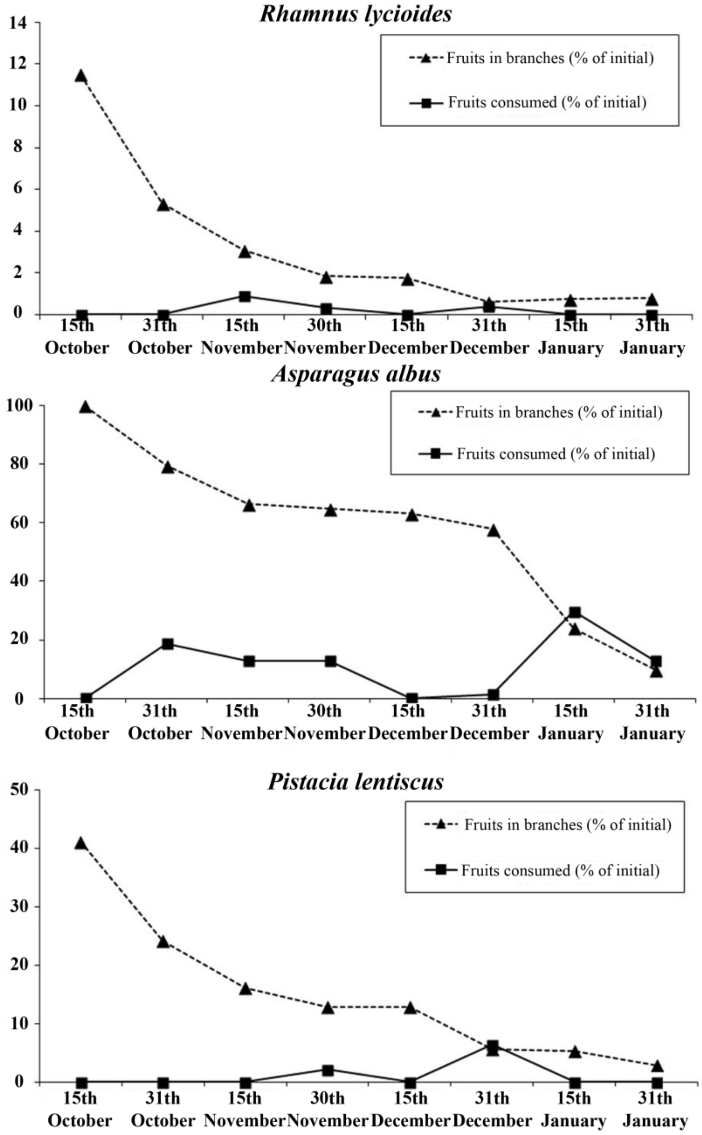
Finally, a principal components analysis (PCA) was performed to display gradients of bird movement between forest and adjacent systems, and to relate it with the patterns of seed flow detected. Ordination of mist-netted bird data, characterized by their direction of capture, was carried out through CANOCO 4.5 (ter Braak and Šmilauer, 2002).

6.3. Results

6.3.1 Fruit availability and consumption along the bird wintering period.

The mean availability of fruits of *R. lycioides* at the start of the fruiting season in 2010 was estimated at 1,704.39 ± 716.6 fruits per individual shrub. By October 15th only 11.5% of those fruits remained in the shrubs (196.00 ± 82.40). On the same date, the shrubs on which we observed bird feeding activity in 2011 had an average of 200 fruits per individual (visual estimates with an accuracy of ±50 fruits). The estimates of fruits consumed during the period under consideration, represent very low percentages of the initial standing crop. For *R. lycioides* (Figure 6.2) less than 12% of the initial fruit crop was still available for birds by the start of October, and consumption was less than 2% (around

15% of the remaining crop) during the following two months. For *A. albus*, at least in 2010, seed removal from shrubs was much more important, although in 2011 consumption events involving this species were rarely recorded, at least in the shrubs surrounding the focal *Rhamnus* individuals. For *P. lentiscus*, more than half of the crop is readily consumed by birds in late summer, and although 40% of the fruits remain in the shrubs in autumn, only some 2% of the initial crop is exploited in mid-winter.


Rhamnus lycioides

Asparagus albus

Pistacia lentiscus

Figure 6.2: Standing fruit crop of the three shrub species studied along the study period in 2010, and fruits estimated to have been consumed by birds on the same dates (fruits counted on the previous visit minus those collected in trays placed under selected branches), both expressed as percent of the initial fruit crop estimated at the start of the fruiting season of each species.

6.3.2. Characterization of the frugivorous bird community

In previous winter surveys (Table 6.1), 5 legitimate frugivores and up to 9 illegitimate ones were recorded. In spring, these numbers were 5 and 6, respectively. The relative abundance of both legitimate and illegitimate frugivores seemed to decrease from the early winter period onwards. The total number of bird species recorded in the winter of 2011 was 36. Of these, 14 were frugivores *sensu stricto* (Table 6.2).

(a)

Illegitimate frugivores	**May density** (individuals/ha)	**June density** (individuals/ha)	**Mean spring density** (individuals/ha)
Aegithalos caudatus	0,75	1,75	1,25
Carduelis carduelis	0,00	0,50	0,25
Carduelis chloris	1,25	1,75	1,50
Parus major	1,25	0,75	1,00
Passer domesticus	1,50	2,00	1,75
Serinus serinus	3,50	2,75	3,13
TOTAL:	**8,25**	**9,50**	**8,88**
Legitimate frugivores	**May density** (individuals/ha)	**June density** (individuals/ha)	**Mean spring density** (individuals/ha)
Sturnus unicolor	0,50	0,00	0,25
Sylvia hortensis	0,50	0,50	0,50
Sylvia melanocephala	2,00	3,50	2,75
Sylvia undata	1,00	0,00	0,50
Turdus merula	1,50	1,00	1,25
TOTAL:	**5,50**	**5,00**	**5,25**

(b)

Ilegitimate frugivores (seed predators)	**Frequency (early winter)** N=2	**Frequency (mid-winter)** N=4	**Frequency (late winter)** N=3
Aegithalos caudatus	1,00	0,50	0,40
Carduelis cannabina	0,50	-	-
Carduelis chloris	0,50	1,00	0,67
Fringilla coelebs	-	0,50	-
Loxia curvirostra	0,50	-	-
Parus ater	0,50	-	-
Parus major	1,00	0,50	0,67
Passer domesticus	-	-	0,33
Serinus serinus	0,50	1,00	0,67
Legitimate frugivores	**Frequency (early winter)**	**Frequency (mid-winter)**	**Frequency (late winter)**
Erithacus rubecula	1,00	0,75	-
Phoenicurus ochruros	0,50	0,50	-
Sylvia melanocephala	1,00	1,00	0,33
Sylvia undata	1,00	-	-
Turdus merula	1,00	0,50	0,67

Table 6.1: Previous estimates of frugivore abundance in the bird assemblage of the study area: (a) breeding densities recorded in the spring of 2007 (point counts with delimited radius); (b) relative frequency (number of surveys in parentheses) calculated during vegetation sampling carried out in the winters 2007-2008. The divisions considered within the wintering season are early (October-November), mid (December-January) and late (February-March) winter.

Illegitimate frugivores (seed predators)	**Density** (ind/ha)	**±SE**
Carduelis cannabina	0.30	0.11
Carduelis carduelis	0.08	0.04
Carduelis chloris	0.21	0.06
Fringilla coelebs	2.70	0.69
Parus cristatus	0.01	0.01
Parus major	0.98	0.12
Serinus serinus	2.35	0.45

Legitimate frugivores	**Density** (ind/ha)	**±SE**
Erithacus rubecula	3.14	0.28
Phoenicurus ochruros	0.48	0.15
Sylvia atricapilla	0.45	0.17
Sylvia melanocephala	4.60	0.33
Sylvia undata	0.85	0.12
Turdus merula	0.70	0.12
Turdus philomelos	0.70	0.23

Table 6.2: Abundance of the frugivore bird species (legitimate and illegitimate) recorded during the winter sampling period of 2011 (overall mean of point counts with delimited radius).

Among the 14 strict frugivores, half were legitimate and half illegitimate (Table 6.2). Total winter bird density in the study area in 2011 was 25.94±5.39 individuals/ha (mean of the six sampling dates ± SE), of which frugivores account for 17.55±2.88 ind/ha. Frugivores represent just over 40% of the total richness and over 60% of forest species *sensu lato* (Table 6.3). The winter density of legitimate frugivores in 2011 (10.92±1.40 ind/ha) doubled on average the spring values of 2007 (around 5 ind/ha), and something opposite occurred with illegitimate frugivores (6.63 ±1.48 ind/ha in winter 2011 *vs* around 9 ind/ha in spring 2007). The spring density of frugivores is in any case relevant since it can indicate the pressure exterted by birds on fruit resources in the late breeding period (Jordano, 1985; Herrera, 2004).

	Whole bird assemblage	*Passeriformes*	Forest species (*s.l.*)	Legitimate frugivores	Illegitimate frugivores
Total richness (n° sps)	34	25	23	7	7
%		73.53	67.65	20.59	20.59
% of forest sps s.*l.*				30.43	30.43
% of frugivores				50.00	50.00
Total density (ind/ha)	25.94	24.46	24.32	10.92	6.63
%		94.29	93.75	42.09	25.56
% of forest sps s.*l.*				44.90	27.26
% of frugivores				62.22	37.78

Table 6.3: Composition and relative richness of frugivore species with regard to the whole forest bird assemblage recorded during the winter sampling period of 2011.

6.3.3 Fruit consumption by legitimate frugivores (direct estimation based on bird foraging activitiy)

The shrubs under observation were visited by 5 species (*S. melanocephala, S. undata, E. rubecula, Phoenicurus ochruros* and *Phylloscopus collybita*), of which the first four are legitimate frugivores, and the last one an occasional consumer. Fruit consumption could be confirmed for the three first ones, and only presumed for the latter. Only two out of these five species visited *P. lentiscus* (*S. melanocephala* and *E. rubecula*). Consumption was observed in 37.5% of bird visits, and presumed in 85.0%. Illegitimate frugivores were not recorded during any of the observation periods.

In the case of *R. lycioides*, the consumption confirmed by direct observation (conservative figure) was 0.012±0.006 fruits per minute, while the maximum possible consumption (or 'optimistic' figure, assuming full success) was 0.028±0.010 fruits per minute (Table 6.4). These values correspond to the activity of the full frugivore assemblage. Most confirmed and presumed consumption was done by *S. melanocephala* (73.3-75.0%) followed by *E. rubecula* and *S. undata* (13.3% of the conservative total).

Pistacia lentiscus		**Fruits removed/min**		**Total fruits consumed (2 months)**	
		Mean	SE	Mean	SE
Visiting species	*Sylvia melanocephala*	0.000	0.000	-	-
	Sylvia melanocephala+P	0.013	0.009	2,261.07	1,544.61
	Erithacus rubecula	0.000	0.000	-	-
	Erithacus rubecula+P	0.013	0.009	2,261.07	1,544.61
	Sylvia undata	0.000	0.000	-	-
	Sylvia undata+P	0.000	0.000	-	-
TOTAL	CONSUMED	0.000	0.000	-	-
	+PRESUMED	0.025	0.017	4,522.15	3,089.22

Rhamnus lycioides		**Fruits removed/min**		**Total fruits consumed (2 months)**	
		Mean	SE	Mean	SE
Visiting species	*Sylvia melanocephala*	0.009	0.003	188,819.71	73,223.88
	Sylvia melanocephala+P	0.021	0.005	463,466.57	113,403.18
	Erithacus rubecula	0.002	0.001	34,330.86	24,179.82
	Erithacus rubecula+P	0.003	0.002	68,661.71	33,922.95
	Sylvia undata	0.002	0.001	34,330.86	24,179.82
	Sylvia undata+P	0.002	0.001	34,330.86	24,179.82
TOTAL	CONSUMED	0,012	0,006	257,481.43	121,583.52
	+PRESUMED	0,028	0,010	566,459.14	171,505.95

Table 6.4: Rates of fruit removal (min-1) from the sample of shrubs under observation by the visiting bird frugivores, and extrapolated total consumption for the study area during the two months of study (October-November). For each bird species the first row represent the observed ('conservative estimate') and those plus the presumed ('optimistic estimate') consumption values. Only the species with confirmed consumption are included.

During the two months of observation, *Rhamnus* fruits were removed in quantities between 42.89±20.25 (conservative figure) to 102.94±37.15 ('optimistic' figure) per shrub, considering eight hours of foraging activity (four in the morning and four in the evening). This is a moderate value that represents on average 6.7-16.1% of the mean standing crop at the start of the study period (650 fruits per observed shrub). Since this crop is reduced to less than a quarter (100 fruits) by the second fortnight of November, the remainder has to be assigned to other fruit fates (falling locally or being removed by other animal consumers).

Extrapolated consumptions for all the local *R. lycioides* population (211.76 ± 65.23 ind/ha) would represent 257,481.43±121,583.52 fruits for realized consumption and 566,459.14±171,505.95 for maximum possible consumption (Table 6.4).

As regards *P. lentiscus*, the two frugivores visiting that species exhibited the same rate of fruit removal (0.013±0.009 fruits/min, only presumed consumption could be estimated). The sum of their activity accounted for 4,522.15±3,089.22 fruits (Table 6.4). This figure is much lower than the number of *Rhamnus* fruits consumed, but would represent all the estimated fruit crop of the focal shrub at the start of the study period.

GLMMs show significant differences in fruit production between shrubs located in the patch interior and those located in both edge types (agricultural and other), production being higher in the former (Table 5). Similar results are found when analyzing consumption, although in this case significant differences were found only in 2010 (Table 6.5).

	2010							
	Production				Consumption			
	FA	FO	FU	**PI**	FA	FO	FU	**PI**
FA				**				**
FO				**				
FU								
PI								

	2011							
	Production				Consumption			
	FA	FO	**FU**	**PI**	FA	FO	FU	PI
FA			***	**				
FO			***	**				
FU								
PI								

Table 6.5: Results of GLMMs relating Rhamnus fruit production and consumption variables with the spatial zonation of the study area (four groups of shrubs, corresponding to small sub-catchments with specific locations within the Forest Park). Differences of the the left class of the predictor with regard to the upper one (in favor of that indicated in **bold**) is shown by asterisks, with significance levels set at 0.01 (***) and 0.05 (**). Codes for spatial subdivisions are: Forest- Agricultural (FA), Forest-Other (FO), Forest-Urban (FU), Patch interior (PI).

A marginal relationship was found between the number of visits and the shrub biovolume (adjusted r^2=0.29; p=0.077; Figure 6.3). The number of visits and the presumed consumption (but not the effectively recorded one) increased significantly at intermediate or intermediate-to-high fruit densities (p<0.05, Kruskal-Wallis test). None of the variables regarding fruit consumption and rate of visitation was significantly related with the location of the shrub regarding the type of forest edge (p>0.1 in all cases).

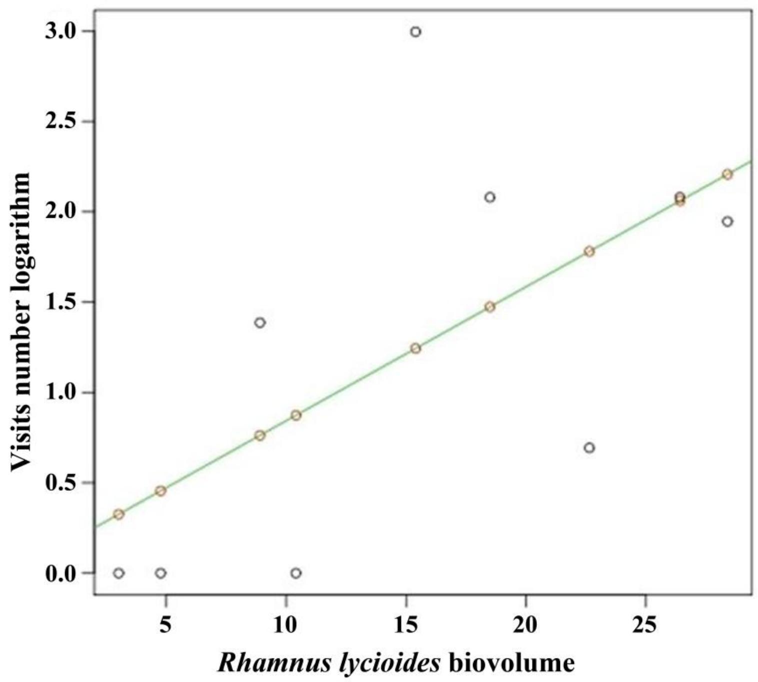


Figure 6.3: Relationship between the number of bird visits to Rhamnus shrubs and their biovolume (regression is marginally significant, $p<0.1$)

Significant differences in the number of seeds collected under perches (pine trees and shrubs) where found in most paired comparisons, both for total seeds (of all shrub species) and for those of R. lycioides only (Table 6.6). In general, patch interior perches where those that received a lower number of seeds, and forest-agricultural edges those with more seeds deposited.

	All seeds				Seeds of *Rhamnus lycioides*			
	FA	FO	FU	PI	FA	FO	FU	PI
FA		***	***	***		***	***	***
FO				***			***	***
FU				***				***
PI								

Table 6.6: Results of GLMMs relating seed rain with the spatial distribution of perches (same divisions used for shrubs in table 3). Differences of the left class of the predictor with regard to the upper one (in favor of that indicated in bold) are shown by asterisks, with significance levels set at 0.01 (***) and 0.05 (**). Data are pooled for 2010 and 2011.

6.3.4 Indirect estimation of potential Rhamnus lycioides fruits consumption by legitimate frugivores (based on their energy requeriments)

The three most abundant legitimate frugivores, *S. melanocephala* (4.60±0.33 ind/ha) and *E. rubecula* (3.14±0.28 ind/ha) and *S. undata* (0.85±0.12 ind/ha) had a total requirement of 6572,93 kj. Consequently they would need to consume between 526,809.26±44,332.23 and 353,419.97± 29,517.26 fruits of *Rhamnus* to meet their energy requirements during the months under study (Table 6.7), depending on the degree of frugivory of their diets. The degree of frugivory reflects the relative importance of fleshy fruits in the species' diets (Herrera, 2004). It was set at the minimum value of the range (65-100%; Jordano, 1987) for *S. undata*, a species for which a mean value was not reported by the first author. Considering an available crop of 1.16 million *Rhamnus* fruits by the start of the study period (12% of the 9.69 million estimated by Zapata *et al*., 2014, for the whole area), these three bird species take between 45 and 30% of the standing crop available during that period. Even if the consumption of the less abundant legitimate frugivores is added, the overall figure does not exceed the 50% of the available fruit resource.

Bird species	*S. melanocephala*	*E. rubecula*	*S. undata*
Winter density	4.60	3.14	0.85
SE	0.33	0.28	0.12
Wintering population (individuals)	123.56	84.34	22.83
SE	8.86	7.52	3.22
Weight (g)	12.50	17.30	9.10
Daily bird requirements in kJ (2,77 kJ/g)	34.63	47.92	25.21
Total bird requirements in kj (61 days)	2,112.13	2,923.18	1,537.63
% frugivory	70.00	60.00	65.00
Fruits required (100% frugivory)	2,050.61	2,838.04	1,492.84
Fruits required (specific % frugivory)	1,478.49	1,753.91	999.46
Required fruit consumption (100% frug)	253,364.77	239,361.41	34,083.07
SE	18,176.17	21,344.33	4,811.73
Required fruit consumption (sp % frug)	182,676.00	147,925.35	22,818.62
SE	13,105.02	13,190.80	3,221.45
Estimated consumption (observation)	188,819.71	34,330.86	34,330.86
SE	73,223.88	24,179.82	24,179.82
% of required (100% frug)	74.52	14.34	100.73
% of required (sp % frug)	103.36	23.21	150.45
Estimated consumption (+presumed)	463,466.57	68,661.71	34,330.86
SE	113,403.18	33,922.95	24,179.82
% of required (100% frug)	182.92	28.69	100.73
% of required (sp % frug)	253.71	46.42	150.45
Required *vs* consumed (100% frugivory)	R>E; R<E+P	R>>E; R>>E+P	R≈E; R≈E+P
Required *vs* consumed (sp % frugivory)	R≈E; R<E+P	R>>E; R>>E+P	R<E; R<E+P

Tabla 6.7: Calculations of expected Rhamnus fruit consumption by the three most abundant wintering frugivores (S. melanocephala, E. rubecula and S. undata) during the two months of the study, based on their energy requirements and considering two degrees of frugivory (100% and the species' specific percent). Ph. collybita has been excluded from the calculation because consumption was not confirmed by observations. Bird weights were obtained from Herrando & Brotons (2001) for S. melanocephala and from http://genomics.senescence.info/species/ for the other two species.

6.3.5 Seed dispersal analysis from mist-netting and faecal samples

Only 20% of the droppings collected from caught birds contained seeds (max. 3 seeds/dropping). Furthermore, these seeds belonged exclusively to *A. albus* and A. *horridus*.

The bird–ordination through PCA displays information on the different areas between which frugivores move (Figure 6.4). The cumulative variance of the two first axes (Principal Components I and II) was 92.82%. *Turdus merula* and *T. philomelos* seem to prefer moving between forest areas and cropland, while *P. ochruros* and *Sylvia atricapilla* preferently move from the forest patch to the urban gardens. This is in accordance with the preference of these species for peripheral forest habitats, as shown by their average densities in these zones (Figure 6.5). Thus, the horizontal axis (PC I) of the ordination depicts a gradient from urban edges to agricultural/dry grassland ones. In turn, the positions of of *S. melanocephala* and *E. rubecula* display a gradient of habitat preferences from interior to peripheral forest habitat (PC II, vertical axis). However, the ordination is not conclusive as regards potential seed transport, given the very low representation of seeds in faeces of birds caught in patch edges, which suggests a very weak dispersal from the patch interior. Samples from perches showed a much higher presence of seeds: 58.82% of the samples contained seeds, and the maximum number of seeds in a faecal sample was 25. In this case, the predominant species was *R. lycioides* although *A. albus* and *A.horridus* appear too.

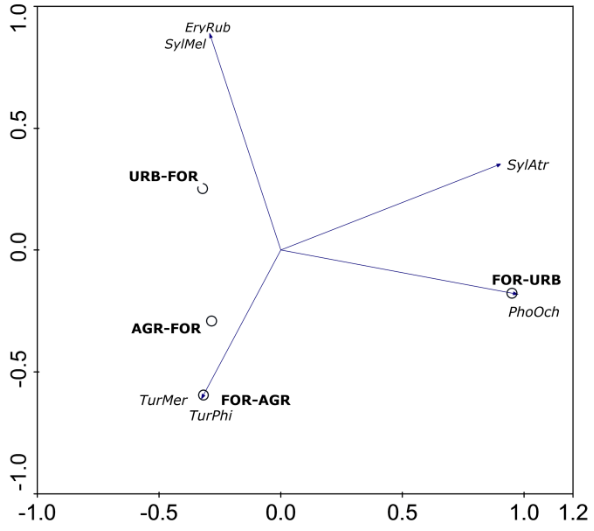


Figure 6.4: Ordination of bird movements across different types of forest patch edges, performed through Principal Component Analysis of the number of birds caught in mist-nets in each transition.

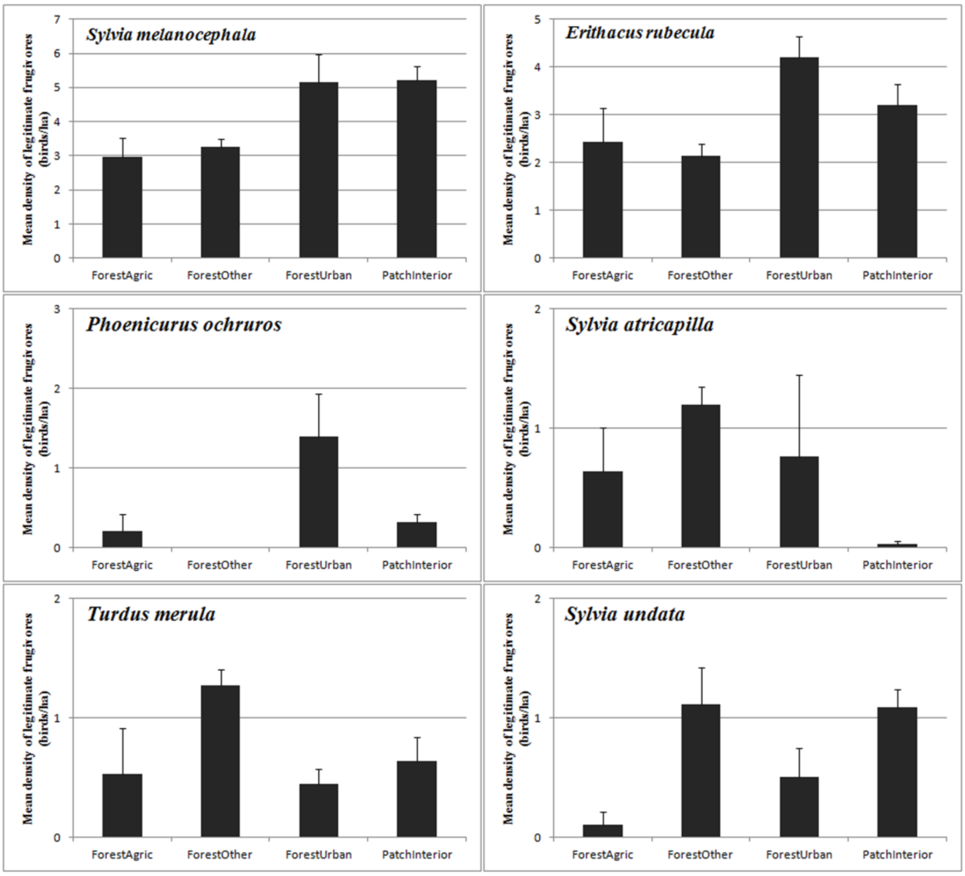


Figure 6.5: Mean density of the most abundant legitímate frugivores recorded in point count census (species with less than 1 ind/ha in all habitats excluded).

6.4. Discussion and conclusions

Mutualistic interactions between frugivorous birds and Mediterranean shrubs are essential for the conservation of ecological and biological diversity (Tellería *et al.*, 2005), but are poorly studied in semiarid ecotonal areas (García *et al*., 2010; Zapata *et al*., 2014). Under the limiting environmental conditions characteristic of Mediterranean semiarid landscapes, it is essential to assess the significance of bird frugivory, its spatial and temporal variation, and its relationship with the availability of fruits. The focus has to be put on the critical areas or periods regarding the conservation of birds (and the services provided by them), as well as on the consequences for key plant processes (e.g. local regeneration, dispersal and colonization of new areas).

This study represents only a first attempt to evaluate bird-shrub interactions in semiarid Mediterranean landscapes, where human influence on forest habitat (landscape disturbance and climatic change) can result in critically limiting conditions for biodiversity-based values and services (Esteve-Selma *et al*., 2010; García-Llorente *et al*., 2012). However, we are confident that the results can provide a general framework for the interpretation of dispersal processes and a preliminary assessment of the services delivered by bird frugivores in the context on forest conservation and management.

Our main findings are that fruit resources seem enough for the birds overwintering in the study area, even in the case that alternative food resources (invertebrates) were so scarce to force birds to keep a strictly frugivore diet. Although the distribution of fruit production and consumption within the patch favours interior habitats (Table 6.5), seed deposition is more important in peripheral ones, suggesting a trend for birds to feed in the former but rest preferently in the latter (Table 6.6). However, seed export from the patch to surrounding habitats seems irrelevant.

6.4.1 Fruit resources and frugivorous population

Local fruit resources, although important and probably sufficient for local wintering frugivore populations, are scarcer than in more typical Mediterranean habitats, where *R. alaternus* and *P. lentiscus* can produce from a few thousand to several tens of thousands of fruits per individual shrub (Jordano, 1989b; Bas *et al.*, 2005). At the start of October, wintering frugivores have access to around one million *Rhamnus* fruits (1,704.39 ± 716.61 per individual shrub; Zapata *et al.,* 2014), approximately half of which are consumed according to the more optimistic estimate of feeding success.

Although it is expected that bird communities change in response to the availability and quality of fruits (Jordano, 1985; Rodríguez de los Santos *et al.*, 1986; Cuadrado *et al.*, 1995), this does not always occur, as pointed by Herrera (2004). The lack of a direct relationship could indicate that the responses of bird communities differ according to the type of habitat. In fruit–frugivore interactions high variability is the norm, but the composition of the bird assemblage studied seems to give some support to the fruit-tracking theory, which predicts that changes in the abundance of consumers are linked to changes in the abundance of fruit resources in time and space (Burns, 2004).

6.4.2 Consumer behaviour

Individual fruit consumption may seem low (i.e. less than one fruit per half an hour of feeding activity in the best case). This will represent 16 fruits/day or 960 fruits during the two months studied. Few researches have assessed fruit consumption in similar ecosystems. Traveset (1994) cites a consumption of 0.1 mature fruits of *P. terebinthus* per minute (three times higher than our best estimate), considering the activity of the total community of frugivorous. If the consumption by illegitimate frugivores is subtracted (53.1% in his study), the figure is lowered to 0.05 fruits per minute, approaching the maximum possible consumption of our study area. Anyway, consumption in both years studied was lower compared to other areas. This could be due to the low density of shrub species with much higher lipid concentrations, like *P. lentiscus* (>58%) or *Olea europaea* var. *sylvestris* (>20%), compared to the less than 5% of *R. lycioides* (Herrera, 1984). This could force birds to concentrate in food sources energetically and nutritionally more profitable, like garden plants or insects. Birds are able to measure the quality of the fruits discriminating those with low nutrient content, and can also assess fruit lipid content from their texture, being able to distinguish contents from 1% to 2% (Schaefer *et al.,* 2003).

As seen in section 3.3, the number of visits increases with shrub biovolume (Figure 6.3). However, birds prefer shrubs with medium or high-medium densities of fruits. One possible explanation is that higher fruit densities increase competition (intraspecific and interspecific), and hence interference. On the other hand, although high densities allow a shorter stay in the shrub, predators could expect more bird visits to shrubs with high fruit densities (Murray, 1987; Guitián *et al*., 2000).

There is a great variation in the percentage of energy requirements that can be covered by *R. lycioides* fruit, depending if the observed or maximum potential consumption is taken as a reference. In the case of *S. melanocephala* these percentages range from some 70-100 (considering a degree of frugivory of 70%) for realized consumption, and 180-250 (considering a 100% of frugivory) for 'optimistic' consumption (Table 6.7). Even under the conservative estimate, foraging activity seems sufficient to supply birds with their winter energy requirements. This fact gives importance to the method of direct observation for quantifying fruit consumption by birds in a determined habitat, always in combination with indirect estimations based on fruit removal from shrubs (see section 3.3), or on bird requirements, particularly if a rapid assessment is needed (i.e., in environmental impact studies).

The study years were wetter than average (290.5 mm), particularly 2010 with a total precipitation of 414 mm, but also 2011 (309.46 mm). The standing *Rhamnus* crops at the start of the wintering period (similar among years as it has been seen), could then also represent higher-than-average fruit resources. Inter-annual climatic variability is a characteristic feature of Mediterranean areas (Di Castri *et al*., 1981; Mazzoleni *et al*., 1992). Therefore, the shortage of *Rhamnus* (or *Pistacia*) fruits can represent a major limitation for birds on years with less precipitation (Tellería and Perez-Tris, 2004).

The absence of apparent seed export to surrounding habitats can be related to the fact that fruit resources have been sufficient inside the patch and far-reaching foraging movements were unnecessary, which reduced the chance of seeds to be transported outside it. Territorial defense of feeding habitats can also be invoked to explain the internal distribution of birds (Tellería and Perez-Tris, 2004). In summary, birds seem to range close to the patch edges but apparently do not trespass them and consequently do not transport seeds through (at most some of *Asparagus*).

Regarding the influence of the temporal pattern of seed consumption on shrub regeneration, the effective dispersal (intra and extrapatch) of *R. lycioides* seeds could experience a limitation, related with their survival and germination (Schupp *et al*., 2010). Only a small fraction is available for endozoochorous dispersal at the best time for germination. By the start of the study the majority of the seeds had already been consumed and thus dispersed earlier, during a much dryer period, and when the risk of predation by illegitimate frugivores is higher, diminishing their chance of survival and establishment. Seeds deposited in suitable microhabitats in early winter would have more options to survive than those dispersed in late summer and autumn, and consequently could contribute more to shrub regeneration and to the colonization of new areas.

6.4.3 Alternative models of fruit consumption. Implications for seed dispersal and potential application to management.

GLMMs show that seed production and consumption are concentrated in interior habitats of the patch, while deposition is more widespread along the patch (but quite weak outside it). We have also seen a different distribution of the legitimate frugivores with regard to patch edges (Figure 6.4) and also within the patch (Figure 6.5). The two numerically dominant species (*S. melanocephala* and *E. rubecula*) favour interior areas, but are also well distributed in other areas of the patch. Of the remainder species, *P. ochruros* and *S. atricapilla* favour forest-urban edges.

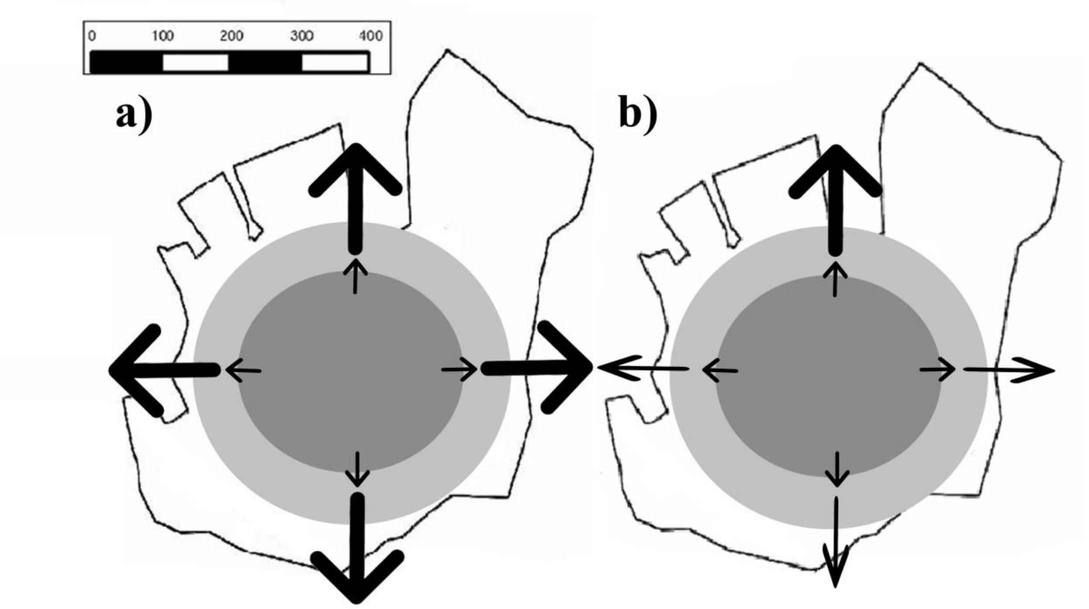


Figure 6.6: Alternative models proposed to describe bird movement patterns, and relative importance of seed dispersal directions within the patch and outside it.

Two spatial models of seed dispersal (Figure 6.6), related to bird distribution and movement, emerge from these patterns. They seek also to explain the pattern of seed deposition within the patch, which follows a different distribution, with part of the seeds being deposited under selected interior shrubs and the remainder under non-feeding perches (close to other shrubs scattered throughout the patch). This suggests that some birds (particularly of the dominant species) can exploit specific fruit patches, while forcing other species (and some individuals of the former) to feed in a less concentrated manner in peripheral areas. In this regard, the possibility that there is a specialization of individual birds is proposed. Some would set in the internal zones of the patch, and others on the peripheral areas.

The resulting models probably would apply more to years with food limitation (i.e., with less rainfall and consequently lower fruit crops). The first one explains seed dispersal assuming a patch-concentrated consumption (Figure 6.6a). The internal circle represents the area exploited by individuals taking advantage of the richer feeding patches, and the outer belt the area where other individuals have to rely on a lower and sparser fruit crop (which could force them to exploit other fruit sources, eventually outside the patch). The small arrows represent the dispersing movements within the patch, that would be very limited because individuals tend to settle near the consumption areas (patches). So, most of the seeds would be deposited close to the parent plants in short-distance movements, increasing intra- and interspecific competition and reinforcing the internal heterogeneity patterns.

Larger arrows represent potential dispersal movements from the periphery of the forest patch to adjacent areas. Now, seed export can be greater as the birds have less fruits available and would have to search more in adjacent areas (although such transport has been anecdotal in our study). Dispersal from the patch periphery towards its interior would be low, since territorial individuals will prevent the entry of others.

The second model explains seed dispersal assuming interior patch consumption and a more intensive use of the gardens as a food resource (Figure 6.6b). This could be performed by territorial interior birds in periods of food scarcity, and by preferently peripheral individuals or species as a routine. The meaning of are the same in this case, but the arrows change. Now, dispersion will be displaced mainly northwards (urbanization), as a consequence of the garden areas' attraction for feeding. Therefore, any potential export of seeds of forest species ingested by birds would go to artificial areas (which can be interpreted as propagule sinks) while the risk of exotic propagule import will be increased.

These models can be relevant for the regeneration of the forest ecosystem itself, and for the potential recovery of adjacent non-forest areas. Seed deposition close to feeding spots could deliver either an ineffective dispersal or a non-homogeneous distribution, reinforcing forest heterogeneity. However, the greatest limitation seems to be the weak potential for seed export towards potentially colonisable habitats (e.g., abandoned cropland) where heterogeneity would not be the greatest problem, but dispersion must be intense enough to generate effective colonization foci.

To enhance the role of frugivorous birds in forest ecosystem restoration, ancillary techniques need to be taken. We propose: (i) low density reforestations to promote a more homogeneous dispersion, and (ii) the installation of artificial perches to favour seed deposition in selected microhabitats or areas.

Restrictive environmental conditions imposed by global change in semiarid areas will result in an increase of scrublands at the expense of trees (Peñuelas *et al.*, 2004; Valladares, 2007; Mangas *et al.*, 2008). Restoration policies need to consider the species distribution models, but also take into account the variations in species interactions that global change will promote (Urban *et al.*, 2012). Similarly, fostering the transition from tree reforestations to open scrublands, will favour the integration in the bird assemblages of these landscapes, of bird species with more restricted distribution in Europe (Birdlife International, 2004), increasing the conservation interest of these areas (Zapata and Robledano, 2014).

6.5. Acknowledgements

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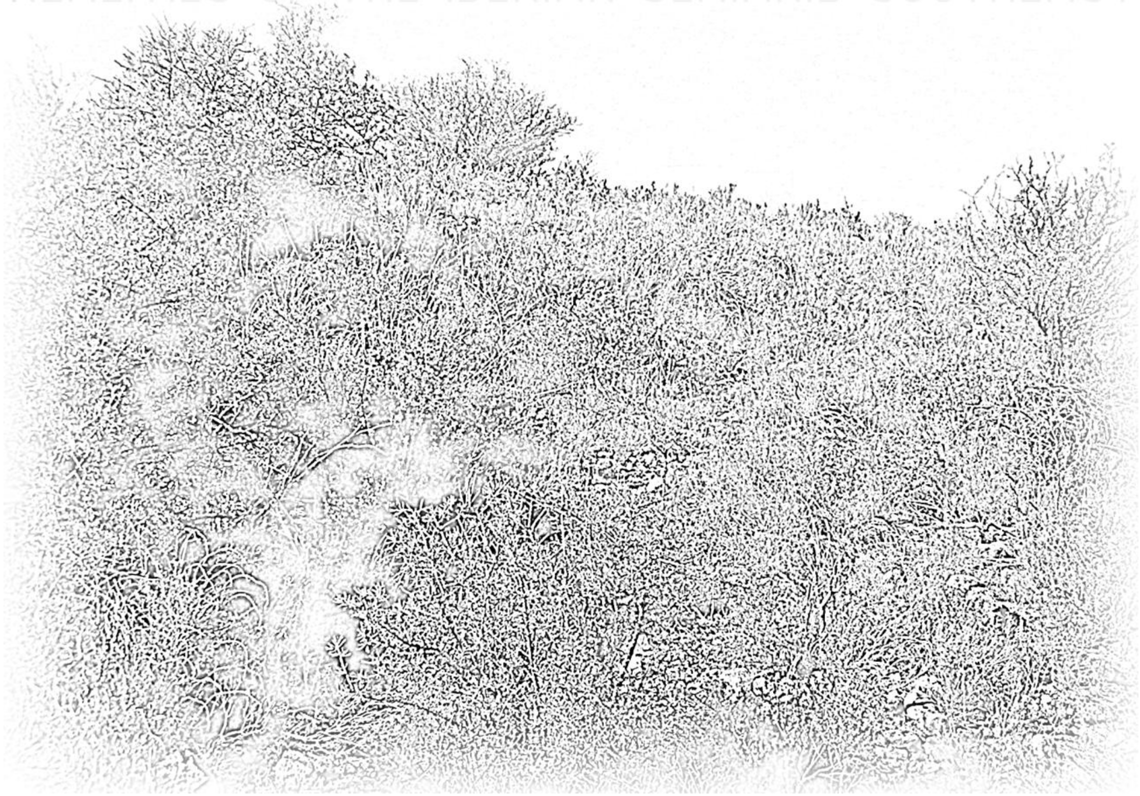
REALITIES OF THE IBERIAN SEMIARID SOUTHEAST



7. Realities of the Iberian semiarid southeast

Regardless of the results that have been exposed in this thesis, there are associated problems related to semiarid southeastern Iberian forest ecosystems. Such problems which might initially seem theoretical threats, have become realities in recent decades and are in full force today.

Mostly, these problems represent human management failures arising from inadequate reforestation policies, wrong urban planning and lack of application of ecological knowledge.

Consequently, this section that precedes the conclusions, shows real examples that highlight the value of the research on which this thesis is framed, In this respect, studies gathered herein address the basis for such failures, and are in some way advancing the course of some of the problems currently being manifested.

7.1 Urban planning and management failures

Several of the aforementioned studies have pointed some effects on different biodiversity groups, owing to the edge effect caused by urbanization. Whatever the contribution of such effect to the biodiversity patterns found in each patch (the subject of our research), there are quotidian visual evidences of this effect derived from poor urban planning and implementation. This would include the selection of urban resort placement (in relation to forest areas), the internal arrangement of built and green spaces, their environmental assessment (and the predictive ecological basis for it), and the consequent monitoring plans to assess and correct unexpected impacts.

For example, dumping of waste, simplification of the vegetation, introduction of invasive exotic species, road mortality, etc. (Figure 7.1), illustrate failures in all these stages of the urban planning process.

Figure 6.1: Examples of edge effects and other fragmentation-derived impacts.

Particularly, urban planning without sound ecological basis has been usual in these areas. This has caused the total isolation of forest patches and irregular shapes in other, that increase the edge effect. As a result, some patches have minimum areas, which may explain their inability to maintain species with higher requirements of space (Figure 7.2).

Figure 7.2: Isolation of patches by urbanizations.

7.2 Ecological failures

The lack of ecological basis does not only undermine urban planning and environmental assessment, but also the management of natural remnants embedded in the urban matrix, often as green spaces or urban parks (local nature reserves). The simplification of the ecosystem caused by urban fragmentation has been already discussed. It is easy to see a general trend towards vegetation homogenization and ruderalization once fragmentation has occurred, and extinctions of species (due to size, shape and edge effects) leave but the more widespread and tolerant species (Figure 7.3). The same happens with the biodiversity of the patch in general.

Figure7.3: Examples of vegetation simplification

Monospecific conifer plantation has been another common intervention in Mediterranean semiarid areas. The internal structure and composition of these areas confer them a lower biological richness and ecological biodiversity than scrub woodlands. The latter are best adapted to current climate semiarid conditions and seem better prepared for the new restrictive conditions imposed by climate change (Figure 7.4).

Figure7.4: Some patches with high pine densities.

7.3 Management failures

The reforestation techniques, particularly those applied in the more recent decades, came not only to disrupt the ecological functions inside the reforested patches, but often also to decrease the landscape value of these areas. This visual impact is the result high pine densities, aligned planting designs, soil preparation with heavy machinery (terracing), monospecific plantations, etc (Figure 7.5)

Figure7.5: Dense aligned designs, terraces and monospecific plantations in pine-dominated landscapes

Current investment in environmental protection and management has decreased due to global economic crisis that some countries (especially Mediterranean ones) have suffered in a particularly severe way. Therefore, any policy that advocates the use of passive restoration techniques must be prioritized over classical measures economically less profitable (Table 7.1). Similarly, ecological and forest research to improve the knowledge about such techniques should be a priority to improve their implementation and reduce costs in the long term.

Budget items	**Total investment period 2003/2012 (€)**	**Annual average (€)**
Forest management	27,249,692	2,724,969
Ecological and forest research	3,282,830	328,283
Restoration and improvement of the natural environment	32,826,132	3,282,613
Environmental restoration	19,495,733	1,949,573

Table 7.1: Reforestation *budgets* in Murcia Region during the last decade (Source: Estrategia Forestal de la Región de Murcia)

7.4 Reforestation failures

Nowadays, pine stands that are present in forest patches are dying due to the attack of a wood borer beetle (*Orthotomicus erosus*) that has spread due to the weakening of pinewoods growing under Mediterranean semiarid climate. A severe drought episode has placed many *P. halepensis* forests in conditions that seem definitely outside their ecological niche, especially when pine density is not suited to those conditiones. This has exacerbated the weakness of tree formations being exposed to such attacks. Climatic changes and extreme climatic events are expected to increase the occurrence and virulence of abiotic pests like *O. erosus* in European forests[1] (Figure 7.6).

Figure 7.6: Attack of wood borer beetles on stands of Pinus halepensis weakened by drought.

Moreover, the decay of pines allows the shrubs that benefit from their passive and active facilitation (i.e., use them as perch for seed dispersers, and as nurse plant), to succeed in the interaction. In most cases, pines are seen as competitors but in some instances the shrubs can grow up under their canopy taking advantage of the shadow and greater humidity of such microhabitat. When extreme events occur (like drought periods or pest outbreaks), the death of the tree results in a competitive release in favor of the already established shrubs. Following this process, passive restoration techniques (use of trees as perches) could be combined with the selective removal of trees already having performed this role, to get more persistent forest formations (Figure 7.7)

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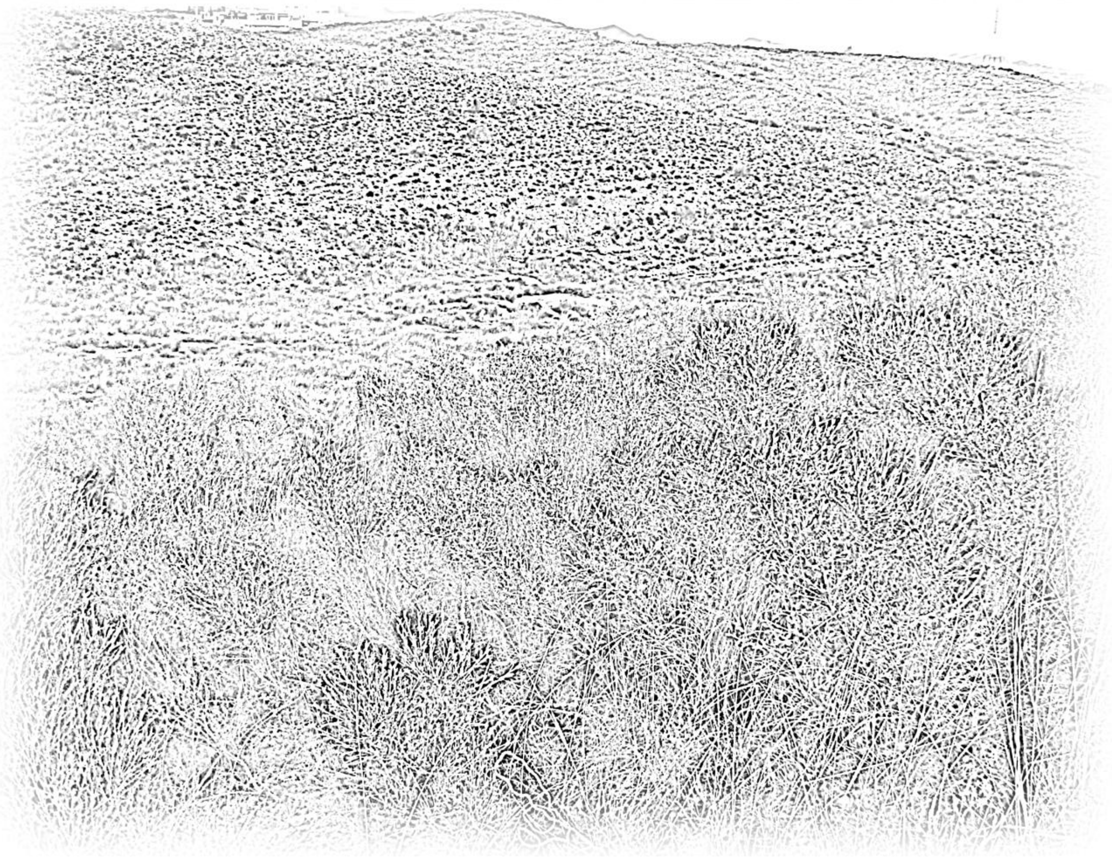
[1] Spathelf, P. Van Der Maaten, E., Van Der Maaten-Theunissen, M., Campioli, M., Dobrowolska, D. 2014. Climate change impacts in European forests: the expert views of local observers. Annals of Forest Science, 71 (2): 131-137.

Figure 7.7: Transient facilitation process by decaying or cleared pine trees

GENERAL CONCLUSIONS



8. GENERAL CONCLUSIONS

The general conclusions of this thesis can be divided between pure research conclusions, extracted from the discussion of the chapters' results, and applied conclusions (recommendations/proposals). Despite their interdependent character, the latter fall approximately under three main topics.

8.1 Research conclusions

- In the southeast of the Iberian Peninsula (Western Mediterranean), the interaction between current urban development and fragmented natural landscapes is one of the main determinants of forest biodiversity. As a result of its particular climate and diverse lithology, and the changing land uses superimposed on the natural biophysical setting, an ecologically and culturally rich rural landscape has evolved.

- In recent decades, many semi-natural remnants of that landscape have been directly occupied by human activities (urbanization, infrastructure, intensive agriculture), managed to increase forest cover, or subject of indirect effects from fragmentation, causing changes in local species composition and variations in their regional distribution areas.

- Intrinsic (size, shape, habitat type and historical management) and extrinsic factors (main fragmentation agent, surrounding land uses, connectivity) condition the biodiversity of forest patches, defined in a broad sense as any remnant of vegetation dominated by woody plants, whether arboreal, shrub-like, or a mixture of both, and independently of its origin (natural *vs* reforested), use, or management.

- Such remnants, usually in the form of isolated patches, local patch networks (*archipelagos*), or marginal fragments disconnected from edges of the forest matrix of main mountain systems, are candidate to take part in municipal systems of protected areas or green spaces (urban forests).

- Birds and woody vegetation qualify as useful indicators of the biodiversity value of in these Mediterranean forest fragments. Although with differences in terms of ease of collection and previous expertise required, the information they provide is of great value for extensive surveys and quick evaluations of the type demanded by environmental impact and land planning assessments.

- Moreover, complementarity has been established between the indication provided by different taxocenoses (e.g. woody plants *vs* forest birds), which calls for a multitaxa assessment (including these and other compositional, structural and functional indicators) when addressing the biodiversity of intensively man-managed landscapes.

- As shown by the lack of correspondence between the legal status of a forest fragment and its conservation value, heterogeneous rural mosaics retaining a variety of forest patches, spread along piedmonts and lowlands between large mountain areas with continuous forest landscapes, can be relevant for the conservation of biodiversity at local and regional scales.

- The internal features of forest patches are also important for their biodiversity, independently of size, spatial location or landscape setting. The negative correlation between floristic value and pine density also supports the relevance of site (internal) management variables related to the habitat structure in explaining the biodiversity of the forest remnants of these landscapes.

- Studies performed in fragmented semiarid forest landscapes from a multiscale and multispecies approach, point to different effects of urbanization, depending on the biodiversity indicator considered. Vegetation seems more vulnerable due to its higher sensitivity to fragmentation (low mobility and barriers to dispersal). For birds, changes in community composition and diversity occur as a result of edge effects and the intrinsic characteristics of each isolated fragment (patch size, direct urban influence…).

- Regarding spatial scales, a landscape-patch-microhabitat approach allows the identification of management measures to mitigate the impact of urbanization, and the establishment of guidelines to preserve biodiversity in areas facing urban growth directly. Considering such a range of scales is essential to ensure that, in urban-fragmented contexts, forest remnants are able to fulfil their ecological, economic and social functions.

- Classification and indicator value analyses organized shrubland and forest areas in separate groups representing different levels of urban affection, with higher values of plant conservation indexes in the former (defined by the lack of a tree stratum). Ornithological value changed with the scope of the conservation index, open and shrubland areas scoring higher in the European index than in national ones. Higher bird diversity occurred in shrubland and forest patches far from urban or agricultural developments.

- In Mediterranean-type areas, water limitation confers to woody non-arboreal formations (scrublands) a capital importance. There, fleshy-fruiting shrubs provide food resources for vertebrates (namely birds), giving rise to key mutualistic plant-animal interactions. These are still poorly studied in marginal Mediterranean areas like the semiarid Iberian Southeast, despite their importance for shrub dispersal and forest regeneration.

- The hypothesis that birds direct dispersal of towards the pine tree stratum of semiarid forest patches was supported by studies combining estimates of fruit removal and seed rain. Only a small fraction of the seeds were deposited under pines, but a disproportionate number of adult shrubs were under the tree layer, although canopy cover is typically sparse.

Pines seemed to provide a favorable regeneration niche, at least during the germination and establishment phases.

- Fruit resources seem enough for the birds overwintering in semiarid Mediterranean forest remnants. Although fruit production and consumption within the studied patch was biased towards interior habitats, seed deposition was more important in peripheral ones, suggesting a trend for birds to feed in the former but rest preferently in the latter. Alternative spatial models are proposed to explain the potential consequences for shrub dispersal and forest regeneration.

8.2 Applied conclusions

Regarding ***future urban developments***:

- The distribution of built areas must be carefully planned when it affects natural areas, attempting to maintain buffer areas and to minimize in this way, edge effects. Similarly, it should be pursued to avoid total isolation of patches or to create more permeable zones between them and urban spaces.

- Land acquisition as a result of urban development (through compulsory donation of natural or green spaces) should be an opportunity for public authorities to get strategic locations that would perform the conservation, buffer or restoration roles attributed to fragmented forest patches and related habitats in this thesis.

- The design of public green spaces inside urbanizations should incorporate native species to improve ecologic permeability and connectivity, while becoming a source of propagules for close forest patches. Encouraging local population to use native species in their private gardens would have similar effects.

With respect to ***reforestation policies and methods***:

- A change in current financial policies and incentives for reforestation is necessary. Till now, financial aid regulations have imposed the reforestation of private lands with trees (with species of the *Pinus* genus in a high proportion). While this can be acceptable in wetter sections (i.e. those of mesomediterranean bioclimatic areas), in drier semiarid ones (thermomediterranean), a case-by-case study would be necessary.

- This research has sought to incorporate to the management framework of semiarid Mediterranean forest formations, the use of ecological interactions as a way to optimize resources and save costs. In this respect, the preliminary study of vertebrate dispersal as a passive restoration technique has helped to identify some recommendations and cautions.

- Considering that frugivore birds often seem to disperse seeds inside the same patch where they have consumed them, dispersal can be ineffective to restore neighbouring non-forested areas. However, it can be useful for internal patch restoration, as a way to modify tree-dominated forest habitats towards scrub woodland formations.

- Private gardens of the urban areas and households are attractive to frugivore birds, which could produce a significant flow of seeds between natural habitat and these gardens. This fact highlights the importance to use native species in private gardens.

- Raising the general awareness about the importance of shrublands and open woodlands as the most representative landscape of semiarid Mediterranean areas is necessary, beginning with a with a firm and adequately explained restoration of underdeveloped tree forestations towards these more resilient and biologically rich formations

Finally, in relation to ***internal patch management***:

- In this thesis, a wide typology of forest units has been studied, ranging from detailed (site) to landscape-scale (patch systems) scales, in order to determine biodiversity patterns and their environmental determinants. Therefore, a specific survey and environmental diagnosis of each forest patch or estate incorporated to the local natural network is essential for determining the correct action to take for its management.

- Ideally, such survey and diagnosis should be followed by a management plan, and by a monitoring scheme to assess its performance. All these aspects should be incorporated as a compulsory requirement of urbanization plans and their environmental assessment, for any land donation to municipalities as a result of urban development projects

- According to the environmental assessment recommended, in all those patches with high densities of pine trees but inadequate conditions for their development, the number should be reduced through selective thinning. This operation should be directed to weakened or dying specimens, particularly those that have acted as nurse plant for shrubs, which will favour the transition towards a scrub woodland physiognomy.

RESUMEN Y CONCLUSIONES GENERALES

9. Resumen general

9.1 Resumen general

La biodiversidad ha evolucionado en la Cuenca Mediterránea a lo largo de la historia geológica y de las distintas fases de colonización humana. Diferentes etapas de enriquecimiento relacionadas con fenómenos tectónicos y climáticos, y sus consiguientes procesos de migración y refugio, han dado como resultado un paisaje diverso que califica como uno de los puntos calientes de biodiversidad mundial. En las últimas décadas, los procesos de desarrollo urbanístico se han venido promoviendo de manera continua a lo largo de todo este espacio, aprovechando su clima privilegiado y la singularidad de su paisaje y patrimonio cultural.

La interacción entre estos dos tipos de desarrollo, natural y urbano, es el origen de uno de los principales problemas a los que se enfrenta actualmente la biodiversidad mediterránea. En este aspecto, el sureste de la Península Ibérica no ha sido una excepción. Debido a la predominancia de un clima semiárido particular, y a su diversidad litológica, son numerosas las especies que han evolucionado en esta zona o han encontrado en ella nichos donde establecerse. Muchos de estos hábitats han sido ocupados directamente por construcciones humanas haciendo totalmente incompatible su persistencia. En otros casos, lo que el desarrollo urbano ha provocado ha sido la fragmentación de las zonas naturales ocasionando cambios en la composición local de especies y variaciones en sus áreas de distribución regional.

Centrándonos en los principales actores de la fragmentación de zonas naturales en el Sureste de la Península Ibérica, no solo debemos fijarnos en los procesos urbanos, sino que también en otros tipos de actividades como la agricultura. Las consecuencias para los ecosistemas son muy diferentes dependiendo del tipo de fragmentación. Del mismo modo, las consecuencias también variarán dependiendo de si el ecosistema fragmentado se restaura con especies o comunidades dentro de su rango potencial de crecimiento o si, por el contrario, el hombre está tratando de utilizar especies o formas de vida en condiciones cerca del límite de su nicho potencial.

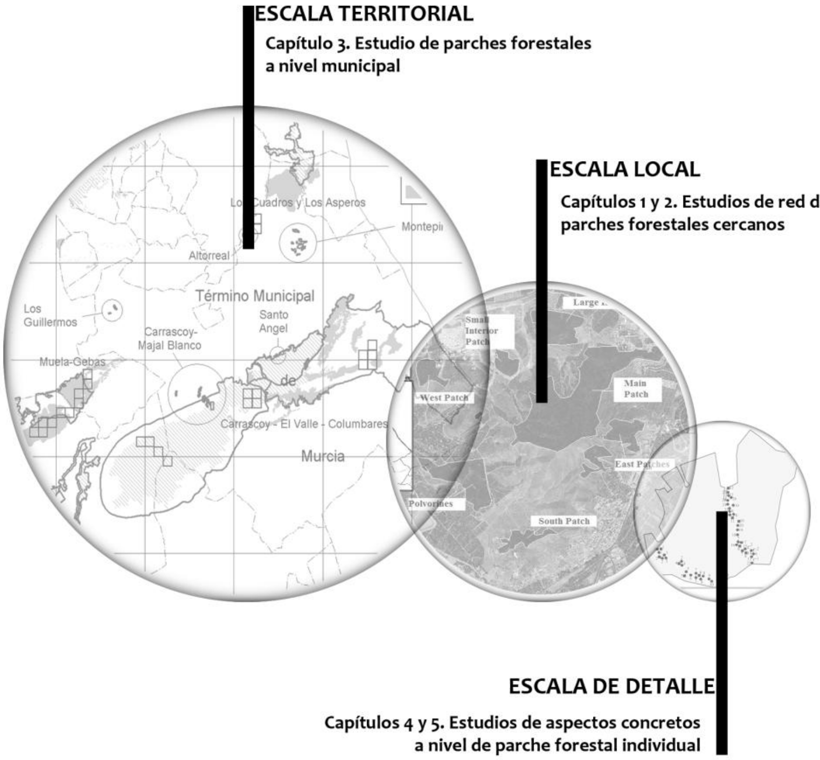
Sobre esta base, la presente tesis expone avances en la comprensión de las comunidades biológicas que habitan estas zonas fragmentadas, mostrando los efectos del tipo de fragmentación sobre los fragmentos naturales dependiendo del tipo de ecosistema predominante en ellos, estudia el desarrollo de algunas interacciones biológicas en estos sistemas así como su posible uso como técnicas de restauración pasiva, y por último propone medidas de gestión para el desarrollo natural y la restauración de estas zonas.

Como bioindicadores de las comunidades que se desarrollan en estos fragmentos forestales, en el presente trabajo se han elegido las aves y la vegetación leñosa. En el caso de la avifauna, se necesita de amplia experiencia para su identificación y registro, pero al mismo tiempo, la información que nos aportan es de gran importancia ya que la mayoría de especies de este grupo se sitúan en los eslabones más altos de las redes tróficas. En el caso de la vegetación, su muestreo es más sencillo, y su importancia radica en ser uno de los pilares estructurales básicos sobre el que se organizan el resto de comunidades, así como uno de los componentes más importantes del paisaje.

Esta tesis ha desarrollado sus estudios en parches forestales de naturaleza diversa distribuidos por todo el municipio de Murcia, localizado en el Sureste de la Península Ibérica (Oeste del Mediterráneo). El clima predominante es Mediterráneo Semiárido con precipitaciones anuales que rondan los 300 mm y temperatura media de 18,6 °C, con distribución irregular de las precipitaciones aunque tendentes a concentrarse en Otoño-Invierno.

En esta zona, las políticas de reforestación ejecutadas a mediados del siglo XX plagaron los montes de prácticas agresivas con plantaciones monoespecíficas de Pino Carrasco (*Pinus halepensis* Miller.). Las condiciones climáticas semiáridas pueden ser limitantes para el desarrollo de esta especie, por ello el resultado son a menudo masas arbóreas poco desarrolladas y con escasa representación de otras especies vegetales. Esta simplificación del ecosistema resulta ser problemática a nivel ecológico, pero también a nivel social por la baja calidad paisajística que confiere a estos lugares y la baja aceptación por la sociedad.

La presente tesis queda estructurada en 5 capítulos que abordan la problemática de la fragmentación desde diferentes escalas. Los capítulos 1 y 2 se basan principalmente en buscar patrones y respuestas a una escala local atendiendo a redes de parches forestales. De esta manera se buscan influencias de la conectividad o aislamiento entre los parches así como respuestas de las especies. En el capítulo 3 se amplía la escala a un nivel territorial estudiando un conjunto de parches de diferente naturaleza y fisionomía, dispersos por todo el municipio, para definir la respuesta de la biodiversidad en función de sus características intrínsecas y extrínsecas. Por último, los capítulos 4 y 5 acercan el foco a una escala de detalle que permite estudiar los procesos de dispersión como interacciones ecológicas a nivel de parche forestal, buscando además medidas de gestión de potencial aplicación directa.



ESCALA TERRITORIAL

Capítulo 3. Estudio de parches forestales a nivel municipal

ESCALA LOCAL

Capítulos 1 y 2. Estudios de red de parches forestales cercanos

ESCALA DE DETALLE

Capítulos 4 y 5. Estudios de aspectos concretos a nivel de parche forestal individual

Los resultados del **capítulo 1** nos muestran que no existe una correlación entre el status legal de una mancha forestal y el valor de conservación de la misma. De esta manera se han obtenido valores de índices de conservación mayores en manchas forestales no protegidas o con una categoría de protección menor (municipal) que en otras con protección regional o superior. Se ha obtenido una relación negativa entre el valor florístico y la densidad de pinos, lo que nos hace pensar en una reducción del pinar para conseguir una formación forestal auto-sustentable. Sin embargo, esta relación negativa no se encontró para los índices de conservación de aves, por lo que existe una complementariedad entre índices que debería tenerse en cuenta en la protección y gestión de los fragmentos.

El **capítulo 2** trabaja a la misma escala, y los principales resultados obtenidos muestran un efecto diferente de la fragmentación por urbanización dependiendo del grupo biológico estudiado. En referencia a esto, la vegetación aparece como el grupo más vulnerable y sensible a la fragmentación. En el caso de las aves, los cambios en la composición de las comunidades y su diversidad ocurren como resultado del efecto borde creado por la urbanización. El efecto será diferente dependiendo de las características intrínsecas de cada parche. Por ello, a nivel de paisaje habría que identificar medidas de gestión que mitigasen o amortiguasen los efectos de la urbanización. Estas líneas de gestión deberían también de tenerse en cuenta a la hora de construir nuevas zonas urbanas que fragmenten el paisaje.

Escalando a un nivel más territorial, el **capítulo 3** nos muestra un mayor valor de conservación para la vegetación en parches de matorral mediterráneo que en zonas forestales. En el caso de las aves, dependiendo del ámbito geográfico-administrativo del índice de conservación considerado (nacional o Europeo) se obtienen unos resultados diferentes. Por ejemplo, zonas de fisonomía esteparia o de matorral obtienen mayores valores de conservación para índices europeos debido a la escasez de este tipo de paisajes en el territorio comunitario, y el consecuente mayor valor de las especies asociadas. Lo contrario ocurre con los índices de conservación nacional donde estos paisajes son más comunes y se valoran más las especies de carácter forestal. La mayor diversidad de especies aparece en parches de matorral arbolado. Sobre la base de todo ellos, se proponen medidas de gestión a aplicar a los parches forestales del Mediterráneo semiárido. Se apuesta por el clareado de pinos reduciendo su concentración en aquellos parches con altas densidades (normalmente inadecuadas para las condiciones climáticas imperantes). También se indica la necesidad de promover las interacciones naturales como medidas de restauración pasiva y como forma de conectar parches cercanos aislados. Además, se recomienda la protección de zonas no forestales (matorrales abiertos de aspecto estepario o matorrales arbolados) para completar la diversidad paisajística protegida bajo normativas locales o regionales.

Los capítulos 4 y 5 aproximan el foco a un nivel de mayor detalle. Buscan ahondar en el conocimiento del comportamiento de la dispersión en parches forestales y en el uso de esta interacción en la restauración de zonas degradadas o mejora de zonas naturales poco desarrolladas.

El **capítulo 4** se centra en conocer el funcionamiento de la dispersión dentro del parche y en el papel los ejemplares arbóreos de *Pinus halepensis* como perchas y posteriores plantas nodrizas de arbustos mediterráneos de frutos carnosos. Fijándose en *Asparagus albus, Rhamnus lycioides* y *Pistacia lentiscus* como especies de fruto carnoso, se observó un mayor consumo de frutos de *A. albus* que de las otras dos especies. Sin embargo, la dispersión dirigida hacia los pinos fue mayor para las semillas de *R. lycioides*. Por otro lado, más de la mitad de los arbustos adultos del fragmento forestal se encuentran bajo la influencia de la cobertura arbórea (mucho más si consideramos también los ejemplares arbóreos muertos o lo tocones). Este valor cobra más importancia si tenemos en cuenta que la cobertura arbórea del parche es muy escasa. Por lo tanto, si las condiciones de estos micrositios son buenas para la germinación y establecimiento de plántulas, de nuevo se apoya la reducción de la densidad de pinar pero dirigiéndola a aquellos ejemplares (entre los debilitados o moribundos) bajo cuya influencia aparezcan ejemplares de arbustos.

Por último, el **capítulo 5** se centra en estudiar más a fondo la dirección de la dispersión de semillas por aves, y en conocer el comportamiento de éstas en periodos limitantes de disponibilidad de frutos. Se observó que la producción y el consumo de frutos eran mayores en el interior del parche, e igualmente se vio una dispersión dirigida a lugares específicos dentro del propio parche. Con todo ello, se propusieron dos modelos de dispersión aplicables a parches forestales fragmentados en años o épocas con escasez de semillas. Un primer modelo propone un importante consumo interno por individuos ya establecidos en el parche (podrían ser los de mayor edad) y que no tienen necesidad de desplazarse fuera de los límites de la mancha. De esta manera, consumo y dispersión quedan concentrados en el interior del parche, y sólo los individuos no dominantes serían desplazados a la periferia o zonas cercanas pudiendo contribuir al transporte más lejano de semillas. El segundo modelo propone un comportamiento similar en el interior del parche, pero en este caso con una influencia importante de las zonas urbanas como fuente de recursos alimenticios y de agua. Por lo tanto, la dispersión dirigida hacia las zonas ajardinadas (que actuarían como sumideros) carecería de importancia ecológica y de igual manera se incrementaría el riesgo del transporte de propágulos de especies exóticas al interior del parche. Con esto, se propone de nuevo la reducción de la densidad de pinar en el interior del parche para contribuir a una dispersión más homogénea, y la instalación de perchas artificiales como atrayentes de aves en zonas periféricas para favorecer su restauración pasiva.

9.2 Conclusiones generales

Las conclusiones generales de la presente tesis se pueden dividir en conclusiones meramente científicas y extraídas de las correspondientes conclusiones de cada capítulo; y discusiones aplícadas y de gestión (recomendaciones/propuestas).

9.2.1 Conclusiones científicas

- En el sureste de la Península Ibérica (Mediterráneo Occidental) la interacción entre los desarrollos urbanos recientes y la fragmentación de los paisajes naturaleses uno de los principales determinantes de la biodiversidad forestal. Como resultado de un clima particular, una litología diversa, y los cambios en los usos del suelo que se han impuesto a la configuración biofísica natural, el paisaje ha evolucionado hacia un paisaje rural de gran riqueza ecológica y cultural.

- En las últimas décadas, muchos remanentes semi-naturales del paisaje han sido directamente ocupados por las actividades humanas (urbanizaciones, infraestructuras, agricultura intensiva), manejados con el fin de incrementar su cobertura forestal, o sujetos a efectos indirectos de la fragmentación. Esto ha casudao cambios en la composición local de especiesy variaciones en su distribución regional.

- Factores intrinsecos (tamaño, forma, tipo de hábitat y manejo histórico) y extrínsecos (principal agente fragmentador, usos de suelo adyacentes, conectividad) condicionan la biodiversidad de los parches forestales definidos en un sentido amplio como cualquier remanente de vegetación dominado por plantas leñosas. Estos parches pueden ser arbóreos, matorrales o mezcla de ambos, y con independencia de su origen (naturales o reforestados), el uso o la gestión.

- Estos remanentes normalmente aparecen como parches aislados, archipiélagos locales de parches o fragmentos marginales desconectados de la matriz forestal o del sistema montañoso. Estos parches son candidatos para formar parte del sistema municipal de áreas protegidas o espacios verdes (bosques urbanos).

- Aves y vegetación leñosa han sido utilizados como indicadores de biodiversidad en fragmentos naturales Mediterráneos. Aunque con diferencias, son grupos fácil de muestrear con un conocimiento de base, y proveen una información de gran valor para evaluaciones rápidas.

- Además, se ha encontrado una complementariedad entre índices de conservación de estos dos grupos nos aseguran una información multitaxón (incluyendo también otras variables de composición y estructurales del parche forestal) a la hora de abordar la gestión de estos paisajes transformados por el hombre.

- La falta de relación entre el nivel de protección legal de un parche y su valor de conservación, la heterogeneidad que aportan los mosaicos rurales que conservan parches forestales, la distribución de estos a lo largo de piedemontes o áreas montañosas, son aspectos relevantes para la conservación de la biodiversidad a escala local y regional.

- Las características internas de los parches forestales son también importantes para la biodiversidad interna de los mismos, independientemente de su tamaño, localización espacial o configuración paisajística. La corelación negativa entre el valor florístico y la densidad de pinar también sostiene la importancia del manejo interno del parche pudiendo explicar en gran medida la biodiversidad asociada a estos remanentes naturales.

- Estudios llevados a cabo en paisajes forestales semiáridos fragmentados desde un punto de vista de multiescala y multiespecífico, anotan diferentes efectos de la urbanización dependiendo del indicador de biodiversidad considerado.La vegetación aparece como más vulnerable debido a su alta sensibilidad a la fragmentación (baja movilidad y aparición de barreras en la dispersión). Para aves, se han visto cambios en la composición y diversidad como resultado del efecto borde y de las características intrínsecas de cada parche aislado (tamaño del parche, influencia urbana,…)

- Atendiendo a las escalas, una aproximación a nivel de paisaje, parche y microhabitat permite identificar medidas de manejo para mitigar el impacto de la urbanización y el establecimiento de líneas de gestión para preservar la biodiversidad en áreas naturales afectadas de manera directa por el crecimiento urbano. Considerando tal rango de escalas, es esencial asegura que, en un contexto de fragmentación urbana, los remanentes forestales puedan desarrollar todas sus funciones ecológicas, económicas y sociales.

- Las clasificaciones y las especies indicadoras organizan matorrales y áreas arbóreas en grupos separados representando diferentes niveles de afección urbana, con mayores valores de conservación vegetal en las áreas de matorral. El valor ornitológico cambia con el carácter Europeo o Nacional del índice de conservación. Zonas abiertas y de matorral obtienen mayores valores en los índices de conservación Europeos que en los Nacionales. Se han encontrado mayores diversidades de avifauna en parches alejados de los desarrollos urbanos o agrícolas intensivos.

- En las zonas Mediterráneas, la limitación hídrica confiere a la vegetación leñosa no arbórea una importancia capital. Así, los arbustos de frutos carnosos surten de recursos alimenticios a numerosas especies de vertebrados (especialmente a aves) siendo la clave en las relaciones mutualistas de interacción planta-animal. Exiten pocos estudios en zonas marginales Mediterráneas, así como en el Sureste Semiárido de la Península Ibérica, a pesar de su importancia para la dispersión de los arbustos Mediterráneos y la regeneración forestal.

- La hipótesis de que las aves dirigen la dispersión hacia los ejemplares de pinar en los parches forestales semiáridos viene apoyada por estudios que combinan el consumo de frutos y la lluvia de semillas. Sólo una pequeña fracción de las semillas consumidas son depositadas bajo la influencia de un ejemplar de pino, pero una alta proporción de arbustos adultos se encuentran bajo la copa del pinar, aunque estos pinos aparezcan de manera escasa. Por lo tanto, pareciera que los pinos proveyeran de un nicho favorable para los arbustos, al menos durante las fases de germinación y establecimiento.

- La cantidad de frutos disponibles durante los periodos invernales parece ser suficientes para la comunidad de aves. Aunque la producción y consumo de frutos dentro del parche forestal estudiado estaba dirigida hacia el interior del hábitat, la deposición de semillas fue mayor en la parte periférica, sugiriendo una tendencia de las aves de ingerir frutos en el interior pero defecar en las zonas más cercanas al borde. Dos modelos de dispersión espacial han sido propuestos para explicar las consecuencias de la dispersión de frutos y orientar el uso de esta interacción en la regeneración forestal.

9.2.2 Conclusiones aplicadas

Se pueden agrupar en tres grupos principales atendiendo a su aplicación en diferentes temáticas. Con ellos, aplicadas a los futuros desarrollos urbanísticos, se puede concluir:

- La distribución de las areas construidas debe ser planificada con extremo cuidado sobre todo cuando afecte a zonas naturales. Debe intentar mantener zonas buffer o de tampón para así minimizar el efecto borde. De la misma manera, se debería evitar el aislamiento total de parches y crear zonas permeables en los espacios urbanos.
- La adquisición de terrenos a través de medidas compensatorias derivadas de desarrollos urbanísticos debería ser una oportunidad para las autoridades públicas para conseguir localizaciones estratégicas que mejoraran la conservación de espacios limítrofes al desarrollo. De esta manera se priorizaría la creación de las zonas tampón ya comentadas y se mejoraría la conservación de los parches naturales.
- El diseño de los espacios verdes públicos dentro de las urbanizaciones debería abogar por el uso de especies nativas que mejoraran la permeabilidad ecológica y la conectividad, pudiendo actuar también como fuentes de propágulos para los hábitats naturales fragmentados. Además, invitar a la población de las urbanizaciones a que usen especies nativas en sus jardines privados pudiendo tener un efecto similar.

Con respecto a las políticas de reforestación y a las metodologías utilizadas, se puede concluir que:

- Se necesita un cambio en las políticas de financiación actuales y en las subvenciones e incentivos para las reforestaciones. Actualmente, las subvenciones para la reforestación de terrenos privados imponen el uso de especies arbóreas (con altas proporciones de especies del género *Pinus*). Esto sería aceptable en zonas más húmedas (como en zonas de clima mesomediterráneo), pero en terrenos semiáridos (termomediterráneo) se necesitaría un estudio de caso por caso.
- La presente tesis busca la incorporación en el manejo de formaciones forestales semiáridas de interacciones ecológicas como manera de optimizar recursos y reducir costes. A este respecto, el estudio preliminar de dispersión de frutos por aves como técnica de restauración pasiva ha ayudado a proponer algunas recomendaciones.
- Considerando que las aves frugívoras a menudo parecen dispersar semillas dentro del mismo parche en el que consumen frutos, la dispersión puede no ser efectiva para restaurar zonas periféricas degradadas. Sin embargo, si puede ser útil para la restauración interna del parche, como una manera de modificar el hábitat dominado por ejemplares arbóreos hacía una formación de matorral dominante.
- Los jardines privados de las áreas urbanas son muy atractivos para las aves frugívoras, lo cual puede producir un significante flujo de semillas entre la zona natural y los espacios privados ajardinados. Este hecho pone de manifiesto la importancia del uso de especies nativas en estos jardines.
- Crear conciencia sobre la importancia de los matorrales y bosques abiertos como el mejor representante de los paisajes Mediterráneos es necesario, comenzando con una

restauración firme y adecuada de las zonas forestales poco desarrolladas hacia estas formaciones más resilientes y biológicamente más ricas.

Finalmente, en relación al manejo interno de cada uno de los parches:

- En la presente tesis, se ha estudiado una amplia tipología de zonas forestales fragmentadas a diferentes escalas (desde el detalle hasta el paisaje), para así determinar el comportamiento de la biodiversidad y los factores ambientales en los mismos. Por lo tanto, una diagnosis minuciosa de cada parche forestal o zonas incorporadas al dominio público sería esencial para determinar las acciones correctas a realizar en cada caso.
- Idealmente, esta diagnosis debería ir acompañada de un plan de manejo y un biomonitoreo que evalue estos cambios. Todos estos aspectos deberían ser incorporados a los planes urbanísticos o desarrollados cuando se dé la adquisición de terrenos compensatorios por desarrollos urbanísticos.
- De acuerdo con estas recomendaciones, en todos los parches con alta densidad de pinar y con condiciones ambientales inadecuadas para su desarrollo, el número de ejemplares debería ser reducido. Esta acción debería ir dirigida a aquellos individuos débiles o muertos, especialmente a aquellos que hayan actuado como planta nodriza de arbustos mediterráneos, favoreciendo así la transición hacia una fisionomía de matorral.

Entre dos hileras de olivos,
bajo el abrasador sol de la mañana
y de un mochuelo de atenta mirada,
emprendí melodioso el camino.

Tierras de margas, calizas y esquistos,
plantas sedientas, aves que cantan,
almas repletas de calma,
pueblos de mar, monte y ramblizos.

Me sentí ufano;
he aquí ante el lugareño y el foráneo
la profunda belleza del paisaje Mediterráneo.

Carlos Martínez Hernández