UNIVERSIDAD POLITÉCNICA DE MADRID
ESCUELA TÉCNICA SUPERIOR DE INGENIERÍA DE
MONTES, FORESTAL Y DEL MEDIO NATURAL

**SCIENTIFIC BASIS FOR SUSTAINABLE
MANAGEMENT OF BARBARY SHEEP (*Ammotragus
lervia*) POPULATIONS IN PROTECTED AREAS OF
SEMIARID ENVIRONMENTS**

DOCTORAL DISSERTATION

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Ingeniero de Montes

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“Yo vengo de una raza de pastores que perdió su libertad cuando perdió sus ganados y sus pastos”

Julio Llamazares,

“Pursue some path, however narrow and crooked, in which you can walk with love and reverence”

Henry-David Thoreau

*“Ouço todos os dias,
De manhãzinha,
Um bonito poema
Cantado por um melro
Madrugador.
Um poema de amor
Singelo e desprendido,
Que me deixa no ouvido
Envergonhado
A lição virginal
Do natural,
Que é sempre o mesmo, e sempre variado.”*

Miguel Torga

ABSTRACT

Ungulate preferences for woody species and its implications for plant regeneration are key aspects to determine sustainability thresholds of ungulate populations and to define adequate management and conservation measures. This research aims to improve our understanding on **Barbary sheep** (*Ammotragus lervia*) interactions with vegetation in a protected area of a semiarid environment to provide new insights in the ecology and management of ungulate-dominated systems. The thesis is structured in four studies that provide valuable science-based data on the ecology and management of both the Barbary sheep populations and their habitat in SE Spain.

First, we analyzed plant preferences on widespread woody species. We found that browsing intensity on **widespread woody species** was low in broadleaved trees and negligible in conifers. Highly preferred and sparse shrub species were the most vulnerable and might be severely affected by medium to high Barbary sheep densities. **Second**, we compared the herbivore impact on widespread vs. threatened (rare) species and found that, contrary to our hypothesis, the herbivore impact on **threatened woody species** was significantly higher than on widespread woody plants, probably due to their overall higher palatability. The use of threatened woody plants revealed to be very useful as a warning signal of unsustainable densities of large herbivores. **Third**, we analyzed the effects of regeneration silviculture treatments on the Barbary sheep use, woody plant diversity and plant regeneration in pine (*Pinus* spp.) forests. We found that recruitment levels were significantly higher

in **regeneration silvicultural plots** as compared to intact (control) plots only for saplings and juveniles (>130 cm high). We did not find any clear patterns indicating that treated areas suffered heavier browsing damage across all woody plant species. Ungulates caused significantly greater rubbing damage in treated areas but did not use the treated areas more often than the control plots, indicating that small-scale treatments (≤ 0.75 ha) are compatible with low to moderate ungulate densities. **Fourth**, we examined the use of man-made structures (**supply points and firebreaks**) by Barbary sheep and analyzed the derived effects on vegetation (diversity, regeneration and fuel load). We found that ungulate habitat use decreased with longer distances to structures. Areas close to the structures (<50 m) showed greater plant damage (browsing, trampling and rubbing) and lower fuel load than areas located farther away (100 m). Firebreaks were less susceptible to ungulate damage as compared to supply points and, thus, an appropriate combination of both structures can help distribute more efficiently the ungulate effects throughout the system according to our management goals. The **integrative approach** of these four studies provides researchers and managers with a scientific basis towards sustainability, conservation and management of ungulate-dominated environments in a Mediterranean context where biodiversity enhancement, fire prevention and overall sustainability represent nowadays strong priorities.

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1. INTRODUCTION

1.1 Integrative approach

Ungulates are a diverse group of mammals widely distributed across the globe. Wild and domestic ungulates include ecologically and economically important species, some of them highly charismatic, whose preservation and sustainable use is desirable. Most ungulates are large-sized animals that feed on plant material (herbivores) and, therefore, play a crucial role on ecosystem functioning (Weisberg & Bugmann, 2003; Blondel, 2006; Royo *et al.*, 2010). They are considered both keystone and engineer species due to their disproportionate effect on the ecosystem relative to their abundance (Jones *et al.*, 1997; Rooney & Waller, 2003; Vázquez & Simberloff, 2003; Derner *et al.*, 2009) and their ability to modify and create ecological communities (Hobbs, 1996; Augustine & McNaughton, 1998; Gordon *et al.*, 2004; Beguin *et al.*, 2016).

In the early twentieth century, ungulate populations decreased dramatically (Ripple & Larsen, 2000; Osuna *et al.*, 2006; Haanes *et al.*, 2010). Both *in* and *ex situ* conservation measures such as introductions and re-introductions of ungulates, were implemented since then in order to face the rapid species extinction rates. This extinction phenomenon is expected to increase under the current climate change context (Pritchard *et al.*, 2012; Svenning *et al.*, 2016). Despite the increasing loss of ungulate species, overall wild ungulate populations have reached today the highest

densities ever recorded in many areas (Porter, 1992; Perea *et al.*, 2014), mostly as a result of rural abandonment and land use change (Rey-Benayas *et al.*, 2007; San Miguel *et al.*, 2010). For instance, deer (Cervidae) and wild boar (*Sus scrofa*, Linnaeus, 1758) numbers have increased exponentially in many European and North American countries (Côté *et al.*, 2004; Massei *et al.*, 2015). Only through an integrative scientific-based approach, we will be able to meet certain management objectives, which should prioritize the overall conservation of the system. In highly diverse and unpredictable systems such as the Mediterranean (Myers, 1990; Myers *et al.*, 2000; Blondel, 2006), this ultimate conservation target usually requires a wide array of management practices that go from the preservation of fragile and singular habitats/species to the prevention of animal diseases or destructive wildfires.

Both factors, the increased rates of wild ungulates and the ungulates “assisted migration or colonization” (McLachlan *et al.*, 2007; Hunter, 2007) may modify previous scenarios balance, creating new situations where ungulate-ecosystem relationships are unknown or unexpected. Even though ungulates represent a driving force for terrestrial ecosystem dynamics (e.g., affecting soil biogeochemical cycles or sustaining trophic chains), herbivory (i.e., ungulate feeding on the vegetation cover) might be the most important effect on ecosystems functioning (Nugent *et al.*, 2001; Danell *et al.*, 2003; Faison *et al.*, 2016a). These herbivore-vegetation relationships usually represent the baseline for the conservation of certain ecosystems, as herbivore impact on woody vegetation defines the ecological sustainability limit in

woody-dominated systems (Rooney & Waller, 2003; Morellet *et al.*, 2007; Gill & Morgan, 2010; Perea *et al.*, 2014, 2015).

Apart from understanding how ungulates influence trophic cascades, it is essential to analyse how wild ungulates interact with human activities in order to optimize production and reduce human-wildlife conflicts (Di Marco *et al.*, 2014, 2015; Svenning *et al.*, 2016). Human necessities modify management objectives, affecting the sustainability limit and the ecosystem structure and functioning (Mysterud, 2006). Therefore, the following three elements and the interactions between them need to be evaluated when defining sustainability thresholds: ungulate population requirements, vegetation conservation and human needs (Reimoser *et al.*, 1999; Beguin *et al.*, 2016).

An integrative scientific approach is proposed in the following chapters to evaluate the interaction among the target ungulate populations, the vegetation as a whole (including plant recruitment) and the conservation objectives of a Protected Area (Morellet *et al.*, 2007; Perea *et al.*, 2015).

1.2 Ungulate preferences for plant species

Ungulate feeding selectivity on grasses and woody species has been extensively studied in order to understand their interaction with vegetation composition and structure (Hobbs, 1996; Augustine & McNaughton, 1998) or their population productivity and behaviour (White, 1983; Gebert & Verheyden-Tixier, 2001).

Ungulate preferences for plant species explain these interactions partially (Forsyth *et al.*, 2002) and contribute to understand other herbivory determinants: 1) the ungulate diet type: browser/grazer proportion (Gordon, 2003; Codron *et al.*, 2007) or 2) the feeding behaviour: opportunist, specialist or generalist (Senft *et al.*, 1987; Freschi *et al.*, 2017).

Herbivore preferences for plant species is based on the relationship between plant consumption and plant availability (Manly *et al.*, 2002). Previous studies on woody species have shown that preferences depend mainly on ungulate species, not on population density (Fernández-Olalla *et al.*, 2006), although these preferences may be affected by different factors such as sex ratio, herbaceous resources availability or supplementary feeding (San Miguel *et al.*, 2000; Danell & Bergström, 2002) .

Spatial scale plays an important role in food selection (Senft *et al.*, 1987; Van Beest *et al.*, 2010). It has been suggested that deer select first the patch within the landscape according to the global patch characteristics and later they select individuals according to their specific preferences (Bee *et al.*, 2009). This feeding pattern, shown for other herbivores (Miller *et al.*, 2006), underlines the importance of the surrounding vegetation before selecting individual characteristics. Thus, plant recruits surrounded by more palatable species could have more opportunities to survive, according to Ward *et al.* (2008). However, Bee *et al.* (2009) suggest that opportunities are higher if surrounding vegetation creates an overall non-attractive patch for ungulates. In this sense, the role of non-preferred species as nurse plants for highly preferred seedlings and saplings has been shown in several studies in

Mediterranean environments (Castro *et al.*, 2004; López-Sánchez *et al.*, 2016; Ameztegui & Coll, 2015), specifically non-palatable evergreen shrubs of sufficient size (Perea *et al.*, 2016). Therefore, plant palatability and ungulate preferences play a significant role in facilitating or limiting the survival and growth of some species over others, contributing to changes in vegetation structure and composition (Gill & Beardall, 2001; Didion *et al.*, 2009).

Temporal scale also conditions resource availability and feeding selectivity of ungulates. Intra- and inter-annual variation in food resources contributes to explain the importance of browsing as a major feeding strategy in Mediterranean systems since green grass availability is primarily restricted to spring and autumn (Bugalho & Milne, 2003; Schoenbaum *et al.*, 2017). Nevertheless, browsing intensity can be higher in other seasons, depending on other intrinsic factors of the ungulate population. For instance, a study on juniper (*Juniperus oxycedrus* subsp. *macrocarpa*) herbivory by deer showed unexpected lower browsing levels in summer in Doñana, which might be attributed to the stag movements when searching for more productive habitats (Muñoz-Reinoso, 2016).

1.3 Herbivory and plant regeneration

The plant regeneration process comprises several stages. All of them are necessary to successfully complete the regeneration cycle that ends up with new individuals (Schemske *et al.*, 1994; Jordano & Herrera, 1995; Wang & Smith, 2002). This

process allows the forest perpetuation and it plays an important role in the existence and persistence of each forest species within the ecosystem (Gil *et al.*, 2010). Its success has been defined as “*the minimum seedling density of different height classes that is required to reach a new stand compatible with specific management objectives*” (Beguin *et al.*, 2016).

Regeneration is affected by both intrinsic species characteristics and extrinsic environmental factors. In sexual reproduction, abiotic (light, water, temperature, etc.) factors play a crucial role although some particular processes involved in the regeneration cycle, such as pollination, may require the participation of biotic interactions (e.g., plant-animal mutualisms). Biotic factors are particularly relevant after fruit and seed set when multiple biotic agents interact through key stages and processes such as seed dispersal, seed predation, seedling establishment and survival, for instance through zoochory, granivory and herbivory (Jordano, 2000; Perea *et al.*, 2011). Among them, herbivory has been pointed out as an extensive and determinant process that may limit recruitment when occurring at high intensity (Henkin *et al.*, 2005; Lecomte *et al.*, 2017). However, herbivores do not select all plants or plant parts equally. In fact, herbivores have a selective foraging behaviour that can strongly modify the quantity, the quality and the distribution of fruits, seeds, seedlings and saplings. Therefore, herbivory is a key factor that regulates the recruitment, especially of the most preferred plant species (White, 2012), conditioning plant succession (Muñoz-Reinoso, 2016) and even woody plant composition and diversity (Perea *et al.*, 2017).

Ungulate browsing on fruits and seeds has been proven to reduce seed productivity. For instance, in Mediterranean gum cistus shrub (*Cistus ladanifer* L.) stands, where predation by deer, mice and insects were studied, 42.3% of the plant reproductive structures were removed by ungulates (Lecomte *et al.*, 2017). Ungulate browsing also reduces plant regeneration capacity by decreasing the leaf area (Rhodes *et al.*, 2017) and by modifying seed dispersal and seed deposition through endozoochory or ectozoochory (Manzano & Malo, 2006; Albert *et al.*, 2015). Strong endozoochory effects on seed dispersal has been appointed in several wild ungulate populations, such as white-tailed deer (Niederhauser & Matlack, 2017), roe deer, wild boar or red deer (Pellerin *et al.*, 2016). In red deer populations, endozoochory was more conspicuous for grasses and small herbs in certain studies (Gill & Beardall, 2001), although it was also important for the dispersal of some Mediterranean woody species (Malo & Suárez 1996, 1998; Perea *et al.*, 2013).

Light and water availability are considered two main limiting factors for seedling and sapling survival in Mediterranean environments (Sack, 2004; Gómez-Aparicio *et al.*, 2006), together with their resistance to nutrient shortages, wildfires and herbivory damage (Rodríguez-Calcerrada *et al.*, 2008a,b; Perea & Gil 2014; López-Sánchez *et al.*, 2016). Although herbivory causes important damages, it also contributes to regeneration success in certain cases, as it has been reported in pine populations where heavy browsing on plant competitors benefitted the study pines (Osem *et al.*, 2011). Browsing effects on seedling and sapling distribution and survival has been extensively documented (Castro *et al.*, 2004; Pérez-Ramos & Marañón, 2008;

Beguin *et al.*, 2016). Several studies have shown that browsing caused more important effects on saplings as compared to seedlings (Gill & Morgan, 2010; Kuijper *et al.*, 2010). This greater ungulate effect on saplings was also found for other sources of damage such as rubbing, a less studied process (Massei & Bowyer, 1999; Castro *et al.*, 2004; Charco *et al.* 2016).

The intensity and extent of ungulate damage on seeds, seedlings and saplings are highly dependent on ungulate number of species or ungulate densities. The diversity of ungulate species showed to affect woody systems regeneration, probably due to the lack of functional redundancy among herbivore species (Faison *et al.*, 2016b; Bakker *et al.*, 2016). Similarly, moderate ungulate densities seem to allow seedlings survival (Mitchell & Kirby, 1990) but high ungulate densities show negative effects over recruitment, e.g.: seed production (Lecomte *et al.*, 2016), basal regrowth of oaks (Henkin *et al.*, 2005) or oak saplings (Ruiz-Mirazo & Robles, 2012). It seems that only when ungulate densities are low or intermediate, vegetation management (e.g., through silviculture) is able to ensure recruitment of palatable tree species (Beguin *et al.*, 2016). In fact, some studies highlight that extensive grazing at sustainable densities is a plausible management to improve shrub diversity in Mediterranean environments (Mendes *et al.*, 2015). Thus, managers should look for appropriate indicators of browsing and grazing sustainability in ungulate-dominated environments to ensure forest persistence and vegetation conservation (Morellet *et al.*, 2007; Perea *et al.*, 2015).

1.4 Indicators of sustainable/unsustainable ungulate populations

Sustainability of ungulate populations can be estimated through multiple indicators. Most indicators are based on animal health and population dynamics (Seddon *et al.*, 2007; Ewen *et al.*, 2012) and very rarely on other variables beyond the target animal species (Morellet *et al.*, 2007; Perea *et al.*, 2015). Some authors consider that sustainability approaches should account for other ecosystem components such as ungulate top-down effects on ecological communities (Leibold *et al.*, 1997; Martin *et al.*, 2010). These ecological effects are usually easy to monitor and provide more accurate and objective values (Fernández-Olalla *et al.*, 2006; Garzón-Machado *et al.*, 2010; Kuijper *et al.*, 2010; Perea *et al.*, 2014, 2015). Therefore, indicators that are sensitive to increasing ungulate pressure and affect the ecosystem as a whole (e.g. herbivory) may represent useful tools to address sustainability. For instance, highly preferred woody species can help establish sustainability thresholds based on the overall browsing damage and/or their regeneration probability (Fernández-Olalla *et al.*, 2006; Perea *et al.*, 2015). Nevertheless, sustainability indicators should also consider other variables, such as the management system (Bernués *et al.*, 2005) or the interactions with other conservation/management objectives (Papanastasis *et al.*, 2017).

According to Dale and Beyeler (2001), adequate indicators are those that easily capture changes in complex systems but remain sufficiently simple to be routinely

monitored. Therefore, an integrative sustainability indicator would involve the use of appropriate ecological parameters that fully evaluate the interaction between the target ungulate populations and the ecosystem as a whole (Morellet *et al.*, 2007; Perea *et al.*, 2015). In systems dominated by woody plants such as forests, woodlands or shrublands, adequate indicators of ecological sustainability should account for the persistence (e.g., regeneration success) of the woody plant diversity (Perea *et al.*, 2015). At high densities, selective foraging by large herbivores can limit the recruitment of the most preferred woody species (Miranda *et al.*, 2011; White, 2012; Perea *et al.*, 2014). In addition, heavy browsing on endemic and threatened woody taxa could, eventually, contribute to local extinction and the loss of both functional and species diversity, leading to irreversible ecological and conservation consequences (Olden, 2006; Rooney, 2009; Perea *et al.*, 2014). Therefore, special attention should be paid to threatened or rare taxa due to their sensitivity and vulnerability to possible local or regional extinction. A comparison between widespread vs. rare species indicators should be also evaluated in order to determine the costs and benefits of using one or another.

1.5 Ungulate-adapted forest management

The current high ungulate populations in most temperate areas, together with the increasing anthropogenic pressure on forests, demand multifunctional management approaches, capable of creating synergies among different objectives and policies

(Brun, 2002; McGranahan & Kirkman, 2013). Forest persistence in Mediterranean systems requires that forest management is coordinated with wild ungulate management and wildfire preventive silviculture to achieve sustainability (Morellet *et al.*, 2007; Mason *et al.*, 2010; Heinze *et al.*, 2011; Perea *et al.*, 2015).

Ungulate and forest regeneration management have evolved in parallel to the increasing understanding of biotic interactions and the results of the adapted management practices. Ungulate effects should go beyond simple “game damages” (Reimoser *et al.*, 1999; Weisberg & Bugmann, 2003), in order to develop new synergistic ungulate-adapted silvicultural practices (Kuijper, 2011; Apollonio *et al.*, 2010; Burney & Jacobs, 2013; Edenius *et al.*, 2014; Faison *et al.*, 2016a). Ungulate effects on tree regeneration are among those more extensively studied (Danell *et al.*, 2003; Kuijper *et al.*, 2010; Perea & Gil, 2014; López-Sánchez *et al.*, 2016; Beguin *et al.*, 2016). Therefore, ungulate-adapted silvicultural practices may create the “oportunity window” for seedlings and saplings that are strongly affected by browsing (Reimoser, 2003; Didion *et al.*, 2009) or by rubbing (on young and mature trees), a less studied effect in Mediterranean environments (Nielsen *et al.*, 1982; Ramos *et al.*, 2006; Gerhardt *et al.*, 2013; Charco *et al.*, 2016).

Regeneration silviculture is usually based on the creation of regeneration gaps in the forest canopy that affect partially shelter availability and habitat heterogeneity -two of the most important factors in explaining herbivore distribution (Gordon *et al.*, 2004)-. Forest gaps seem to be attractive to ungulates (Wattles & DeStephano, 2013), as the number, diversity and growth rate of seedlings and saplings is higher than in

the surrounding canopy (Kuijper *et al.*, 2009; Royo *et al.*, 2010). However, lower forage quality within the forest gaps could be compensated by a greater volume of lateral branches (Edenius *et al.*, 1993, Hartley *et al.*, 1997). In these gaps, the herbivores effects are varied and related to plant location within the gap or its surroundings (Kellner & Swihart, 2016). However, we still lack understanding of how regeneration silvicultural treatments affect ungulate use, browsing and rubbing damage and overall plant regeneration and diversity. Further studies are therefore needed to improve the current knowledge on silvicultural practices to establish appropriate ungulate-adapted management measures.

Browsing is, in fact, the most studied inference process of ungulates over the understorey vegetation after silvicultural practices (Gill, 1992; Weisberg & Bugmann, 2003; Vospernik & Reimoser, 2008). Several studies have analyzed the effects of browsing on tree regeneration to assess forest management (Zamora *et al.*, 2001; Weisberg & Bugmann, 2003; Perea *et al.*, 2016; Faison *et al.*, 2016a). Browsing over tree regeneration was found to be highly dependent on: 1) the treatment in the canopy (e.g. logging), 2) the stem density/height in the stand and, 3) the species palatability (Heikkila & Harkonen; 1996; Ameztegui & Coll, 2015; Faison *et al.*, 2016a). To reduce browsing pressure on young trees, different techniques have been tested, for instance, using felled pines as supplemental food (Månsson *et al.*, 2010). In this regard, the presence of shrub cover plays a crucial role in facilitating tree recruitment in Mediterranean environments (Pugnaire *et al.*, 1996;

Callaway & Davis 1998; Gómez-Aparicio *et al.*, 2008; Perea & Gil, 2014) and warrants further research within the scope of forest silviculture and management.

Silviculture for wildfire prevention has multiple aims that may affect forest structure and composition differently from those practices applied in regeneration silviculture. Therefore, coordination among all management objectives is required to achieve an integrative objective that meets the implementation of multifunctional management. Silviculture for wildfire prevention attempts to reduce woody fuel load and create vertical and horizontal discontinuity in forest structure. In this sense, silvicultural practices can be enhanced by ungulate browsing/grazing effects over vegetation (Lovreglio *et al.*, 2014), for instance by reducing understory woody phytovolume in forests (Mancilla-Leytón *et al.*, 2013) or firebreaks (Ruiz-Mirazo *et al.*, 2011).

Firebreak networks are common wildfire silvicultural measures to reduce wildfire spread risk and to facilitate extinction works (Agee *et al.*, 2000). They modify forest species composition and structure, as well as the distribution of ungulates by changes in food and shelter availability (Andreassen *et al.*, 2005; Miller *et al.*, 2009). Wildfire prevention techniques should consider that areas subject to grazing or browsing show a decrease in phytovolume, vegetation flammability or even annual costs as compared to manual clearing of firebreaks in ungrazed areas (Mancilla-Leytón *et al.*, 2013). This herbivory effect on firebreaks has been documented and analysed in Mediterranean forests, either by wild herbivores (Dufour-Dror, 2007) or livestock (Robles *et al.*, 2009; Ruiz-Mirazo *et al.*, 2011; Ruiz-Mirazo & Robles, 2012; Bianchetto *et al.*, 2015; Bashan & Bar-Massada, 2017). Due to their linear design,

firebreaks behave as barriers for wildfires, but also as corridors that facilitate terrestrial animal movements (Kuefler *et al.*, 2010) and build ecological networks throughout the forest (Joubert *et al.*, 2016). As potential diversity connectors, previous studies have shown that firebreaks are used by frugivorous mammals and birds, enhancing shrub species seed dispersal and the following recruitment and establishment along their verges (Suárez-Esteban *et al.*, 2013a, Suárez-Esteban *et al.*, 2013b). Although large ungulates avoided firebreaks or dirt tracks for defecation, the response has shown to be species dependent (Suárez-Esteban *et al.*, 2016). Further studies are therefore needed to clarify the use of firebreaks by wild ungulates and their effect on the surrounding matrix (e.g. browsing damage, regeneration abundance, forest structure, fuel load, etc.) particularly as the distance to the structure increases.

Importantly, one of the most common management measures in ungulate-dominated environments consists of supplementary feeding, water provision or addition of mineral blocks in the so-called supply points. It is therefore essential to understand how wild ungulates make use of these supply points and the derived effects on the system (soil, vegetation structure and diversity) to distribute and use them efficiently and in a sustainable manner. Previous studies have shown that different attributes of the supply points (e.g., size, time of use, quality) and their distribution along the landscape may affect ungulate habitat use, foraging behavior and sociobiology (Sahlsten *et al.*, 2010). Supply points are indeed attracting points that, when widely

and homogenously distributed in the landscape, can contribute to reduce ungulate damage on highly sensitive species, areas or growth stages (Månsson *et al.*, 2010).

The spatial distribution of supply points can enhance or reduce ungulate damage on forest vegetation, included browsing, rubbing or trampling damage as it has been appointed for grass-dominated open spaces within the forest (Smith *et al.*, 2007; Perea *et al.*, 2014; Månsson *et al.*, 2015). However, ungulate damage might reach unsustainable levels around these points if their nutritional balance is not considered (Timmons *et al.*, 2010; Miranda *et al.*, 2015; Felton *et al.*, 2017) or if it causes heavy browsing, trampling or rubbing pressure on certain woody plants (Cooper *et al.*, 2006; Timmons *et al.*, 2010). High concentration of wild ungulates can also facilitate the spread of diseases (Sorensen *et al.*, 2014) - also to domestic livestock (Carrasco-Garcia *et al.*, 2016) - and increase inter and intra-specific competition, resulting in stress and poorer body conditions (Ossi *et al.*, 2017).

Therefore, it is imperative to evaluate the actual use of different supply points by wild ungulates and determine the effects of wild ungulate use on plant damage, fuel load and woody plant regeneration and diversity. Particularly interesting is the analysis of how ungulate use and effects vary along a distance gradient from the supply points in order to improve our understanding on the differential ungulate effects at increasing distances from management infrastructures. All this will contribute to provide scientific-based guidelines on the multipurpose management of protected areas and in most ungulate-dominated systems.

1.6 Wild ungulate populations in protected areas

Protected areas (12.7% of the planet's land surface area) are effective tools to preserve species and habitats worldwide (Geldmann *et al.*, 2013), showing higher biodiversity levels than in non-protected areas (Gray *et al.*, 2016). Their resilience is clearly linked to their socioeconomic environments (Cumming *et al.*, 2015; Ens *et al.*, 2016), where thorough planning and management is essential to coordinate human activities and conservation goals. This coordination requires a more integrative approach that should be science-based to assure its effectiveness (Apollonio *et al.*, 2017). For instance, D'Amico *et al.* (2016) showed that optimizing the overall road network within Doñana Natural Park could restore almost entirely (91%) the potential habitat quality for red deer and wild boar at a Regional scale, whose presence probability had been reduced 40% for red deer and 55% for wild boar.

The lack or absence of ungulate natural predators in many protected areas has strongly modified ungulate behaviour, losing fear to humans (Brown *et al.*, 2012) and requiring careful management to avoid overabundance, human conflicts or important impacts on natural vegetation persistence (Nugent *et al.*, 2011; Found, 2017). Sometimes, human activity within a protected area deteriorates its biodiversity or results detrimental for certain species. For instance, Fernández & Gómez (2012) showed that preserving grazing as a traditional use in a protected area

increased ungulate damages on *Erysimum popovii*, balancing out any positive effects of the protected area on pollination.

The positive effects of protected areas on wild ungulate conservation seem to be unclear, although antipoaching measures showed positive effects (Geldmann *et al.*, 2013). Administrative borders modify ungulate behaviour (Benoist *et al.*, 2013) but may be insufficient and less important than coordinating management inside/outside the protected area (Meisingset *et al.*, 2018). Management coordination and long term monitoring of ungulate distribution are even more essential in migratory ungulates (Thirgood *et al.*, 2004; Western *et al.*, 2009; Anderwald *et al.*, 2015).

Therefore, managers, researchers and policy-makers should carefully consider and monitor wild ungulate populations in protected areas to assure the conservation of the whole system. Overabundant population may endanger other protected or non-protected organisms and may limit or modify important ecological processes that are essential for the preservation of all biodiversity levels (alfa, beta and gamma) and facets (taxonomic, genetic and functional diversity). Similarly, the absence or low levels of herbivores may contribute to the loss of protected and ecologically-important habitats, for instance EU grasslands of Natura 2000 Network (San Miguel *et al.*, 2010), or create a depauperate trophic chain with the loss of important ecological processes and interactions such as seed dispersal, prey-predator interactions or herbivory (Bartuszevige & Endress, 2008; Mattioli *et al.*, 2011; Moleón *et al.*, 2014; Gennet *et al.*, 2017). This doctoral thesis takes place in a semi-

arid protected area of SE Spain (Sierra Espuña Natural Park, Murcia province) and will hence focus on the long-term conservation of woody plant diversity in relation to wild ungulate populations.

1.7 The case of Barbary sheep in Sierra Espuña Natural Park

Barbary sheep or aoudad (*Ammotragus lervia* Pallas, 1777) are large caprin ungulates (males up to 160 kg of body weight; females up to 90 kg), native to large areas of semi-arid and arid mountain regions of northern Africa. The species is threatened as a vulnerable species by the IUCN (Cassinello *et al.*, 2008) and was introduced in 1970 in Sierra Espuña Natural Park (Murcia province) with the main purpose of adding game diversity (San Miguel *et al.*, 2011). It was introduced given the ecological resemblance (semiarid climate and mountainous topography) between Sierra Espuña and the Barbary sheep native areas (Valverde, 2004; San Miguel *et al.*, 2011). It is therefore considered an exotic ungulate species in Sierra Espuña and throughout Spain (Piñero & Luengo, 1992; Acevedo *et al.*, 2007; Garzón-Machado *et al.*, 2010). Some authors argue that wild ungulate populations, well-adapted to the target environment (e.g., Barbary sheep in semiarid conditions) may represent an opportunity to take up the ecological role played by the disappearing livestock (San Miguel *et al.* 2011). However, specific studies are needed to address the effects of introduced wild ungulates on the environment (e.g., conservation of woody vegetation). In addition, further research is necessary to establish sustainable ungulate densities and appropriate management techniques that ensure biodiversity

conservation. More details on Barbary sheep ecology and behavior as well as Sierra Espuña description and characterization can be found in sections 3.1 and 3.2 (Study area and species, respectively).

1.8 Research motivation and justification

Risks that threaten natural systems with ungulate populations are higher in semiarid environments, where water and vegetation cover are scarce and expected to be reduced under the current climate change context (Duncan *et al.*, 2012). This anthropogenic climate change is expected to favor an increase of semiarid conditions worldwide as a consequence of temperature increase and probable rainfall decrease (Huang *et al.*, 2016; Schlaepfer *et al.*, 2017), which will also modify ungulate effects on biodiversity (Araujo & Rahbek, 2006, Levinsky *et al.*, 2007). Ungulates could even increase negative climate-change impacts over vegetation in dry lands, depending on the herbivore species or the plant-herbivore shared history (Maestre *et al.*, 2016).

Protected areas usually show higher biodiversity levels (Gray *et al.*, 2016) and their resilience is directly related to their socioeconomic environment (Cumming *et al.*, 2015). Therefore, their conservation and management aims require multifunctional and integrative approaches that should be science-based to assure its effectiveness (Apollonio *et al.*, 2017). Within this context, understanding the effects of wild herbivores and their associated management practices (silviculture, supply points,

etc.) on plant composition, vegetation structure and regeneration growth is essential to define adequate management practices towards the persistence of woody systems and the overall conservation of ungulate-dominated areas.

In addition, it is imperative to address the role played by introduced ungulates such as the Barbary sheep in environments previously dominated by livestock grazing. Such is the case of many areas in the Northern Hemisphere where rural abandonment is occurring at increasing rates (Caballero *et al.*, 2009; San Miguel *et al.*, 2010; Navarro & Pereira, 2015). It is therefore far from clear whether introduced wild ungulate populations can take up or even improve the ecological role traditionally played for millennia by the disappearing livestock or, conversely, it represents a serious threat for biodiversity conservation. The context of this research is particularly timely given the current expansion of wild ungulates across the globe and the increasing disappearance of extensive livestock rearing. The case of the introduced Barbary sheep in Sierra Espuña Natural Park represents a unique opportunity to address these questions given their long-term effects (since 1970) on the system and the possibility of evaluating different management practices (e.g., regeneration silviculture, supply points, firebreak network) in an ungulate-dominated area.

2. OBJECTIVES

The overall aim of this thesis is to provide scientific basis for sustainable management of Barbary sheep populations in a protected semiarid environment. In particular, this research thesis aims to understand Barbary sheep (*Ammotragus lervia*) interactions with woody vegetation to develop sustainability thresholds and define adequate management measures. The integrative approach of this research pursues multiple objectives within a multifunctional management framework (Fig. 2.1), including the overall preservation of woody systems (shrublands, forests and woodlands) and the implementation and use of appropriate management structures (firebreaks and supply points) and practices (regeneration silviculture and supplementary feeding).

Fig. 2.1 Diagram of the integrative approach proposed for this research thesis

The general aim includes the following specific objectives (Fig. 2.2):

- (i) Define Barbary sheep browsing preferences for widespread plant taxa and evaluate its impact on the conservation of semiarid woody plant communities.
- (ii) Examine the effects of an introduced population of Barbary sheep on the threatened woody taxa and compare them with those obtained for widespread taxa in order to establish possible indicators of sustainable/unstable ungulate populations.

Fig. 2.2 Summary of the specific objectives pursued in this research thesis. Each objective will be fully developed in chapters 4-7, respectively and a final discussion (chapter 8) will relate and integrate the main results.

- (iii) Analyze the effects of Barbary sheep populations on woody plant communities (diversity, structure and regeneration) following silvicultural treatments (reproductive fellings) of the main tree species (*Pinus* spp., *Quercus* spp. and *Juniperus* spp.).
- (iv) Investigate Barbary sheep habitat use and effects on vegetation (plant damage, fuel load, diversity and regeneration) along a distance gradient from management structures: firebreaks and supply points (water, salt and food supply points).

3. MATERIAL AND METHODS

3.1 Study area

The work was carried out in Sierra Espuña Regional Park, a 17,804 ha protected area located in southeastern Spain (Murcia province; 2°4'-2°14' N, 37°47'-37°57' W). The elevation varies between 300 and more than 1,500 m above sea level. The climate is Mediterranean semiarid, with long summer droughts (>4 months). Annual rainfall is usually low, between 200 and 500 mm with strong seasonal and annual variability. Temperatures are warm, without frosts (thermo-Mediterranean thermotype), in the lower areas, but relatively cold in winter in the upper slopes and summits (supra-Mediterranean thermotype), where continentality is high and frosts are frequent between November and February. The lithological substrate is basic, usually limestone, dolomite or marle. The vegetation is dominated by pines (*Pinus halepensis* Mill., *P. pinaster* Ait., *P. nigra* Arn.), holm oak (*Quercus rotundifolia* Lamk.), kermes oak (*Quercus coccifera* L.), shrub-like junipers (*Juniperus phoenicea* L., *J. oxycedrus* L.), with extensive patches of evergreen shrubs (*Rosmarinus officinalis* L., *Lithodora fruticosa* (L.) Griseb, *Thymus vulgaris* L., *Cistus* sp.). Xerophytic perennial grasslands, mostly dominated by *Macrochloa tenacissima* (L.) Kunth or *Helictotrichon filifolium* (Lag.) Henrard, are also widespread all over the area. The upper parts are covered by low cushion-shaped thorny scrub formations under scattered pines and junipers. The vegetation shows high levels of biodiversity, as a consequence of both the high number of vascular species and the high rate of endemisms (Alcaraz *et al.*, 2008). It also includes some protected habitat types included in the European Natura 2000 Network, such as

natural and seminatural grasslands (Habitat codes 6110, 6220, 6170, 6420), Mediterranean scrubs (1430, 4090, 5210, 5330) and forests (9340, 9540) (Alcaraz *et al.*, 2008).

Fig. 3.1 Location of the study area (Sierra Espuña)

3.2 Study species: *Ammotragus lervia* (Pallas, 1777)

Taxonomically, the Barbary sheep or aoudad (*Ammotragus lervia* Pallas, 1777) is the only species in the genus *Ammotragus* Blyth, 1840. It is included in the Caprinae subfamily, showing a phylogenetic position intermediate between sheep and goats but closer to goats. In fact, *Ammotragus* has been appointed as an ancestor of *Capra* genus (Casinello, 1998).

Currently, there are six subspecies with different distributions along Northern Africa, although a close extinct species, *Ammotragus europaeus*, was present in southern France and southeastern Spain (Andalusia and Murcia) 1.2 million years before

present, in the inferior Pleistocene (Moullé *et al.*, 2004). At present, several ex-situ areas (outside their natural range) support populations of Barbary sheep, mostly in Spain and North America (Cassinello *et al.*, 2008; IUCN 2015), where they were introduced in the past with the main goal of increasing sport hunting diversity (Gray, 1985; Cassinello, 1998; San Miguel *et al.*, 2010).

The Barbary sheep is a medium sized ungulate (males up to 160 kg of body weight, and 100 cm of wither height, and females up to 90 kg body weight and 90 cm wither height in the study area). Barbary sheep are adapted to water shortage and show generalistic/opportunistic feeding habits, eating grasses, forbs or shrubs according to the seasonal availability but preferring grasses (Cassinello, 1998; San Miguel *et al.*, 2010; Miranda *et al.* 2012). In the Canary islands, important browsing damages on native flora species have been appointed, maybe due to a lack of green grasses most part of the year (Luengo & Piñero, 1992; San Miguel *et al.*, 2011).

Barbary sheep prefer mountainous areas from 200 m to 4100 m (Cuzin, 2003). In previous studies, the species selected areas with low winter precipitation regime, high altitudes, high slopes and forests (Cassinello *et al.*, 2006; Acevedo *et al.*, 2007). Habitat preferences and groups dynamics vary depending on the season. According to Johnston (1980), the Barbary sheep prefers woodylands in summer, grasslands in autumn and winter and protective rocky slopes in spring, the main breeding season. Regarding group dynamics, out of the rutting season (September-December), male groups (5-15 individuals) are separated from groups constituted by females and

juveniles, usually bigger (15-50 individuals). Adult males join female groups in the rutting season and form groups of 100 (150) individuals in the study area (Cassinello, 2000; San Miguel *et al.*, 2010).

The Barbary sheep, as other bovidae species, reaches sexual maturity when males and females are 18 months old approximately (San Miguel *et al.*, 2011), although sexual maturity can be reached at 9 months for males and 14 months for females (Cassinello, 1997). Its mean gestation period is 160 days, similarly to goat and sheep. Females can give birth twice a year and twins are rather frequent (25-38% in the study area), also triplets (2% in the study area) (García, 2001; García-Morell, 2005). These data show the great growth potential of Barbary sheep populations, what could explain partially their expansion in southeast Spain, that was studied in order to prevent potential conflicts with other ungulate native species, as Iberian ibex (*Capra pyrenaica*), or endemic flora (Cassinello *et al.*, 2006; Acevedo *et al.*, 2007; Miranda *et al.*, 2012).

Barbary sheep are an example of a threatened large ungulate (caprid), whose native populations in Northern Africa are currently decreasing (Cassinello *et al.*, 2008; IUCN, 2015). However, their off-site populations in southeast Spain expanded from Sierra Espuña Natural Park through Murcia and other regions (Alicante, Almería or Granada) to a large extent because of introductions and translocations (Cassinello, 2017). It has been appointed that these off-site populations could serve as ex-situ preserves for the species (Valverde, 2005), especially when an important population

decline (>10%) is expected in its native habitat over the next 15 years (Cassinello *et al.*, 2008; IUCN, 2015). Today, there is a strong social and scientific debate whether to exterminate some introduced ungulate species or maintain them in low density populations as they may also provide some important socioeconomic resources (e.g. ecotourism and hunting; San Miguel, 2011). In addition, it has been argued that low populations of large herbivores, including Barbary sheep, could take up some ecological roles traditionally played by the disappearing livestock (goats and sheep), favouring the conservation of some protected grassland communities (San Miguel *et al.*, 2010). Previous research shows contrasting results on the effect of Barbary sheep over its habitat. For instance, Sánchez-Zapata *et al.* (2010) showed its role as a source of food for other protected species and Sánchez-Gómez *et al.* (2011) suggested heavy browsing in the study area as one of the main possible risks for Mediterranean threatened woody species.

3.3 Data collection

Some studies of this research thesis have a common methodology for the data collection of some variables. We describe in this section general methods and material regarding the following variables that are found in Chapters 4-7: Plant availability; browsing damage; browsing preferences; other damages; ungulate relative density; regeneration level; forest structure; understory structure and physical environment.

Table 3.1. Data collected by study and the corresponding chapter. The plus sign (+) indicates the variables collected in each study: Plant availability (AV); Browsing damage (BD); Other damages (OD); Browsing preferences (BP); Ungulate relative density (UD); Regeneration level (RL); Forest structure (FS); Understory structure (US); Physical environment (PE)

Study	Chapter	AV	BD	BP	OD	RL	UD	FS	US	PE
Widespread vegetation	4	+	+	+			+			+
Threatened flora	5	+	+	+	+	+	+			+
Regeneration silviculture	6	+	+	+	+	+	+	+	+	+
Firebreaks/supply points	7	+	+	+	+	+	+		+	+

Plant availability

To estimate woody plant availability, we recorded the abundance of all woody plant species in each plot (Table 3.2). The area sampled in each plot was 78.5 m² (a 5m radius circular plot) that corresponded to the minimum area concept to survey woody plant communities in our study area (Braun-Blanquet, 1951). The abundance of each species was estimated with the Braun-Blanquet abundance scale (Braun-Blanquet 1951; Guisan & Harrell 2000), using the ground cover percentage of each plant species in the plot. The relative availability of each species in each plot was calculated by dividing its average canopy cover by the sum of those of every species present in the plot.

Browsing damage

Browsing impact on each woody species was estimated by analyzing browsing intensity using a 6-rank (0-5) degrees methodology (Table 3.2): 0=No browsing evidence; 1=Light browsing, with <10% of the twigs browsed; 2=Low browsing, with 10-30% of twigs browsed; 3=Intense browsing (30-60% of the twigs); 4=Heavy browsing (>60% of the twigs), with clear modification of plant shape; 5=Maximum browsing (>90% of the twigs browsed). Only woody cover lower than 2 m high (accessible to the ungulate) was considered. More details about this 0-5 rank can be found in Perea *et al.* (2015). Unsustainable browsing was defined when browsing intensity was >3 since it clearly limits regeneration success and plant growth (Perea *et al.*, 2014, 2015).

Browsing preferences

Browsing preferences were studied by comparing the utilization of each woody species with its availability through an adaptation from the forage ratio, generally called the selection index (Manly *et al.*, 2002), using the following expression (Fernández-Olalla *et al.*, 2006; Perea *et al.*, 2014, 2015):

$$w_{ij} = \frac{o_{ij} * p_{ij} / \sum_{i=1}^n o_{ij} * p_{ij}}{p_{ij} / \sum_{i=1}^n p_{ij}} = \frac{o_{ij} * \sum_{i=1}^n p_{ij}}{\sum_{i=1}^n o_{ij} * p_{ij}}$$

where w_{ij} : forage ratio or preference (selection) index for the species i in the j plot; o_{ij} : browsing degree of the species i in the j plot; p_{ij} : ground cover percentage of the species i in the j plot, and n : number of woody species present in the j plot. The final preference index for each species was the average of those calculated for every plot where the species is present.

Other damages

Other damages different from browsing, as trampling and rubbing (presence-absence data) were recorded for each species. Trampling was identified when the vegetation was knocked down with no sign of browsing or rubbing. Rubbing damage was recorded when the trunk or the main stem was damaged (usually with scars) or debarked.

Regeneration level

Additionally, in each plot we also recorded natural regeneration of all woody species through a 4-rank degree [0 = no seedlings were found; 1= few isolated seedlings were found (<10); 2= numerous isolated seedlings were found (>10), and 3 = very abundant regeneration, high-density patches of seedlings].

Ungulate relative density

To estimate the habitat use (relative density) of Barbary sheep in each of the previous plots (5 m-circular) we used pellet counting (Table 3.2). Only faecal pellet groups

containing six or more pellets were recorded following Smart *et al.* (2004a) and Acevedo *et al.* (2010).

Table 3.2 Criteria used to collect woody plant utilization and availability in the sample sites

Score	Availability	Utilization: Browsing degree 6-rank classification	Barbary sheep relative abundance index
0	0 < Ground cover < 10% Occasional presence	No browsing evidence	Total absence of traces of Barbary sheep presence.
1	0 < Ground cover < 10% Presence not occasional	Light browsing, with <10% of the twigs browsed	Slight traces of Barbary sheep presence. Few and scattered droppings.
2	10 < Ground cover < 25%	Low browsing, with 10-30% of twigs browsed	Clear, although scant, traces of Barbary sheep presence: 2-3 groups of droppings.
3	25 < Ground cover < 50%	Intense browsing (30-60% of the twigs)	Clear, but not abundant, evidences of Barbary sheep presence: 4-5 groups of droppings and usually one or more trails.
4	50 < Ground cover < 75%	Heavy browsing (>60% of the twigs), with clear modification of plant shape	Abundant evidences of Barbary sheep presence: 5-10 groups of droppings and usually several trails.
5	Ground cover > 75%	Maximum browsing (>90% of the twigs browsed)	Abundant evidences of excessive Barbary sheep density: more than 10 groups of droppings, several trails. Usually some bare soil due to trampling.

Forest structure

To characterize the forest structure we recorded the DBH (diameter at 130 cm height) of all standing trees in a 10 m-radius circular area, taken from the center of the plot to calculate Basal Area. Young plants of pines, holm oaks and shrub-like junipers were quantified in a concentric 5 m-radius area and classified in 5 categories according to height and DBH: 0= annual sprout; 1= plant height <30 cm; 2= plant

height between 30 and 130 cm; 3= plant height >130 cm and DBH<2.5 cm; 4= plant height>130 cm and DBH of 2.5-7.5 cm.

Understory structure

To estimate the understory structure we also measured plant height for each woody species in each plot. That way, plant height and ground cover (see above) determined understory structure.

Physical environment

Slope direction (aspect) and slope steepness (inclination) of each plot was also recorded using an accurate GPS (GARMIN etrex ® 10, 2014), digital topographic maps (1:25000), ortophotos (50 cm resolution) and Geographical Information Systems (software QGis 2.18). Digital maps and ortophotos were obtained from the Regional Geographic Institute (www.murcianatural.org).

3.4 Data analysis

All the analyses were performed using the R programming environment (R Core Team 2016; <http://www.r-project.org/>). All the statistical analyses are explained in chapters 4-7 in detail. The main statistical models used in each study are summarized in Table 3.3.

Table 3.3 Models that best fitted the data in each Chapter indicating the response variables, the predictors, the variables included in the random part of the model and the model type. Models were classified as: LMM: Linear Mixed Model; CLMM: Cumulative Link Mixed Model; GLMM: Generalized Linear Mixed Model and GLM: Generalized Linear Model.

Chapter	Response variable	Model type	Predictors	Random
4. Widespread vegetation	Preference index	LMM	Ungulate relative density	Site
			Plant species	
			Ungulate relative density x Plant species	
	Browsing damage	CLMM	Ungulate relative density	Site
			Plant species	
			Preference classes	Site
5. Threatened woody flora	Browsing damage	CLMM	Preference index	Site
			Plant cover	
			Ungulate relative density	
	Regeneration occurrence	GLMM	Preference index	Site
			Plant cover	
			Ungulate relative density	
	Regeneration abundance	CLMM	Preference index	Site
			Plant cover	
			Ungulate relative density	
6. Regeneration silviculture	Conifers Young Plants	GLMM	Treatment	Site
			Tree species	Years after treatment
			Treatment x Tree species	
	Browsing damage	CLMM	Treatment	Site
			Tree species (<i>P. halepensis</i>)	Years after treatment
			Tree species (<i>Q. ilex</i>)	
			Treatment x Tree species (<i>P. halepensis</i>)	
			Treatment x Tree species (<i>Q. ilex</i>)	
	Rubbing damage	GLMM	Treatment	Site

3. Material and methods

			Tree species	Years after treatment
			Treatment x Tree species	
	Ungulate relative density	GLMM	Treatment	Site
	Shannon applied to DBH classes	GLMM	Treatment	Site
	Variance of plot-level DBH	GLMM	Treatment	Site
	Shannon applied to tree regeneration categories	GLMM	Treatment	Site
	Shrub cover percentage	GLMM	Treatment	Site
	Shannon index	GLMM	Treatment	Site
	Species richness	GLMM	Treatment	Site
7. Firebreaks/ Supply points	Ungulate relative density. Supply points	GLMM	Distance (10-50-100m)	Site
	Ungulate relative density. Firebreaks	GLMM	Distance (10-50-100m)	Site
	Browsing damage. Supply points	CLMM	Distance (10-50-100m)	Site/Plant species
	Browsing damage. Firebreaks	CLMM	Distance (10-50-100m)	Site/Plant species
	Trampling/rubbing damage. Supply points	GLMM	Distance (10-50-100m)	Site/Plant species
	Trampling/rubbing damage. Firebreaks	GLMM	Distance (10-50-100m)	Site/Plant species
	Species richness. Supply points	GLMM	Distance (10-50-100m)	Site/Plant species
	Species richness. Firebreaks	GLMM	Distance (10-50-100m)	Site/Plant species
	Phytovolume. Supply points	GLM	Distance (10-50-100m)	
	Phytovolume. Firebreaks	GLM	Distance (10-50-100m)	
	Regeneration occurrence. Supply points	GLMM	Distance (10-50-100m)	Site/Plant species
	Regeneration occurrence. Firebreaks	GLMM	Distance (10-50-100m)	Site/Plant species
	Regeneration abundance. Supply points	CLMM	Distance (10-50-100m)	Site/Plant species
	Regeneration abundance. Firebreaks	CLMM	Distance (10-50-100m)	Site/Plant species

4. BARBARY SHEEP PREFERENCES FOR WOODY WIDESPREAD SPECIES

Based on: Fernández-Olalla, M., Martínez-Jáuregui, M., Perea, R., Velamazán, M., San Miguel, A. 2016. Threat or opportunity? Browsing preferences and potential impact of *Ammotragus lervia* on woody plants of a Mediterranean protected area. *Journal of Arid Environments*, 129, pp. 9-15.

(Annex I)

Ungulate browsing on woody vegetation is strongly determined by animal preferences and plant palatability (Forsyth *et al.*, 2002; Danell & Bergström, 2002; Fernández-Olalla *et al.*, 2006; Perea *et al.*, 2014). Browsing damage can be used as an estimator of plant consumption and, thus, can be used to determine preferences as the ratio between food consumption and food availability for each plant species (Manly *et al.*, 2002). In addition, browsing damage *per se* can be a suitable proxy to define the ecological sustainability for wild ungulate populations, particularly in systems dominated by woody plants (Bianchetto *et al.*, 2015; Perea *et al.*, 2015). In the following chapter, we analyze Barbary sheep browsing damages on woody species in the study area in order to understand its preferences for widespread woody plants and, therefore, evaluate vegetation conservation status and propose plant indicators for the conservation of woody plant communities, according to the first objective defined in Chapter 2. Here we also describe material and methods that are specific to this study and, therefore, are not included in Chapter 3.

4.1 Specific materials and methods

Data collection

Sampling consisted of 45 field surveys. As major plant communities are distributed

according to altitude and slope direction and thus to thermotypes (thermo-, meso-, and supra-Mediterranean) in the study area (Alcaraz *et al.*, 2008), field surveys evenly covered the whole range of altitude and slope directions represented in the study area (Table 4.1). Each survey was a 5-m radius circle (see Chapter 3). The variables examined in each survey were woody species cover, browsing damage and Barbary sheep relative density. The collection of those variables is shown in the general Material and Methods described in Chapter 3. The calculation of the forage ratio (i.e., preference index) is also shown in Chapter 3.

Table 4.1: Distribution of number of field surveys by altitude class and estimated thermotype (Alcaraz *et al.*, 2008). %: Percentage of the total area of the Regional Park corresponding the each altitude class. N: Number of field surveys located on each altitude class. %sample: Percentage of field surveys for each thermotype.

Altitude (m)	Thermotype (estimated)	%	N	%Sample
260-700	Thermo-Mediterranean	30	12	26.6
701-1200	Meso-Mediterranean	59	25	55.6
1201-1580	Supra-Mediterranean	11	8	17.8

Data analysis

An average preference index was calculated for every species present in our field surveys. However, statistical inference was carried out only with those woody species present in a minimum of 10 surveys (19 species, see Tables 4.2 and 4.3). We

carried out two separate analysis for the dependent variables “preference index” (model 1) and “browsing intensity” (model 2), respectively.

On the basis of “preference index”, we estimated classes of species similarly preferred by Barbary sheep. In addition, we analysed whether Barbary sheep preferences and “browsing intensity” depended on the Barbary sheep relative density. All analyses were performed with R 3.1.2 (R Core Team, 2014).

First, we fitted a linear mixed effect model with normal error distribution to explain the variable “preference index” using “nlme” library (Pinheiro *et al.*, 2015). Fixed effects were plant species and Barbary sheep relative density index within each plot and its first order interaction. The random-effect part of the model entered via plot effects on the intercept. We did model simplification by removing predictors or interaction terms with no significant explanatory power. To achieve the minimal adequate model we compared the models using Akaike's Information Criterion (AIC), considering only the most supportive models, those with $AIC \leq 2$ (Burnham & Anderson, 2002). We continued simplification with random variables and found that the best-fitting model did not include the random structure, so we used a generalized linear model with normal error distribution and identity link in the simplest model. Subsequently, post-hoc analyses using Duncan multiple range test and library “agricolae” (de Mendiburu, 2014) were carried out to define groups of species similarly selected by Barbary sheep (i.e., preference classes). To study whether the

browsing intensity depended on Barbary sheep density and species we fitted a Cumulative Link Mixed Model (CLMM) with a logit link function where the response variable was browsing intensity (ordinal variable), using “ordinal” library (Christensen, 2015). We used the same independent variables as in model 1. Similarly, we fitted another CLMM using the preference classes instead of the species as explanatory variable. The best model was chosen following the AIC criteria explained above.

4.2 Results

Ninety-two woody species (12 trees, 37 shrubs, 40 low shrubs and 3 vines) were recorded in our field surveys (Table 4.2). None of them was listed in the Habitat Directive. Conifers were represented by 2 shrub and 4 tree species. Twenty-seven species were present only in one survey while 19 were present in 10 or more (Table 4.2). Twenty-five species (31.5%) showed no evidence of having been browsed by the Barbary sheep while 16 (17.4%) showed average unsustainable browsing levels. However, only three of them were present in more than 2 surveys. Only one species (*Tetraclinis articulata*) is endangered at a national scale, but was introduced by planting in our study area, where it is not native. Of all the endangered species at a regional scale, only two (*Fumana fontanesii* and *Rhamnus alaternus*) showed unsustainable browsing levels. However, the first species was present only in one survey. Two more endangered species at a regional scale (*Acer monspessulanum* and

Ephedra nebrodensis) showed a browsing intensity of 3.0; of all the rest, two showed a 2.0 degree and the rest were under 2.0.

Seventeen surveys had a Barbary sheep relative density (habitat use) of 1; 12 of 2, 6 of 3, 7 of 4 and 3 of 5 (Mean $\pm$ SE: 2.46 $\pm$ 1.37; CV = 0.56). The statistical inference was carried out on 19 species (Table 4.2; Table S1 in Annex V). The minimal adequate model to explain preference index was a linear model with species as the unique explanatory variable (Adjusted R-squared: 0.5475). Results showed that Barbary sheep density index was not statistically significant ($p > 0.05$), as well as neither the interaction term, which means that Barbary sheep preferences are not significantly affected by its stocking rate. Post-hoc test results show several solutions (Mean Square Error: 0.8783895, alpha = 0.05, see Table 4.3). Results classify species following a preference index order and clearly show that certain species such as *Lithodora fruticosa* and *Staelina dubia* are, each, a group by itself, probably as a consequence of their high preference indexes.

The best model explaining Barbary sheep browsing was a cumulative link mixed model (AIC = 591.46) where the factor preference class was statistically significant ($p < 0.01$ for each class). In addition, browsing intensity was significantly and positively affected by Barbary sheep density (Estimate = 1.0228, Std. Error = 0.1450, $p < 0.01$). Plot remained in the simplest model as a random variable (Variance = 0.5684, Std.Dev. = 0.7539). Abundant species with a high preference index, such as

4. Browsing preferences

Lithodora fruticosa, *Staezelina dubia*, *Rosmarinus officinalis* or *Quercus rotundifolia*, showed sustainable browsing levels (browsing intensity < 3).

Table 4.2. Average values of Availability (Canopy cover percentage), Utilization (Browsing intensity) and Preference Index (Forage Ratio) for woody species found in at least 10 field surveys. (N) Number of surveys in which the species is present. (C.S.) Conservation status: EN: Endangered; VU: Vulnerable; SI: Special interest; UUR: Utilization under regulation.

SPECIES	N	C.S.	Average canopy cover (%)	Average utilization (Browsing damage) (0-5)	Average preference index (0-5)	Preference index St. Dev
<i>Artemisia campestris</i> L. subsp. <i>glutinosa</i> (J. Gay ex Besser) Batt. Batt and Trab.	14		11.8	1.6	1.172	1.04
<i>Cistus albidus</i> L.	31		13.8	0.2	0.219	0.62
<i>Cistus clusii</i> Dunal	18		11.1	0.6	0.918	1.33
<i>Daphne gnidium</i> L.	14		3.6	0.1	0.066	0.25
<i>Dorycnium pentaphyllum</i> Scop.	16		4.7	0.2	0.175	0.49
<i>Genista valentina</i> (Willd. Ex Sprengel) Steudel	10		11.5	0.0	0.000	0.00
<i>Juniperus oxycedrus</i> L.	38	SI	13.0	0.1	0.069	0.25
<i>Juniperus phoenicea</i> L.	16	SI	12.2	0.0	0.000	0.00
<i>Lithodora fruticosa</i> (L.) Griseb	17		8.1	2.9	4.186	2.23
<i>Phlomis lychnitis</i> L.	16		5.3	0.3	0.305	0.97
<i>Pinus halepensis</i> Mill.	40	UUR	39.5	0.1	0.022	0.10
<i>Pistacia lentiscus</i> L.	18	UUR	7.5	0.2	0.175	0.47
<i>Quercus coccifera</i> L.	22	UUR	18.9	0.8	1.010	0.81
<i>Quercus rotundifolia</i> Lam.	17	SI	7.5	1.2	1.383	0.98
<i>Rhamnus lycioides</i> L.	16		5.6	0.4	0.315	0.53
<i>Rosmarinus officinalis</i> L.	41		30.3	1.5	1.883	0.97
<i>Satureja obovata</i> Lag.	10	UUR	4.5	0.9	0.690	0.98
<i>Staezelina dubia</i> L.	19		5.7	2.2	3.019	1.92
<i>Thymus vulgaris</i> L.	33	UUR	8.0	0.7	0.623	1.02

Table 4.3. Results of Duncan's multiple range test. Final preference classes of woody species for the Barbary sheep used in the analysis. Only species that have been found on 10 or more surveys are considered (a-g and 5-0 shows a decreasing order of preference)

SPECIES	Mean	Std. Dev.	Rank	Preference
				Classes
<i>Lithodora fruticosa</i>	4.186	2.2346	a	5
<i>Staehelina dubia</i>	3.019	1.9247	b	4
<i>Rosmarinus officinalis</i>	1.883	0.9697	c	3
<i>Quercus rotundifolia</i>	1.383	0.9768	cd	2
<i>Artemisia campestris</i>	1.172	1.0382	de	2
<i>Quercus coccifera</i>	1.010	0.8127	de	2
<i>Cistus clusii</i>	0.918	1.3279	def	1
<i>Satureja obovata</i>	0.690	0.9770	efg	1
<i>Thymus vulgaris</i>	0.622	1.0181	efg	1
<i>Rhamnus lycioides</i>	0.315	0.5330	fg	1
<i>Phlomis lychnitis</i>	0.305	0.9720	fg	1
<i>Cistus albidus</i>	0.219	0.6195	g	0
<i>Pistacia lentiscus</i>	0.175	0.4743	g	0
<i>Dorycnium pentaphyllum</i>	0.175	0.4889	g	0
<i>Juniperus oxycedrus</i>	0.069	0.2520	g	0
<i>Daphne gnidium</i>	0.066	0.2459	g	0
<i>Pinus halepensis</i>	0.022	0.0979	g	0
<i>Genista valentina</i>	0.000	0.0000	g	0
<i>Juniperus phoenicea</i>	0.000	0.0000	g	0

4.3 Discussion

The Barbary sheep forages on a very wide variety of woody species: from small chamaephytes to trees, although average browsing intensities are usually moderate. Only 10 species (10.2%) showed average browsing intensity above 3.0, which defines sustainability (Perea *et al.*, 2014, 2015). However, only four of them were present in more than two surveys, so further research would be needed to provide

solid inference on their individual response to Barbary sheep browsing. None of the abundant species (present in ≥ 10 surveys) showed unsustainable browsing, and only two of them showed moderate browsing intensity (≥ 2 ; Table 4.2). These results reveal no evidence of major problems regarding actual vegetation structure and composition in our experimental conditions. However, severe browsing damages have been observed on individuals of some sparse shrub species typical of late successional stages (Alcaraz *et al.*, 2008), such as *Rhamnus alaternus* and *Rubus gr. ulmifolius*. Therefore, further research is required on threatened taxa or scarce but highly preferred shrub species, to guarantee that there is no critical threat to any of them regarding sustainable stocking rates. Wild ungulate stocking rates should not result in the local extinction or serious degradation of any species (Fernández-Olalla *et al.*, 2006; Mysterud, 2006; Perea *et al.*, 2014, 2015), especially in protected areas.

Density of the Barbary sheep in the study area does not appear to be uniform (CV = 0.56), probably due to its different habitat preferences throughout the year. Maximum browsing intensity was found only in surveys with high values of Barbary sheep density, which should be avoided as much as possible through management.

Maximum browsing intensity (level 5) was found in some sites for several native (*Osyris quadripartita*, *Lithodora fruticosa*, *Thymus vulgaris*, *Medicago suffruticosa*, *Rhamnus alaternus*, *Rhamnus infectorius* and *Lonicera splendida*) and introduced species (*Prunus dulcis*, *Corylus avellana* and *Opuntia ficus-indica*). Browsing

damage to pines and junipers were zero or negligible, even in surveys with Barbary sheep density levels of 5. However, the cultivated *Prunus dulcis* was severely affected by browsing, even in two surveys with low Barbary sheep density levels (2). That situation demonstrates that conifers are highly rejected species for the Barbary sheep (damages are negligible even with the highest Barbary sheep density), that *Quercus rotundifolia* is a preferred species (damages depend upon Barbary sheep density) and that *Prunus dulcis* is a highly preferred tree species because it is severely browsed even with low Barbary sheep density. Indeed, browsing damages on this particular species result in serious conflicts with farmers from adjoining areas. However, *Olea europaea*, which is a highly preferred species for many wild and domestic ungulates, showed sustainable browsing, even in surveys with Barbary sheep density levels of 4. On the other hand, some highly preferred species (e.g. *Lonicera splendida*) seem to hide in (are facilitated by) other rejected woody species (e.g. *Juniperus phoenicea*) so their browsing damage is probably under-estimated in our study. The most preferred species were abundant and widespread chamaephytes (small shrubs), such as *Lithodora fruticosa* or *Stachelina dubia*, which do not seem to be severely threatened by the Barbary sheep (Table 4.2).

Two species (*Acer monspessulanum* and *Fumana fontanesii*) are seriously endangered in SE Spain (Murcia Region) but not at broader scales: the Iberian Peninsula and northern Africa, respectively. Although they were present only in one survey, both showed a rather high browsing intensity. The most endangered one,

Fumana fontanesii, a small shrub, might be endangered by Barbary sheep trampling. However, unpublished evidence suggests that it might also be favored, through endozoochorous dispersal by the Barbary sheep. *Acer monspessulanum* is a broad leaf, deciduous tree species that is not endangered at a regional (Mediterranean) level. Evidence from other studies suggests that its palatability might be similar to that of other deciduous trees (Perea *et al.*, 2015), so further attention should be paid to this species.

Some woody species of our study area are cultivated, the most abundant being *Prunus dulcis*, *Olea europaea* and *Opuntia ficus-indica*. *Opuntia* seems to be heavily browsed as a consequence of both its high water content and the palatability and nutritive value of its cladodes and fruits, which are available in summer, the worst season regarding high quality forage availability. Other plant species, such as *Citrus aurantium*, *Mespilus germanica*, *Corylus avellana* or *Arundo donax*, seem to have been heavily browsed only as a consequence of their occasional occurrence in surveys with very high Barbary sheep density.

Some shrub and tree species that are both scarce and representative of late successional stages show medium to high levels of browsing utilization. Therefore, their conservation status (following the Habitat Directive scores) could become unfavourable-bad should the Barbary sheep stocking rate increase over today's figures. That seems to be the case for *Acer monspessulanum*, *Arbutus unedo*, *Colutea*

arborescens, *Lonicera splendida*, *Olea europaea* subsp. *europaea* var. *sylvestris*, *Phillyrea angustifolia*, *Quercus faginea*, *Rhamnus alaternus*, *Rubus ulmifolius* or *Viburnum tinus*, among others. Therefore, a close and periodic monitoring of these particular species would be desirable to ensure sustainable stocking rates.

Preference index classes appear to be robust since they were not affected by Barbary sheep relative density. However, as might be expected, the observed browsing intensity of woody species seems to be positively related to Barbary sheep relative density indexes. That situation provides empirical evidence for the establishment of clear limits between sustainable and unsustainable browsing levels and, therefore, sustainable and unsustainable wild ungulate stocking rates.

Wild ungulate sustainable stocking rates for high nature value rangeland should not be based only on considerations about rangeland productivity or nutritive quality vs. ungulate nutritive requirements. Sustainable livestock stocking rates between 0.2 and 1.3 goats/ha/yr (fairly equivalent to 10-62 Barbary sheep/km²/yr) have been estimated for similar environments in SE Spain (Robles & Passera, 1995). However, regarding wild ungulates, special attention should be paid to their effect on the structure and composition of woody vegetation, especially in arid and semiarid natural environments (vs. agricultural land), where green grass is scarce or lacking throughout most of the year (San Miguel *et al.*, 2010). Damage on protected or scarce woody species could be regarded as a suitable threshold between sustainable

and unsustainable browsing (Perea *et al.*, 2014, 2015). Hence, possible unsustainable browsing damage could arise for those species that show both high preference index and low density. Furthermore, following Dale and Beyeler (2001), some woody species could be used as indicators should they fulfill two requirements: high preference index (since they are highly sensitive to browsing damage) and moderate abundance or wide distribution (since scarce species are difficult to find). In our study, species such as *Stachelina dubia* or *Quercus rotundifolia* could be suitable indicators of browsing impact.

The Barbary sheep appears to behave as a mixed-feeder (Hofmann, 1989; Dannel & Bergström, 2002) or grazer (Miranda *et al.*, 2002). It usually grazes but also browses on a great variety of woody species when green herbage is scarce or lacking. Unlike other native wild ungulates, such as red deer (*Cervus elaphus* L.) or Spanish ibex (*Capra pyrenaica* Schinz), which show a certain preference for late successional woody species (García-Gonzalez & Cuartas, 1989; Martínez, 2002; Fernández-Olalla *et al.*, 2006; Perea *et al.*, 2014, 2015), the Barbary sheep seems to behave as a generalist browser. Its feeding behavior seems to resemble that of the native “segureña” sheep breed, which usually feeds on early successional woody species, such as *Rosmarinus officinalis*, *Anthyllis cytisoides* or *Coronilla juncea* (Correal *et al.*, 2006). This foraging behavior is especially interesting in Mediterranean areas, since long summer drought, and sometimes winter cold, result in long periods of green herbage shortage and, thus, increase browsing intensity. Negative effects on

endemic woody legumes and other species have been described in the Canary Islands (Piñero & Luengo, 1992; Garzón-Machado *et al.*, 2010), allegedly because of the lack of green herbage under the Canary pine forests.

The Barbary sheep is undoubtedly an alien species in Spain. The decline of extensive small livestock rearing in southeastern Spain has left an almost empty (but necessary) ecological niche that could be partially occupied by the Barbary sheep, as the eminent Spanish naturalist and researcher Valverde (2004) pointed out when he recommended the introduction of the species in the study area. Furthermore, as a result of its mixed feeding behavior and its tolerance to hot, arid climates, the Barbary sheep could be better adapted to the predicted climate change (Araujo & Rahbek, 2006) in southeastern Spain than most native ungulates. As a complementary result, the Spanish population could improve the conservation status of the endangered Barbary sheep at a global scale. Therefore, the Barbary sheep might be considered as an opportunity for this particular region. However, it may not be suitable for other regions without arid or semiarid conditions where native wild ungulates are already present, or in areas of high endemism (e.g. Canary Islands) where its negative impact on endangered flora has been reported (Piñero & Luengo, 1992; Garzón-Machado *et al.*, 2010). However, it could also become a threat, should its populations become overabundant. Its effect on palatable and widespread woody species could be a suitable indicator of sustainability. A wise management of both the Barbary sheep's populations and its habitat (water sources, forests, rangelands and

grasslands) could help reduce potential problems and enhance synergies.

5. BARBARY SHEEP POPULATIONS AND CONSERVATION OF THREATENED WOODY FLORA

Based on:

Velamazán, M., San Miguel, A., Escibano, R., Perea, R., 2017. Threatened woody flora as an ecological indicator of large herbivore introductions. *Biodiversity and Conservation*, 26(4), pp.917-930. (Annex II)

Threatened woody species usually require a previous, correct risk analysis in order to define the specific conservation measures. Ungulate damages (e.g., rubbing, browsing and trampling), particularly at high levels, can limit plant regeneration and persistence of the most vulnerable and low resistant species. Some other extrinsic factors such as availability of preferred food (e.g. green grass), plant location (e.g. near attractive points for ungulates), plant palatability relative to the neighbor plants and rock cover (Zamora *et al.*, 2001; Miranda *et al.*, 2011; Perea *et al.*, 2015) can indeed affect overall damage on the threatened plant species. In this chapter we analyze damages of Barbary sheep on threatened woody flora populations and compare them with those obtained for widespread woody species to better estimate sustainable populations of Barbary sheep and establish sustainability thresholds in the study area, according to the second objective described in Chapter 2. We describe material and methods that are specific to this study and, therefore, are not included in Chapter 3.

5.1 Specific materials and methods

Data collection

We studied all woody species occurring in the study area (Sierra Espuña Regional Park) that were at highest risk according to the Regional Catalog of Protected Flora

(Order 50/2003, 30th of May; www.borm.es). Fourteen species were studied, 7 vulnerable and 7 in danger of extinction (Table 5.1). Using the protected vegetation cartography of the Regional Park, we selected between 3 and 11 locations of each threatened species, depending on their availability (abundance) in the field (Table 5.1). A total of 86 populations of threatened species were examined (Table 6.1). For each population, 3-5 individuals were randomly selected, recording the following information: plant height, diameter at breast height (dbh), debarking/rubbing damage (presence/absence) and browsing intensity. For plants below 1.3 m, it was recorded the basal diameter or the plant diameter in cushion-shaped shrubs. The field procedures to measure or estimate the study variables are described in the general material and methods (Chapter 3).

Data analysis

We used a Cumulative Link Mixed Model (CLMM) to analyse the effect of the study variables on the browsing intensity of the threatened woody species. CLMM's allow for regression methods similar to linear models while respecting the ordered categorical nature of our observations (Greene & Hensher, 2010; Christensen & Brockhoff, 2013). Browsing intensity for each plant species in each plot was the response variable (ordinal variable). Fixed effects (predictors) were relative habitat use of Barbary sheep (pellet count), preference index of each species and plant cover. Only woody cover lower than 2 m high (accessible to the ungulate) was considered.

Plot was included in the model as a random effect. The model was fitted by the Laplace approximation with a Probit link function, using the “clmm” function within the “ordinal” package (Christensen, 2013) for R 3.1.0 software (www.r-project.org). We used a model averaging approach to summarize all the competing models and to make stronger inference since model-averaged parameters behave in a more stable manner (Burnham & Anderson, 2002; Freckleton, 2011). We first fitted the maximal model, containing all the predictors. Then, we performed model comparison of all possible models by using the AIC weights. For model comparison we used the “dredge” function within the “MuMIn” package of R. Finally, we obtained the model-averaged coefficients as well as the importance of each predictor (from 0 to 1) by using the “model.avg” function of “MuMIn”. To establish a species ranking of conservation priority we defined categories (groups) of threatened species that were significantly different in browsing damage, using the command “relevel” for multiple comparisons of the R software.

To analyze the occurrence of natural regeneration of each protected woody species we used a Generalized Linear Mixed Model (GLMM). The response variable was binary (presence vs. absence of regeneration; binomial error distribution). Predictors were the same as in the model above: preference index, plant cover and habitat use (pellet counts) of Barbary sheep. Plot was also included in the model as a random effect. The model was fitted by the Laplace approximation with a logit link function according to our data properties (Bolker *et al.*, 2009), using the “glmer” function

within the “lme4” package for R 3.1.0 software. To present the results of the logistic regression for the significant predictors we created a combined graph (histograms plus logistic curves), following the recommendations of Smart *et al.* (2004b). To obtain these graphs we used the “plot.logi.hist” function from the “pobbio” library (De la Cruz Rot, 2005). Finally, to analyze the abundance of regeneration we used another CLMM. The response variable was abundance (scale 0-3; ordinal variable). The structure of fixed and random effects was the same as in the previous models. The model was fitted as the first CLMM (see above). We used the model averaging approach for both regeneration models (GLMM and CLMM) following the same procedures as in the first model.

5.2 Results

Seven threatened species (50%) showed unsustainable levels of browsing, with browsing intensity mean ≥ 4 (Table 5.1). Based on the differences in browsing damage across the threatened species we established a conservation priority rank, obtaining 8 different groups (Table 5.1). The group 1 comprises five species (35.7% of the total), representing the species with highest conservation priority (Table 5.1).

Eight species (57.1%) showed no evidences of natural regeneration in, at least, half of their populations (Table 5.1), with very low regeneration abundances that varied from 0 to 0.66 in the 0-3 scale (Table 5.1). Four species (28.6%) showed higher evidences of natural regeneration (>50% of the populations), with average

abundances varying from 0.50 to 1.75 in the 0-3 scale (Table 5.1). Finally, the regeneration of two dwarf species could not be assessed due to difficulties in distinguishing life stages (adults vs. sprouts; Table 5.1).

Ten threatened species (71.4%) showed evidences of rubbing/debarking damage (Table 5.1). Only three of them (21.4%) were frequently affected, with more than 10% of their individuals showing clear damage (Table 5.1). *Sorbus aria* was, by far, the most affected plant species, with 25% of the individuals showing rubbing or debarking damage (Table 5.2).

The mean preference index across all threatened species was 3.10 ± 1.96 (Table 5.1). Preference index was the most important variable in explaining browsing intensity (Table 5.2). Thus, higher preference index (i.e. higher preference of the threatened species with regard to the accompanying species in the sampled population) was associated to higher levels of browsing (Table 5.2; Fig. 5.1). Plant cover was also an important and significant factor (Table 5.2). Sites with lower plant cover showed higher browsing degrees (Table 5.2; Fig. 5.1). Relative use (pellet count) of Barbary sheep was the least important factor and only showed a marginally significant effect on browsing intensity (Table 5.2).

5. Threatened woody flora

Table 5.1. Summary of the results for the threatened woody species and the impact of Barbary sheep on them.

Plant species	PL*	N**	Browsing intensity (Mean±SE)	CPR (§)	Preference index (Mean±SE) †	Natural regeneration*** (%)	AA (0-3)	RD (%)‡
<i>Acer monspessulanum</i>	EN	10	5.00±0.00	1	2.74±0.71	10.0%	0.10	2.1
<i>Coronilla glauca</i>	V	3	5.00±0.00	1	5.60±1.15	28.6%	0.57	9.7
<i>Cotoneaster granatensis</i>	EN	7	5.00±0.00	1	3.49±0.93	33.3%	0.66	4.3
<i>Erica erigena</i>	EN	3	2.67±0.88	5	1.44±0.50	66.6%	1.33	0.0
<i>Fraxinus angustifolia</i>	EN	6	5.00±0.00	1	3.51±0.94	50.0%	0.50	11.4
<i>Fumana fontanesii</i>	EN	4	3.75±0.25	4	8.63±3.80	100.0%	1.75	16.2
<i>Genista longipes</i>	V	3	1.67±0.88	7,8	0.92±0.46	33.0%	0.33	6.2
<i>Phillyrea media</i>	EN	3	4.50±0.50	2	1.85±0.46	0.0%	0.00	0.0
<i>Prunus prostrata</i>	V	6	1.17±0.40	8	2.12±0.82	-	-	4.5
<i>Quercus faginea</i>	V	4	4.25±0.48	3	3.05±0.96	75.0%	1.25	3.6
<i>Salix pedicellata</i>	V	7	1.71±0.49	6,7	1.02±0.20	72.7%	1.27	7.9
<i>Sorbus aria</i>	EN	11	3.73±0.29	4	1.99±0.41	42.8%	0.43	25.0
<i>Thymus serpylloides</i>	V	10	2.00±0.33	6	4.15±1.25	-	-	0.0
<i>Ulmus glabra</i>	V	5	5.00±0.00	1	2.86±0.87	20%	0.20	0.0

(*) Protection Level (PL): EN=endangered; V=vulnerable.

(**) Number of populations for each threatened species.

(§) Conservation Priority Rank (CPR) based on significance differences in browsing levels; 1-8=Highest-lowest priority according to study data and analysis; Species with the same number showed no significant differences.

† Values >1 indicates positive selection with regard to accompanying woody plants in each population.

(***) Percentage of populations with evidences of natural regeneration (%) and (AA) average abundance (scale 0-3). Species with symbol (-) indicates that regeneration (sprouts) could not be clearly distinguished from adults.

‡ Debarking/Rubbing damage (RB) Percentage of all individuals with rubbing damage

Table 5.2. Summary of the CLMM model averaging (n=7 models) to analyse the factors affecting browsing intensity by the Barbary sheep. Predictors are shown in a decreasing order of relative importance. (*) Relative importance represents the sum of the AIC weights across all the models where the fixed effect occurs, with values from 0 (minimum importance) to 1 (maximum importance).

Predictors	Relative Importance*	Averaged Estimate	Standard Error	z-value	P-value
Preference index	1.00	0.684	0.050	13.54	<0.001
Plant cover	0.93	-0.020	0.009	2.08	0.038
Habitat use (pellet counts)	0.89	0.058	0.033	1.73	0.083

Fig. 5.1. Relationship of browsing intensity with plant cover (left) and preference index (right). Browsing intensity is given in an increasing 0-5 scale. Each box shows the median (band in the middle of the box) and the first and third quartiles (edges). Whiskers represent the lowest and highest datum within the 1.5 interquartile range of the lower and upper quartile.

Fig. 5.2. Fitted logistic regression curves showing the predicted probability (left axis) and observed frequency (right axis) of plant regeneration for an increasing habitat use of Barbary sheep. The histograms represent the observed data (1 = presence of sprouts; 0 = absence of sprouts) and the line shows the predicted probability. Habitat use is given in number of pellet groups per plot (=78.5 m²).

Relative habitat use (pellet count) by Barbary sheep was the most important variable in explaining both occurrence (GLMM model) and abundance (CLMM model) of regeneration, with relative importance values of 0.96 and 0.98, respectively (Table 5.3). Thus, habitat use by Barbary sheep showed statistically significant effects on both occurrence and abundance of regeneration (Table 5.3). Lower habitat use by Barbary sheep increased the predicted probability of natural regeneration (Fig. 5.2). Populations with 0-5 pellet groups per plot (~0-6 pellet groups per 100 m²) showed twice higher predicted probability of regeneration than populations with 10-15 pellet

groups (~13-19 groups per 100 m²; Fig. 5.2) and 5-fold higher probability than populations with 20-25 pellet groups (~26-32 groups per 100 m²; Fig. 5.2). Finally, preference index and plant cover were factors of lower importance, with no significant effects on either the occurrence or the abundance of natural regeneration (Table 5.3).

Table 5.3. Summary of the model averaging ($n=16$ models) to analyse the factors affecting both the occurrence (GLMM) and the abundance (CLMM) of natural regeneration for the studied threatened species. Predictors are shown in a decreasing order of relative importance. (*) Relative importance represents the sum of the AIC weights across all the models where the fixed effect occurs, with values from 0 (minimum importance) to 1 (maximum importance).

Response variable	Model type	Predictors	Relative Importance*	Averaged Estimate	Standard Error	z-value	P-value
Presence of regeneration	GLMM						
		Habitat use (pellet counts)	0.96	-0.107	0.053	13.54	0.043
		Preference index	0.28	0.008	0.048	0.18	0.854
		Plant cover	0.27	-0.0001	0.005	0.03	0.974
Abundance of regeneration (scale 0-3)	CLMM						
		Habitat use (pellet counts)	0.98	-0.054	0.017	3.25	0.001
		Plant cover	0.31	0.009	0.003	0.28	0.781
		Preference index	0.27	-0.001	0.022	0.06	0.952

5.3 Discussion

The results reveal that 50% of the threatened plant species showed what we classified as unsustainable levels of browsing, with 35.7% of the species under the highest possible level of browsing. These values contrast with those obtained for common species in the study area, using the same methodology (Fernandez-Olalla *et al.*, 2016), where only 6.5% of the species showed unsustainable values (≥ 4) and none showed the highest level of browsing. Contrary to our hypothesis, the herbivore impact on threatened woody species was significantly higher than that found on common and widespread woody species. Here we showed that the conservation status of some threatened species, based on damage and regeneration survival, clearly represents a bottleneck for the regeneration and ecological sustainability of the system. The impact of browsing on common vs. rare taxa may differ considerably and, therefore, threatened and protected taxa should be especially considered when monitoring the effect of introducing dominant herbivores. Effects of large herbivores on common plant species should be therefore taken with caution since they could be low or negligible but may mask unsustainable densities for some rare and threatened flora taxa or protected habitats. This differential effect on threatened vs. common taxa may be related to the differences in the rarity or the preference index (palatability). Overall, for all threatened species, the mean preference index was 3.10 (Table 5.1), much higher than that found for common species (0.85; Fernández-

Olalla *et al.* 2016). This higher preference for threatened taxa might be related to the fact that most threatened species are late-successional plant species, usually with less aromatic and secondary compounds than early-successional plants. This agrees with previous studies in the Mediterranean region where other large herbivores (deer) also showed higher preferences for late-successional woody plants (Perea *et al.* 2014).

Results also reveal an important limitation to consider when using large herbivore introductions as a conservation tool: the effect of the target threatened species on other protected or threatened organisms. This is particularly remarkable for dominant herbivores in areas with high and endemic plant diversity such as the Mediterranean basin (Myers, 1990). Although the study area is probably one of the most suitable sites for introducing and preserving the threatened Barbary sheep in Europe, due to its location and relatively similar semi-arid ecological conditions, further monitoring of plant populations is required in order to achieve a sustainable conservation program that also preserves other legally protected and threatened organisms. In fact, the use of surveying threatened woody taxa through ecological indicators (e.g. herbivore damage and regeneration success) could indeed serve as a suitable partial proxy for the assessment of the overall ecological sustainability of woody systems under ungulate populations pressure. In that way, the applicability of assessing sustainability by surveying threatened or rare woody flora could be extended to any systems dominated by woody plants such as forests, shrublands or woodlands.

Barbary sheep is clearly an exotic species in SE Spain. However, it may play an important ecological and conservation role (San Miguel *et al.*, 2010). This includes the conservation of protected grassland communities by replacing the roles traditionally played by the disappearing livestock (San Miguel *et al.*, 2010), the reduction of wildfire risk in these Mediterranean fire-prone areas (Ruiz-Mirazo *et al.*, 2011), the conservation of threatened ungulates with a decreasing distribution range (Valverde, 2005; Fernandez-Olalla *et al.*, 2016) or its role as a source of food for other protected species (Sánchez-Zapata *et al.*, 2010). In fact, this ex-situ population could provide individuals for future reintroductions in suitable areas of Northern Africa and could also serve as an important genetic resource.

Examples of recently extinct wild herbivores have shown the importance and the cost of losing genetic diversity (Pérez *et al.*, 2002; Frankham, 2005; Folch *et al.*, 2009). Here we show how to improve the ecological sustainability in the long term: low ungulates densities and appropriate habitat management practices that ensure the conservation of the natural vegetation, including the most sensitive (threatened) woody plant species. However, in other areas with high woody plant endemism and no historical presence of large herbivores (e.g. Canary Islands –Macaronesian region-) the impact of exotic ungulates such as Barbary sheep, even at low densities, may cause an important biodiversity loss (Piñero & Luengo, 1992; Garzón-Machado *et al.*, 2010). Thus, selection of potential areas for ex-situ introductions of large herbivores should clearly consider evolutionary history and plant endemism.

5.4 Conservation and management implications for large herbivore introductions

By assessing the herbivore damage and the probability of natural regeneration we were able to obtain a conservation priority ranking. The ranking provided would give an idea of what species should be monitored regularly and undergo specific protection and management measures first. Populations at highest risk (first priority; Table 5.1) should be therefore strictly protected from browsing (e.g. by fencing) to ensure their viability and, hence, avoid possible local extinction.

The results also showed that plant cover and preference index were important factors determining browsing damage by large herbivores. Thus, populations with higher preference index and lower cover of accompanying woody species will be more exposed to higher levels of browsing damage. This agrees with previous studies showing the importance of plant cover (Senn *et al.*, 2002; Baraza *et al.*, 2006; Perea *et al.*, 2015) and the relative preference (palatability) of the target plant species (Perea *et al.*, 2014; Fernandez-Olalla *et al.*, 2016). Populations with low shrub cover (e.g. surrounded by grasslands or with higher rock cover) should be particularly considered in conservation management. Other essential management practices will involve the avoidance of water points and supplementary feeding in and around the areas housing the populations of the threatened plant species. In doing so, the relative density of large herbivores would diminish considerably, increasing the probability

of successful regeneration (Fig. 5.2). In fact, a reduction from 20-25 pellet groups per plot to 0-5 pellet groups will result in an increase in the probability of successful regeneration from, approximately 10% to 50% (Fig. 5.2). Thus, a straightforward advice would be to keep relative densities of large herbivores as low as possible in the areas surrounding the protected populations. Relative densities lower than three pellet groups per plot (~ 4 pellet group per 100 m^2) will ensure an average regeneration probability of, at least, 50% in the protected woody species and could work as a suitable conservation threshold. Current population control (hunting) could also help reduce herbivory pressure on the threatened plant populations by, for instance, increasing the hunting effort in the areas surrounding the target populations or in those showing highest browsing damage. Thus, a sustainable management would involve not only monitoring animal population trends but also the relative use by large herbivores (relative densities) and the dynamics of the threatened woody plant populations, particularly those that are at highest risk. Monitoring herbivory damage and regeneration success might be more challenging and costly for threatened taxa in comparison to common taxa but it seems to represent a highly sensitive limiting factor for ecological sustainability. The absence or a significant reduction in population viability (regeneration and herbivore damage) of the highly threatened plant species would serve as a warning signal of unsustainable densities of large herbivores.

**6. BARBARY SHEEP EFFECTS ON FOREST
REGENERATION: USE OF SMALL-SCALE
SILVICULTURAL TREATMENTS**

Based on:

Velamazán, M., San Miguel, A., Escribano, R., Perea, R., 2018. Compatibility of regeneration silviculture and wild ungulates in a Mediterranean pine forest: implications for tree recruitment and woody plant diversity. *Annals of Forest Science*, 75(1), p.35. (Annex III)

Ungulate effects on forest regeneration have been extensively studied, especially the ungulate browsing impact on seedlings and saplings. Browsing on young plants is, in fact, considered an important limitation for forest regeneration in most ungulate-dominated environments. In this chapter, we address the third specific objective of this research thesis (see Chapter 2) and, hence, analyze the effects of Barbary sheep populations on woody plant communities (diversity, structure and regeneration) following silvicultural treatments (reproductive fellings) of the main tree species (*Pinus* spp., *Quercus* spp. and *Juniperus* spp.). We, thus, aim to examine to what extent wild ungulate populations are compatible with regeneration silviculture in semiarid environments of southeastern Spain by evaluating ungulate use, plant damage, tree regeneration and structure and biodiversity of woody plant communities in felled and control areas. Here, we describe material and methods that are specific to this study and, therefore, not included in the general materials and methods shown in Chapter 3.

6.1 Specific materials and methods

Forest management and study sites

The main aim of the Forest Management Plan for Sierra Espuña Natural Park, approved in 2001, was to develop sustainable forest management according to

semiarid condition limits. An adaptive silvicultural framework was implemented to enhance forest persistence and make it compatible with small-scale forest harvesting, biodiversity conservation and park visitors.

The reproductive fellings analyzed in this study were organized in a group selection system to fulfil the following aims: (1) ensure and improve natural regeneration of the main tree species (*Pinus* spp., *Quercus* spp. and *Juniperus* spp.), (2) diversify forest structure, and (3) increase woody plant diversity of both tree and shrub species. The first implementation plan began in autumn 2001 and continued until 2010. In the group selection system defined for regeneration achievement, management practices consisted of felling all mature pines within a stand dominated by *Pinus* spp. Gap size, where cutting was carried out, varied from 0.05 to 0.75 ha depending on environmental restrictions and biodiversity requirements. In parallel, coppicing with standards was applied to *Quercus* spp. in the felled areas, and woody residues were added to the soil through chipping (Velamazán *et al.*, 2006).

We selected 6 zones with similar environmental conditions (Meso-mediterranean thermotype) where *Pinus halepensis* was the dominant species. A total of 17 sites were selected within the 6 zones. Each site comprised two plots: the regeneration gap (with silvicultural practices) and the control area (with no management). Plots of each site were separated by at least 30 m from each other to avoid edge effects. We also ensured that the control plot had the same ecological conditions (slope, aspect, topography) and similar stand density and structure as the treated plot before cutting.

Data collection

Field sampling was carried out in three consecutive years (2013-2015), in late winter and early spring, the best season for quantifying browsing damage in Mediterranean woody species (Perea *et al.*, 2014). Each site was measured only once within this 2013-2015 period, 6-14 years after treatment. Shrub cover, browsing/rubbing damage, shrub regeneration and Barbary sheep relative density were estimated in both control and felled plots following the general methods in a 5 m-radius circular plot (see chapter 3). We also recorded the DBH (diameter at 130 cm height) of all standing trees in a 10 m-radius circular area and young plants of pines, holm oaks and shrub-like junipers were quantified in a concentric 5 m-radius area, where we also measured browsing/rubbing damages over these species according to the description shown in Chapter 3.

Data analysis

All the analyses were performed using the R programming environment (R Core Team 2016; <http://www.r-project.org/>). To analyze regeneration abundance we used a Generalized Linear Mixed Model (GLMM) where number of young plants was the response variable (count data; Poisson error distribution with a log link function). Treatment (control vs. cutting) and tree species were the predictors. Site and age after treatment were included in the random structure of the model. We used a Cumulative Link Mixed Model (CLMM) to compare the browsing effects between treated and control plots. CLMM's allow for regression methods similar to linear models while

respecting the ordered categorical nature (0-5 rank) of our observations (Greene & Hensher, 2010; Perea *et al.*, 2015). Browsing damage for each species in each sample site was the response variable (ordinal variable). We included treatment (control vs. cutting) and plant species as the only predictors. Site and plant species were included as random effects. This allowed us to better specify the variance structure and control for possible differences among plant species (e.g. palatability) and sites. The model was fitted by the Laplace approximation with a Probit link function, using the “clmm” function within the “ordinal” package (Christensen, 2013) for R 3.1.0 software (www.r-project.org). We used another GLMM to analyze rubbing damage (presence vs. absence of impact; binomial error distribution and logit function) with the same predictors and random effects. Relative density of Barbary sheep (pellet counts) was analyzed with another GLMM, where number of pellet groups was the response variable (count data; Poisson error distribution) with treatment (control vs. cutting) as the only predictor in the model.

We compared structural/species diversity between control and treated plots (pairwise). Structural measures included shrub cover and stand tree densities (through basal area). We estimated the stand basal area from the DBH of all standing trees within each plot. We measured structural heterogeneity through: (1) variance of plot-level DBH for adult trees (>7.5 cm DBH), (2) Shannon diversity applied to 5 cm DBH classes for adult trees, and (3) Shannon diversity applied to tree regeneration categories (del Río *et al.*, 2016; Fahey *et al.*, 2015). We calculated two woody

species alpha-diversity indicators (Shannon index and species richness) and a beta-diversity index (Jaccard) for presence- absence data (Koleff *et al.*, 2003) to compare more intuitively changes in species composition across all woody plants. Generalized Linear Mixed Models (GLMM) with different transformations and family distributions were used to analyze the following response variables: Shannon applied to DBH classes (squared root transformation; family=gaussian), variance of plot-level DBH (squared root transformation; family=gaussian), Shannon applied to tree regeneration categories (family= gamma), shrub cover percentage (family=binomial), Shannon index (family=gaussian) and species richness (family=poisson).

We used the Jaccard similarity index (Jaccard, 1912; Koleff *et al.*, 2003), which has a minimum value of zero (completely different communities) and a maximum of 1 (identical communities in terms of species presence/absence) using the “vegan” package of R. Similarly, we calculated all the possible pairwise Jaccard indexes across the 17 control areas [$C(17, 2)=136$ pairwise combinations] and compared them with those obtained between the treated and control areas to see if there is higher species similarity across the control areas (at different ecological conditions) than between treated and control areas of each site. To statistically validate these comparisons we performed Exact Permutation Tests estimated by Monte Carlo (9,999 replications), using the library “perm” and the function “permTS” of the R software.

6.2 Results

Regeneration abundance

Overall, the abundance of tree regeneration was significantly greater in felled areas than in control areas (Table 6.1; Fig. 6.1). Regeneration abundance was higher in felled areas for all regeneration categories except for category 1 (seedlings), which showed similar values (Table 6.2). Regeneration categories 3 and 4 (plants with height >1.3 m) showed the greatest abundance difference between treated and control areas, with regeneration densities seven times greater in the treated areas (Table 6.2). Most of the tree regeneration (all categories) was comprised of *Pinus halepensis* young plants (42.88%), followed by *Quercus ilex* (25.92%) and *Juniperus oxycedrus* (30.08%).

Table 6.1 Summary of the Generalized Linear Mixed Model (GLMM) to analyze the effect of treatment (regeneration fellings) on the abundance of regeneration of the three main species (*Pinus halepensis*, *Quercus ilex*, and *Juniperus oxycedrus*)

Predictors	χ^2	d.f.	P
Treatment	10.943	1	0.0009
Tree species	0.010	2	0.9952
Treatment x Tree species	0.222	2	0.8951

Table 6.2 Summary of the regeneration values (mean $\pm$ SE) for each plant category (1-4) and for each paired site (treated vs. control plot). N=Total number of individuals; BD=Browsing damage; R=Rubbing percentage (%). Plant categories: 1= plants <30 cm high; 2= plants 30-130 cm high; 3= plants >130 cm high and DBH<2.5 cm; 4= plants >130 cm high and DBH of 2.5-7.5 cm.

Category	Treated areas			Control areas		
	N	BD	R	N	BD	R
1	86	0.83 $\pm$ 0.10	1.16 $\pm$ 0.01	93	0.56 $\pm$ 0.09	0.00 $\pm$ 0.00
2	177	0.81 $\pm$ 0.08	13.56 $\pm$ 0.03	84	0.58 $\pm$ 0.08	0.00 $\pm$ 0.00
3	134	0.42 $\pm$ 0.05	51.49 $\pm$ 0.04	19	0.23 $\pm$ 0.21	5.26 $\pm$ 0.05
4	28	0.36 $\pm$ 0.16	64.29 $\pm$ 0.09	4	0.50 $\pm$ 0.29	50.00 $\pm$ 0.29

Fig. 6.1 Boxplot representing the abundance of tree regeneration (all categories) in felled and control areas. Each box shows the median (band in the middle of the box) and the first and third quartiles (edges). Whiskers represent the lowest and highest datum within the 1.5 interquartile range of the lower and upper quartile

Browsing damage on regeneration

Browsing damage on young plants was generally low in both control and felled areas (Table 6.2). CLMM models revealed that, overall, there were no significant differences in browsing damage between the felled and control areas (Table 6.3). Browsing damage on *Q. ilex* was significantly higher than in *Pinus* and *Juniperus* (Table 6.3). In addition, browsing damage on young plants was only significantly higher in the felled areas for *Quercus ilex* (browsing damage of 1.83 ± 0.11 for the treated areas vs. 0.92 ± 0.12 for the control plots). *Pinus halepensis* showed greater browsing damage in the control areas (0.21 ± 0.06 for the treated areas vs. 0.44 ± 0.21 for the control; Table 6.3) as compared to *Juniperus oxycedrus* and *Quercus ilex* (Table 6.3). *Juniperus oxycedrus* showed very similar values of browsing damage in the treated and control plots (0.37 ± 0.06 for the treated areas vs. 0.36 ± 0.05 for the control plots).

Table 6.3 Summary of the Cumulative Link Mixed Model (CLMM) to analyze the effect of treatment (regeneration felling) on the browsing damage of the three main tree species (*Pinus halepensis*, *Quercus ilex*, and *Juniperus oxycedrus*). Values of tree species are shown against *Juniperus oxycedrus*. Bold type indicates statistical significance ($P < 0.05$)

Predictors	Factors levels	Coeff.	SE	z value	P
Treatment		0.311	0.294	1.057	0.290
Tree species	<i>P. halepensis</i>	- 0.093	0.221	- 0.423	0.672
	<i>Q.ilex</i>	1.675	0.186	9.005	< 0.0001
Treatment x tree species	<i>P. halepensis</i>	- 1.052	0.459	-2.287	0.022
	<i>Q.ilex</i>	1.438	0.346	4.153	< 0.0001

Rubbing damage

The percentage of trees damaged by rubbing was consistently higher in felled areas than in control areas ($\chi^2_1 = 5.92$; $P = 0.015$; Fig. 6.2). Interestingly, occurrence of rubbing damage increased with regeneration size in both treated and control areas (Table 6.2). No rubbing damage was found at all for the smallest regeneration categories (plants <130 cm) in control areas (Table 6.2). Rubbing damage was mainly focused on *Pinus halepensis* (98.26% of all rubbed young plants; $\chi^2_2 = 22.91$; $P < 0.001$).

Fig. 6.2 Proportion of young plants (%) with rubbing damage in felled and control areas. Whiskers represent standard error.

Relative use by Barbary sheep

We counted a mean of 4.29 ± 1.18 faecal pellet groups per plot in felled areas and 3.53 ± 1.02 in control areas. The GLMM analysis revealed that ungulate relative density did not significantly respond to the treatment ($\chi^2_1 = 0.02$; $P = 0.892$).

Forest structure

The mean basal area in felled areas was almost half ($15.34 \pm 2.78 \text{ m}^2/\text{ha}$) that in the control areas ($29.15 \pm 2.46 \text{ m}^2/\text{ha}$). Stand structural diversity indicators also showed significant differences between the control and felled areas. Mean diameter variance

was higher in the felled areas (66.02 ± 14.51 in felled areas vs. 48.02 ± 6.28 in the control areas; $\chi^2_1 = 17.54$; $P < 0.001$) whereas the Shannon index applied to 5 cm DBH classes was greater in control areas (0.68 ± 0.14 in felled areas vs. 1.09 ± 0.13 in control areas; $\chi^2_1 = 74.30$; $P < 0.001$).

Regeneration structural diversity estimated by Shannon index applied to the regeneration categories was significantly higher in the felled areas than in the control plots (1.09 ± 0.05 vs. 0.67 ± 0.10 ; $\chi^2_1 = 273.68$; $P < 0.001$; Fig. 6.3). Finally, total shrub cover in treated areas was, approximately, 10% greater than in control areas ($57.35 \pm 6.57\%$ vs. $289\ 48.21 \pm 7.51\%$; $\chi^2_1 = 17.54$; $P = 0.004$).

Fig. 6.3 Regeneration structural diversity estimated by Shannon index for felled and control areas.

Whiskers represent standard errors.

Woody plant diversity

A total of 41 woody species were found in the 34 plots, with 26 species in at least three plots (Table 6.4). Neither Shannon Index ($t=0.948$; $P=0.350$) nor species richness ($t=-0.863$; $P=0.394$) were significantly different between the felled and control areas. The Jaccard similarity index (beta-diversity index) between treated and control plot (pairwise analysis; $n=17$ pairs) was 0.497 ± 0.033 . However, the Jaccard similarity index between all control areas (pairwise analysis; $n=136$ pairs of control areas) was 0.754 ± 0.138 , indicating significantly higher similarity in woody plant composition across control plots than between treated and control plots on the same site ($P < 0.001$ estimated by 9,999 Monte Carlo replications).

Browsing damage on woody plants

We compared browsing damage on each woody plant species and six plant species showed unsustainable browsing levels (browsing damage >3 ; Table 6.4). Two of them (*Lonicera* sp. and *Bupleurum fruticosum*) showed unsustainable browsing damage in both areas, felled and control (Table 6.4). Overall, five and three species showed unsustainable browsing levels in control and treated areas, respectively (Table 6.4).

6. Regeneration silviculture

Table 6.4. Comparison of the number of individuals (N), ground cover (%), browsing damage (BD) and preference index (PI) for the woody species that were present in treated and control plots. Only species that were found in, at least, three surveys are shown. Values indicate mean $\pm$ SD.

Plant species	Treatment (felled areas)				No treatment (control areas)			
	N	Cover (%)	BD	PI	N	Cover (%)	BD	PI
<i>Argyrolobium zanonii</i>	4	1.50 $\pm$ 0.00	1.50 $\pm$ 0.65	0.03 $\pm$ 0.02	1	1.50 $\pm$ 0.00	1.00 $\pm$ 0.00	0.03 $\pm$ 0.00
<i>Bupleurum fruticosum</i>	1	1.50 $\pm$ 0.00	5.00 $\pm$ 0.00	0.18 $\pm$ 0.00	2	1.50 $\pm$ 0.00	4.50 $\pm$ 0.50	0.47 $\pm$ 0.36
<i>Cistus albidus</i>	15	12.70 $\pm$ 3.70	0.53 $\pm$ 0.22	0.07 $\pm$ 0.03	13	5.40 $\pm$ 2.50	0.31 $\pm$ 0.17	0.02 $\pm$ 0.01
<i>Cistus salviifolius</i>	2	2.80 $\pm$ 1.30	1.50 $\pm$ 1.50	0.07 $\pm$ 0.07	1	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Coronilla juncea</i>	4	6.10 $\pm$ 4.60	2.00 $\pm$ 0.71	0.30 $\pm$ 0.20	3	7.20 $\pm$ 2.80	3.33 $\pm$ 0.33	0.51 $\pm$ 0.24
<i>Coronilla minima</i>	5	2.20 $\pm$ 0.70	3.60 $\pm$ 0.40	0.08 $\pm$ 0.03	2	1.50 $\pm$ 0.00	2.00 $\pm$ 0.00	0.03 $\pm$ 0.00
<i>Daphne gnidium</i>	5	1.50 $\pm$ 0.00	0.25 $\pm$ 0.25	0.01 $\pm$ 0.01	7	1.40 $\pm$ 0.10	0.29 $\pm$ 0.18	0.04 $\pm$ 0.04
<i>Dorycnium hirsutum</i>	6	1.50 $\pm$ 0.00	0.17 $\pm$ 0.17	0.01 $\pm$ 0.01	4	1.50 $\pm$ 0.00	1.33 $\pm$ 0.88	0.04 $\pm$ 0.03
<i>Dorycnium pentaphyllum</i>	5	1.80 $\pm$ 0.30	1.67 $\pm$ 0.33	0.07 $\pm$ 0.03	6	1.70 $\pm$ 0.20	1.86 $\pm$ 0.51	0.16 $\pm$ 0.10
<i>Erinacea anthyllis</i>	3	7.70 $\pm$ 6.20	0.67 $\pm$ 0.33	0.04 $\pm$ 0.02	5	1.50 $\pm$ 0.00	0.80 $\pm$ 0.37	0.03 $\pm$ 0.01
<i>Genista valentina</i>	5	17.20 $\pm$ 11.30	2.40 $\pm$ 0.51	0.28 $\pm$ 0.12	3	3.80 $\pm$ 1.20	3.33 $\pm$ 0.88	0.39 $\pm$ 0.15
<i>Helianthemum</i> sp.	4	1.50 $\pm$ 0.00	1.00 $\pm$ 0.41	0.01 $\pm$ 0.01	4	1.50 $\pm$ 0.00	0.75 $\pm$ 0.25	0.01 $\pm$ 0.01
<i>Juniperus oxycedrus</i>	15	11.50 $\pm$ 4.10	0.36 $\pm$ 0.13	0.10 $\pm$ 0.07	15	20.70 $\pm$ 4.60	0.31 $\pm$ 0.13	0.17 $\pm$ 0.09
<i>Juniperus phoenicea</i>	1	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	3	6.00 $\pm$ 4.50	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Lithodora fruticosa</i>	2	1.50 $\pm$ 0.00	2.00 $\pm$ 2.00	0.01 $\pm$ 0.01	1	1.50 $\pm$ 0.00	3.00 $\pm$ 0.00	0.04 $\pm$ 0.00
<i>Lobularia</i> sp.	3	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	2	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Lonicera</i> sp.	3	1.50 $\pm$ 0.00	3.67 $\pm$ 0.88	0.03 $\pm$ 0.00	4	1.90 $\pm$ 0.40	4.00 $\pm$ 0.71	0.08 $\pm$ 0.03
<i>Ononis</i> sp.	2	1.50 $\pm$ 0.00	1.00 $\pm$ 1.00	0.04 $\pm$ 0.04	3	2.00 $\pm$ 0.50	1.33 $\pm$ 1.33	0.09 $\pm$ 0.09
<i>Phlomis lychnitis</i>	3	2.80 $\pm$ 0.80	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	4	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Pistacia lentiscus</i>	1	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	6	4.90 $\pm$ 2.60	1.20 $\pm$ 0.37	0.06 $\pm$ 0.03
<i>Quercus coccifera</i>	12	21.30 $\pm$ 7.70	2.50 $\pm$ 0.19	0.40 $\pm$ 0.11	14	18.10 $\pm$ 8.30	1.71 $\pm$ 0.35	0.20 $\pm$ 0.08
<i>Quercus ilex</i>	10	4.90 $\pm$ 1.20	2.83 $\pm$ 0.31	0.27 $\pm$ 0.12	7	3.30 $\pm$ 1.80	2.50 $\pm$ 0.50	0.15 $\pm$ 0.08
<i>Rhamus lycioides</i>	2	2.30 $\pm$ 0.80	1.50 $\pm$ 0.50	0.06 $\pm$ 0.03	5	3.80 $\pm$ 1.60	1.80 $\pm$ 0.37	0.11 $\pm$ 0.06
<i>Rosmarinus officinalis</i>	14	14.10 $\pm$ 2.90	0.79 $\pm$ 0.15	0.23 $\pm$ 0.06	12	13.50 $\pm$ 3.40	0.92 $\pm$ 0.15	0.23 $\pm$ 0.07
<i>Teucrium</i> sp.	7	1.70 $\pm$ 0.20	0.75 $\pm$ 0.53	0.02 $\pm$ 0.01	12	2.10 $\pm$ 0.20	0.29 $\pm$ 0.24	0.01 $\pm$ 0.01
<i>Thymus</i> sp.	9	1.90 $\pm$ 0.30	0.22 $\pm$ 0.15	0.01 $\pm$ 0.00	6	2.30 $\pm$ 0.60	0.17 $\pm$ 0.17	0.01 $\pm$ 0.01

6.3 Discussion

Our results show that, overall, regeneration abundance was higher in treated areas where regeneration fellings were applied. This positive effect of silvicultural treatments on natural regeneration was particularly strong for larger young plants (>130 cm high), which indicates that silvicultural practices mostly favored the survival and development of large saplings but not that of small saplings and seedlings (<130 cm high), suggesting that germination and establishment also took place in the intact (control) areas. This agrees with other studies that emphasize the positive effect of releasing young plants from competition by removing adjacent trees (Rodríguez-Calcerrada *et al.*, 2008), particularly for light-demanding tree species (Zavala *et al.*, 2011).

The results did not support our prediction regarding ungulate habitat use. Although we found greater number of pellets in felled areas, the difference was not significant, contrary to other studies (Kuipjer *et al.*, 2009; Wattless & Stefano, 2013), which state that canopy gaps are visited by ungulates more often. The non-significant difference in our study could be due to the small size of the forest gaps, as their characteristics are probably not different enough from the forest matrix to become significantly more attractive for herbivores. However, as expected, we did find greater ungulate effect (particularly rubbing) in the treated areas (Table 6.2). Interestingly, occurrence

of rubbing damage was proportionally higher than occurrence of browsing when comparing treated and control plots. This proportionally higher rubbing effect might be related to the greater abundance of large saplings and juveniles (>130 cm high) in the treated areas, which were the preferred tree size for rubbing. In fact, no individuals below 130 cm high were found with rubbing damage in the control plots. Trees of intermediate size, such as saplings and juveniles, have also been demonstrated to be preferred for rubbing by other wild ungulate (e.g. red deer, *Cervus elaphus*) in similar Mediterranean pine forests (Charco *et al.*, 2016). These results do not contradict previous studies that found more rubbing damage on intermediate size trees since they considered all diametric classes (regeneration and mature trees) in the stand (Johansson *et al.*, 1995; Massei & Bowyer, 1999; Ramos *et al.*, 2006; Charco *et al.*, 2016). Our results also highlight that rubbing damage by ungulates might be more important than browsing damage for the recruitment and growth of some species (e.g. pine trees in this study). Only for *Quercus* species did we find significantly greater browsing damage in treated areas as compared to control plots. These findings reveal the importance of discriminating between rubbing and browsing damage and the differential ungulate preference for conifers (mostly used for rubbing) in comparison to other species (mostly used for browsing).

As expected, the mean basal area in the treated areas was reduced by the implementation of regeneration fellings although structural diversity indicators for

adult trees responded differently, depending on the index used. However, the structural regeneration diversity, estimated by Shannon index across regeneration categories, was higher in treated areas as compared to the surrounding forest canopy. We suggest that this positive effect was mostly due to the silvicultural treatment *per se* (increase of light and reduction of competition) and not to the ungulate pressure since we found very low browsing damage on pines (the dominant tree species) in line with previous studies in this area (Fernández- Olalla *et al.*, 2016).

Apart from the influence on regeneration structure, silvicultural treatments did not significantly affect any woody plant diversity indexes (both species richness and Shannon). However, Jaccard similarity indexes revealed an approximately 50% replacement (species turnover) for woody plants. Interestingly, species turnover was even higher between control-treated areas on the same site than between control areas on different sites, suggesting that silvicultural treatments promote beta-diversity at a local scale.

Finally, we found unsustainable browsing damage (browsing score ≥ 3) on six shrub species (*Bupleurum fruticosum*, *Genista valentina*, *Coronilla juncea*, *Coronilla glauca*, *Lithodora fruticosa* and *Lonicera* spp.), that are mostly considered highly preferred plant species for browsing by *Ammotragus lervia* (Fernández-Olalla *et al.*, 2016 and Table 6.4). However, we did not find any clear patterns indicating that shrubs in treated areas suffered significantly heavier browsing damage. In fact, we

found a higher number of species with unsustainable browsing damage in the control plots (five species) than in treated areas (three species). This might be due to the lower shrub cover and, thus, lower food availability, in control (closed-canopy) plots as compared to treated areas but further studies on the relationship between forest practices, shrub cover and browsing would be desirable.

6.4 Conservation and management implications

This study confirms that small-scale forest interventions are suitable practices to improve biodiversity, as they promote structural heterogeneity with positive effects on trees and understory vegetation (Torrás & Saura, 2008). We highlight the fact that forest gaps smaller than 0.75 ha appear to be small enough to avoid higher habitat use by ungulates but sufficiently large to increase sunlight and recruitment growth. Although small-scale regeneration silviculture does not seem to increase woody plant species richness, it does increase beta-diversity and provides structural diversity and understory heterogeneity by replacing many woody species that thrive in the adjacent closed-canopy areas. In addition, small-scale interventions contribute to increased shrub cover, which is considered essential to facilitate tree recruitment in dry environments (Pugnaire *et al.*, 1996; Castro *et al.*, 2004) particularly in ungulate-dominated areas (Perea & Gil, 2014). Therefore, small-scale interventions might benefit from shrub encroachment to increase favourable microsites for seedlings and, hence, enhance the protection against ungulates.

In order to reduce the possible detrimental effect of wild ungulates (rubbing and browsing) we recommend selecting the regeneration areas sufficiently far away from supply points or attractive foraging areas (water, mineral correctors, feeding areas, crops, pastures, etc.) since relative high densities of wild ungulates have been proven to decrease regeneration probability and woody plant diversity (Perea *et al.*, 2015; Velamazán *et al.*, 2017). The use of standing dead trees or the addition of artificial decoy posts might represent an adequate alternative to reduce the rubbing impact of ungulates on saplings and juveniles, which seems to be the most deleterious effect of small-scale interventions in ungulate-dominated environments. Further studies should analyze the effect of forest gap size on ungulate habitat use as well as the consequences for woody plant diversity and tree regeneration in order to estimate the most appropriate scale for intervention in areas where wild ungulates are abundant.

**7. BARBARY SHEEP USE OF FIREBREAKS AND
SUPPLY POINTS: EFFECTS ON PLANT DAMAGES,
FUEL LOAD, WOODY PLANT DIVERSITY AND
REGENERATION**

Based on:

Velamazán, M., San Miguel, A., Escibano, R., Perea, R., 2018. Use of firebreaks and artificial supply points by wild ungulates: Effects on fuel load and woody vegetation along a distance gradient. *Forest Ecology and Management*, 427, pp.114-123. (Annex IV)

Supply points (water, salt and food) and firebreaks distribution condition shelter, food and water through the landscapes, modifying ungulate behaviour and effects over vegetation (Gordon *et al.*, 2004; Andreassen *et al.*, 2005; Miller *et al.*, 2009; Sahlsten *et al.*, 2010; Suárez-Esteban *et al.*, 2013, 2014). In this Chapter, according to the fourth objective defined in Chapter 2, we investigate ungulate habitat use around firebreaks and supply points along a distance gradient (10-50-100 m). We also analyze the effects of Barbary sheep on plant damage, fuel load (phytovolume), woody species diversity and regeneration along the distance gradient from both management structures (supply points and firebreaks) to distribute and use them efficiently and in a sustainable manner throughout the landscape. We describe material and methods that are specific to this study and, therefore, not included in the general material and methods (Chapter 3).

7.1 Specific materials and methods

The firebreak network

The firebreak network in Sierra Espuña includes more than 40 km of linear structures (15-60 meters width) where thinning, pruning and shrub clearing (selective brush out) are implemented periodically to keep wildfire risk at low levels. The network

spreads mainly across the meso-Mediterranean thermotype (700-1200 m a.s.l.), where pine forests are the dominant vegetation and interruption of forest fuels is more necessary.

Supply points

Three main types of supply points for wild ungulates are distributed throughout the Natural Park: water, salt and food supply points. Water supply points consists of small artificial ponds (0.5-15 m²) near natural springs or where water is added regularly. Salt supply points consist of a large block of salt available throughout the year. Food supply points are small crop areas (cereals such as wheat –*Triticum* spp.-, oat –*Avena* spp.-, or barley –*Hordeum* spp.- and legumes such as vetch –*Vicia* spp.-or French honeysuckle -*Hedysarum coronarium*-) located within the scrublands or forests to provide a highly nutritious food source (grain, herbs, grass) in a browse-dominated area. The distance between supply points is highly variable but they are separated, at least, 500 m from each other.

Data collection

Field sampling was carried out in five consecutive years (2013-2017), mainly in late winter and early spring, the best season for quantifying browsing damages in Mediterranean woody species (Cook & Stubbendieck, 1986; Perea *et al.*, 2014).

Forty line transects of 100 m long were distributed randomly throughout the firebreak network (20 transects; distance between them > 150 m) and 20 supply

points (6 feeding points, 4 water points, 3 salt points and 7 mixed points where combinations of water, food and salt are provided). All supply points were built or established at least 15 years ago. Firebreak transects were established perpendicular to the firebreak line and transects from the supply points were established radially (one transect per supply point). Along the length of each transect, we ensured the same ecological conditions (same aspect and inclination). Each transect consisted of three 5 m-radius circular plots at 10, 50 and 100 meters, respectively (distance gradient), from the firebreak or supply point boundary (n= 120 plots). In each plot we recorded plant cover (%) and height for each species, as well as browsing, rubbing and trampling damage following the general material and methods explained in chapter 3. Regeneration abundance and ungulate relative density (habitat use) was also estimated following the methodology explained in Chapter 3.

Data analysis

To analyze relative habitat use by ungulates as a function of the distance to firebreak or supply point, we used a Generalized Linear Mixed Model (GLMM) where number of pellet groups was the response variable (count data; Poisson error distribution). Distance to the firebreak or supply point (10, 50 and 100 m) was the only predictor. Site (each particular transect) was included in the random structure of the model. We used a Cumulative Link Mixed Model (CLMM) to analyze browsing intensity on woody plants as a function of the distance to the firebreak or supply points. CLMM's

allow for regression methods similar to linear models while respecting the ordered categorical nature of our observations (Greene & Hensher, 2010; Christensen & Brockhoff, 2013). Browsing intensity (i.e., browsing damage) for each plant species in each plot was the response variable (ordinal variable). Distance to the firebreak or supply point was the only predictor. Species and site were also included in the model as a random effect to control for possible intrinsic differences across plant species following Perea *et al.* (2015). The model was fitted by the Laplace approximation with a Probit link function, using the “clmm” function within the “ordinal” package (Christensen, 2013) for R 3.1.0 software (www.r-project.org).

To analyze the occurrence of other impacts different from browsing (i.e., trampling or rubbing) of each woody species we used a Generalized Linear Mixed Model (GLMM). The response variable was binary (presence vs. absence of impact; binomial error distribution). Predictor was the same as in the model above: distance. Species and site were also included in the model as a random effect. The model was fitted by the Laplace approximation with a logit link function according to our data properties (Bolker *et al.*, 2009), using the “glmer” function within the “lme4” package for R 3.1.0 software. To present the results of the logistic regression we created a combined graph (histograms plus logistic curves), following the recommendations of Smart *et al.* (2004). To obtain these graphs we used the “plot.logi.hist” function from the “pobbio” library (De la Cruz Rot, 2005).

We compared woody species richness along the distance gradient with a GLMM. We used the same predictor and random structure as in the previous analysis (count data; Poisson error distribution). To measure changes in species composition along the gradient of each site we calculated a beta diversity index for presence-absence data (Koleff *et al.*, 2003). We used the Jaccard similarity index (Jaccard, 1912; Southwood & Henderson, 2000; Koleff *et al.*, 2003), which has a minimum value of zero (completely different communities) and a maximum of one (identical communities in terms of species presence/absence) using the “vegan” package of R (Oksanen *et al.*, 2013).

We used a Generalized Linear Model (GLM) to evaluate whether fuel load of woody species varied with the distance to firebreaks and supply points (family=Gamma; link=log). Fuel load in each plot was estimated as phytovolume (Casals *et al.*, 2016) calculated as the product of plant ground cover and plant height for each woody species. Probability of regeneration occurrence was analyzed with another GLMM (binomial distribution) where presence-absence of regeneration was the response variable. The model had the same structure as in the previous GLMM's. Abundance of regeneration was analyzed with another CLMM where regeneration abundance (0-3 scale) was the response variable, with the same structure as in the previous CLMM.

7.2 Results

Use of firebreaks and supply points by Barbary sheep

GLMM models revealed that, overall, there were differences in the number of pellet groups along the distance gradient from supply points and firebreaks. The number of pellet groups significantly decreased at 100 m ($Z=-1.179$; $P=0.002$) but not at 50 meters ($Z=-0.526$; $P=0.168$) from the supply points boundary. Significant differences were obtained for firebreaks at 100 meters ($Z=-1.348$; $P<0.001$) and at 50 meters ($Z=-0.516$; $P<0.001$) from the firebreak boundary. Higher pellet count values (mean $\pm$ SD) were found at any distance from the supply points (10.79 $\pm$ 0.56 at 10 m; 6.56 $\pm$ 0.35 at 50 m; 4.75 $\pm$ 0.34 at 100 m) as compared to firebreaks (7.04 $\pm$ 0.42 at 10 m; 5.13 $\pm$ 0.29 at 50 m; 2.90 $\pm$ 0.24 at 100 m).

Browsing intensity

CLMM showed that browsing intensity increased with shorter distances to the supply points and firebreaks (Fig. 7.1), obtaining significant differences only between 10 and 100 meters distance, and not between 10 and 50 meters (Table 7.1). Supply points showed, on average, 1.8 times greater browsing damage than firebreaks at any distance from the boundary (Fig. 7.1).

Table 7.1. Summary of the CLMM model comparing the average browsing damage along a distance gradient from firebreaks and supply points. Values are shown against 10-m distance plots.

Distance	Firebreaks			Supply points		
	Estimate±SE	Z-value	P-value	Estimate±SE	Z-value	P-value
50 m	-0.2390±0.2065	-1.158	0.24703	-0.2461± 0.1981	-1.242	0.2141
100 m	-0.5568±0.2092	-2.662	0.00777**	-0.5202± 0.1994	-2.608	0.0091**

Fifty-six woody species were found along the transects but only 39 woody species were found in more than three plots (Table 7.2). Three taxa (*Hedera helix* L., *Lonicera* sp., and *Bupleurum* sp.) showed unsustainable browsing levels (Browsing degree >3) at any distance from firebreaks or supply points. Overall, we found greater proportion of species with unsustainable damage at 10 m (35% for both firebreaks and supply points) than at 50 m (31% and 25% for firebreaks and supply points, respectively) and 100 m (11% and 19% for firebreaks and supply points, respectively; Table 7.2). Fifteen species (26.7%) showed unsustainable browsing levels at certain distances along the transects (Table 7.2).

Fig. 7.1 Barplot showing the average browsing degree (0-5) along the distance gradient. 0=No browsing evidence; 1=Light browsing, with <10% twigs browsed; 2=Low browsing, with 10-30% of twigs browsed; 3=Intense browsing (30-60% of the twigs); 4=Heavy browsing (>60% of the twigs), with clear modification of plant shape; 5=Maximum browsing (>90% of the twigs browsed). Different letters above the bars indicate significant differences ($P < 0.05$). Whiskers represent standard error.

7. Firebreaks / supply points

Table 7.2a Comparison of the number of plots (N), ground cover (%) and browsing degree (BD) for the woody species that were present in firebreaks along the distance gradient. Values indicate mean $\pm$ SD.

FIREBREAKS		10			50			100	
Sp	N	Cover	BD	N	Cover	BD	N	Cover	BD
<i>Anthyllis cytisoides</i>	2	25.25 $\pm$ 24.75	4.00 $\pm$ 1.00	1	5.00 $\pm$ 0.00	5.00 $\pm$ 0.00	4	6.50 $\pm$ 4.27	3.75 $\pm$ 0.63
<i>Artemisia</i> sp.									
<i>Bupleurum</i> sp.	2	2.75 $\pm$ 2.25	5.00 $\pm$ 0.00	5	1.90 $\pm$ 0.78	3.60 $\pm$ 1.00	3	0.50 $\pm$ 0.00	4.33 $\pm$ 0.67
<i>Cistus albidus</i>	15	7.97 $\pm$ 2.12	1.13 $\pm$ 0.31	16	3.22 $\pm$ 0.65	0.63 $\pm$ 0.20	12	4.04 $\pm$ 0.98	0.50 $\pm$ 0.19
<i>Cistus clusii</i>	2	0.50 $\pm$ 0.00	2.50 $\pm$ 2.50	2	0.50 $\pm$ 0.00	1.00 $\pm$ 1.00	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Coronilla juncea</i>	2	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	2	1.75 $\pm$ 0.25	2.50 $\pm$ 1.50
<i>Coronilla minima</i>	5	1.90 $\pm$ 0.92	4.00 $\pm$ 0.55	1	5.00 $\pm$ 0.00	5.00 $\pm$ 0.00	2	3.25 $\pm$ 1.75	0.50 $\pm$ 0.50
<i>Cistus salviifolius</i>	2	12.75 $\pm$ 12.25	1.50 $\pm$ 0.50	1	5.00 $\pm$ 0.00	2.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Daphne gnidium</i>	4	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	1.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	1.50 $\pm$ 1.50
<i>Dorycnium hirsutum</i>	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00			
<i>Dorycnium pentaphyllum</i>	5	2.40 $\pm$ 0.86	2.80 $\pm$ 0.73	6	0.50 $\pm$ 0.00	2.17 $\pm$ 0.70	3	5.75 $\pm$ 4.76	1.75 $\pm$ 1.18
<i>Erinacea anthyllis</i>							1	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Fumana</i> sp.	4	0.50 $\pm$ 0.00	1.25 $\pm$ 0.63						
<i>Genista scorpius</i>	1	0.50 $\pm$ 0.00	1.00 $\pm$ 0.00	1	8.00 $\pm$ 0.00	1.00 $\pm$ 0.00	1	5.00 $\pm$ 0.00	2.00 $\pm$ 0.00
<i>Genista umbellata</i>									
<i>Hypericum ericoides</i>	2	0.50 $\pm$ 0.00	1.00 $\pm$ 1.00				1	1.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Helianthemum</i> sp.	2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	5	0.80 $\pm$ 0.30	0.00 $\pm$ 0.00	2	1.75 $\pm$ 0.25	0.00 $\pm$ 0.00
<i>Hedera helix</i>	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00						
<i>Juniperus oxycedrus</i>	14	9.07 $\pm$ 3.34	0.64 $\pm$ 0.20	10	13.10 $\pm$ 4.58	0.60 $\pm$ 0.16	15	18.03 $\pm$ 4.94	0.27 $\pm$ 0.12
<i>Juniperus phoenicea</i>				1	0.50 $\pm$ 0.00	1.00 $\pm$ 0.00			
<i>Lithodora suffruticosa</i>	3	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	5	0.50 $\pm$ 0.00	3.80 $\pm$ 0.40	2	0.50 $\pm$ 0.00	2.50 $\pm$ 0.00
<i>Linum</i> sp.	2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00				1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Lonicera</i> sp.	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	4.50 $\pm$ 0.50	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00
<i>Olea europaea</i>	3	2.17 $\pm$ 0.83	2.67 $\pm$ 1.33						
<i>Ononis</i> sp.	3	1.33 $\pm$ 0.83	5.00 $\pm$ 0.00				1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Phlomis lychnitis</i>							1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Pistacia lentiscus</i>	6	1.25 $\pm$ 0.75	3.00 $\pm$ 0.77	8	6.88 $\pm$ 1.55	1.13 $\pm$ 0.35	9	7.44 $\pm$ 2.11	1.22 $\pm$ 0.32
<i>Quercus coccifera</i>	15	29.57 $\pm$ 6.29	2.53 $\pm$ 0.33	16	29.72 $\pm$ 4.52	2.63 $\pm$ 0.15	15	29.17 $\pm$ 5.35	2.47 $\pm$ 0.24
<i>Quercus rotundifolia</i>	4	15.13 $\pm$ 8.83	3.25 $\pm$ 0.48	4	11.25 $\pm$ 4.73	3.25 $\pm$ 0.48	2	20.00 $\pm$ 0.00	3.00 $\pm$ 0.00
<i>Rhamnus lycioides</i>	3	3.67 $\pm$ 3.17	1.00 $\pm$ 0.00	6	5.67 $\pm$ 2.36	1.50 $\pm$ 0.56	5	2.30 $\pm$ 1.10	1.20 $\pm$ 0.20
<i>Rosmarinus officinalis</i>	19	26.42 $\pm$ 5.61	1.47 $\pm$ 0.26	20	19.78 $\pm$ 4.71	1.15 $\pm$ 0.24	20	26.55 $\pm$ 4.76	1.00 $\pm$ 0.21
<i>Rosa</i> sp.									
<i>Retama sphaerocarpa</i>	2	0.50 $\pm$ 0.00	4.00 $\pm$ 1.00	1	0.50 $\pm$ 0.00	3.00 $\pm$ 0.00	2	7.50 $\pm$ 2.50	2.00 $\pm$ 1.00
<i>Staeheleina dubia</i>	1	5.00 $\pm$ 0.00	5.00 $\pm$ 0.00	2	1.25 $\pm$ 0.75	1.50 $\pm$ 1.50	1	0.50 $\pm$ 0.00	2.00 $\pm$ 1.00
<i>Satureja</i> sp.	1	0.50 $\pm$ 0.00	1.00 $\pm$ 0.00						
<i>Teucrium</i> sp.	3	1.33 $\pm$ 0.83	0.33 $\pm$ 0.33	4	1.63 $\pm$ 1.13	0.75 $\pm$ 0.75	2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Thymus</i> sp.	15	1.57 $\pm$ 0.79	0.20 $\pm$ 0.20	15	1.83 $\pm$ 0.76	0.13 $\pm$ 0.07	17	3.12 $\pm$ 1.67	0.18 $\pm$ 0.18

7. Firebreaks / supply points

Table 7.2b Comparison of the number of plots (N), ground cover (%) and browsing degree (BD) for the woody species that were present in supply points along the distance gradient. Values indicate mean $\pm$ SD.

SUPPLY POINTS		10			50			100	
Sp	N	Cover	BD	N	Cover	BD	N	Cover	BD
<i>Anthyllis cytisoides</i>							3	4.50 $\pm$ 2.84	2.00 $\pm$ 0.58
<i>Artemisia</i> sp.	2	15.25 $\pm$ 14.75	2.00 $\pm$ 0.00	1	5.00 $\pm$ 0.00	2.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	0.50 $\pm$ 0.50
<i>Bupleurum</i> sp.	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00						
<i>Cistus albidus</i>	11	4.59 $\pm$ 1.81	1.36 $\pm$ 0.51	9	6.72 $\pm$ 1.41	0.78 $\pm$ 0.36	8	2.88 $\pm$ 0.60	0.38 $\pm$ 0.38
<i>Cistus clusii</i>	3	2.00 $\pm$ 1.50	0.67 $\pm$ 0.67				2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Coronilla juncea</i>							1	0.50 $\pm$ 0.00	3.00 $\pm$ 0.00
<i>Coronilla minima</i>				2	0.50 $\pm$ 0.00	2.00 $\pm$ 2.00	4	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Cistus salviifolius</i>									
<i>Daphne gnidium</i>	2	2.75 $\pm$ 2.25	2.00 $\pm$ 2.00	3	3.00 $\pm$ 0.00	1.00 $\pm$ 0.58	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00
<i>Dorycnium hirsutum</i>	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	4	0.50 $\pm$ 0.00	2.00 $\pm$ 1.22	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00
<i>Dorycnium pentaphyllum</i>	2	0.50 $\pm$ 0.00	3.00 $\pm$ 2.00	2	0.50 $\pm$ 0.00	1.00 $\pm$ 1.00	2	2.75 $\pm$ 2.25	1.00 $\pm$ 1.00
<i>Erinacea anthyllis</i>	3	21.00 $\pm$ 9.00	1.33 $\pm$ 0.88	4	25.00 $\pm$ 15.41	1.75 $\pm$ 0.63	6	29.42 $\pm$ 13.57	1.17 $\pm$ 0.48
<i>Fumana</i> sp.									
<i>Genista scorpius</i>	1	10.00 $\pm$ 0.00	1.00 $\pm$ 0.00	2	9.00 $\pm$ 6.00	0.50 $\pm$ 0.50	2	31.75 $\pm$ 1.25	1.50 $\pm$ 1.50
<i>Genista umbellata</i>	1	10.00 $\pm$ 0.00	4.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	2.00 $\pm$ 0.00	2	6.50 $\pm$ 3.50	1.50 $\pm$ 0.50
<i>Hypericum ericoides</i>	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00						
<i>Helianthemum</i> sp.	3	0.50 $\pm$ 0.00	2.67 $\pm$ 1.33	4	0.50 $\pm$ 0.00	1.25 $\pm$ 0.95	5	1.00 $\pm$ 0.50	1.40 $\pm$ 0.75
<i>Hedera helix</i>	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00
<i>Juniperus oxycedrus</i>	14	13.32 $\pm$ 3.54	2.00 $\pm$ 0.42	14	18.29 $\pm$ 4.29	1.00 $\pm$ 0.26	15	16.73 $\pm$ 3.61	0.73 $\pm$ 0.18
<i>Juniperus phoenicea</i>	3	2.00 $\pm$ 1.50	0.00 $\pm$ 0.00	3	6.83 $\pm$ 4.28	1.33 $\pm$ 0.88	4	4.75 $\pm$ 3.47	0.50 $\pm$ 0.50
<i>Lithodora suffruticosa</i>	2	0.50 $\pm$ 0.00	2.50 $\pm$ 1.50	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	4	1.75 $\pm$ 0.72	3.50 $\pm$ 0.50
<i>Linum</i> sp.							1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Lonicera</i> sp.	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00						
<i>Olea europaea</i>	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00			
<i>Ononis</i> sp.	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Phlomis lychnitis</i>	6	0.92 $\pm$ 0.42	0.00 $\pm$ 0.00	4	1.75 $\pm$ 0.72	0.00 $\pm$ 0.00	3	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Pistacia lentiscus</i>	6	11.33 $\pm$ 4.52	2.17 $\pm$ 0.48	5	3.80 $\pm$ 1.76	2.60 $\pm$ 0.87	4	15.00 $\pm$ 5.40	1.00 $\pm$ 0.41
<i>Quercus coccifera</i>	8	15.25 $\pm$ 5.40	3.38 $\pm$ 0.38	11	20.86 $\pm$ 6.36	2.91 $\pm$ 0.44	7	28.00 $\pm$ 9.31	2.71 $\pm$ 0.52
<i>Quercus rotundifolia</i>	8	8.63 $\pm$ 4.61	3.63 $\pm$ 0.53	10	8.70 $\pm$ 3.90	3.30 $\pm$ 0.50	7	5.64 $\pm$ 2.72	2.57 $\pm$ 0.69
<i>Rhamnus lycioides</i>	3	6.00 $\pm$ 2.08	3.33 $\pm$ 0.67	3	6.83 $\pm$ 3.17	2.33 $\pm$ 1.33	5	3.80 $\pm$ 1.76	2.40 $\pm$ 0.68
<i>Rosmarinus officinalis</i>	12	15.50 $\pm$ 3.58	1.17 $\pm$ 0.42	12	31.71 $\pm$ 6.65	0.83 $\pm$ 0.27	14	35.75 $\pm$ 6.90	0.79 $\pm$ 0.19
<i>Rosa</i> sp.	3	5.33 $\pm$ 2.33	4.00 $\pm$ 1.00	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00	3	6.83 $\pm$ 3.17	4.00 $\pm$ 1.00
<i>Retama sphaerocarpa</i>	1	0.50 $\pm$ 0.00	4.00 $\pm$ 0.00	3	8.33 $\pm$ 3.33	1.00 $\pm$ 0.58	4	5.88 $\pm$ 2.44	2.00 $\pm$ 0.82
<i>Staeheleina dubia</i>	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	2	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00
<i>Satureja</i> sp.	3	2.00 $\pm$ 1.50	2.33 $\pm$ 1.45	2	0.50 $\pm$ 0.00	5.00 $\pm$ 0.00	1	0.50 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Teucrium</i> sp.	5	0.50 $\pm$ 0.00	2.00 $\pm$ 2.00	4	1.25 $\pm$ 0.11	0.25 $\pm$ 0.11	6	0.50 $\pm$ 0.00	0.83 $\pm$ 0.83
<i>Thymus</i> sp.	9	2.39 $\pm$ 1.05	0.33 $\pm$ 0.22	11	1.91 $\pm$ 1.00	0.00 $\pm$ 0.00	12	2.67 $\pm$ 1.53	0.00 $\pm$ 0.00

Other damages on vegetation

The combined predicted probability graph shows that trampling and rubbing decrease with distance to the firebreaks and supply points (Fig. 7.2). However, no significant differences were found on trampling or rubbing damage between 10 and 50 m, either for firebreaks ($Z=-1.445$; $P= 0.1485$) or supply points ($Z=-0.4180$; $P= 0.270$). We did find significant differences between 10 and 100 m for both firebreaks ($Z=-0.867$; $P= 0.029$) and supply points ($Z=-1.118$; $P= 0.004$). Supply points showed approximately 33% greater probability of damage than firebreaks at any distance from the boundary (Fig. 7.2).

Fig. 7.2. Fitted logistic regression curves showing the predicted probability (left axis) and observed frequency (right axis) of plant damages different from browsing (i.e., trampling and rubbing) along an increasing distance to firebreaks and supply points. The histograms represent the observed damages at the top (1 = presence of damages) and the observed absence of damages at the bottom (0 = absence of damages) and the line shows the predicted probability.

Fuel load

In firebreaks, fuel load (woody species phytovolume) was significantly higher at 100 m as compared to 10 m and 50 m (Fig. 7.3) but no significant differences were found between 10 and 50 m ($Z=-0.048$; $P=0.273$). However, for supply points, fuel load increased for plots located at 50 meters ($Z=0.471$; $P<0.001$) and at 100 meters ($Z=0.548$; $P<0.001$) as compared to those located at 10 m (Fig. 7.3). Fuel load between 10 and 100 m only increased 20% in firebreaks but 78% in supply points (Fig. 7.3).

Fig. 7.3 Barplot showing fuel load (phytovolume) variation along the distance gradient for firebreaks (left) and supply points (right). Different letters above the bars indicate significant differences ($P<0.05$). Whiskers represent standard error.

Biodiversity indicators

Woody species richness was significantly lower in plots located at 10 m from supply points as compared to plots located at 50 and 100 m (Table 7.3). However, for firebreaks, differences were only significant between plots located at 50 and 10 m and marginally significant between 10 and 100 m (Table 7.3). Species richness of supply points was slightly lower (7.8-8.8 species per plot) than that of firebreaks (8.5-9.3 species per plot).

Table 7.3. Summary of the GLMM model comparing species richness along a distance gradient from firebreaks and supply points. Values are shown against 10-m distance plots.

Distance	Firebreaks			Supply points		
	Estimate±SE	<i>t</i> -value	<i>P</i> -value	Estimate±SE	<i>t</i> -value	<i>P</i> -value
50 m	0.044±0.006	6.94	3.88e-12***	0.086± 0.010	8.77	<2e-16***
100 m	0.010±0.006	1.60	0.109	0.038± 0.010	3.82	0.000133***

Beta-diversity (mean Jaccard similarity indexes) for all possible combinations of plots at 10, 50 or 100 m within each transect (pairwise analysis) was similar for both firebreaks and supply points (values from 0.80 to 0.87; Table 7.4). Mean Jaccard similarity index between plots (at the same distance) across all transects did not show any significant change for either firebreaks or supply points (Table 7.4; diagonal values). Overall, the mean Jaccard similarity index between transects of supply points was lower (range of 0.65-0.71) than those across transects of firebreaks (range of 0.78-0.82; Table 7.4).

Table 7.4. Summary of the results regarding Jaccard beta diversity indexes for firebreaks (FB) and supply points (SP) along the distance gradient. Values outside the diagonal represent the beta diversity indexes between 10-50, 50-100 and 10-100 m plots within each transect. Values in the diagonal 10-10, 50-50 and 100-100 indicate the beta diversity indexes across transects of FB (left) and SP (right) at 10, 50 and 100 m, respectively. All values are Mean $\pm$ SE.

	10	50	100	SP
10	0.78 $\pm$ 0.01	0.70 $\pm$ 0.01	0.82 $\pm$ 0.03	10
50	0.80 $\pm$ 0.04	0.81 $\pm$ 0.01	0.82 $\pm$ 0.04	50
100	0.87 $\pm$ 0.03	0.87 $\pm$ 0.03	0.65 $\pm$ 0.01 0.82 $\pm$ 0.01	100
FB	10	50	100	

3.6 Woody species regeneration

Predicted probability of regeneration occurrence increased with distance to firebreaks (Fig. 7.4) but showed no relationship with distance to supply points (Fig. 7.4). No significant differences in regeneration abundance (0-3 scale) were found along the distance gradient for either firebreaks or supply points ($P>0.05$ for all comparisons; Fig 7.5). Only nine species (16.1% of all woody species) showed natural regeneration in at least three plots in both firebreaks and supply points (Table 7.5).

Fig. 7.4 Fitted logistic regression curves showing the predicted probability (left axis) and observed frequency (right axis) of woody species regeneration along an increasing distance to supply points or firebreaks. The histograms represent the observed damages at the top (1 = presence of regeneration), the observed absence of damages at the bottom (0 = absence of regeneration) and the line shows the predicted probability.

7. Firebreaks / supply points

Table 7.5. Comparison of the number of plots (N) and mean regeneration abundance (R) along a distance gradient for woody species that at least had three plots with natural regeneration in firebreaks (FB) and supply points (SP). Regeneration value (R) varies from 0 (no regeneration) to 3 (very abundant regeneration). Values indicate mean $\pm$ SD.

Sp	SP					
	10		50		100	
	N	R	N	R	N	R
<i>Anthyllis cytisoides</i>					3	0.67 $\pm$ 0.67
<i>Cistus albidus</i>	11	0.91 $\pm$ 0.21	9	1.00 $\pm$ 0.23	8	0.75 $\pm$ 0.16
<i>Dorycnium pentaphyllum</i>	2	0.00 $\pm$ 0.00	2	0.50 $\pm$ 0.50	2	0.00 $\pm$ 0.00
<i>Juniperus oxycedrus</i>	14	0.43 $\pm$ 0.14	14	0.71 $\pm$ 0.19	15	0.60 $\pm$ 0.16
<i>Pinus halepensis</i>	15	0.40 $\pm$ 0.16	15	0.47 $\pm$ 0.13	16	0.44 $\pm$ 0.13
<i>Pistacia lentiscus</i>	6	0.00 $\pm$ 0.00	5	0.40 $\pm$ 0.24	4	0.50 $\pm$ 0.29
<i>Quercus coccifera</i>	8	0.88 $\pm$ 0.30	11	1.18 $\pm$ 0.26	7	1.29 $\pm$ 0.42
<i>Quercus rotundifolia</i>	8	0.88 $\pm$ 0.30	10	1.20 $\pm$ 0.29	7	0.86 $\pm$ 0.26
<i>Rosmarinus officinalis</i>	12	0.58 $\pm$ 0.19	12	0.50 $\pm$ 0.23	14	0.71 $\pm$ 0.19

Sp	FB					
	10		50		100	
	N	R	N	R	N	R
<i>Anthyllis cytisoides</i>	2	1.00 $\pm$ 1.00	1	1.00 $\pm$ 0.00	4	0.25 $\pm$ 0.25
<i>Cistus albidus</i>	15	0.80 $\pm$ 0.20	16	0.81 $\pm$ 0.19	12	1.08 $\pm$ 0.19
<i>Dorycnium pentaphyllum</i>	5	0.00 $\pm$ 0.00	6	0.83 $\pm$ 0.40	4	0.25 $\pm$ 0.25
<i>Juniperus oxycedrus</i>	14	0.50 $\pm$ 0.14	10	0.60 $\pm$ 0.22	15	0.47 $\pm$ 0.22
<i>Pinus halepensis</i>	20	0.68 $\pm$ 0.15	20	0.65 $\pm$ 0.17	20	0.70 $\pm$ 0.14
<i>Pistacia lentiscus</i>	6	0.33 $\pm$ 0.33	8	0.25 $\pm$ 0.16	9	0.22 $\pm$ 0.15
<i>Quercus coccifera</i>	15	1.27 $\pm$ 0.32	16	1.44 $\pm$ 0.30	15	1.60 $\pm$ 0.24
<i>Quercus rotundifolia</i>	4	1.25 $\pm$ 0.95	4	1.00 $\pm$ 0.58	2	2.00 $\pm$ 0.00
<i>Rosmarinus officinalis</i>	19	0.42 $\pm$ 0.14	20	0.40 $\pm$ 0.13	20	0.40 $\pm$ 0.13

Fig. 7.5 Barplot showing mean regeneration abundance (0-3 scale) along the distance gradient. Different letters above the bars indicate significant differences ($P < 0.05$). Whiskers represent standard error.

7.3 Discussion

Ungulate habitat use

Our results show that firebreaks worked as attractive corridors for wild ungulates as they were more used than the surrounding land. This agrees with other studies that show how firebreaks represent grassy areas for wild ungulates and extensive livestock, offering greater food quality and abundance within the shrubland matrix (Dopazo & Suarez, 2004; Ruiz-Mirazo *et al.*, 2011). Although other studies have also showed the attractive effect of soft linear structures for most frugivorous birds and

mammals, this pattern was not found for certain wild ungulate species such as wild boar and deer (Suárez-Esteban *et al.*, 2013a). This suggests that the effectiveness of linear structures to attract ungulates may strongly depend on the surrounding matrix and its food quantity and quality. In areas where woody plants are dominant, firebreaks represent an alternative, highly nutritious food supply where herbaceous vegetation is predominant (Ruiz-Mirazo *et al.*, 2011).

As expected, the results also confirm that supply points are attractive areas for wild ungulate populations (Grosman *et al.*, 2011), with an increasing habitat use as distance to supply points decreases. However, differences were not significant between 10 and 50 m from the supply point, which indicate a non-linear relationship where ungulate use is rather high and constant in an area of about one hectare around the point. Importantly, wild ungulate use was considerably higher in supply points than in firebreaks for any distance along the gradient. Therefore, supply points and their surrounding land are also more sensitive to possible ungulate damage and, hence, managers should avoid locating supply points in areas where threatened or sensitive plant species occur. Supply points represent heavily used areas by ungulates that can serve (1) as suitable areas to monitor the type and intensity of ungulate damage (e.g., browsing, trampling or rubbing) and (2) as attractive and high-resource patches to modify wild ungulate distribution and, thus, to reduce damages on forest regeneration or threatened plant populations (Gordon *et al.*, 2004).

Effects on plant damage

Higher ungulate use indeed implies changes in browsing patterns. Changes in browsing intensity were particularly noticeable at 100 meters distance from supply points and firebreaks, where a significant reduction in browsing pressure was observed. These results partly agree with Månsson *et al.* (2015) where a strong reduction in browsing pressure was found at distances shorter than 135 meters from food plots. However, other studies have detected damages at larger scales in Scandinavian forests (Felton *et al.*, 2017) that could be explained by man-made interventions such as the proximity of forest gaps or the longevity of the feeding program (Milner *et al.*, 2014).

Trampling and rubbing are typical damages associated to ungulate movements through the forest. These damages depend on ungulate densities and movements but were not found to be species-dependent in this study. Similarly to browsing damage, trampling and rubbing occurred at much lower frequency with shorter distances to supply points (1.8 times higher occurrence at 10 m than at 100 m) and firebreaks (1.7 times higher at 10 m than at 100 m). Therefore, we confirm that all types of plant damage by ungulates (browsing, trampling and rubbing) increased with distance to supply points and firebreaks. Importantly, all types of damage were higher for supply points as compared to firebreaks at any distance, which highlights a greater differential effect of supply points. However, both firebreaks and supply points

showed similar patterns along the distance gradient. Their abundance and distribution throughout the forest landscape should be strongly considered in forest management and conservation plans, particularly in areas dominated by highly palatable plant species or threatened plant taxa given their sensitivity and vulnerability to ungulate damage (Velamazán *et al.*, 2017).

Regarding vegetation dynamics, man-made structures attracting ungulates seem to favor earlier successional stages. The lack of species belonging to later stages of plant dynamics (e.g. legume shrubs, trees or tree-like shrubs) has been previously shown as a good indicator of unsustainable browsing pressure (Tremblay *et al.*, 2006; Perea *et al.*, 2014). These plant communities, dominated by short, scattered distributed shrubs could enhance wildfire prevention. In addition, these ungulate-attracted points, when appropriately distributed across the landscape, may work as a useful management tool to spread ungulate use homogeneously, avoiding high local densities and detrimental and unsustainable overuse. Conversely, the increased pressure around supply points could favor natural and semi-natural grasslands protected under the 92/43/CEE Directive (San Miguel *et al.*, 2010) and therefore they can also be used for the conservation of other non-forest habitats and to increase overall habitat heterogeneity.

Effects on fuel load

Fuel load (phytovolume) is an important indicator of fire risk that was significantly reduced at shorter distances from supply points and firebreaks. Control of fuel load by ungulates has been analysed in previous studies within firebreaks (Jauregui *et al.*, 2007; Mancilla-Leitón *et al.*, 2013). However, our results also reveal a differential effect of linear structures as compared to supply points. We found that effects of ungulates on fuel load were stronger for supply points, probably because firebreaks are long linear structures that occupy vast areas while supply points concentrate higher ungulate pressure in less extensive areas. Thus, ungulate effects on vegetation structure around firebreaks seem to be less efficient in controlling fuel load in agreement with our previous results that showed greater ungulate use and plant damage for supply points. Therefore, establishing supply points within the firebreak network could help increase ungulate habitat use and thus favor fuel reduction in fire-prone areas where maintenance of firebreaks is unlikely or costly. Further studies should evaluate the combined effects of firebreaks and supply points since their additive or synergistic effect can be of great use to inexpensively reduce fuel loads and maintain firebreaks.

Effects on woody plant diversity and regeneration

Overall, woody species richness increased with distance to supply points and

firebreaks but, surprisingly, beta-diversity values remained similar along the distance gradient. Thus, contrary to our expectations, changes in plant communities along the distance gradient were relatively small between all pairwise comparison along the distance gradient (10-50, 50-100 and 10-100 m) as compared to other ungulate-dominated environments where plant composition is strongly affected by increasing browsing pressure (Gordon *et al.*, 2004; Mancilla-Leyton *et al.*, 2014; Perea & Gil, 2014). Previous studies show that medium to moderate levels of herbivory contribute to increase heterogeneity and species turnover (Škornik *et al.*, 2010; García-Moreno *et al.*, 2014; Papanastis *et al.*, 2017). However, at high ungulate densities, browsing pressure usually causes biotic homogenization of plant communities (Rooney & Waller, 2003; Rooney, 2009; Perea *et al.*, 2014), which could explain why changes in species composition were small. In fact, at heavy ungulate pressure, preferred plant species are eventually replaced by non-preferred species (White, 2012; Perea *et al.*, 2014). Our results reveal that supply points and firebreaks showed lower abundance (ground cover) of preferred species (e.g., *Lithodora fruticosa* and *Stachelina dubia*) as compared to other non-attracted sites (random plots) in the same study area (Fernández-Olalla *et al.*, 2017). Supply points and firebreaks are attractive areas subject to high ungulate pressure that may not be able to reflect changes in plant composition due to high woody plant homogenization along the study gradient. However, homogenization and simplification of plant communities might be highly scale-dependent. For instance, ungulates at moderate densities, may homogenize

small and high-resource patches but may contribute to heterogenization at larger scales (e.g. landscape), by creating highly diverse mosaics of distinct plant communities (e.g. intensively browsed, less browsed and non-browsed communities), which enhances overall biodiversity and the provision of ecosystem services (Papanastis *et al.* 2017). Supply points and firebreaks can therefore be used to generate moderate, medium and high ungulate densities patches throughout the landscape and thus increase habitat heterogeneity and the conservation of EU protected habitats, which strongly depend on moderate grazing pressure (San Miguel *et al.*, 2010).

Our results also show an overall lack of plant regeneration along the distance gradient, which was particularly pronounced for supply points where ungulate use was higher. Thus, only nine species (16.1% of all woody plants) showed natural regeneration in at least three plots in both firebreaks and supply points. These regenerating species were mostly non-preferred plants (Fernández-Olalla *et al.*, 2017) or had a strong ability to regenerate vegetatively (e.g. root sprouting). This regeneration failure of some species over others indeed favours biotic homogenization (Rooney and Waller, 2003; White, 2012; Perea *et al.*, 2014). Further studies should evaluate longer distance gradients in order to detect possible changes in plant regeneration and their associated shifts in woody plant communities.

Supply points and firebreaks are attractive areas for wild ungulates and represent

useful tools to manage wild ungulate populations, modifying ungulate habitat use and their associated effects on woody vegetation. Both supply points and firebreaks showed greater ungulate use and lower fuel load as distance to boundaries decreases but effects were rather constant within one hectare around the structures. Only at 100 m, fuel load and plant damage decreased significantly for both firebreaks and supply points, indicating a non-linear relationship along the distance gradient. Woody species richness also increased with distance to firebreaks and supply points but changes in plant composition (beta-diversity) remained similar along the gradient mostly due to limited regeneration and overall heavy use by ungulates, which favored biotic homogenization of plant communities but further research should undertake the effects of supply points/firebreaks on heterogeneity at a broader scale. Supply points showed significantly greater ungulate effects as compared to firebreaks at any distance for all the study variables (i.e., greater habitat use, plant damage, fuel load reduction, woody species richness and beta-diversity changes). Supply points and firebreaks are useful tools to identify and monitor ungulate damage on woody systems but might not be adequate to evaluate effects on plant species composition unless farther distances, beyond 100 m, are considered. Managers should carefully consider the location and distribution of firebreaks and supply points throughout the landscape given their differential capacity to attract ungulates and their associated effects on woody plant communities. Fire-prone areas within the landscape can benefit from the combined use of firebreaks and supply points to reduce fuel load at a

very low cost (e.g. adding salt lick points).

8. GENERAL DISCUSSION

The results of this research contribute to improve our understanding on Barbary sheep interactions with woody vegetation and allow us to fulfil the main research objective of providing science-based guidelines for the conservation of ungulate-dominated woody systems. They also reveal that vegetation analysis through different parameters (plant preferences, plant damages, biodiversity indicators and natural regeneration) is highly useful to obtain sustainability thresholds and establish adequate management measures at different levels and scales. For instance, it helped establish management practices for the target animal populations (appropriate relative densities) or the management of habitats (e.g., structure and cover of forests, scrubland communities and populations of threatened plant species), and even the management of forest and ungulate infrastructures (e.g. firebreaks and supply points). Learning about the Barbary sheep effects on flora and vegetation allowed us to better understand the influence of wild ungulates (large herbivores) on the ecosystem and the derived effects of management on the conservation of semiarid environments. Despite this research took place in a protected area, it also contributes to establish scientific-based management practices in multifunctional systems where social, economic and environmental objectives should be met. This general discussion is structured along the main milestones of this research, connecting the specific objectives established in Chapter 2.

8.1 Plant damage and preferences for woody plants

We analyzed three different damages on vegetation in order to understand ungulate effects on woody plants: browsing, rubbing and trampling. Browsing has been extensively studied worldwide, particularly in Mediterranean areas, where it represents a major feeding strategy, as green grass availability is primarily restricted to spring and autumn (Bulgalho & Milne, 2003; Zamora & Matías, 2014; Schoenbaum *et al.*, 2017). Rubbing or debarking is considered one of the most common and important type of damage by domestic and wild herbivores, which can eventually cause tree death (Mountford & Peterken, 2003; Charco *et al.*, 2016). Trampling effects, studied frequently through soil properties measures (Moret-Fernández *et al.*, 2011; Pulido *et al.*, 2016), has shown to be an important factor to determine vegetation characteristics in ungulate-dominated environments (Stavi *et al.*, 2009, Macci *et al.*, 2012; 2015; Stein *et al.*, 2016). Our results focus on the three indicators and show that browsing damage was more conspicuous and widespread and, hence, had a stronger relationship with the study variables.

The relationship between browsing damage (utilization) and woody plant cover (availability) allowed us to determine Barbary sheep preferences for woody species. Barbary sheep was found to forage on a very wide variety of woody plant species: from small chamaephytes to trees, although average browsing intensities were usually moderate. Barbary sheep seemed to behave as a mixed-feeder that predominately grazes but it also browses on a great variety of woody species when

green herbage is scarce or lacking (Fernández-Olalla *et al.*, 2016). Its feeding behaviour would be similar to that of fallow deer (*Dama dama*) and opposite to ungulate species that require higher quality forage (e.g. Roe deer - *Capreolus capreolus*; Ferretti *et al.*, 2008).

A key objective of this research was to understand ungulate preferences for conifers, as most part of the study area is covered by pine species (*Pinus halepensis* Mill., *P. pinaster* Ait., *P. nigra* Arn.) or shrub-like junipers (*Juniperus phoenicea* L., *J. oxycedrus* L.). Conifers showed to be highly rejected species for the Barbary sheep (damages are negligible even with the highest Barbary sheep density). This was confirmed in the regeneration silvicultural plots dominated by *Pinus halepensis*, where damages on conifer regeneration (both on *Pinus* and *Juniperus*) were mainly caused by rubbing (Velamazán *et al.*, 2018a). Similar results, indicating low browsing preference for conifers have been recorded in other areas dominated by other Caprinae species, such as the Iberian ibex (*Capra pyrenaica*) in Sierra Nevada (Zamora *et al.*, 2001), Sierra de Cazorla (Martínez, 2002b) and Sistema Central (Perea *et al.* 2015).

An important aspect of Barbary sheep population and management is the intensity and extent of Barbary sheep damage on cultivated plants. The cultivated *Prunus dulcis*, that is probably the most widespread and affected crop in the study area, was found to be heavily browsed. The domestic olive trees (*Olea europaea*), less widespread in the study area, showed sustainable browsing, even in surveys with high Barbary sheep density levels. On the contrary, *Opuntia* - usually in old orchards

near forest houses - seemed to be heavily browsed as a consequence of both its high water content and the palatability and nutritive value of its cladodes and fruits, which are available in summer, the worst season regarding high quality forage availability. Other cultivated plant species, such as *Citrus aurantium*, *Mespilus germanica*, *Corylus avellana* or *Arundo donax*, seemed to have been heavily browsed only as a consequence of their occasional occurrence in surveys with very high Barbary sheep density. These preferences are an important source of human-wildlife conflicts that require management measures to reduce or avoid ungulate damages in the farmlands surrounding the forests or scrublands. Some measures have been analyzed in this study, as the implementation and use of supply points in the forest (far from orchards) or the increase of hunting effort to reduce overall ungulate densities (Velamazán *et al.*, 2017; 2018b). Hunting for fear in the farmland could also be a suitable tool to study in order to avoid or reduce ungulate damages in the farmlands (Ciuti *et al.*, 2012; Cromsigt *et al.*, 2013; Norum *et al.*, 2015).

As it might be expected, the observed browsing intensity of woody species was positively related to Barbary sheep habitat use or relative densities. We found maximum browsing intensity on widespread vegetation in surveys with high values of Barbary sheep density, which should be avoided as much as possible through management (Fernández-Olalla *et al.*, 2016). Our results show that to achieve the ecological sustainability, populations should be at low densities and under appropriate management practices that could ensure the conservation of the natural

vegetation, including the most sensitive (threatened) woody plant species (Velamazán *et al.*, 2017).

Intermediate-low ungulates densities seem to be the most adequate for biodiversity conservation in the study area, supporting the Intermediate Disturbance Hypothesis (Connell, 1978), that showed to be a key measure for the conservation of different species: butterflies (Bartoňová *et al.*, 2017), dung beetles (Treitler *et al.*, 2017), ground beetles (Kaltsas *et al.*, 2013), spiders (Zakkak *et al.*, 2014), bats (Rainho *et al.*, 2010) or plants (Ljubičić *et al.*, 2014; Dutoit *et al.*, 2016). However, this hypothesis showed to be irrelevant in other areas with high woody plant endemism and no historical presence of large herbivores (e.g., Canary Islands—Macaronesian region-) where the impact of exotic ungulates such as Barbary sheep, even at low densities, caused an important biodiversity loss (Piñero & Luengo, 1992; Garzón-Machado *et al.*, 2010).

Our findings also reveal the importance of discriminating between browsing and other damages (e.g. trampling or rubbing) and the differential ungulate preference for certain species depending on the type of damage. For instance, we found a different selection for rubbing, where conifers were highly preferred (Velamazán *et al.*, 2018a), as compared to other species mostly used for browsing, such as most of the threatened woody species (Velamazán *et al.*, 2017).

The analysis on threatened species showed rubbing differences among woody species, although most damages were caused by browsing. It showed evidences of rubbing/debarking damage in most species (71.4% of all species analyzed). However,

only three of them (21.4%) were frequently affected, with more than 10% of their individuals showing clear damage. These results were highly species-dependent and *Sorbus aria* was, by far, the most affected plant species, with 25% of the individuals showing rubbing or debarking damage (Velamazán *et al.*, 2017). This strongly agrees with Iberian ibex rubbing damage in Sierra de Guadarrama (Perea *et al.*, 2015) where *Sorbus* spp. were heavily affected. Trees on the Rosaceae family seem to be highly preferred for rubbing/debarking damage. In fact, heavy debarking by livestock has been identified as a main threat for the conservation of farmlands (Fischer *et al.*, 2010) or the persistence of traditional orchards dominated by Rosaceae trees (López-Sánchez *et al.*, 2018).

Our results on regeneration silvicultural plots highlight that (1) rubbing damage by ungulates might be more important than browsing damage for the recruitment and growth of some species (e.g. pine trees in our study) and (2) rubbing damage can be highly dependent on the size/age of the plants. There was a proportionally higher rubbing effect in treated plots that might be related to the greater abundance of large saplings and juveniles (>130 cm high) in the treated areas, which were the preferred tree size for rubbing. In fact, no individuals below 130 cm high were found with rubbing damage in the control plots (Velamazán *et al.*, 2018a). Trees of intermediate size, such as saplings and juveniles, have also been demonstrated to be preferred for rubbing by other wild ungulates (e.g. red deer, *Cervus elaphus*) in similar Mediterranean pine forests (Charco *et al.*, 2016). Therefore, pine regeneration might be mostly limited by rubbing damage rather than browsing although the frequency

and intensity of damage was rather low in our study system. However, it may affect tree growth and wood quality in productive tree forests. Addition of natural/artificial pine posts (Charco *et al.*, 2016) or preservation of a protective shrub cover around seedlings and saplings (Ramos *et al.*, 2006; Perea & Gil, 2014) have been suggested as adequate management practices to reduce rubbing damage on sensitive flora or individuals. Leaving natural decoy posts in small-size silvicultural treatments, that showed to increase shrub cover and regeneration (Velamazán *et al.*, 2018a), might be an adequate practice for pine regeneration. Fencing large areas (e.g. the stands treated with regeneration fellings or other silvicultural treatments) seems to be an expensive and difficult practice to implement through the forest but it would be the most suitable practice for localized individuals/populations showing regeneration problems (Charco *et al.*, 2016; Velamazán *et al.*, 2017).

Trampling was the third type of ungulate damage that we recorded and it was mostly addressed by presence/absence data given the difficulties on establishing degrees of intensity or cover. Trampling modifies microtopography and reduces soil penetration depth, which conditions vegetation growth and distribution (Stavi *et al.*, 2009). Other analyses in the study area showed no effects of trampling on soil physical properties but changes in electrical conductivity, nutrient concentration, microbial activity and microbial communities around supply points (Pascual-Rico *et al.*, 2018). Our results on rubbing and trampling damage align with the hypothesis that ungulate use and damage decrease with distance to man-made structures that are highly attractive for ungulates such as supply points and firebreaks (Velamazán *et al.*, 2018b). Reduction

in species richness and fuel load (ground cover) might be related to increasing ungulate damage, including trampling. However, we were unable to link reduction in fuel load to specific ungulate damage (trampling, browsing, rubbing). Further studies should address trampling damage more thoroughly since it may directly affect vegetation cover and composition. In some species, trampling has been proved to be very important (e.g. in cattle), more than browsing and rubbing. This may depend on the feeding behavior (grazer vs. browser), the animal movement and the body size (Cumming & Cumming, 2003; Hobbs & Searle, 2005). Given the larger size of Barbary sheep compared to other wild ungulates such as Iberian ibex or mouflon, as well as their high sheep-like movement (San Miguel *et al.*, 2010), these animals will probably cause higher trampling damage than expected.

8.2. Conservation of biodiversity. Application for protected habitats and threatened species

Conservation and enhancement of biodiversity is one of the main management goals, particularly in protected areas where biodiversity conservation is the main priority. The management practices to be implemented in ungulate-dominated environments must therefore ensure and enhance biodiversity at all levels and across all diversity facets (Nugent *et al.*, 2001; Bartuszevige & Endress, 2008; Fernández & Gómez, 2012; Moleón *et al.*, 2014). Our results focus on common man-made interventions typical of most Mediterranean forests such as (1) silvicultural treatments, (2) the creation and maintenance of management structures such as firebreaks and supply

points and (3) specific conservation measures for singular habitats or threatened species within the forest.

Interestingly, we found that silvicultural treatments did not significantly affect any indexes of alpha woody plant diversity (both species richness and Shannon). Only for the Shannon index by 5 cm diameter, we found a significant decrease in structural diversity after treatment, generating greater heterogeneity. In addition, we estimated the structural regeneration diversity, estimated by Shannon index across tree regeneration (mainly *Pinus halepensis*, *Quercus rotundifolia* and *Juniperus oxycedrus*) categories, that was significantly higher in treated areas as compared to the surrounding forest canopy. We suggest that this positive effect was mostly due to the silvicultural treatment *per se* (increase of light and reduction of competition) and not to the ungulate pressure since we found very low browsing damage on pines (the dominant tree species) in line with previous studies in this area (Fernández-Olalla *et al.*, 2016; Velamazán *et al.*, 2018a).

However, beta-indexes calculated as Jaccard similarity indexes revealed, approximately, a 50% replacement (species turnover) for woody plants after the regeneration treatment. Interestingly, species turnover was even higher between control-treated areas of the same site than between control areas of different sites, suggesting that silvicultural treatments promote beta diversity at a local scale (Velamazán *et al.*, 2018a). These results are in contrast with those found for firebreaks and supply points, where beta-diversity index showed no significant

changes along the distance gradient (Velamazán *et al.*, 2018b). This could be related to the sudden creation of a gap in the forest through fellings, which increases sunlight and the proliferation of shade-intolerant woody species (mostly light-demanding), increasing overall diversity and heterogeneity (Busing & White, 1997; Torras & Saura, 2008) and compensating higher damages by ungulates near the firebreak/supply point. However, the beta-diversity homogeneity and the slightly increase in species richness may be better explained by woody plant homogenization along the study gradient due to high ungulate pressure.

In our study on firebreaks/supply points (Velamazán *et al.*, 2018b), low palatable species became more abundant near the attracted areas in line with previous studies, which point out that high densities of wild ungulates favor non-palatable plants (Perea *et al.*, 2014; Fernández-Olalla *et al.*, 2016; Perea *et al.*, 2017). Furthermore, our results reveal that supply points and firebreaks showed lower abundance (ground cover) of preferred species (e.g., *Lithodora fruticosa* and *Staehelina dubia*) as compared to other non-attracted sites (random plots) in the same study area (Fernández-Olalla *et al.*, 2016). Regarding vegetation dynamics, man-made structures attracting ungulates seem to favor earlier successional stages. On the contrary, most threatened woody species in the study area showed to be late-successional plant species, usually with less aromatic and secondary metabolites than early-successional plants and higher preference index (palatability) than common taxa (3.10 vs. 0.85). This could be the reason why they showed unsustainable

browsing pressure (Velamazán *et al.*, 2017), what agrees with previous studies in the Mediterranean region where other large herbivores such as red deer—*Cervus elaphus* (Perea *et al.* 2014)— and Iberian ibex—*Capra pyrenaica* (Perea *et al.* 2015) — also showed higher preferences for late-successional woody plants. This suggests that plant selection might be related to the greater palatability of threatened woody plants regardless of the ungulate species.

These differential effects of ungulates on biodiversity across management treatments (regeneration fellings), man-made structures (supply point/firebreaks) and threatened woody species populations, suggest several conservation measures:

First, our studies confirm that small-scale forest interventions are suitable practices to improve biodiversity, as they promote structural heterogeneity with positive effects on trees and understory vegetation (Torrás & Saura, 2008). We highlight the fact that forest gaps smaller than 0.75 ha appear to be small enough to avoid higher habitat use by ungulates but sufficiently large to increase sunlight and recruitment growth (Velamazán *et al.*, 2018a). Second, areas surrounding supply points and firebreaks seem to be rather homogeneous due to high ungulate pressure (Velamazán *et al.*, 2018b). However, homogenization and simplification of plant communities might be highly scale-dependent. For instance, ungulates at moderate densities, may homogenize small and high-resource patches but may contribute to heterogenization at larger scales (e.g. landscape), by creating highly diverse mosaics of distinct plant communities (e.g. intensively browsed, less browsed and non-browsed communities),

which enhances overall biodiversity and the provision of ecosystem services (Papanastasis *et al.*, 2017). Supply points and firebreaks, as attractive areas for ungulates, could be used to generate moderate, medium and high ungulate densities patches throughout the landscape and thus increase habitat heterogeneity and the conservation of habitats which strongly depend on moderate grazing pressure, such as natural and semi-natural grasslands protected under the 92/43/CEE Directive (San Miguel *et al.*, 2010). In fact, browsing has showed positive effects in restoring shrub-encroached dry grasslands in previous studies (Alrababah *et al.*, 2007; Masson *et al.*, 2015; Elías & Tischew, 2016), improving conditions for small herbivores that feed on grass such as European rabbits, *Oryctolagus cuniculus* (Mancilla-Leytón *et al.*, 2013).

We deduced four important considerations from the analysis on threatened woody species for biodiversity conservation (Velamazán *et al.*, 2017): (1) Firstly, the inclusion of threatened taxa when monitoring herbivory damage showed to be a conservative and integrative indicator of the overall ecological sustainability of the system. The absence or a significant reduction in population viability (regeneration and herbivore damage) of the highly threatened plant species would serve as a warning signal of unsustainable densities of large herbivores. In that way, the applicability of assessing sustainability by surveying threatened or rare woody flora could be extended to any systems dominated by woody plants such as forests, shrublands or woodlands; (2) Secondly, by assessing the herbivore damage and the

probability of natural regeneration in threatened populations, we were able to obtain a conservation priority ranking. The ranking provided an idea of what species should be monitored regularly and undergo specific protection. Populations at highest risk should be therefore strictly protected from browsing (e.g., by fencing) to ensure their viability and, hence, avoid possible local extinction; (3) Thirdly, the results on threatened woody species also showed that plant cover and preference index were important factors determining browsing damage by large herbivores. Thus, populations with higher preference index and lower cover of accompanying woody species will be more exposed to higher levels of browsing damage. This agrees with previous studies showing the importance of plant cover and the relative preference (palatability) of the target plant species. Populations with low shrub cover (e.g. surrounded by grasslands or with higher rock cover) should be particularly considered in conservation management plans; (4) Finally, other essential management practices could involve the avoidance of water points and supplementary feeding in and around the areas housing the populations of the threatened plant species, particularly given the results shown in Chapter 7 (Velamazán et al. 2018b). In doing so, the relative density of large herbivores would diminish considerably, increasing the probability of successful regeneration. In fact, a key finding from our study is that, at least, 100 meters around man-made structures (firebreaks and supply points) will be affected by herbivory pressure and therefore biodiversity sensitive areas or reforestation activities with more palatable species should be avoided (Velamazán *et al.*, 2018b).

8.3 Forest regeneration

Regeneration of woody species is a key process in forest ecology to ensure the whole forest persistence and propose management measures (Wang & Smith, 2002; Gil *et al.*, 2010; Beguin *et al.*, 2016). Our results focused on the effects of ungulate populations on the regeneration process across different regeneration stages: adults (fruit/flower production) and seedlings/saplings.

First, we observed the effects of ungulates on forest regeneration through the browsing damage on adult plants in order to assess overall vegetation conservation. We used a 0-5 score on browsing damage and considered that browsing damage higher than 3 on adult woody plants compromise the ecological sustainability, as it affects growth and flower/fruit production and, therefore, the regeneration cycle (Perea *et al.*, 2014, 2015; Fernández-Olalla *et al.*, 2016). The results revealed no evidence of major problems regarding actual vegetation structure and composition in our experimental conditions. None of the abundant species (found in more than 10 plots) showed unsustainable browsing, and only two of them showed moderate browsing intensity. Only 10 species (10.2%) showed average browsing intensity above 3 and maximum browsing intensity (level 5) was only found in some sites where pellets abundance indicated a very high Barbary sheep density. However, certain scarce and threatened species representative of late successional stages showed medium to high levels of browsing utilization (Fernández-Olalla *et al.*, 2016). As their conservation status could become unfavourable-bad, we focused our

second analysis on the composition and regeneration of these threatened species. In contrast with the regeneration of widespread species, that did not show important regeneration problems, our findings showed 5-fold greater herbivore impact on threatened woody species, which increased the proportion of species with unsustainable herbivory. This analysis, focused on threatened species, revealed that 50% of the threatened plant species showed unsustainable levels of browsing, with 35.7% of the species under the highest possible level of browsing, what contrasted with those obtained for common species in the study area, using the same methodology, where only 6.5% of the species showed unsustainable values (Velamazán *et al.*, 2017). Populations of threatened species are rare and represent attracted points for ungulates as they represent “different” or “new” browse within the shrubland or forest matrix. Therefore, rare species are usually more vulnerable and are earlier indicators of browsing damage than widespread species. This is even accentuated when these threatened or rare species are highly palatable or their ability to regenerate vegetatively is limited (Miranda *et al.*, 2011; White, 2012; Perea *et al.*, 2014, 2015).

Our study on woody species regeneration after silvicultural treatments (regeneration fellings), confirmed previous results on browsing preferences and the consequent effects on fruit production. As expected, we found unsustainable browsing damage (browsing score >3) on six shrub species (*Bupleurum fruticosum*, *Genista valentina*, *Coronilla juncea*, *Coronilla glauca*, *Lithodora fruticosa* and *Lonicera* spp.), that are

mostly considered highly preferred plant species for browsing by *Ammotragus lervia* (Fernández-Olalla *et al.*, 2016). Surprisingly, we did not find any clear patterns indicating that shrubs in treated areas suffered significantly heavier browsing damage. In fact, we found higher number of species with unsustainable browsing damage in the control plots (five species) than in treated areas (three species). This might be due to the lower shrub cover and, thus, lower food availability, in control (closed-canopy) plots as compared to treated areas (Velamazán *et al.*, 2018a). Interestingly, we found that only three woody species showed unsustainable browsing at any distance from supply points and firebreaks. Two of them (*Lonicera* sp. and *Hedera helix*) have been previously considered highly palatable and seem to require plant facilitation (i.e., plant protection against herbivores) by low palatable plants (e.g., *Juniperus phoenicea*) to thrive in ungulate-dominated environments (Fernández-Olalla *et al.*, 2016). *Bupleurum* sp., a low-preferred species, showed considerably higher browsing levels, with no regeneration, near supply points or firebreaks, which indicates that the increasing browsing pressure around firebreaks and supply points could threaten the persistence of many species, including some low-palatable plants (Velamazán *et al.*, 2018b)

When we analyzed seedlings and saplings of threatened species, we found that ungulate relative densities and regeneration presence were related. Relative densities lower than three pellet groups per plot (4 pellet group per 100 m²) could ensure an average regeneration probability of, at least, 50% in the protected woody species and

could work as a suitable conservation threshold. In this regard, population control (hunting or live-trapping) could help reduce herbivory pressure on the threatened plant populations by, for instance, increasing the hunting or trapping effort in the areas surrounding the target populations or in those showing highest browsing damage (Velamazán *et al.*, 2017).

Our results showed no clear pattern for regeneration presence/abundance along the distance gradient from firebreaks or supply points, what could be explained by the overall high ungulate pressure. Nine species showed greater regeneration abundance, mainly low palatable plant species (e.g. Labiateae or Cistaceae species) or those with strong ability to regenerate vegetatively (e.g. root sprouting). In fact, Labiatae and Cistaceae species, usually belonging to early stages of plant succession have been used as nurse shrubs in ungulate-dominated environments (Perea & Gil, 2014) due to their low palatability and ability to spread rapidly. After a long browsing legacy, these non-palatable species become strongly common, indicating diversity loss and biotic homogenization (Rooney & Waller, 2003; Perea *et al.*, 2014). Thus, studies should distinguish regeneration at species level and control for each plant species abundance since overall high abundance of regeneration is not *per se* a suitable indicator of sustainable ungulate populations.

Contrary to threatened species populations, firebreaks or supply points, that showed to be attractive for ungulates, we found similar Barbary sheep use in treated and non-treated areas after the silvicultural treatments. It seems that the forest gap size is big

enough to improve regeneration of light-demanding tree species but small enough to avoid changes on ungulate habitat use and important ungulate damages. Despite the small size of the interventions, small-scale silvicultural treatments seemed to favor pine regeneration abundance, that was higher in treated areas. This positive effect in treated areas was weaker in small saplings and seedlings (<130 cm high), what could suggest that germination and establishment also took place in the control areas and silvicultural treatments improved mainly the survival and development of larger (> 130 cm high) young plants. Larger young plants showed more rubbing damages than the rest of seedlings and saplings but it did not seem to prevent their survival (Velamazán *et al.*, 2018a). Furthermore, small-scale interventions seem to increase shrub cover, what increases favorable microsites for seedlings (López-Sánchez *et al.*, 2016) and protection against ungulates for saplings (Farris & Filigheddu, 2008; Perea & Gil, 2014). The use of standing dead trees or the addition of artificial decoy posts may be an adequate alternative to reduce the rubbing impact of ungulates on saplings and juveniles, which seems to be the most deleterious effect of small-scale interventions in ungulate-dominated environments. In this regard the use of conifer posts is recommended given their rubbing and de-barking preference for these species. Other studies have shown that pine posts are attractive for ungulates to rub their antlers or horns (Charco *et al.*, 2016). Further studies should focus on the preferred diameter by Barbary sheep so that we ensure the maximum effectiveness for this management practice. Here, we showed that posts should have at least 1,30 m high and preferably more than 2,5 cm in dbh (Velamazán *et al.*, 2018a).

Since relative high densities of wild ungulates have proved to decrease regeneration probability and woody plant diversity (Perea *et al.*, 2015; Velamazán *et al.*, 2017) we recommend that the regeneration areas are sufficiently far away from supply points or attractive foraging areas (water, mineral correctors, feeding areas, crops, pastures, etc.), particularly to assure regeneration of threatened preferred species. It seems that to improve the whole forest regeneration cycle, ungulate populations management should be coordinated with forest vegetation management (Beguin *et al.*, 2016) to create a heterogeneous mosaic with different levels of ungulate pressure (Miller *et al.*, 2009) where low palatable species and highly preferred species can regenerate.

8.4 Fire prevention

Wildfires are important disturbances to understand the evolution of certain ecosystems. Naturally caused wildfires showed to play an important role in Mediterranean regeneration (Pausas *et al.*, 2009; Morales-Molino *et al.*, 2018) and historical data show the way they were used by humans to modify vegetation, mostly to favour agricultural lands and to increase grasslands for livestock (Morales-Molino *et al.*, 2017; Camarero *et al.*, 2018). However, wildfire risk is an important concern for forest conservation nowadays (Acacio *et al.*, 2009; Fischer *et al.*, 2016). It is especially important in areas, as the Mediterranean basin, where climate change is expected to rise temperatures and decrease precipitations (Huang *et al.*, 2016; Schlaepfer *et al.*, 2017). Therefore, prevention measures (e.g. firebreak networks) are implemented in order to hamper wildfire spread and facilitate the access of fire

brigades in case of wildfire (Mancilla-Leytón *et al.* 2013). The contribution of well-designed livestock programs to the maintenance of firebreaks has shown good results (Ruiz-Mirazo *et al.*, 2011; Ruiz-Mirazo & Robles, 2012), although our study focused on wild ungulate populations and firebreaks in combination with ungulate management structures (supply points). In this regard, we analyzed ungulate populations use and fuel load variation along a distance gradient from firebreaks/supply points. We used fuel load (woody plant phytovolume) as it is a good indicator of fire risk and can be controlled by livestock (Jauregui *et al.*, 2007; Mancilla-Leitón *et al.*, 2013).

One important result is that firebreaks define areas that attract ungulates, similarly to supply points (Velamazán *et al.*, 2018b), what confirms that the distribution of both man-made structures along the forest landscape is able to modify animal distribution and behaviour, as not only food but also shelter resources and habitat heterogeneity are strongly modified (Gordon *et al.*, 2004; Grosman *et al.*, 2011). The number of pellet groups along the distance gradient shows that firebreaks are more visited by ungulates than their surrounding areas (Velamazán *et al.*, 2018b). It is probably related to the differences with the surrounding matrix, as firebreaks with herbaceous vegetation provide highly nutritious food within woody systems (Ruiz-Mirazo *et al.*, 2011). These differences in food availability or shelter could explain the contrast with Suárez-Esteban *et al.* (2013a), that considered soft linear structures (firebreaks and dirt tracks) and showed this attractive effect for most frugivorous birds and

mammals, but not for certain wild ungulate species such as wild boar and deer.

There is a clear increasing number of pellets as distance to supply points decreases. Differences were only significant at 100 m, which indicates a high ungulate use around the supply points (0-50 m), covering almost an area of 1 ha around the point. Interestingly, relative abundance levels are higher in supply point transects than in firebreaks for any distance along the gradient. This could imply a different habitat use in each case, more intense around supply points. Therefore, supply points and their surrounding land are also more sensitive to possible ungulate damage and, hence, managers should avoid locating supply point in areas where threatened or sensitive plant species occur. Supply points represent heavily used areas by ungulates that can serve as suitable sites to monitor the type and intensity of ungulate damage (e.g., browsing, trampling or rubbing). They can also be used in forest planning to manage wild ungulate populations as resources and habitat heterogeneity are strongly modified (Gordon *et al.*, 2004).

Finally, ungulate effects on vegetation structure around firebreaks seem to be less efficient in controlling fuel load than supply points, at least at the study ungulate densities (<14 ind./km²). The effects are more evident in supply points, probably because firebreaks are long infrastructures that occupy vast areas while supply points concentrate higher ungulate pressure in less extensive areas. Nevertheless, for more concentrated attractive areas such as supply points ungulates were able to cause a strong fuel reduction. Location of supply points within firebreaks structure could

help increase ungulate habitat use and thus favor fuel reduction in fire-prone areas where maintenance of firebreaks is unlikely or costly. Further studies should evaluate the combined effects of firebreaks and supply points since their additive or synergistic effect can be of great use to inexpensively reduce fuel loads and maintain firebreaks. The addition of salt lick points, which are highly inexpensive, along the firebreak network could easily increase the use and maintenance of firebreaks at a very low cost. Further studies should also address possible differences across the variety of supply points (e.g. water, food and salt points) to evaluate the potential to attract ungulates and affect the vegetation at and around the supply points. In that regard, one might expect that water points will be more used in summer, supporting a strong temporal variation in the use of supply points. Similarly, feeding points will be more used in winter and summer, when food availability is lower. Therefore, spatio-temporal variation on the use of supply points is an important aspect to address in future studies.

9. FUTURE RESEARCH TOPICS

New research topics arise from the results of these studies:

1.- Effects on cultivated trees, particularly almond tree (the most sensitive and affected tree), should be addressed from an economic and social point of view to objectively quantify the damage caused by Barbary sheep. Hunting for fear or the use of other deterrent methods (e.g. fencing, plantation of non-preferred plants, etc.) should be evaluated to avoid or reduce ungulate damages in the farmlands.

2.- Behavioral ecology of Barbary sheep remains elusive. Intra- and inter-annual variability in habitat use as well as overall movements (depending on herd size, sex, etc.) across the landscape would provide a valuable information on the use of resources across space and time. Management could be therefore evaluated and improved according to their natural behavior.

3.- Analysis of Barbary sheep effects on other components of biodiversity such as arthropods, herbaceous plants, fungi or micro-organisms would be desirable to provide a more integrative approach of its effect on the system as a whole.

4.- Effects of Barbary sheep on other important processes such as nutrient cycling, seed dispersal, and energy flow would also provide a more holistic view. Affections to other biodiversity facets such as functional and phylogenetic diversity should be also analyzed.

5.- Forest gaps smaller than 0.75 ha appear to be small enough to avoid higher habitat use by Barbary sheep but sufficiently large to increase sunlight and recruitment growth (Velamazán *et al.*, 2018a). Further studies should analyze the effect of forest gap size on ungulate habitat use as well as the consequences for woody plant diversity and tree regeneration in order to estimate the most appropriate scale for intervention in areas where wild ungulates are abundant.

6.- We found lower number of species with unsustainable browsing damage in silvicultural treatments than in the control plots (Velamazán *et al.*, 2018a). This might be due to the lower shrub cover and, thus, lower food availability, in control (closed-canopy) plots as compared to treated areas. Further, more conclusive studies on the relationship between forest practices, shrub cover and browsing would be desirable to confirm this.

7.- Trampling effects were more conspicuous at shorter distance from man-made structures (firebreaks or supply points -Velamazán *et al.*, 2018b). Given the larger size of Barbary sheep compared to other wild ungulates such as Iberian ibex or mouflon, as well as their high sheep-like movement (San Miguel *et al.*, 2010), these animals will probably cause higher trampling damage than expected. Further studies should address trampling damage more thoroughly since it may directly affect vegetation cover and composition.

8.- Our results showed no clear pattern for regeneration presence/abundance along the distance gradient from firebreaks or supply points, what could be explained by high ungulate densities (Velamazán *et al.*, 2018b). Further studies should evaluate longer distance gradients from man-made structures in order to detect possible changes in plant regeneration and their associated shifts in woody plant communities.

9.- Ungulate damages were always higher at supply points than at firebreaks (Velamazán *et al.*, 2018b). Further studies should evaluate the combined effects of firebreaks and supply points and their differential ungulate use across time and space.

10.- We compared firebreaks and supply points, but we did not distinguish among water, food or salt points (Velamazán *et al.*, 2018b). Further studies should also address possible differences across the variety of supply points to evaluate the potential to attract ungulates and affect the vegetation at and around the supply points.

11.- Our results showed woody plant homogenization along the study gradient from firebreaks and supply points (Velamazán *et al.*, 2018b). Further research is needed at greater scales (e.g. landscape) in order to analyse spatial heterogeneity and the diversity and abundance of plant communities (mosaics) that may enrich overall biodiversity and the provision of ecosystem services.

10. CONCLUSIONS

1. Barbary sheep forage on a very wide variety of woody plant species: from small chamaephytes to trees. Small shrubs were the most preferred species and showed, on average, higher levels of browsing damage. However, browsing intensity was low in broadleaved trees and insignificant in conifers. Highly preferred and sparse shrub species might be severely affected by medium to high Barbary sheep densities.
2. The Barbary sheep could become an opportunity for arid environments that lack native wild ungulates and traditional livestock grazing, since it could partially fulfill most ecological roles played by the disappearing livestock. However, it could also become a threat should it become overabundant.
3. Threatened species were preferred over common plants, probably due to their greater palatability. Overall plant cover, including neighboring species, was also an important factor determining browsing damage and, thus, habitats with low ground cover should be particularly considered in conservation plans.
4. The use of threatened woody taxa through the studied ecological indicators (herbivore damage, plant preferences, habitat use and regeneration success) represented a useful tool to assess the sustainability of large herbivore introductions and to establish a priority conservation ranking for threatened plant species. These findings highlight the deleterious effects of overabundant ungulate populations regardless its origin (exotic or native) and the need of monitoring threatened woody

taxa to better estimate the suitability and sustainability of large herbivore introductions.

5. Small-scale forest interventions (< 0.75 ha) promoted advanced regeneration (large saplings and juveniles) and woody plant beta-diversity without increasing ungulate habitat use and detrimental browsing damage. Rubbing damage by ungulates was higher in the treated areas and no effect was found on woody plant alpha-diversity.

6. Ungulate habitat use decreased with distance to structures. Both supply points and firebreaks also showed greater plant damage (browsing, trampling and rubbing) and lower fuel load as distance to structures decreased. Effects were rather constant within 50 m around the structures, indicating a non-linear relationship along the distance gradient.

7. Woody species richness increased with distance to firebreaks and supply points but changes in plant composition (beta-diversity) were small (0.80–0.87 similarity), primarily due to limited regeneration and heavy use by ungulates along the gradient, which mainly favored the regeneration of ungulate-resistant plants (non-palatable or resprouting).

8. Supply points showed significantly greater ungulate effects as compared to firebreaks at any distance for all the study variables. Supply points had, on average, 1.8 times greater browsing damage than firebreaks at any distance from the boundary. Similarly, fuel load between 100 and 10 m from the structures was reduced by 78% in supply points but only by 20% in firebreaks.

9. Fire-prone areas within the landscape can benefit from the combined use of firebreaks and supply points to reduce fuel load at a low cost but regeneration of ungulate-sensitive and threatened taxa should be considered.

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