Universitat Autònoma de Barcelona

Biodiversity conservation: Between protected areas and local communities
A case study in Picos de Europa National Park (northern Spain)

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PhD Dissertation

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Abstract

There is an on-going debate on how to manage protected areas for effective long-term biodiversity conservation. Some authors embrace passive management approaches reducing human intervention in protected areas. This approach may be suitable for restoring natural ecosystems processes in large-scale abandoned areas. However, with a terrestrial surface increasingly dominated by human activities, other authors argue that conservation efforts should also pay attention to the role of humans on natural systems and resolve how to achieve biodiversity conservation without compromising the livelihood of the local communities living near or within to the protected areas. For this school of thought, traditional practices based in common resource management systems can help guaranteeing long-term biodiversity conservation.

This thesis examines traditional practices applied in forest commons and their potential impacts on biodiversity, aiming at identifying human activities that are favorable to biodiversity and that could be therefore used to maintain biodiversity on human-dominated landscapes. To do so, an interdisciplinary methodological approach is applied combining conventional analytical frameworks used in biological conservation science –i.e., direct measures of biodiversity such as species richness and evenness– and social analytical tools –i.e., ethnobiological and historical approaches.

Specifically, this thesis investigates the ecological outcomes of traditional practices applied in forest commons in Spain, a country with long history of forest community-ownership. First, through a review of the literature of the historical evolution of Spanish forest commons, this study examines management practices conducted during the performance of traditional livelihood activities applied by forest-dwelling communities that may have benefitted forest biodiversity and the impacts on biodiversity derived from replacing such practices by other management forms. Second, using a case study, this research explores the effectiveness of formally protecting an area on preserving species diversity compared to traditional management systems allowing local communities use of ecological resources. Data collection included botanical inventories as well as topographic, edaphic, and anthropogenic impact data from 50 0.2-hectares concentric plots distributed

through neighboring forest commons inside and outside a protected area classified as an IUCN category II (National Park). In the final part of the thesis, qualitative data from 42 interviews to residents of the studied area are used to document traditional forest-related management practices shaping regional landscape mosaic and local perceptions of recent landscape changes.

Results from the literature review illustrate that, at the national level, interventionism and privatization of forest commons in Spain during the nineteenth and twentieth centuries had negative consequences for forest biodiversity. At a local level, results of the study case do not support the idea that protected areas hold more biodiversity than surrounding areas and suggest that human factors are important drivers of tree species distribution. Results from this work also help identify a set of traditional management practices favorable to regional landscape patchiness and the maintenance of forest systems. Finally, information from local perception of historical landscape transformation in the study area suggests that local communities might be a valid source of information to monitor ongoing ecological changes.

The results of this dissertation indicate that certain traditional practices carried out in community-based resource management systems in the performance of their traditional activities are biodiversity-friendly. This finding might help in the design of biodiversity conservation efforts linking biodiversity maintenance and local development, which might be particularly relevant in the establishment of protected areas in populated zones.

Keywords: Biodiversity conservation; community-based resource management; ethnobiology; forest commons; protected area; rural history.

Resumen

Existe un debate entorno a cómo gestionar las áreas protegidas para lograr una conservación efectiva de la biodiversidad a largo plazo. Algunos autores adoptan un enfoque de gestión pasiva basado en limitar la intervención humana en áreas protegidas. Dicho enfoque puede ser adecuado para la restauración de procesos ecológicos naturales en extensas áreas abandonadas. Sin embargo, con una superficie terrestre cada vez más ocupada por actividades humanas, otros autores consideran que las iniciativas de conservación deberían considerar también el papel de las personas en los sistemas naturales y tratar de alcanzar la conservación de la biodiversidad sin comprometer el bienestar de las comunidades locales asentadas en el interior o en los terrenos adyacentes a los espacios protegidos. Esta perspectiva considera que las prácticas tradicionales basadas en sistemas de gestión de recursos comunitarios pueden ayudar a garantizar la conservación a largo plazo de la biodiversidad.

Esta tesis examina las prácticas tradicionales llevadas a cabo en montes comunales y su potencial impacto en la biodiversidad, buscando identificar actividades humanas favorables para la biodiversidad y que puedan ser empleadas para mantener la biodiversidad en paisajes dominados por los humanos. Para ello, se aplica un enfoque metodológico interdisciplinar que combina marcos analíticos convencionales empleados en ciencias biológicas –esto es, medidas de biodiversidad directas como riqueza y abundancia de especies– y herramientas analíticas sociales –esto es, enfoques etnobiológicos e históricos–.

Específicamente, esta tesis investiga los impactos ecológicos de prácticas tradicionales llevadas a cabo en montes comunales de España, un país de larga tradición de propiedad comunitaria de montes. Primeramente, mediante una revisión literaria de la evolución histórica de los montes comunales españoles, este estudio examina, por un lado, las prácticas de gestión llevadas a cabo por comunidades rurales como medio tradicional de subsistencia que han podido beneficiar la biodiversidad forestal, y por otro, los impactos en la biodiversidad debidos a la sustitución de estas prácticas por otras formas de gestión. En segundo lugar, mediante el empleo de un caso de estudio, esta investigación explora la efectividad de un área protegida para preservar la diversidad de especies en comparación a sistemas tradicionales de manejo que permiten el uso de los recursos ecológicos por parte de las comunidades locales. La toma de datos de campo incluye inventarios botánicos así como datos topográficos, edáficos y de impacto antrópico en 50 parcelas concéntricas de 0.2 hectáreas distribuidas a lo largo de montes comunales dentro y fuera de un espacio protegido clasificado como categoría II de la IUCN (Parque Nacional). En la parte final de la tesis, los datos cualitativos obtenidos a partir de 42 entrevistas a residentes del área de estudio son empleados para documentar las prácticas tradicionales de gestión forestal que confieren el mosaico paisajístico tradicional y las percepciones locales de cambios paisajísticos recientes.

Los resultados obtenidos de la revisión literaria muestran que, a nivel nacional, la intervención y privatización de los montes comunales en España durante los siglos XIX y XX tuvo consecuencias negativas para la biodiversidad forestal. A nivel local, los resultados del caso de estudio no corroboran la idea de que las áreas protegidas albergan mayor biodiversidad que las áreas no protegidas y sugieren que los factores humanos son importantes condicionantes de la distribución de especies arbóreas. Los resultados de este trabajo también ayudan a identificar una serie de prácticas tradicionales de gestión beneficiosas para la heterogeneidad paisajística regional y el mantenimiento de los ecosistemas forestales. Finalmente, la información recogida a partir de las percepciones locales de la transformación histórica del paisaje en el área de estudio sugiere que las comunidades locales pueden ser una fuente de información válida para el seguimiento de cambios ecológicos.

Los resultados de esta tesis indican que ciertas prácticas tradicionales llevadas a cabo en sistemas de gestión de recursos comunitarios permiten la presencia de especies sin perjudicar el bienestar de las comunidades locales. Estos resultados pueden ser de utilidad para el diseño de iniciativas de conservación de la biodiversidad que busquen tanto el mantenimiento de especies como el desarrollo local, lo cual puede ser particularmente relevante para el establecimiento de áreas protegidas en zonas habitadas.

Keywords: Área protegida; conservación de la biodiversidad; etnobiología; historia rural; montes comunales; sistemas de gestión de recursos comunitarios.

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Chapter I

Introduction

1.1 Statement of the thesis: Biodiversity conservation

The term "biodiversity" refers to the variety of genes, species and ecosystems, occurring in a given system (Redford and Mace, 2018). Human societies depend on biodiversity to obtain essential goods like food, fiber, and potable water and for the correct provision of ecosystem services such as biomass production or nutrient and water recycling (Díaz et al., 2006; Cardinale et al., 2012). Over the past four decades, biodiversity has declined in such alarming rate that the present rate of species loss is estimated to be 1,000 times higher than the background extinction rates typical over Earth's history, occurring across the globe in all taxonomic groups (Chapin et al., 2000). According to Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), the main anthropogenic drivers of biodiversity loss are land use change, over-exploitation and unsustainable use of natural resources, introduction of invasive alien species, climate change, and pollution (IPBES, 2016).

Aware of the importance of biodiversity for human well-being, the international community has put in place diverse agreements to reduce or mitigate the rate of biodiversity loss. The establishment and maintenance of a global network of protected areas has been a key strategy for achieve it (Gaston et al., 2008; Leverington et al., 2010; Geldmann et al., 2013). Over the last decades, an increasing number of protected areas has been established worldwide, with almost 15 per cent of terrestrial and inland water areas and almost 17 per cent of coastal and marine areas under some category of protection in 2018 (UNEP-WCMC, 2018). However, despite the remarkable effort towards the designation of protected areas, global biodiversity continues to decline (Butchart et al., 2010), prompting discussion about the real effectiveness of protected areas to halt species and natural habitats loss (Joppa et al., 2008; Hirschnitz-Garbers and Stoll-Kleemann, 2011; Coetzee et al., 2014).

There are several reasons why the designation of protected areas might not suffice to achieve conservation goals. First, research suggests that granting an area with a protection status does not necessarily leads to the protection of the species and ecosystems on it, as conservation initiatives are not always effective or they fail to be enforced (Dudley et al., 2005). A recent global evaluation of protected areas management effectiveness estimated that only 22 per cent of protected areas have a successful management capacity (Leverington et al., 2010). In other words, many of the world's protected areas exist only as 'paper parks', i.e., areas set aside for protection on paper, but where the lack enforcement does not prevent the realization of activities that affect biodiversity conservation (Anderies and Janssen, 2016).

Second, the limited temporal extent of most reserves, which rarely are more than 100-years old, often relies on management strategies that consider protected areas as static entities, rather than as part of landscape dynamics (Bengtsson et al., 2003). However, over time, natural (e.g., pests and disease, flooding, windthrows) and human disturbances (e.g., fire regime, livestock grazing) influence species composition of natural areas by shaping more-or-less favorable environmental conditions to different species assemblages. For instance, human activities such as thinning, hay cutting or pollarding brought major landscape changes in European deciduous forests shaping a grassland-woodland habitat mosaic (Bradshaw and Hannon, 2006). Awareness that the biodiversity found in protected areas is subject to disturbances is raising among conservation scholars, who advocate for the creation of more resilient ecosystems to face global environmental threats (Bengtsson et al., 2003; Cumming et al., 2015). Along these lines, considerable attention is being given to enhance protected areas resilience by developing integrative management approaches that contemplate interactions between human use of nature and biodiversity maintenance (Berkes and Turner, 2006; Kareiva and Marvier 2012; Cumming et al., 2015).

Third, gene flow is needed to maintain viable populations of species, for which species trapped in an isolated protected area without opportunity of natural exchange of genes suffer from gradual decline (Dudley et al., 2004). Therefore, protected areas isolated from similar habitats have limited usefulness in the long-term conservation of many species unless they are very large (Dudley et al., 2004; Gaston et al., 2008). Implementing strategies to ensure habitat connectivity between protected areas is key for improving conservation effectiveness. In order to achieve this goal, conservationists nowadays seek to reconcile other land uses known to benefit stress-tolerant and habitat-specialist species at a local and landscape level, such as low-intensity agriculture, forestry or agroforestry, with biodiversity conservation to promote a wildlife-friendly matrix suitable for the passage of species dispersal surrounding the protected areas (Kremen, 2015). The debate now refers to how to make the activities in the matrix biodiversity-friendly without impacting human livelihoods and well-being (Babai et al., 2015; Kremen, 2015).

The need to address the limitations of the strategy of using mostly protected areas for the preservation of global biodiversity is reflected on most recent international agreements, such as in the Convention on Biological Diversity, in which Aichi target 11 contemplates including ‘other effective area-based conservation measures’ (OECMs) as figures of protection of species and ecosystems. OECMs refer to territories that effectively conserve biodiversity but which are not recognized as protected areas and have been directly linked to areas conserved by Indigenous peoples and local communities (Jonas et al., 2017). Through traditional management practices, Indigenous peoples and local communities have allowed the persistence of areas of particular importance for biodiversity across the globe, drawing the attention of policy makers and conservationists to reconsider the role of Indigenous peoples and local communities in natural resources use and conservation (Agrawal and Gibson, 1999). Indeed, these territories comprise at least 40 per cent of worldwide biodiversity-rich areas (Garnett et al., 2018). Prior research recognizes the critical role played by the ecological knowledge embedded in Indigenous peoples and local communities’ customary practices to sustain both people and their environments (Gadgil et al., 2003; Berkes and Turner, 2006; Preuss and Dixon, 2012; Hernández-Morcillo et al., 2014). Moreover, scholars and international organizations increasingly recognize the potential of combining Indigenous and local knowledge with scientific research to produce robust and effective conservation outcomes (Riseth, 2007; Preuss and Dixon, 2012; Pardo-de-Santayana et al., 2014; Nash et al., 2016; Periago et al., 2017; Thaman et al., 2017).

In this dissertation, I examine whether specific management techniques carried out by local communities have positive effects for ecosystem and species diversity. Specifically, (i) I use a historical approach to analyze the potential of community-based resource management for forest biodiversity conservation, (ii) I study differences of biodiversity between deciduous forests inside and outside a protected area in a territory traditionally managed by local communities as a common, and then (iii) I identify natural resource management practices that might promote species and habitat diversity at the landscape level. The study area is located in the Liébana valley, a mountain region of Northern Spain that preserved a complex farming system based on traditional land use activities until the mid-twentieth century. The historical dependence of rural mountain societies on their surrounding natural resources strongly engaged them in the monitoring of resources to prevent mismanagement and over-exploitation (Moreno, 1998; Piqueras, 2002). The progressive abandonment of traditional natural resources exploitation that has taken place in the area since the mid-twentieth century, when the rural exodus and the entrance of market economies strongly disrupted the traditional farming system of the region, offers a unique context to analyze the implications for biodiversity of traditional management practices.

1.2 Theoretical background

The protection of large and wild areas from detrimental human activities to achieve biodiversity conservation goes back to 1872, with the establishment of the North American Yellowstone National Park (Zube and Busch, 1990). This approach emphasizes the need to preserve species and natural habitats by the strict regulation of human activities, and it is based on the idea that pristine ecosystems persisted until the very recent past (Denevan, 1992). However, more often than not, protection regulations have been implemented in areas inhabited by Indigenous groups, resulting in the dispossession or displacement of the communities living inside the created reserve and raising conflicts among reserve managers and dwellers that heavily constrain conservation outcomes (West et al., 2006; Riseth, 2007; Hirschnitz-Garbers and Stoll-Kleemann, 2011).

As the failures of exclusionary conservation initiatives became increasingly evident, scholars embraced more integrative framings to include social dimensions in biodiversity conservation efforts (Brown, 2002; Berkes, 2004). Since 1970s, terms such as social-ecological systems, resilience, or cultural landscape have emerged in the conservation literature. While these represent different approaches, they all consider the interdependence of nature and people in the establishment and management of areas where biodiversity conservation should be prioritize and adopt more people-oriented approaches to conservation (Shultis and Heffer, 2016). Among these new narratives, 'community-based conservation' stands as one of the most relevant approaches to implement more socially-sensitive conservation (Berkes, 2007; Hirschnitz-Garbers and Stoll-Kleemann, 2011; Brooks et al., 2013).

Community-based conservation proposes that "if conservation and development can be simultaneously achieved, then the interests of both could be served" (Berkes, 2004:621). According to this approach, management initiatives should focus not only in ecological, but also economic and social outcomes for local communities living near or within protected areas. Despite the wide acceptance of the community-based conservation perspective among scholars, important questions arise as seeking local communities' welfare and development may not be compatible with biodiversity protection (Hayes, 2006). Indeed, studies assessing the efficacy of community-conserved areas reveal that these initiatives often prioritize one outcome, either conservation or development, but rarely deal simultaneously with both (Brooks et al., 2013).

The lack of ability to simultaneously address both objectives is partly attributed to the low capacity of conventional resource-management science to conceive multiple objectives (Berkes, 2007), an aspect that according to some authors could be complemented by the knowledge hold by Indigenous groups and local communities (Folke, 2004; Berkes, 2007). Such knowledge is usually called traditional ecological knowledge (TEK) and has been defined as "the body of knowledge and beliefs about the relations of specific human societies to the local environments in which they live, as well as their local practices for ecosystem use and stewardship" (Hernández-Morcillo et al., 2014:3). Evidence from previous studies suggests that TEK can complement scientific ecological knowledge in the quest to conserve biodiversity and sustainably manage natural resources (Donovan and Puri, 2004) and protected areas (Vizina and Kobei, 2017). Through a number of practices such as social taboos, spiritual beliefs or the establishment of sacred site guardians, Indigenous peoples and local communities have protected their surrounding biological resources (McPherson et al., 2016; Karst, 2017; Samakov and Berkes 2017). Also through their small and intermediate-scale disturbances, such as traditional farming systems, or through a set of site-specific norms to manage resources sustainably, such as restricting overharvest of fuelwood trees, Indigenous groups have also contributed to preserve some species or landscape patchiness (Potee and Ostrom, 2004; Babai et al., 2015). Although nature conservation might not necessarily be the objective of management practices based on TEK, but rather a consequence of them (Berkes et al., 2000), examining which particular practices allow species to persist in Indigenous peoples and local communities territories might generate important insights to integrative conservation approaches.

This dissertation explores whether certain traditional management practices result in resources and ecosystem sustainable management. To that end, I examine a type of community-based resource management system critical to biodiversity conservation and human livelihood whose customary practices are rooted in TEK: traditional community forests or forest commons. The concept "forest commons" refers to woodlands collectively managed by local communities, considering a community as a social group living in a small spatial unit, with a homogeneous social structure, frequent interactions, and shared interests and norms (Agrawal and Gibson, 1999). Forest commons are characterized by having defined boundaries and legal enforceable property rights and by providing resources to the social groups involved in their management (Aryeetey et al., 2012). Nowadays, forests commons constitute 18 per cent of forest area globally, contributing significantly to biodiversity conservation and to the rural household economies of more than a billion people through multiple forest-related products such as fodder, timber, firewood, fruits and game (Chhatre and Agrawal, 2008; Angelsen et al., 2014).

The importance of forest commons to the livelihood strategies of the social groups involved in their management has often resulted in local forms of use and regulation of forest-related resources (e.g., customary rules of use and exclusion) and in local-level governance institutions to regulate their use (e.g., village councils). For instance, Sight et al. (2013:12) identified an informal indigenous institution in eastern Himalaya, the *Kebang*, which was "responsible for taking collective decisions" related to biodiversity conservation. Particularly, these institutions were responsible for the establishment of local norms on the sustainable use of natural resources and for the enforcement of a system sanctioning users who not followed the rules-in-use. Additionally, other social mechanisms, such as religious beliefs or sacred objects, also seem to play a crucial role to forest biodiversity conservation (Joa et al., 2018). In some ethnic communities in China, for example, the only human activity carried out in sacred forest commons is mushrooms collection, which has allowed the persistence of virgin forests to the present day (Yeo-Chang et al., 2012).

In order to identify people's traditional management practices that may be useful for biodiversity conservation in human-dominated landscapes, I apply an interdisciplinary methodological approach that combines conventional analytical frameworks used in biological conservation science –i.e., direct measures of biodiversity such as species richness and evenness– and social analytical tools –i.e., ethnobiological and historical approaches–. Although integrative conservation frameworks encourage interdisciplinary methodological approaches for a better understanding of the interactions between nature and human systems (Berkes, 2004; Bennet et al., 2017), few studies integrate social-ecological approaches on their methodology. This research contributes towards the integrative study of biological and social phenomena using a case-study approach to obtain further in-depth information on interactions between nature and human systems and to determine the potential of traditional management practices for biodiversity conservation.

1.3 Selected case study: the Liébana valley

The Liébana valley was identified as a suitable case for research due to its socioecological characteristics. The communities in Liébana remained mainly self-sufficient until the mid-twentieth century applying a community-based natural resources management system of the forests surrounding them. The progressive abandonment of traditional land uses in the last six decades provides an opportunity to examine whether the loss of certain traditional management practices might have been detrimental to the long-term maintenance of biodiversity. In addition, the recent establishment of a strict protected area in the northwestern municipalities of Liébana, i.e., Picos de Europa National Park, brings the opportunity to test the effectiveness of conservation initiatives to preserve biodiversity.

1.3.1 Ecological and socioeconomic context of the study area

The Liébana valley is located in the southwest of the Cantabria region, in north Spain (Figure 1.1). It comprises a mountain region of 57,400 hectares consisting of steep reliefs, with altitudes ranging from 300 meters above sea level at the bottom of the valley to 2600 meters in the surrounding mountain system, a physiographical feature that results in relative isolation from the adjacent regions.

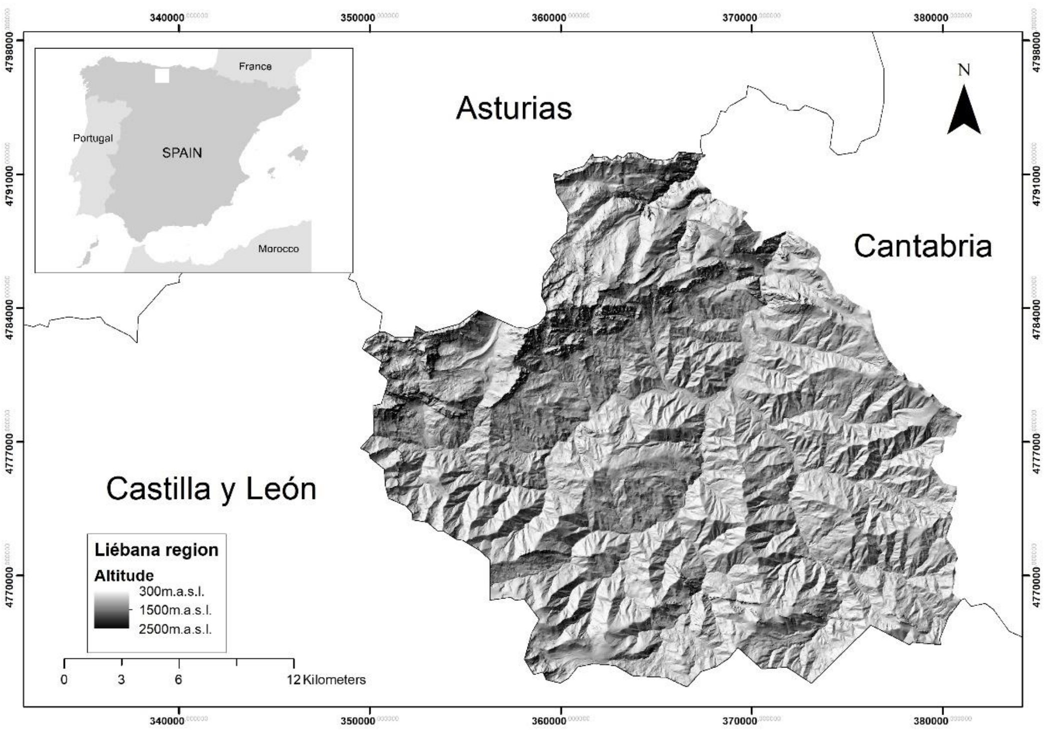

Figure 1.1. Geographical location and land surface elevation of the study area.

The physiography of the territory greatly influences other features such as its climate, vegetation, and hydrology. Thus, the Atlantic climate of the Cantabria region turns into a Mediterranean climate in the bottom of the valley and into subalpine and alpine zones in its highest altitudes. This range of climatic conditions promotes the establishment of different forest ecosystems covering the altitudinal range, from perennial and deciduous lowland forests of oak (*Quercus* spp.) or beech (*Fagus sylvatica*), to coniferous montane stands of yew (*Taxus baccata*). Physiography also influences the basin's hydrology, forming a drainage network divided in four river systems that converge in Potes, the geographic and administrative center of the region (ETSIM, 1978).

From a socioeconomic perspective, the geographical isolation of the Liébana valley until the first third of the twentieth century resulted in the maintenance of a subsistence strategy among local communities, who until recently produced their own food and energy using resources from their immediate surroundings (Arbeo, 2012). Until 1960, agriculture and livestock farming were the main economic activities pursued by Liébana's families, who cultivated wheat, rye, oats, legumes, potatoes, and maize. Agricultural production also included apple and pear orchards and vineyards, which were located around human settlements, and meadows for forage production on steeper slopes or areas more distant from the villages (López, 1978; Castañón and Frochoso, 2007).

Livestock farming consisted on extensive grazing of cattle and herds in pastureland and summer highlands. However, this economic activity was subordinated to the agricultural farming system, focusing in raising draft livestock for agricultural work and not for dairy or meat production. For instance, the *Lebaniega* cattle breed was optimal for agricultural tasks and transportation but was minimally profitable for production of milk and meat. Liébana's livestock also included sheep, goats, pigs and horses (Table 1.1).

Table 1.1. Livestock census in Liébana in 1865 (adapted from Arbeo, 2012)

Livestock	Human consumption	Agricultural task	Transport	Reproduction and farming	Total
Cattle	182	3.438	53	5.491	9.184
Sheep	2.206	-	-	13.669	15.875
Goat	1.495	-	-	8.187	9.682
Pigs	2.391	-	-	3.918	6.309
Horses	-	4	356	374	734

Forest-related resources also played an important role in pre-industrial Liébana, providing fodder to domestic livestock, fruits, game and fishing to human consumption, timber for house construction, and tools that were sold to the neighboring region of Castilla y León in exchange of cereal (Ezquerra and Gil, 2004). Altogether resulted in a grassland-woodland habitat mosaic on Liébana's landscape consisting in dispersed human settlements immediately surrounded by cereal crops and orchards, followed by a mixture of arable lands, meadows and woodlands in the lowlands, substituted by highland pastures and grasslands in the upper parts of the region (López, 1978).

Liébana's traditional landscape persisted until the mid-twentieth century, when a combination of demographic (i.e., rural exodus) and economic issues (i.e., integration to national market economy) resulted in the progressive abandonment of traditional land uses in the area. For instance, local farmers adopted a more intensive livestock farming system to respond to national market demands of meat and dairy products (Corbera, 2006). Moreover, since 1960s, the economic activity of Liébana has moved to the tertiary sector, particularly rural tourism, and a large part of the territory is nowadays occupied by touristic sector demands for accommodation and transport infrastructures and leisure activities (González, 2016).

1.3.2 Land tenure organization and communal use of natural resources

Communal property is a key feature in the Liébana region, with almost 80 per cent of the territory under this land tenure regime, which historically was oriented to extensive livestock herding, firewood collection, and timber harvesting for housing or caving (Arbeo, 2012). Local rules over Liébana's communal resources date back at least to the fifteenth century, being the oldest regulations documented in the Cantabria region. These local-level regulations responded to the particular geographical features of the region and, subsequently, to the economic activities pursued by Liébana's families (Pérez-Bustamante and Baró, 1988).

Shared resources were governed by local institutions known as village councils, or *concejos*, which were participatory assemblies involved in the local forms of use and regulation of the resources (López, 1978). Rules issued by village councils had a direct influence in the local economic activities through the intervention of products, such as strict local rules issued during the sixteenth and seventeenth centuries limiting the trade of wood wheels with the neighboring regions. It was compulsory for community members to attend the village council assemblies, as well as shift turns to carry out certain duties including the punishment of illegal felling practices or the monitoring of the construction of shieling huts for livestock on the common summer pastures (Figure 1.2) (Pérez-Bustamante and Baró, 1988).

Figure 1.2. Shieling huts or *invernales* in the summer pastures of Bejes (Liébana, Cantabria). Photo credits: S. Guadilla-Sáez.

Nowadays, *concejos* in Liébana are regulated by Law No. 6/1994 of Cantabrian Regional Government, consisting of one Council President and two Council members elected by the neighborhoods themselves that still have authority over the management and use of the resources in their territories. For instance, the management interventions applied by the regional forestry administration intervention in Liébana's forest commons need the approval of the neighborhood councils.

1.3.3 Nature protection and social conflicts: Picos de Europa National Park

Nature conservation policy in Spain begins with the promulgation of the first Spanish National Park Act in December 1916 and follows the North American model of preserving natural areas by preventing human interventions within them (Voth, 2007). In 1918, Spain declared its first national park in the western part of Picos de Europa mountain range (Asturias region), a natural landscape that, in contrast to the undisturbed, almost pristine American areas, had been altered by humans for five millennia (Rico, 2006). The declaration of a National Park in local communities' ancestral lands –almost 94 per cent of the designated territory was communal property– came with the restriction of traditional practices of pastoralism, timber harvesting and hunting, and consequently led to conflicts between communities and conservationists during the twentieth century (Castañón y Frochoso, 2007).

In 1995, conflicts with local communities increased due to National Park's enlargement to the adjacent territories of Picos de Europa mountain range, an enlargement that actually included human settlements inside park's boundaries (Voth, 2007). Three municipalities of Liébana were included in this extension: Camaleño, Tresviso and Cillorigo de Liébana (Figure 1.3). According to Law No. 4/1989 of Spanish Government, a management plan to implement conservation strategies and to regulate land uses inside the reserve through zonation should be implemented in a maximum of one-year delay after the extension. However, the management plan was not approved until 2002 (Royal Decree No. 384/2002 of Spanish Government). The plan, however, was repealed in 2005 due to a legal action taken by local communities living within Picos de Europa buffer zone (Spanish Government, 2005), resulting in the lack of a management capacity of the park that persists to present day.

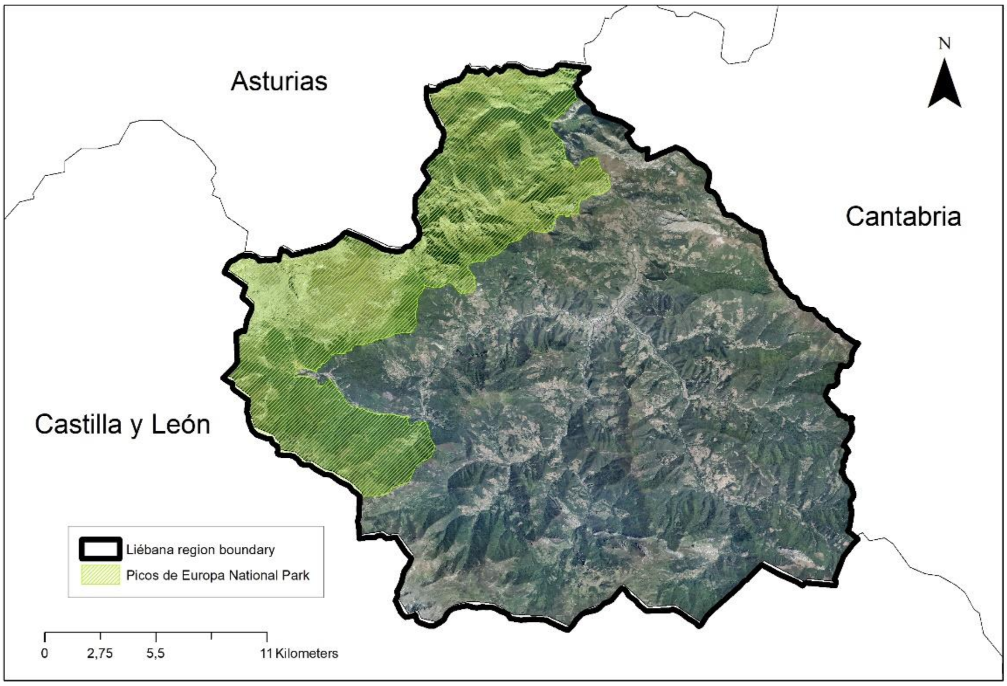

Figure 1.3. Location of Picos de Europa National Park in the Liébana region.

1.4 Aims and outline of the dissertation

The overall aim of this dissertation is to get a better understanding of the role that traditional management practices carried out in forest commons might hold to foster biodiversity conservation and sustainable management, while allowing local communities use of ecological resources. To do so, I formulated three specific objectives that guided the empirical research.

The specific objectives are:

1. at the national level, to examine the potential of community-based management for preserving diverse, biodiversity-rich forest ecosystems (Chapter II);

2. at the regional level, to assess the effectiveness of protected areas in maintaining species diversity in comparison to neighboring unprotected sites (Chapter III); and

3. at a landscape level, to identify traditional management practices potentially beneficial to species and habitat diversity (Chapter IV).

This dissertation is structured on this general introduction, three chapters that report results from empirical research, and a final general discussion.

After this general introduction, Chapter II explores the implications for biodiversity conservation of replacing a community-based resource management system by private and public ownership and management forms. The chapter is based in an extensive historical analysis of the replacement of traditional community forest management by private and state management in Spain. This chapter corresponds to the article 'Community-based approaches to improve nature conservation: the example of Spanish forest commons', submitted to the journal *Forest Policy and Economic* in January 2019.

Chapter III quantifies the biodiversity status in three types of neighboring temperate deciduous forests in Liébana comparing forest types under protected and non-protected status. I provide empirical data that allows to analyze the role of protected areas for forest biodiversity persistence and analyze the particular ecological and anthropogenic variables that explain tree species distribution in the study area. This chapter corresponds to the paper 'Biodiversity conservation effectiveness provided by a protection status in temperate forest commons of north Spain', published in the journal *Forest Ecology and Management* in February 2019 (Guadilla-Sáez et al., 2019).

Chapter IV analyses local perceptions of land use changes occurred in Liébana since the mid-twentieth century due to the abandonment of traditional practices and the locally perceived ecological implications of such changes. I compare qualitative information provided by local informants with information from previous empirical studies in the area to document i) landscape historical dynamics and ii) the local knowledge embedded in traditional management reported to be favorable to the maintenance of biodiversity. This chapter corresponds to the article 'The role of traditional management practices in shaping a species-rich habitat mosaic in a mountain region of north Spain', submitted to the journal *Land Use Policy* in March 2019.

Chapter V provides a brief discussion of the key findings of this research and its main theoretical and methodological contributions. It also includes the practical implications of this research and suggests potential areas for future work.

Finally, Appendix I presents a list of supporting publications to this PhD research and Appendices II-III further supplementary information.

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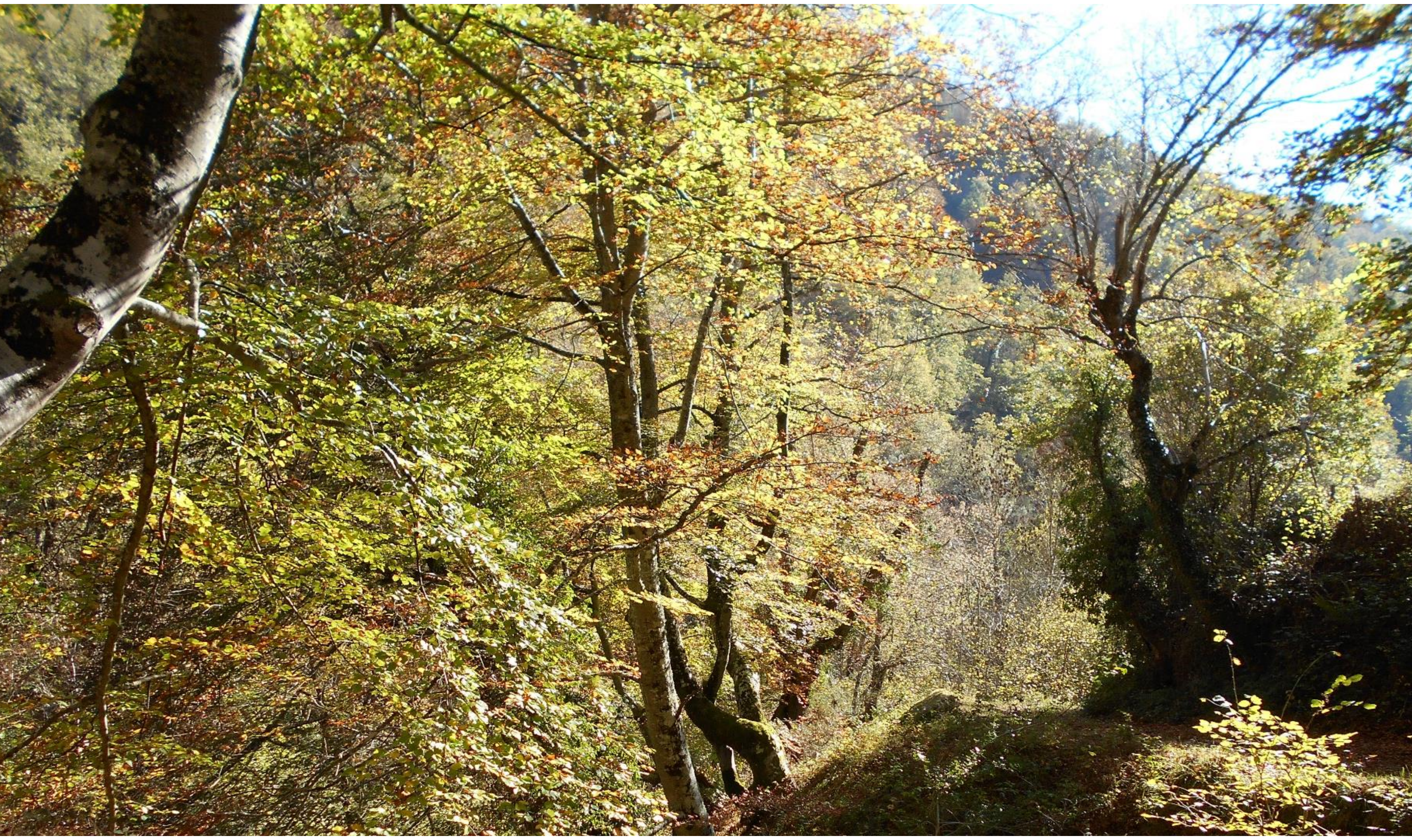
Chapter II



Mixed broadleaf woodland in forest commons of Liébana (Cantabria, Spain). Photo credits: S.Guadilla-Sáez

Community-based approaches to improve nature conservation: the example of Spanish forest commons

This chapter corresponds to the article: Guadilla-Sáez, S., Pardo-de-Santayana, M. & Reyes-García, V. Community-based approaches to improve nature conservation: the example of Spanish forest commons. In review. *Forest Policy and Economics*

Paper I

Community-based approaches to improve nature conservation: the example of Spanish forest commons

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Abstract

To date, the backbone official instrument for biodiversity conservation has been the establishment and maintenance of a system of legally recognized protected areas. However, the effectiveness of this network of protected areas to halt biodiversity loss is much debated, and therefore approaches to reconcile other land uses with biodiversity conservation are increasingly being adopted. Here, we use a historical approach to analyze the potential of community-based resource management for promoting biodiversity friendly and economically profitable management systems. Specifically, we examine the ecological implications of the historical replacement of traditional community-based systems by other types of land ownership and management systems. Particularly we focus on the process of privatization and interventionism of traditional community-ownership forests that took place in the nineteenth and twentieth centuries in Spain. Our review indicates that the replacement of traditional community-based management systems might have had negative consequences for forest biodiversity. On the short term, privatization resulted in the cut of the woodlots acquired to reimburse the cash value of the purchase. Moreover, interventionism resulted in the arise of illegal practices, partly as opposition to the dismantling of historical use rights, partly because local communities lost their authority to sanction illegal uses. On the long term, the abandonment of traditional forest-related management practices led to the densification and homogenization of the rural landscape mosaic, increasing the wildfire risks and reducing biodiversity associated to ecosystems dependent on human practices. Our interpretation of historical process finds support in current biodiversity distribution. Notably, in rural areas where –due to a strong local opposition to the appropriation process– community ownership and use rights had been restored, forests commons overlap with important areas for conservation. These results further support the idea that community-based resource management can provide useful insights for designing conservation strategies that complement the network of protected areas.

Keywords: Biodiversity conservation; community forests; historical review; rural history; sustainability.

2.1 Introduction

The term 'biodiversity' refers to the great variety of life forms and the high diversity of interactions and processes that occur at the many levels of biological organization (McElhinny et al., 2005). Despite its relatively recent introduction in the scientific literature –Walter Rosen was the first to use the word in 1986–, the term has been widely adopted by the general public. Over the last decades, biodiversity conservation has become a target for many international organizations, as reflected in the signature of diverse international agreements for biodiversity conservation. Moreover, understanding what are the many threats to biodiversity (e.g., growing human population, land use change, overuse of natural resources, environmental degradation) and finding strategies to mitigate them are major sources of social concern (McElhinny et al., 2005).

To date, the backbone official instrument for biodiversity conservation has been the establishment and maintenance of a system of legally recognized protected areas (Gaston et al., 2008; Gray et al., 2016). However, the efficacy –in terms of biodiversity maintenance– of the different protection categories is much debated (Hayes, 2006; Geldmann et al., 2013; Coetzee et al., 2014). Several studies suggest that, overall, protected areas do help to protect biodiversity, although researchers have also noted that granting an area the 'protected' status does not necessarily leads to biodiversity protection, as regulations designed to protect biodiversity are not always effective or not sufficiently enforced (Bruner et al., 2001; Dudley et al., 2004; Dudley et al., 2005). While some conservationists argue that the solution to that just lies in ensuring compliance with regulations, others posit that efforts should also be directed to maintain biodiversity outside the physical boundaries of protected areas. Following this logic, approaches oriented to conserve biodiversity beyond the network of designated areas are increasingly being adopted worldwide (Poiani et al., 2000; Mathur and Sinha, 2008; Guadilla-Sáez et al., 2019).

Of particular interest are approaches aiming to reconcile extractive land uses, such as agriculture, forestry or mining, with biodiversity conservation, as these approaches could complement protected areas (Kremen, 2015). This perspective seems to have been widely adopted by the international forestry community, which increasingly advocates for implementing management systems that combine economic benefits and sustainable use of forest-related resources (Hernando et al., 2010). Accordingly, several authors have analysed whether multipurpose forest management strategies actually provide profitable forest-related uses without compromising biodiversity conservation. Although it has already been established that owners' willingness to combine nature-oriented and economic uses of forests is determinant for the establishment of conservative management objectives (Nielsen et al., 2017; Bergstén et al., 2018; Pynnönen et al., 2018; Weiss et al., 2018), many of these studies do not include considerations on the effects of land ownership type on biodiversity. Moreover, the few studies that include proprietorship in the analyses mainly focus on comparing public and private forest ownership forms, despite the literature stressing the potential of traditional community ownership for guaranteeing long-term forest resources conservation (Agrawal and Gibson, 1999; Ostrom, 1999).

Traditional community forest ownership, i.e., community forests or forest commons, refers to woodlands collectively managed by local communities, considering community as a social group living in a small spatial unit, with a homogeneous social structure, frequent interactions, and shared interests and norms (Agrawal and Gibson, 1999). Understanding how local communities manage their forests is important to global biodiversity, as traditional community-based management comprise at least 40 per cent of worldwide biodiversity-rich areas (Garnett et al., 2018). Here, we contribute to the analysis of the ecological outcomes of forest commons ownership by examining the historical evolution of collective property regimes in Spain, a country in Western Europe. During the eighteenth and nineteenth centuries, the transition from the medieval period to modernity in Europe brought the establishment of a political and economic framework that introduced the concept of property law to previous feudal land tenure regimes, in which lands held in common were considered as public property (Izquierdo, 2007). Because traditional community ownership does not fit well in the private vs. public dichotomy, forests commons had to be classified as private, public, semi-private or semi-public, depending on the regional context (see Weiss et al., 2018). The heterogeneous land tenure change occurring in Western Europe at the time offers a unique context to analyse the woodland landscape dynamic resulting from the replacement of forest commons' traditional management systems by other land ownership forms.

In particular, we examine the historical evolution of forest commons in Spain, a region with long history of forest community-ownership (Montiel, 2007). In pre-industrial Spain, the use of local forest-related resources was essential in guaranteeing peasants' subsistence, especially in mountain areas (Piqueras and Sanz, 2007). Overtime, Spanish local communities developed formal –e.g., local ordinances– and informal –e.g., cultural practices– norms and rules to manage forest commons and prevent from overuse (Moreno, 1998; Linares, 2000; Serrano, 2014). However, the political and economic framework established by the late eighteenth century in the country did not recognize community ownership (Caballero, 2015) and the establishment of this new framework resulted in a heterogeneous evolution of forest commons ownership in this country.

Within the general aim of understanding the potential that community-based resource management holds to foster a sustainable use of ecological resources, we examine the different woodland landscape dynamics resulting from the heterogeneous dismantling process of forest commons in Spain, to better understand the ecological consequences of replacing traditional community ownership by other forms of land ownership. In addition, we provide a succinct description of the multiple legacies of community-ownership forests recognized in the contemporary Spanish legal code, and the nature conservation interest of these categories.

2.2 Historical evolution of forest commons in Spain

2.2.1 Initial records of the commons

Woodlands collectively managed in Spain dates back, at least, to the Middle Ages, when the territories that now constitute current Spain experienced a process of human resettlement and land use redistribution associated to the Christian Reconquest (Pardo and Gil, 2005; Montiel, 2007). From the eighth to the fifteenth century, medieval kings granted land privileges to the Christian settlers who displaced Muslim populations from the newly gained territories. Such strategy created a special type of common tenure, in which settlers, organized in village councils or *concejos*, collectively managed land concessions consisting of meadows, woods, and streams. As the management of such common lands and resources was not officially regulated, with time, users developed a set of informal rules adapted to local social-ecological conditions that became widely accepted by community members (Mangas, 2013; Blanco, 2014).

Such rules were mainly orally transmitted and rarely written down until the thirteenth century, when *las Siete Partidas*, or the Seven Divisions, a legal code compiled by Alfonso X the Learned of Castile, refers to the management of the commons stating that 'mountains and pastures and all places similar (...) belong to the common. Every man who is a resident can make use of them' (Law IX, Title XXVII, Third Partida) and 'Cities and towns can own fields and other lands (...) although these are property of all inhabitants, nevertheless, each one of them cannot separately and individually make use of them' (Law X, Title XXVII, Third Partida) (Burns, 2012). From the thirteenth century onwards, the *concejos* issued local ordinances to define the condition of resident and guide the long-term conservation of resources in common use by all residents (Moreno, 1998; Arango, 2009). For example, in eastern Spain, local regulations dating from 1271 were issued to avoid the entrance of non-resident livestock herds and the ploughing of forest commons (Piqueras, 2002).

Later, in the fifteenth century, the Catholic Monarchs issued a decree aiming to regulate the use of forest commons. According to Wing (2015), the new decree shows crown's intention in regulating forest use at the same time that recognized local municipalities, handicraftsmen, and shepherds needs of forest resources, as well as farmers' interests to extend their arable lands by ploughing local woodlands and pastures. In an attempt to be adapted to preserve tree canopy layer while fulfilling local communities' livelihood needs, the regulations issued in the royal decree promoted a sustainable use of forest-related resources. For instance, one regulation prevented veteran trees from excessive cutting, so while it allowed cutting branches for firewood and carving, it regulated that this could only be done to the extent that it did not impede new growth.

From the sixteenth century onwards, crown's regulation of woods management intensified. Spanish monarchs' concern regarding the decrease of forested lands, along with Spanish navy's high requirements of timber, resulted on the promulgation of several royal ordinances limiting woodlands use by local populations. In 1518, for instance, a royal decree on 'Formation of new forest plantations and ordinances to conserve old and new forests' called for the designation of local guards to defend against the cutting of trees (Rey, 2004; Wing, 2015). Spanish Monarchy attempts to prevent woodlands depletion by issuing protective ordinances continued during the seventeenth and eighteenth centuries (Ramos, 2007). This is, for example, the case of the 1748 Forest Ordinances, which forbade cutting trees marked by and for the navy and authorized the expropriation of lands suitable for forest nurseries. Thus, between the sixteenth and the eighteenth centuries, crown's forest regulations shifted from defending the use of forest commons to increasingly restricting it.

2.2.2 Dismantling the commons

The nineteenth century largely resulted in the breakdown of the traditional community system in Spain. Following the liberal movement spread through Europe at the end of the Old Regime, two major reforms were enacted during that period, both with important effects on communal lands. First, Spanish earliest written Constitution was issued in 1812. Despite the long historical tradition of forest commons in Spain, the new legal code did not include community ownership as a form of property (Caballero, 2015). Rather, *concejos* were replaced by larger and hierarchically dependent municipalities, the Town Councils (Serrano, 2014). In other words, ancient ordinances were not recognized by the new legal framework, which implied that, from then forth, the different laws governing the commons –and specifically those related to woodlands– only recognized Town Councils as valid intermediaries between villagers and public administrations. Consequently, local residents, represented by the *concejos*, were not authorized to profit from their woodlands except through Town Councils (Serrano, 2014). However, given that the 1812 Constitution was repealed only two years after it was promulgated, common lands survived this first political attempt of abolishing them.

The second reform, this one with real important effects on common lands, was the disentailment policy (*Desamortización*) issued by the Minister of Finances Mendizábal in 1836-1837 and which continued until the twentieth century. Originally, this policy aimed to increase the number of rural small landowners by releasing to the market land properties that were, in liberal terms, lying stagnant (Arango, 2009). The process also aimed at decreasing the social influence of the Catholic Church, forcing the sale of ecclesiastical properties. However, the sought improvement of land distribution was not achieved, as vast quantities of property were acquired by an increasingly dominant bourgeoisie (Arango, 2009). In 1855, new disentailing policies affected public lands, many of them held in common, which imposed municipalities the sale of their own lands through public auctions (Beltrán, 2015). In such context, and particularly in mountain areas of central and northern Spain where rural communities feared to loose local resources that were essential for their everyday life, inhabitants organized themselves and pooled capital to collectively bid in the auctions and acquire disentailed forest commons for themselves (Medrano et al., 2013). However, in many occasions, the same fear resulted in individual appropriation of forest commons, typically by enclosing (i.e., delimiting common lands within a surface demarcated, for example, on a cadastral map) and ploughing (i.e., transforming forestland into crop fields) (Rotherham, 2013).

Disentailment policies resulted in the individual and State appropriation of common lands (Caballero, 2015). The process had a great impact in forest commons, as this type of property was widespread in the countryside. For instance, in the northern regions of Spain, such as La Rioja or Castilla y León, community property at that time represented more than three quarters of the mountain areas (Moreno, 1998; Rey, 2004). As result, before disentailment, there were ten million hectares of public mountain areas[1] in Spain, mainly integrated by municipal properties that included forest commons; and the amount of public woodlands sold to particulars during 1855-1924 is estimated to be five million hectares (Laso and Bauer, 1964; Pérez-Soba, 2013).

2.2.3 State interventionism and people's resistance

Given the Spanish political and social instability, enclosing and ploughing of forest commons and other illegal practices flourished during the nineteenth century, largely by fear to disentailment (GEPC, 2004). Other factors, such as the high demand of timber by naval shipbuilding, charcoal industries, and population increase in rural areas, also contributed to the depletion of forest resources (Liaño and García, 2003; Arango, 2009; Beltrán, 2015).

Deforestation raised authorities' awareness, who reacted to prevent further deterioration, but aimed at doing so with a top-down scientific-based forestry approach (Liaño and García, 2003; Parrotta and Trosper, 2012). In line with this aim, the Spanish School of Forest Engineers was founded in 1846. In this School, professionals were trained

[1] Note that since 1864, when Spanish first classification of woodland areas was carried out, forest commons were considered to belong to municipalities, that is, they were considered public properties.

using the theories developed by the German Forestry Faculty of Tharandt, who promoted the idea that States should assume the management of woodland areas through a bureau of forest technicians (Linares, 2000). Following this logic, the Spanish Corps of Forest Engineers was created in 1853. Interestingly, the first request made to this corps was to produce a list that included all country's public woodlands to be exempted from disentailment. The classification generated a 'Catalogue of Public Utility Woodland' that dates from 1864. Such catalogue introduced a novel type of woodland property: Public Utility Woodland or *Montes de Utilidad Pública*, which included municipal forest commons exempted from sale during the disentailing process (Sieira, 1956). Once the Catalogue classified a municipal forest as Public Utility Woodland, its monitoring was transferred to the State Forestry Administration (Sieria, 1956; GEHR, 1999). The inclusion implied that, from then forth, villagers had to ask for the approval of the Forest Administration to obtain goods from forest commons. Thus, the Liberal State initiated a process of enclosing of communal resources and increased its influence in the management of the forest commons exempted from the privatization policies (Beltrán, 2015)

The great monitor exerted over forest commons by the State gave rise to tensions between forest authorities and villagers (Cobo et al., 1992; Linares, 2000). Still, until the nineteenth century, rural communities often contested State actions with protests like illegally felling trees or enclosing common lands. During the nineteenth century, however, local strategies to recover traditional rights lost during the disentailment policies were more complex, including a combination of individual illegal actions such as ploughing or fires, and organized legal actions like the collective purchase of communal lands in public auctions (Linares, 2000; Piqueras and Sanz, 2007; Valbuena-Carabaña et al., 2010).

The opposition to the abolition of the historical communal property and use rights provoked numerous conflicts during the nineteenth and twentieth centuries. Still, as Soto et al. (2007) remark, it is convenient to distinguish between conflicts generated by the loss of ownership rights and conflicts generated by the loss of traditional community use rights. On the one side, the Spanish State did not recognize community ownership; moreover, a royal order issued in 1848 denied any possibility of community ownership, and forest commons' ownership rights were transferred to municipalities (Caballero, 2015). Several instances of conflicts related to the loss of traditional community property rights have been documented particularly in northwest Spain, where forest commons have a private collective ownership origin (Cuadrado, 1980; Caballero, 2015).

On the other side, rural inhabitants were concerned by the loss of use rights in common lands (Cobo et al., 1992; Soto et al., 2007). As mentioned, forests hold resources that were critical to rural livelihood, notably for the poorest peasants. Disentailment policies, along with State monitoring of forest commons included in the Catalogue of Public Utility Woodlands, resulted in the decrease of the forest area that local communities were able to use (Cobo et al., 1992). In addition, the forest management system adopted by State forest technicians –based on the assumption that some traditional uses, such as grazing or prescribed burns, were incompatible with long-term conservation of forest cover– limited traditional forest-related practices (Serrano, 2005). However, partly due to the key role of these traditional practices on local livelihood (Cobo et al., 1992; Balboa, 1999), but also as a means of protest (Piqueras, 2002; GEHR, 1999), these uses continued to be carried out by rural communities. Moreover, as the new legal framework dismissed *concejos*' authority to sanction illegal uses, activities such as the enclosing and ploughing of forest commons proliferated (Serrano, 2005). And so, the penalization of traditional uses, rather than resulting in forest cover preservation, seems to have had the opposite effect (Campos et al., 2013).

Overall, the disentailing process negatively affected forest conservation for two reasons. First, privatized forests were logged, as private owners were inclined to compensate the cash value of their purchase (Laso and Bauer, 1964; Ezquerra and Gil, 2008). Second, local opposition to the cessation of forest commons' historical uses resulted in the proliferation of illegal practices that were not longer monitored by the *concejos* (Serrano, 2005).

2.2.4 Rural adaptation to the limitation of forest commons traditional use

Thus, Spanish rural landscapes entered the twentieth century drastically deforested (Valbuena-Carabaña et al., 2010). During the twentieth century, the State Forest Administration focused its efforts on reversing the degradation trend and restoring the vegetation cover of public woodlands through afforestation policies (GEHR, 1999). During the first third of the century (1901-1939), afforestation focused on protective outcomes, such as to prevent periodic flooding, for which fast growing tree species, like *Pinus* species, were used (Valbuena-Carabaña et al., 2010). Later, from 1940 to 1986, afforestation shifted to an intensive silvicultural treatment in which fast growing tree species that could be harvested in less than ten years, such as *Populus* and *Eucalyptus* species, were favoured over traditional ones (GEPC, 2004; Ramos, 2007; Valbuena-Carabaña et al., 2010).

Another measure taken by Spanish public administration to avoid further degradation of forested landscapes was to adopt a more conservationist interventionism in those forests included in the Catalogue of Public Utility Woodlands. Forest commons management, monitored by the State Forest Administration since 1863, aimed to prevent traditional uses in catalogued woodlands. This intervention was done under the argument that some practices, such as logging, firewood collection or small ruminant livestock grazing, were incompatible with the long-term maintenance of forest cover (Cobo et al, 1992; Linares, 2000; Montiel, 2007). To that end, from the beginning of the nineteenth century, the access and use to public woodlands became regulated through forest management plans, which –in an attempt to reduce peasants' use of forest resources– were often overly restrictive regarding traditional practices (Parviainen, 2006; Linares, 2007; Johann et al., 2012).

Both interventions –afforestation policies and limitations to traditional forest uses– along with the dismantling of forest commons, resulted in a decline in the use of woodlands by rural communities. The intensive afforestation created very specific ecological systems that were not connected to local productive systems, impeding the multiple-use of forest resources (GEPC, 2004). Additionally, and partly due to restrictions in the use of forest resources, local communities increasingly abandoned forest-based activities shifting to other economic activities in response to national market demand (Balboa, 1999; GEHR, 1999).

The disentailment process –which continued until the first decades of the twentieth century– also encouraged the penetration of a market-based economy in agricultural production, as the enclosing and ploughing of common lands allowed farmers to enlarge their productive capacity. However, farmers' illegal appropriations of common lands was a heterogeneous phenomenon in the Iberian peninsula, because of in those areas where farming practices competed with other land uses, such as extensive livestock grazing, conflicts among community members limited the enclosing process (Beltrán, 2015). Thus, in semi-arid Mediterranean areas of Spain, where environmental conditions are favourable for agriculture, cropland was favoured at the expense of the dehesas, a woodland-pasture managed in common in the past (Linares, 2000; Campos et al., 2013). In contrast, in Mediterranean continental areas, with environmental conditions less favourable for farming cultivation, summer pastures were favoured instead, as a mean of guaranteeing the basement for traditional stockbreeding, frequently managed through systems of agrarian collectivism (Montiel, 2007). Similarly, northern areas located in the Atlantic ecosystem did not experience the enlargement of arable land at the expense of forestlands. The higher production capacity of Atlantic areas due to their humid conditions but with a sharp relief and deficient communications resulted in the intensification of agricultural productivity of these lands without resorting to the expansion of crops (GEPC, 2004).

2.2.5 Contemporary trends in traditional community forests

From mid-twentieth century onwards, the enclosing process declined. The rural crisis associated with depopulation, agricultural and livestock intensification and mechanization, and the abandonment of traditional activities, took out pressure from the arable land supply leading to a progressive natural vegetation succession of abandoned lands (Rotherham, 2013; Viedma et al., 2015). These changes led to the densification and homogenization of the traditional rural landscape mosaic, which resulted in an impoverishment of forest biodiversity because of the transformation of woodland to shrubland and in an increasing risk of wildfires due to a higher fuel load and continuity (Loepfe et al., 2010).

The abandonment of local forest management seems also to have had negative ecological outcomes to natural habitats dependent on traditional practices. For instance, the progressive decline of high species diversity in chestnut groves (*Castanea sativa* Mill.) has been attributed to the abandonment of human management practices on these stands such as grazing, regular pruning, or periodical understory burns (Gondard et al., 2006; Guitián et al., 2012). Another traditional practice with documented positive effects for biodiversity conservation that suffers from abandonment is the traditional pruning that used to be carried out in woodland-pastures systems of *Quercus* species. Traditional pruning of *Quercus* spp. makes compatible farming and herding with the persistence of a canopy layer, at the same time that allows the persistence of veteran trees, which are key for saproxylic fauna and flora and as a habitat niche for cavity-nesting birds and wood-inhabiting fungi (Olea and San Miguel-Ayanz, 2006; Siitonen and Ranius, 2015). On the one side, the cessation of *Quercus* spp. traditional management is linked to the woody encroachment of these habitats. On the other, traditional management is being substituted by more intensive systems, such as commercial conifer forestlands, with the consequent loss of flora and fauna associated to *Quercus* forests (Taboada et al., 2006). Another traditional practice worth mentioning for its positive ecological outcomes is the transhumance, an ancestral pastoral practice consisting of seasonal moving of livestock to graze on higher pastures in summer, which arguably contributes to species biodiversity by increasing landscape complexity through the creation of grassland-woodland habitat mosaics (Oteros-Rozas et al., 2012; Orlandi et al., 2016).

Thus, overall, the State Forest Administration initial assumption that decreasing traditional forest-related practices would result in forest conservation seems to have had the opposite effect in the long-term. In fact, the abandonment of traditional uses meant the encroachment of forest habitats and the simplification and homogenization of rural landscape mosaic shaped by traditional management, negatively affecting biodiversity conservation and increasing fire hazard risks.

2.3 Multiple legacies in current legal framework of Spanish forest commons

From the historical account detailed above, it can be assumed that the persistence of collective ownership of forests has been closely related to the environmental conditions of the different geographical areas of Spain (Beltrán, 2015). Thus, in southern areas where agricultural lands were favoured at the expense of forest commons, enclosing rates was higher than in mountainous regions of north Spain, in which forest-related goods were a major support to the rural livelihoods. In addition, Spanish northern rural communities fought for legal recognition of their traditional community forests' rights, which were restored by the 1957 Forestry Act that recognized a particular type of collective woodlands known as neighbour woodlands or *montes vecinales en mano común* (Cuadrado, 1980). Also in some regions of northern and central Spain, where local communities' collectively purchased forest commons put up for sale during disentailment, this type of legacy of community ownership has been recognized, when Spanish legal framework catalogued them as *pro indiviso* forests in an additional disposition to 2003 Forestry Act.

As result, most recent legal Spanish Forest Act (Law No. 21/2015 of Spanish Government) distinguishes between three different categories of community-ownership forests: (1) Forest commons, (2) Partners' woodlands, and (3) Neighbour woodlands.

2.3.1 Forests commons

Forest commons, or *montes comunales* in Spanish, are conformed by former forest commons that survived the privatization wave or –in other words– that were considered as exempted from disentailment during the nineteenth century classification carried out by the Corps of Forest Engineers. Forest commons typically belong to municipalities, but their use corresponds to local communities (Sieira, 1956). The management of forest commons is ruled by ordinances approved by residents, with the forestry administration exerting its influence by monitoring the commoners' (i.e., users of the commons) access to grazing, firewood, and other forest-related goods (Balboa, 1999).

Forest commons are the most abundant community forests in Spain, with presence in all regions of the country. They used to feature a regulated spatial planning in which commoners organized themselves to carry out traditional practices (Couto and Gutiérrez, 2012). Examples of these practices include the *vecería*, a communal pastoral activity consisting in shifting turns among community members to move a common herd to graze in forest commons, which was habitual in Spanish northern regions until recent times (González, 2001). According to Vázquez (2016), apart from reducing agricultural workload, the *vecería* system also strengthened social relations in rural communities. The same author posited that, although the persistence of this collective organization system was undesirable by liberals, local requests to preserve it were frequently accepted, as commoners argued that individual pastoral systems are not viable for the poorest peasants, likely resulting in cattle mismanagement and uncontrolled grazing.

A remarkable example of forest common with positive ecological impact is the Urbión Forest in Castilla y León, north Spain. In this region, thirteenth century local ordinances enforcing local communities' right to make use of forest goods have been legally endorsed until the present, for which traditional community management was not affected by exclusionary policies. Interestingly, the Urbión Forest conforms nowadays is the most extensive continuous wooden area on the peninsula, with *Pinus* spp., *Quercus* spp., *Fagus sylvatica* L. and *Juniperus thurifera* L. stands (Segur et al., 2014). These high nature value habitats overlap now with Cañón del Río Lobos Natural Park and several Natura 2000 sites.

2.3.2 Partners' woodlands

Partners' woodlands, or *montes de socios*, are a second type of community-ownership woodlands that survived the disentailing policies. Partners' woodlands were created during the nineteenth century through the association of neighbours that pooled economic resources to buy the forests that they had traditionally managed in common (Montiel, 2005). This category constitutes a type of ownership form in which the forest is private property, but owned by a group of people who collectively bought it, resulting in a great number of co-owners (Mangas, 2013; Medrano et al., 2013).

Most of partners' woodlands originated from the collective response of rural communities in forested provinces of inner Spain such as Burgos and Soria, where a strong local opposition to forest commons' usurpation took place (Piqueras, 2002; Montiel, 2007). Partners' woodlands can be also found in other north and inner provinces of Spain, referred by a large variety of names –e.g., *montes del común* (woodlands of the common), *sociedad del monte* (society's woodland), *monte de la sociedad de vecinos* (neighbours' society woodland)–, which reflects their past abundance and importance (Medrano et al., 2013). Nowadays, more than 1,500,000 hectares of forestland have the status of partners' woodlands, although a legal framework for the management of this type of property was not issued until the 2015 Spanish Forestry Act.

From an ecological perspective, partners' woodlands can also provide important insights for designing sustainable conservation strategies. For instance, the economic profit obtained from the traditional practice of quotas or *suertes*, in which woodland is divided in cutblocks among residents for timber harvesting, is known to increase commoners' interest in forest conservation, reflected in the internal regulations issued to prevent illegal uses such as logging or burning (Gogeascoechea, 1999). Nowadays, several partner's woodlands overlap with designated as Natura 2000 areas, such as Sierra de Cabrejas in Castilla y León that constitutes the largest *Juniperus thurifera* forest in Europe (Pecurul-Botines et al., 2014).

2.3.3 Neighbour woodlands

Neighbour woodlands, or *montes vecinales en mano común* (hereafter MVMC, for their acronym in Spanish), were originated during the second half of the twentieth century from the social resistance of Galician peasants, northwest Spain, to disentailment policies (Couto and Gutiérrez, 2012). MVMC are a type of ownership form in which the forest is private property, owned by all neighbours of a particular local community, so that status of neighbouring is required to obtain forest ownership and use rights (Caballero, 2015). The term 'neighbour' refers to the representative of a family or house granted with property rights on the forest common due to the condition of resident (GEPC, 2004).

Noteworthy, MVMC have a very different historical evolution than forest commons and partners' woodlands (Balboa, 1999). MVMC have a private ownership origin, in which ownership collectively belong the local community members (Cuadrado, 1980; Caballero, 2015). This particular private ownership form was not recognized by the liberal legislation, which in 1848 denied any possibility of community-ownership form in Spain and transferred MVMC's ownership rights to municipalities. However, in Galicia, more than in other regions, the livelihood dependence of forest resources led to a strong, long-term rural resistance to the appropriation process (Piqueras, 2002; GEPC, 2004). The combination of both factors –an original private ownership form and strong local opposition to the dismantling process– resulted in the early legal recognition of the community-ownership form in Galicia through the inclusion of MVMC in the 1957 Spanish Forestry Act, and through the promulgation in 1968 of a specific Forestry Act returning MVMC ownership rights to Galician local communities. In 1975, the legal framework of MVMC was extended to the neighbouring northwest provinces of Zamora, León, Asturias and Cantabria (Cuadrado, 1980; Blanco, 2014).

Nowadays MVMC represent one-third of the total surface of Galician forests, covering approximately 673,000 hectares, with more than 2,800 community-owners rule the management of MVMC based on traditional norms (Arango, 2009; IDEGA, 2013; Caballero, 2015). Due to the 1968 legal endorsement, neighbours have been able to traditionally manage forest resources for many decades, which has resulted in many high valued ecosystems that currently overlap with different protection categories, as for example *Sobreiras do Faro*, where commoners themselves apply for the designation of their MVMC as Natural Protected Area (Couto and Gutiérrez, 2012).

4.5 Conclusion

This paper has provided a historical examination of community-ownership tenure regimes in Spanish woodlands and their potential impacts on forest biodiversity maintenance. Our results document how local communities created norms and institutions regulating and monitoring the multiple uses of forest commons to prevent resources depletion back in the Middle Ages. The replacement of traditional community governance systems occurring during the process of privatization and state interventionism of communal lands in the nineteenth and twentieth centuries in Spain had negative consequences for forest cover maintenance. On the one side, forests acquired by private owners were cut to compensate the cash value of their purchase. On the other, forest considered as public suffered from illegal uses no longer sanctioned by local institutions.

In the long-term, traditional uses abandonment lead to a simplification and homogenization of rural landscape mosaic, associated to a decrease of biodiversity and an increase of fire hazard risk. Interestingly, in geographical areas showing a stronger opposition to forest commons' dismantling policies and where traditional community ownership rights were restored earlier, we found several instances of forests commons that currently overlap with important areas for conservation. These results further support the idea that community-based management can hold useful insights for the maintenance of diverse, high ecological valued ecosystems, while allowing local communities the use of natural resources. Further research should aim to identify which particular management practices traditionally applied by local communities in forest ecosystems of Spain had been favorable to biodiversity and economically profitable at the same time.

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Chapter III

Picos de Europa National Park. Photo credits: S.Guadilla-Sáez

Biodiversity conservation effectiveness provided by a protection status in temperate forest commons of north Spain

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Paper II

Biodiversity conservation effectiveness provided by a protection status in temperate forest commons of north Spain

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Abstract

The establishment and maintenance of protected areas is the backbone of global conservation strategies to halt biodiversity loss. However, despite the more than 200,000 legally designated protected sites worldwide, the rate of species extinction has not decreased, for which some debate the real effectiveness of protected areas to preserve biodiversity. Using data from tropical areas, many studies have attempted to test the effectiveness of protected areas by comparing species richness in protected and neighbouring unprotected sites, without reaching a consensus. Here, we extend this line of research with data from temperate deciduous forests inside and outside Picos de Europa National Park and Biosphere Reserve (North Spain). Specifically we compare data from mixed broadleaved woodlands, beech forests (*Fagus sylvatica* L.) and Pyrenean oak (*Quercus pyrenaica* Willd.) forests. We conducted botanical inventories and recorded ecological data from 25 0.2-hectares concentric plots distributed in forest commons inside the reserve and from other 25 similar plots established in neighbouring not protected forest commons. Data were used to construct a set of ecological indicators and evaluated using modelling methods. We found no significant differences in species composition between plots in protected and non-protected forest commons, likely due to the similar management criteria applied in both land uses. We found less active management outside the protected area, which helps to maintain stands in a semi-natural state. In contrast, we observed the presence of silvicultural treatments inside the protected area, although these treatments were non-intensive, promoting vegetation composition associated to late-successional ecosystems. We only detected significant differences between plots inside and outside the protected area when relation between species richness was analysed with reference to forest habitat type. Precisely, plots of beech forests inside Picos de Europa were more homogenous than plots outside the protected area, which may indicate that management practices inside the protected area do not favour tree species diversity. Non-intensive silviculture management in beech forests inside Picos de Europa seems to promote the presence of the dominant tree species *Fagus sylvatica* L., which in the absence of perturbations is characterized by conforming monospecific vegetation communities. Overall, our results do not support the idea that protected areas hold more biodiversity than surrounding forest commons. Conservation treatments applied in protected areas should promote the presence of species associated to disturbances, particularly in stands tending to homogeneous species composition at late-successional stages, as this may enhance their resilience under the current rapid global changes.

Keywords: Anthropogenic disturbances; biodiversity indices; protected areas; species richness; temperate deciduous forests.

3.1 Introduction

With 234,468 Protected areas (PA) already established worldwide (IUCN, UNEP-WCMC, 2017), site designation rate is considered one of the most remarkable conservation successes of the twentieth century (Gaston et al., 2008). If this trend continues, the goal of 17 per cent coverage for terrestrial and inland waters by 2020 under Aichi Biodiversity Target 11 would be achieved (Gannon et al., 2017). However, in numerous cases, site designation has followed an exclusionary approach –referred to as 'neo-protectionist' or 'fortress conservation' approach (Wilshusen et al., 2002; Brockington, 2002)– resulting in the displacement and dispossession of communities residing in the newly protected site, and often led to contested actions (Laudati, 2010; Mahapatra et al., 2015). Moreover, even when less controversial 'conservation-centric' initiatives have been applied to the establishment of a PA, such as protected sites created in partnership with local people, conflicts with local residents have still arisen, especially when the establishment of a PA have prevented local users from the management of the surrounding natural resources (West et al., 2006). As the economic basis of many indigenous peoples and local communities is closely dependant on the goods obtained from neighbouring natural areas, particularly forests (Angelsen et al., 2014), when restrictions to local use are applied without providing suitable alternative livelihood options, struggles are likely to appear (Mahapatra et al., 2015), constituting a significant shortcoming to PA conservation efforts (Andrade and Rhodes, 2012).

There is evidence of better biodiversity conservation outcomes from PA management strategies integrating local economic activities than from strictly conservation PA management regimes (Oldekop et al., 2016), for which many conservationists nowadays embrace a more integrative perspective for the establishment and management of PA (Shultis and Heffer, 2016). Notions such as social-ecological systems, resilience, or cultural landscape are holistic approaches which consider people as part of their surrounding environment rather than mere passive users of landscape biophysical components. People modify their living and adjacent territories, sometimes causing the depletion of natural resources, but sometimes coevolving with nature in the benefit of a sustainable use and promotion of a dynamic mosaic of ecosystems at the landscape level. Along these lines, scholars increasingly advocate for considering PAs as 'social-ecological systems' (Hirschnitz-Garbers and Stoll-Kleemann, 2011; Cumming et al., 2015; Mathevet et al., 2016), a conceptual framework that contemplates both the social and ecological aspects of the system as equally important (Berkes, 2017). Under the social-ecological systems approach, the social and the natural systems are indeed coupled subsystems that co-evolve, which implies that societies are able to adapt to perturbations in the environment and vice versa. Given the rapid global changes occurring nowadays, the coevolving capacity offered by the social-ecological systems approach brings to light the idea that societies have the opportunity to face environmental challenges without compromising long-term sustainability of ecosystems (Berkes et al., 2003).

The growing scholarly emphasis on conservation efforts outside the physical boundaries of PAs focuses on reconcile management practices from land uses, such as farming or forestry, with biodiversity conservation (Kremen, 2015). Research on practices that may be both favourable to biodiversity and economically profitable has significantly increased attention to the potential that community-based resource management may bring to foster a sustainable use of ecological resources (Xu and Melick, 2007; Larson et al., 2016). Through practices such as clearing, livestock grazing, or swidden agriculture, humans have modified landscapes for millennia, partially replacing the ecological functions that megaherbivores used to play in shaping vegetation structures of terrestrial biomes (Sandom et al., 2014; Bocherens, 2018). While not all of these small and intermediate-scale disturbances have a positive effect in preserving biodiversity, overall, management practices applied by communities seem to enhance biodiversity through the creation of a mosaic of ecosystems (Agrawal and Gibson, 1999; Guèze et al., 2015).

Aiming to further understand the conservation outcomes that may result from community resource management, here we study a community- based regime that can be considered a social-ecological systems: forest commons. Forest commons are characterized by having clearly defined boundaries and legal enforceable property rights and by providing resources to a variety of social groups that are usually involved in their management (Aryeetey et al., 2012). The importance of forest resources to support rural household economies strongly engaged local communities in the monitoring of their surrounding woodlands to prevent mismanagement and overexploitation, resulting in the implementation of management techniques that allowed the co-existence and long-term maintenance of diverse forest uses and habitats (Parrotta and Trosper, 2012; Kirby and Watkins, 2015). Nowadays, forests commons constitute 18 per cent of global forest area and appear to contribute significantly to biodiversity conservation (Chhatre and Agrawal, 2008).

In this work, we analyse the role of forest commons in long-term biodiversity persistence and the effectiveness of protected areas in this type of community-based regime. We do so by comparing a set of ecological and anthropogenic features observed in forest commons plots inside and outside a PA classified as an IUCN category II (National Park), a very restrictive protection category with regard to human activities (Gray et al., 2016; Hewitt et al., 2016). Our study has three main goals. First, we test whether plots inside and outside the PA differ in their ecological characteristics (i.e. topography, edaphic factors), and how this relates to species richness. We based our null hypothesis on the general assumption that we will find the same tree species abundance and evenness in plots inside and outside the PA. The second goal of this research is to analyse the effect of the human intervention on species distribution in plots under protected and unprotected sites, an analysis performed by linking the variables measuring human disturbances (i.e. plot isolation, silvicultural systems) with tree species occurrence. Based on previous case studies highlighting the association of anthropogenic disturbance with species richness in human-dominated landscape (Guèze et al., 2015; Mod et al., 2016), our hypothesis is that anthropogenic disturbances can induce changes in species composition that would result in more heterogeneous species assemblages of the studied forest communities. Finally, the third goal of this paper is to study the relationship between tree species composition, and particularly species diversity, and a) spatial distribution and b) forest management approach. For this goal we compare three different forest habitat types occurring inside and outside the study PA, in an attempt to quantify the conservation outcomes resulting from the protection status between habitats of the same land use type, i.e. temperate deciduous forests.

3.2 Methods

3.2.1 Study site

We conducted the study in the *Liébana* valley (57,500 hectares), a wide depression located in the southwest of the Cantabria region, in northern Spain. High elevation differences characterize the region, with altitude ranges from 330 meters to 2600 meters above sea level. Liébana is surrounded by hills, a geographical feature that results in relative geographical isolation with the neighbouring areas and a high number of habitat types of unique ecological value. The bottom of the valley presents a Mediterranean microclimate with less rainfall than the rest of the Cantabria region; at higher altitudes we find an Atlantic climate (Rescia et al., 2008). Liébana's mean temperature varies from 7.9 to 20.8 °C and the annual average rainfall varies from 700 to 1500 mm in the mountainous parts (ETSIM, 1978). Topographic and climatic differences result in a very heterogeneous landscape with a wide array of vegetation types.

From a land tenure perspective, common property is a key feature in the Liébana region, with almost 80 per cent of its territory under this regime (Arbeo, 2012). More than three quarters of the woodlands in Liébana are forest commons (Ministerio de Medio Ambiente, 1997–2007). The area has a long tradition of human intervention, as reflected in the large number of local ordinances regulating forest uses since the fifteenth century (Pérez-Bustamante and Baró, 1988). Management activities have shaped the structure of the local ecosystems and their species composition, at the same time that they have allowed the persistence of high valuable habitats, to the point that 60 per cent of Liébana's forest commons are currently under some category of protection, including the *Picos de Europa* National Park (hereafter Picos de Europa) in the northwest of the region.

Picos de Europa was the first Park designated in Spain, in 1918, following the North American conservationist model that advocated for the preservation of wilderness areas by preventing human interventions. It also applied a State-led management (Hirschnitz-Garbers and Stoll-Kleemann, 2011; González, 2015). Since its designation, conflicts with local communities in Liébana's neighbouring region arose due to the limitations that the protection status enforced on local uses (e.g., hunting, wood extraction) (Voth, 2007).

Conflicts have shaped the negative perception of Liébana's rural communities regarding the extension of Picos de Europa National Park in 1995, which included three municipalities of Liébana, conforming the second largest site of the Spanish National Parks and notably including human settlements within its boundaries. Although the management criteria established in 1995 aimed at making compatible local traditional uses –like livestock herding or fuelwood collection– with biodiversity conservation (Royal Decree No. 640/1994 of Spanish Government), confrontation with local population, particularly livestock farmers, resulted in the revoke of the National Park management plan in 2005 due to a legal action taken by local communities within Picos de Europa buffer zone (Spanish Government, 2005).

The lack of a valid management plan, specific to the area, difficults the regulation of traditional uses. However, and despite the lack of a specific plan, several conservation initiatives are being undertaken inside the national park in response to national and regional environmental legislation, including the recovery plans for the endangered species brown bear (*Ursus arctos*) and Cantabrian capercaillie (*Tetrao urogallus cantabricus*) conservation. Altogether, this situation makes of Picos de Europa a very suitable site to analyse the effects of human activities and conservation management strategies on species diversity.

3.2.2 Description of the forest communities studied

To test the ecological differences between stands inside and outside the PA, we studied the ecological features of the three more abundant habitats in the forest commons inside Picos de Europa, which are also present outside the PA: (1) Mixed broadleaf woodlands, defined as forest with a variable mixture of at least two native broadleaf species accounting for≥70 per cent of the plot forest cover; (2) Beech forests or forest with *Fagus sylvatica* L. as the dominant tree species, accounting for ≥70 per cent of the plot forest cover; and (3) Pyrenean oak forests, or forest with *Quercus pyrenaica* Willd. as the dominant tree species accounting for ≥70 per cent of the plot forest cover.

3.2.2.1 Mixed broadleaf woodlands

Mixtures of broadleaf species cover most of the territory. Despite some human influence, mixed broadleaf woodlands are considered naturally distributed particularly in areas of contact between different tree formations. The predominant species in the mixed broadleaf woodlands located in upper altitudes of Liébana are *Quercus pyrenaica*, *Fagus sylvatica*, *Ilex aquifolium* L., and *Crataegus monogyna* Jacq. At lower altitudes, the predominant tree species are conformed by *Castanea sativa* Miller, *Fraxinus excelsior* L., *Tilia cordata* L., *Tilia platyphyllos* Scop., *Quercus ilex* L., and *Corylus avellana* L. (ETSIM, 1978).

*3.2.2.2 Beech forests (*Fagus sylvatica *L.)*

Forests of *Fagus sylvatica* are the predominant tree species formation in the Liébana valley and, generally, also the ones with the best conservation status and regeneration rates. These woodlands are shade-tolerant and occupy lands from 600 m to 1700 m in shadow slopes, and from 700 to 1300 m in sunny slopes. Beech woods form close, dense stands, where only shade-tolerant species can grow and where the competing tree species need to take advantage of clearings resulting from felling, browsing animals, or fires. As a result, *F. sylvatica* only appears in combination with other species in boundary areas with other forest types, or in areas where it displaces other species, as in the case of *Quercus pyrenaica*. A sparse understorey could accompany beech forests, composed by *Ilex aquifolium*, *Crataegus monogyna*, *Sorbus aria* (L.) Crantz, and *Corylus avellana*. However, most frequently, the accompanying species are *Pteridium aquilinum* (L.) Kuhn in Kerst and *Rubus* sp., partly due to the increased light following human interventions such as clear cuts (ETSIM, 1978; Godefroid et al., 2005; Kelemen et al., 2012).

*3.2.2.3 Pyrenean oak (*Quercus pyrenaica *Willd.)*

Forests of *Quercus pyrenaica* occur widely in the Liébana valley, with the species also occurring as a shrub in combination with *Corylus avellana*, *Crataegus monogyna*, *Erica* and *Ulex* sp. in mixed broadleaf woodlands. *Q. pyrenaica* is tolerant to a wide range of site conditions, occurring from 400 to 1300 m in altitude on sunny slopes and from 300 to 1200 on shady ones. Despite its frequency as shrub strata, it appears most often as a tree when the forest is cleared, i.e. when human intervention is high (Tárrega et al., 2006), accompanied by *Corylus avellana*, *Crataegus monogyna*, *Erica arborea* L., *Daboecia cantabrica* (Hunds) C. Koch and *Rubus* sp. Occasionally, it also appears in combination with *Prunus spinosa* L., *Erica vagans* L., *Calluna vulgaris* L. Hull, leguminous species like *Ulex europaeus* L., *Genista florida* L., *Cytisus* sp., or *Quercus ilex* (ETSIM, 1978).

3.2.3 Local history of forest commons management

As mentioned, the area has a long tradition of forest commons management. The invasion of Germanic tribes into the Iberian Peninsula in the fifth century brought the concept of woodlands collective property to the northwest areas of the Peninsula, where forest natural resources were used by local communities (Aranda, 1996). Some centuries later, during the Christian Reconquest (eighth to fifteenth century), the communal regime spread to other parts of the Peninsula as a strategy followed by medieval kings to promote the settlement of Christian populations that would displace Muslims from the newly gained territories. Within this process, the crown, ultimate owner of the common lands, granted land concessions –including woodlands– to the new settlers, who organized in village councils or concejos to collectively manage and use natural resources (Behar, 1983; Pardo and Gil, 2005).

Forest commons, or *montes comunales* in Spanish, were the most common type of tenure regime in Spanish woodlands until the nineteenth century, when Europe's transition to capitalism gave rise to the establishment of a new liberal framework that initiated a process of privatisation of communal resources. Despite the long tradition of forest commons in Spain, the liberal framework did not recognize communal ownership and just distinguished between the public and private proprietorship of lands and goods. Under that political context, forest commons were classified as public properties and their management transferred to Spanish State Forest Administration (Beltrán, 2015; Guadilla-Sáez et al., 2017). In the Liébana region, forest commons belong to municipalities, while their use corresponds exclusively to local communities, with the regional forestry administration exerting its influence by monitoring access to grazing, firewood and other forest goods (Balboa, 1999; Pérez-Soba and Solá, 2004). Remarkably, some forest-related stewardship customs still persist today in the study area, such as the neighbourhood councils or *juntas vecinales*, minor local bodies which have replaced former village councils and have legal rights to regulate the use of forest commons (Law No. 6/1994 of Cantabrian Regional Government). As a result, the regional forestry administration management intervention needs the approval from the neighbourhood councils before being applied in forest commons. However, and despite the regional legal enforcement of these minor local bodies role in the management of their common lands, the Picos de Europa Board of Trustees or *Patronato* –i.e. the participatory body aimed to integrate society to management activities and to promote further implications of local residents (Law No. 30/2014 of Spanish Government)–, does not include neighbourhood councils representatives.

3.2.4 Data collection

We used a GIS procedure to randomly select plots in forest patches with, at least, 70 per cent of tree canopy cover according to the Third Spanish National Forest Inventory (Ministerio de Medio Ambiente, 1997–2007). In each forest type, we located the centre of the plots according to a systematic sampling design in the intersection of a 125×125 meters fishnet grid created with ArcGIS version 10.3.1 (ESRI, 2015). From the total possible labels, we selected 50 values using the 'randbetween' option of MS Excel and stored their spatial coordinates in a global positioning system (GPS) device. These values were taken as the centre of each plot. We inventoried 50 circular plots of 25 meters radius (0.2 hectares). Half of the plots were in forest commons inside Picos de Europa (Figure 3.1, white-shaded area) and the other half were in forest commons located outside Picos de Europa and not affected by any other formal category of protection (Figure 3.1, grey-shaded area).

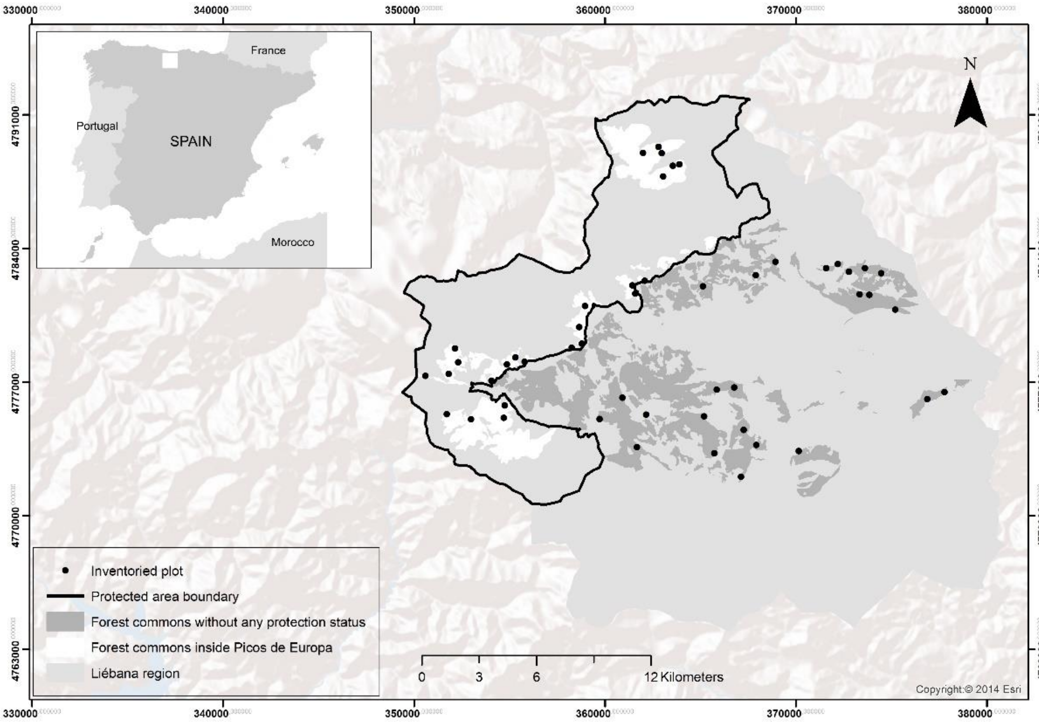

Figure 3.1 Map of the study area, illustrating the location of the 50 plots across the Liébana region (Cantabria, Spain).

We collected data during 2015 and 2016. We recorded a set of ecological parameters in each of the plots that included their general characteristics, the dominant tree species of the stand, stand structure properties, and distribution of the ground vegetation cover (see Appendix II.b). We inventoried the dendrometric characteristics of the 10 adult trees closest to the centre of the plot, where an adult tree were defined as a tree with more than 3 meters height or with a minimum diameter at breast height (1.30 meters) of 7.5 cm. To determine the abundance of each tree species, we identified all tree stems rooted within a sub-plot of 15 meters radius (aprox. 700m^2 in area). We also quantified the topographic variable slope for every plot using a SUUNTO clinometer and collected a surface soil sample to later analyse pH and texture parameters in the laboratory by using a glass electrode in a suspension of 1: 10 soil: distilled water. As additional monitored field measurements, we inventoried other edaphic variables such as soil texture, organic matter thickness, and stoniness. We also recorded the presence of silvicultural treatments such as clear cuttings, brush removal, thinning, or ground improvements within 25 meters radius (0.2 hectares).

We measured several anthropogenic variables associated with plot disturbance, including plot accessibility and linear distance from the plot centre to the nearest village and nearest path (i.e. unpaved roads or trails), calculated in desktop using ArcGIS version 10.3.1 (ESRI, 2015) in high resolution aerial photographs derived from the Spanish Aerial Ortophoto National Plan (PNOA, 2015). We also measured population density of the nearest towns using data from the Spanish National Statistics Institute (INE, 2018).

3.2.5 Ecological indices

Species presence indicators are frequently used to monitor effectiveness of a particular forest management treatment in biodiversity conservation (Canadian Council of Forest Ministers, 1997). For this research, we calculated a set of tree species composition indices to examine the heterogeneity in the composition of the studied communities, including the Shannon index (H'), a separate measure of evenness (J') from the standardization of the Shannon index, Species Richness (D_{Mn}), the complement of the Simpson index ($1 - D$), and the reciprocal form of the Berger-Parker index ($1/d$) (Table 3.1).

Although most studies have focused on the numerical richness to compare tree species diversity between different ecosystems (Hui et al., 2011), surrogate measures, such as stand structure indicators, are increasingly being used to provide a measure of biodiversity in forest communities (Pommerening, 2002). Following this trend, in this study, we have considered the Clark and Evans Index of aggregation (R) to define the distance between neighbouring trees in a forest spatial structure unit (Neumann and Starlinger, 2001). We have also included the Uniform angle (contagion) index (W_i), which tests the regularity of the distribution pattern of the trees, and the species complement for the Mingling index ($1 - M_i$) to test the heterogeneity of species among nearest neighbouring trees (Aguirre et al., 2003; Pommerening, 2002). The spatial unit considered for the estimation of the stand structure indices was the group size of the four nearest neighbouring trees to the reference tree, which is considered the optimum group size for evaluating spatial attributes (Hui et al., 2011).

Table 3.1. List of the diversity and stand structure indices used to analyse the ecological data.		
Index	Formula	Definition
Species composition indices		
Shannon (Magurran, 2004)	$H' = -\sum p_i \ln p_i$	where p_i is the proportion of individuals found in the i species referred to the total number of individuals.
Evenness (Elliot et al., 1997)	$J' = \frac{H'}{H_{max}} = \frac{H'}{\ln S}$	where H_{max} is the maximum level of diversity possible within a given population and S is the total number of species.
Richness (Magurran, 2004)	$D_{Mn} = \frac{S}{\sqrt{N}}$	where S equals the number of different species represented in the sample, and N is the total number of individuals in the sample.
Simpson (Magurran, 2004)	$D = \sum p_i^2$	where p_i is the proportion of individuals in the ith species.
Berger-Parker (Magurran, 2004)	$d = \frac{N_{max}}{N}$	where N_{max} is the number of individuals of the most abundant species and N refers to the total number of individuals.
Stand structure indices		
Clark-Evans (Vorčák et al., 2006)	$R = \frac{\frac{1}{n}\sum_i^n r_i}{0.5 \times \sqrt{\frac{Pl}{n}}}$	where r_i is the distance from the reference tree to its nearest neighbour, n is the number of trees on the sample plot and Pl the area of the sample plot in square meters.
Uniform angle (contagion) index (Pommerening, 2002)	$W_i = \frac{1}{n}\sum_{j=1}^{n} w_{ij}$	where w_{ij} is 1 if the angle with the jth neighbouring tree is lower than the defined standard angle, and w_{ij} equals 0 otherwise. n is the number of trees on the sample plot.
Species Mingling (Hui et al., 2011)	$M_i = \frac{1}{n}\sum_{j=1}^{n} v_{ij}$	where v_{ij} is 1 if the jth neighbouring tree is not the same species as the ith reference tree, and v_{ij} equals 0 otherwise. n is the number of trees on the sample plot.

3.2.6 Statistical analyses

3.2.6.1 Indices computed

We estimated tree species presence and stand structure indicators for each forest habitat type and subjected the resulting data to the Shapiro-Wilk normality test (Table A.1 in Appendix II.a). We used a two-sample t-test for testing for differences between forest habitat types inside and outside Picos de Europa for the normally distributed data and a two sample Wilconxon rank-sum (Mann-Whitney) test for the non-normally distributed data (Tables A.2–A.4). Significance levels (p-values) were adjusted to the number of tests carried out using a standard Bonferroni correction (Zuur et al., 2007). To facilitate the graphical comparison between stands of the three different forest habitats inside and outside the PA, we standardized correlation coefficients, representing in a common scale values obtained. We did all analysis using the STATA software version 13.1 (StataCorp, 2013).

3.2.6.2 Model selection

We considered two explanatory environmental variables in our model: (1) plot topography and (2) soil characteristics. Due to their relevance for the research, we also included as explanatory variables the level of human intervention observed in the plots, including variables that measure (3) plot isolation –i.e. distance to human infrastructures–, and (4) presence of anthropogenic disturbances –i.e. forest management practices–.

As dependent variable, we used the abundance of individuals per tree species estimated by the Shannon index (H'), a simple formula widely used to measure species richness in which the higher the value of H', the greater the species diversity of the studied system (Elliot et al., 1997; Magurran, 2004; Zuur et al., 2007). We evaluated the relative importance of the variables that measure environmental characteristics, plot isolation, and anthropogenic disturbances in explaining tree species distribution by using a Wald test analysis of H' against different models combining the explanatory variables. To select the explanatory variables to be included in the final model, we checked the normality of the variables through a correlation matrix and applying the Shapiro-Wilk normality test (see Table A.5). Based on those results, we selected non-parametric statistics for several of the explanatory variables. To determine relations between the explanatory variables, we applied Pearson's correlation test for the normally distributed variables and Spearman rank correlation test for the variables not-normally distributed (Table A.6). For categorical variables, we used a Pearson's chisquared correlation. After running the regressions, we removed multiple variables due to collinearity, identified by tolerances values approaching 0.1 in these predictors.

The first step for the model selection consisted in fitting the global models of each set of explanatory variables to the data, examining the goodness-of-it of each model with a χ^2 statistical test (Johnson and Omland, 2004). Due to our reduced sample (50 plots), models only included up to four variables at a time, as recommended by Harrell et al. (1996), representing all possible combinations of the variables (excluding interactions) with no model including more than one variable from each general category of variables (i.e. plot topography, soil characteristics, plot isolation, and anthropogenic disturbances). In total, we analysed a total of 479 models. We fit each model to the data by using an Ordinary least squares method and performed model comparison of all possible models by using the Akaike information criterion (AIC) for selecting the best set of explanatory variables in describing the variation of species diversity based on the minimum AIC score (Burnham and Anderson, 2002; Quinn and Keough, 2002; Johnson and Omland, 2004). We calculated the AIC difference (ΔAIC_i) and Akaike weights (w_i) for the ten best ranked models fitting the data to assess the statistical level of support for a given model (Table A.7). Due to the low AIC differences between models, we considered a subset of models with $\Delta AIC_i < 4$ to estimate the relative importance of individual explanatory variables (Burnham and Anderson, 2002). We then repeated the analysis using Richness index as dependent variable.

3.3 Results

3.3.1 Protection status and ecological and human disturbance variables

A one-way ANOVA analysis of variance test showed significant differences in the explanatory and dependent variables measured in forest commons inside and outside Picos de Europa (Table 3.2). Mean values of longitude and latitude showed significant difference in plots inside (mean ± SD: 368965.8 ± 5080.0) and outside (357567.4 ± 4456.5) Picos de Europa, arguably because of the geographical location of the PA in the northwest part of the Liébana region. Our results suggest that there is a significantly thicker layer of organic matter in the soils of the plots outside (7.1 ± 2.5) than in the soils of plots inside the PA (4.8 ± 2.8). Additionally, there is a lower presence of surface stones in the plots outside the PA.

Regarding the anthropogenic variables, plots inside Picos de Europa were more distant to towns (2225.6 ± 907.6) than plots outside the PA (817.2 ± 434.1), likely due to the reduced number of villages that have their municipality boundaries inside the protected area. Nearest town population density presented an average of 66.6 and 37.6 inhabitants, respectively, a situation explained by the presence of two very touristic villages inside Picos de Europa, *Espinama* and *Pido*. We also found significant differences in the implementation of silvicultural practices, with presence of some type of silvicultural systems –regeneration felling, forest cover improvement and/or ground improvements– in 80 per cent of studied plots inside the PA and only in 52 per cent for plots outside it. Specifically, plots inside Picos de Europa had a higher presence of cover improvement treatments (80 per cent) than plots outside (48 per cent). We did not detect significant differences in the presence of regeneration felling or ground improvement practices.

The census of the 50 sample plots led to the identification of a total of 14 families and 17 tree species, with an average number of 2.52 (SD=1.7) and 3.8 (SD=1.5) species per plot in plots inside and outside Picos de Europa, respectively. We observed significant differences between species abundance distributions in Shannon index means across plots. Specifically, the mean Shannon index was lower in plots inside Picos de Europa (0.45 ± 0.5) than in plots outside it (0.86 ± 0.4). We did not find significant differences for the Richness index, which also showed lower values in plots inside (0.32 ± 0.2) than outside Picos de Europa (0.47 ± 0. 3).

Table 3.2. Mean and standard deviation (SD) (or percentage for categorical variables) of variables considered, by location in relation to Picos de Europa. The One-way ANOVA analysis of variance compares the means in plots outside and inside the protected area, in bold significant differences based on their *p-values* (** *p-value* ≤ 0.01; * *p-value* ≤0.05) .

			Pool		Outside		Inside	
Variable	Code	Specification	Mean (±SD)	p-value	N[a]	Mean (±SD)	N	Mean (±SD)
Explanatory variable								
Topography								
Longitude	UTMY	Degrees	363266.6 (±7450.6)	.000	25	**368965.8**** (±5080.0)	25	**357567.4**** (±4456.5)
Latitude	UTMX	Degrees	4779509 (±4620.2)	.042	25	**4778188*** (±4004.1)	25	**4780830*** (±4890.2)
Slope	SLO	Percentage	46.9 (± 12.7)	.276	24	48.9 (±12.5)	25	44.9 (±12.9)
Soil characteristics								
pH	PH	Numeric scale	6.2 (±0.8)	.177	24	6.0 (±0.7)	21	6.4 (±0.95)
Texture	TEX	1- Sandy 2- Loam 3- Clay	45.8% 22.9% 31.2%	.388	23	43.5% 39.1% 17.4%	25	48.0% 8.0% 44.0%
Organic matter	OM	Centimetres	6.0 (±2.9)	.004	25	**7.1**** (±2.5)	25	**4.8**** (±2.8)
Stoniness	STO	1- Without stones 2- Low stony 3- Stony 4- Very stony	26.0% 32.0% 18.0% 24.0%	.043	25	**36.0%*** **36.0%*** **12.0%*** **16.0%***	25	**16.0%*** **28.0%*** **24.0%*** **32.0%***
Isolation								
Distance to path	DIST1	Meters	161.78 (±188.4)	.651	25	149.6 (±154.1)	25	174 (±220.1)
Distance to town	DIST2	Meters	1521.4 (±1000.9)	.000	25	**817.2**** (±434.1)	25	**2225.6**** (±907.6)
Town population	POP	Number of inhabitants	52.7 (±31.7)	.001	23	**37.6**** (±30.6)	25	**66.6**** (±26.3)
Anthropogenic disturbances								
Grazing	GRA	0- No presence 1- Presence	32.0% 68.0%	.071	25	44.0% 56.0%	25	20.0% 80.0%
Silvicultural treatments	SILV	0- No presence 1- Presence	34.0% 66.0%	.037	25	**48.0%*** **52.0%***	25	**20.0%*** **80.0%***

[a] Not available or unclear observations excluded from the analysis.

Table 3.2. (Cont.) Mean and standard deviation (SD) (or percentage for categorical variables) of variables considered, by location in relation to Picos de Europa. The One-way ANOVA analysis of variance compares the means in plots outside and inside the protected area, in bold significant differences based on their *p-values* (** *p-value* ≤ 0.01; * *p-value* ≤0.05) .

			Pool		Outside		Inside	
Variable	Code	Specification	Mean (±SD)	p-value	N[a]	Mean (±SD)	N	Mean (±SD)
Explanatory variable								
Anthropogenic disturbances								
Regeneration felling	FEL	0- No presence 1- Presence	48.0% 52.0%	.093	25	60.0% 40.0%	25	36.0% 64.0%
Cover improvement	COV	0- No presence 1- Presence	36.0% 64.0%	.018	25	**52.0%*** **48.0%***	25	**20.0%*** **80.0%***
Ground improvement	GRO	0- No presence 1- Presence	96.0% 4.0%	1.00	25	96.0% 4.0%	25	96.0% 4.0%
Dependent variable								
Shannon	H'		0.64 (±0.5)	.004	21	**0.86**** (±0.4)	25	**0.45**** (±0.5)
Richness	D_{Mn}		0.39 (±0.3)	.064	21	0.47 (±0.3)	25	0.32 (±0.2)

[a] Not available or unclear observations excluded from the analysis.

3.3.2 Ecological indices

Although we observed higher species diversity in the mixed broadleaf woodlands plots inside Picos de Europa than in the plots outside, we did not find any significant differences in the ecological indices calculated (Figure 3.2 and Table A.2). In contrast, although protected and unprotected *Fagus sylvatica* plots were relatively similar in terms of spatial diversity, the richness and evenness was higher in plots outside (Figure 3.2 and Table A.3). Finally, we do not find any significant differences between *Quercus pyrenaica* plots inside and outside the protected area (Figure 3.2 and Table A.4).

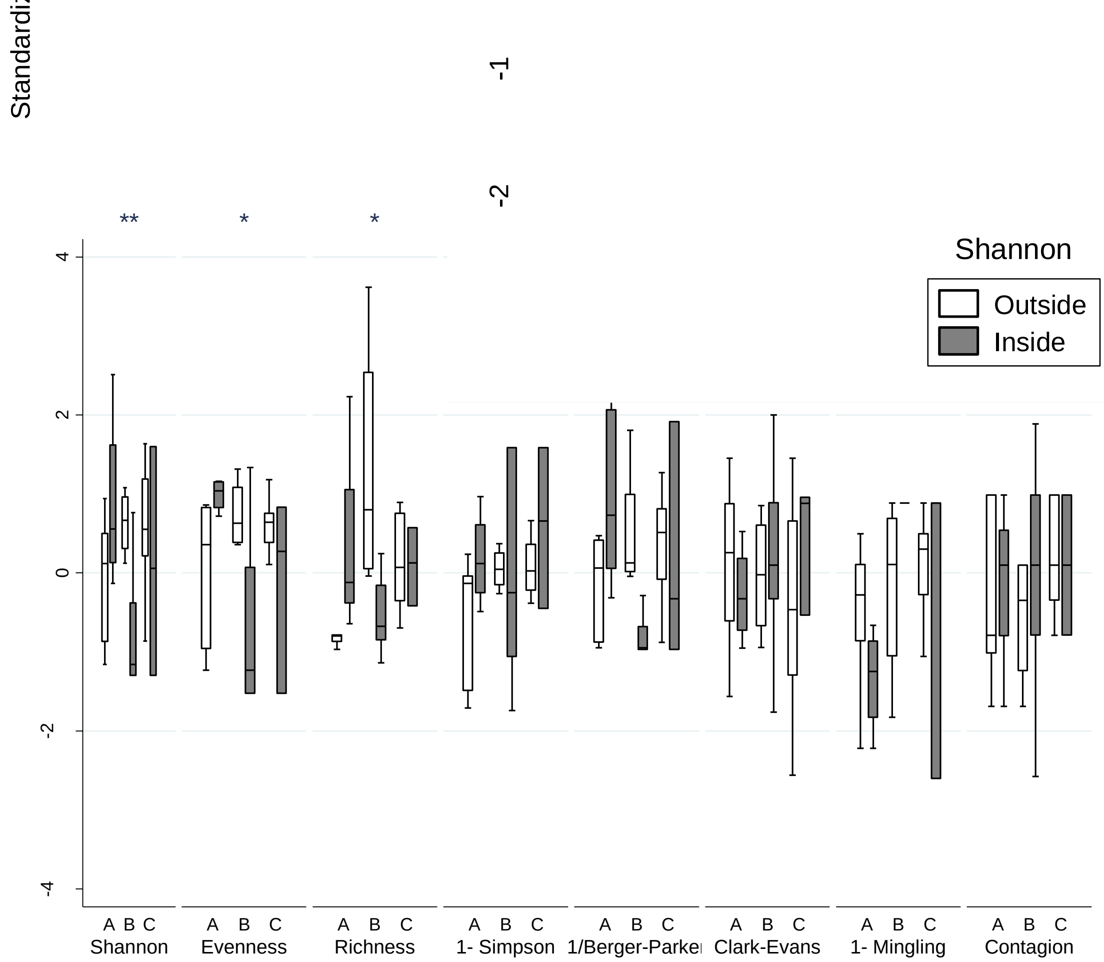

Figure 3.2. Boxplots illustrating Z scores for the five species composition indicators (n=46) and the three stand structure ones (n=49) for mixed broadleaf woodlands (A), *Fagus sylvatica* forests (B), and *Quercus pyrenaica* stands (C). Asterisks indicate significant differences in means between protected and unprotected *Fagus sylvatica* forests based on their *p-values* after Bonferroni correction (** *p-value* ≤ 0.005; * *p-value* ≤ 0.025).

3.3.3 Model selection

According to the AIC, the best-ranked model for explaining the evenness of tree species of forest commons in the Liébana valley is given by Eq. (1):

$$H' = (-0.002) \times \mathrm{SLO} + (-0.003) \times \mathrm{PH} + (-0.0002) \times \mathrm{DIST2} + (-0.196) \times FEL + 1.287 \quad (1)$$

where H' is the Shannon index value of the studied plots. This model had an AIC_c on 47.75 which gave $w_i = 5.71\%$ (Table A.7). The best-ranked model for explaining the species richness of forest commons in the Liébana valley is given by Eq. (2):

$$D_{Mn} = 0.028 \times OM + (-0.0004) \times DIST1 + 0.289 \quad (2)$$

where D_{Mn} is the Richness index value for the studied plots. This model had an AIC_c on 5.22 which gave w_i = 2.33% (Table A.7). In both models Akaike weights have a very low value, indicating uncertainty that these models are the best approximating models to our data (Symonds and Moussalli, 2011). Given this uncertainty, and following McCracken et al. (2015), we tested the dependent variables against the explanatory variables included in the subset of models with ΔAIC_i < 4, estimating the Akaike weights for these candidate explanatory variables and their relative importance across the subset models (Table 3.3).

Table 3.3. Importance of environmental and anthropogenic variables considered across the 50 study plots. Importance was derived using Akaike weights (w_i), reporting in bold the most important variables (with importance ≥0.4). See table 3.2 for variables definition.

	Topography			*Soil characteristics*				*Isolation*			*Disturbances*				
Variable	UTMY	UTMX	SLO	PH	TEX	OM	STO	DIST1	DIST2	POP	GRA	SILV	FEL	COV	GRO
Response = Shannon index. Models where Δ_{AIC}< 4 = 44 of 473.															
Importance	.05	.06	**.63**	**.44**	.29	.04	.08		**1**		.08	.10	.32	.09	.08
Response = Richness index. Models where Δ_{AIC}< 4 = 93 of 473.															
Importance	.18	.17	.20			**.78**	.13	**.40**	**.49**	.02	.07	.11	.16	.12	.17

The analysis shows that the two abiotic variables slope of the plot (SLO) and soil acidity (PH), as well as the anthropogenic variable distance to nearest town (DIST2) are the explanatory variables most associated with tree species evenness. All these variables bear a negative association with the Shannon index thus suggesting that, in general, species evenness is higher in plots with lower slopes, less acid soils, and closest to human settlements. Presence of anthropogenic disturbances as regeneration felling practices (FEL) also show a negative association to species evenness, although the strength of this association is weak.

In addition, the soil characteristic thickness of organic matter (OM) and the anthropogenic variables distance to nearest path (DIST1) and town (DIST2) seem to play an important role in the distribution of vegetation communities. Tree species richness was positively associated to organic matter thickness, indicating that many species prefer soils with a deep organic humus layer. In contrast, the Richness index was negatively associated to plot isolation variables, showing more species diversity in more accessible plots, or plots located closer to human settlements.

3.4 Discussion

Our results generally suggest that tree species composition is less heterogeneous inside than outside the Picos de Europa National Park. Two main factors may explain this result: the dominance of monospecific *Fagus sylvatica* forest inside the protected area and the application of silvicultural systems oriented to promote the presence of beech forests inside the protected area. Our results also suggest that human intervention variables, particularly distance to nearest town, are more important drivers of tree species distribution and diversity in forest commons of the Liébana valley than the abiotic factors considered for the analysis.

3.4.1 Ecological variables

The main goals of this work were to quantify differences in ecological characteristics between forest commons inside and outside Picos de Europa and to analyse how location in relation to the protected area relates to tree species distribution and diversity.

We found that, compared to soils in the surrounding landscape, soils inside Picos de Europa are characterized by a higher abundance of stones and a thinner humus layer, two important soil parameters associated to plant diversity in forests (Cantero et al., 2003; French et al., 2008; Ren et al., 2012). Stoniness is negatively associated to species richness in the study area, a result in line with previous studies in Mediterranean environments (Ceacero et al., 2012). However, according to our best-ranked models, stoniness is not a relevant variable for tree species diversity. In contrast, topsoil organic matter content shows an important effect, particularly on the Richness index, showing a positive association with species composition. This result, which matches previous studies analysing the effect of organic matter mass in temperate deciduous forests in central

Europe (Härdtle et al., 2003), may explain the lower value of the Richness index of plots inside Picos de Europa. However, topsoil organic matter is not associated in a significant way to the Shannon index, the species diversity indicator that actually differed between protected and unprotected plots.

Our best-ranked models also show an association between species richness and pH soil acidity and plot slope. These results match findings observed in earlier studies analysing the effects of edaphic and topographic factors on species richness (Härdtle et al., 2003; French et al., 2008; Mod et al., 2016). Still, these two ecological factors are similar in all study plots, for which they do not help to explain differences on species composition between plots in the protected area and outside it.

Finally, we observed lower Shannon index values in plots inside the PA, indicating a more heterogeneous landscape outside Picos de Europa. A possible explanation for this result is the dominance of *Fagus sylvatica* stands inside Picos de Europa, which contrast with the dominance of mixed stands in plots outside Picos de Europa. As beech forests are characterized for conforming monospecific stands (Krämer and Hölscher, 2009), this homogeneity of tree species might result in low values of the diversity Shannon index. Similar differences have been observed in other studies comparing ecological features of near-natural *Fagus sylvatica* stands and communities with less proportion of one dominant tree species (Hui et al., 2011).

Overall, soil's organic matter seems to be the only ecological factor studied contributing to explain the lower species inside the PA. Yet, soil organic matter is not associated to the Shannon index, the only indicator that significantly differed between plots inside and outside the PA. Thus, overall, our results do not show key association between ecological variables and species richness in Liébana forest commons.

3.4.2 Anthropogenic disturbances

The second objective of this study was to quantify differences in human-dominated disturbances between forest commons under the protected status and forest commons outside it, and to analyse how these factors may influence tree species richness distribution in the study area.

Best-ranked models for Shannon and Richness indices show a significant association between tree species composition and anthropogenic disturbances, particularly plot isolation and presence of silvicultural systems. The most important variables in our models are those measuring distance to the nearest path and village, and, with less direct importance, presence of regeneration felling treatments. These findings dovetail with recent research evaluating variables associated to species diversity, which also emphasize the importance of including anthropogenic disturbances in species diversity analyses, particularly in human-dominated landscapes (Guèze et al., 2015; Mod et al., 2016). Five aspects deserve further discussion.

First, the results of the correlation analysis underscore the significant association between species richness and distance to the nearest human settlement, a finding previously reported in tropical forest of Bolivian Amazon (Guèze et al., 2015) and in pine and oak forests across Mexico (Silva-Flores et al., 2016). The finding, however, contrasts with one study in temperate oaks in Spanish Central Pyrenees showing no association between distance to nearest town and *Quercus* spp. spatial distribution (Kouba et al., 2011). We argue that these differences in findings probably relate to variations in the socioeconomic characteristics in the study sites, which might have resulted in different impacts on the ecological system (Meyer and Crumley, 2011). Thus, during the second half of the twentieth century, the Pyrenean region suffered an important depopulation and a consequent abandonment of human activities, a phenomena that frequently results in the homogenization of the rural landscape mosaic as woody species colonized abandoned lands (Rotherham, 2013; Viedma et al., 2015; Lavorel et al., 2017). Although the Liébana region also suffered from depopulation during that time –i.e. population decreased by 54 per cent from 1950 to 1981 (Reques, 1997)–, the increasing demand for food supply by the neighbouring industrialized areas favoured the specialization in livestock production of Liébana from the 1970s onwards (González, 2001). Traditional livestock farming operations such as hay making, pruning, or grazing on forest commons allowed the maintenance of a grassland-woodland mosaic, and the biodiversity dependent upon these practices, until the early 2000s. Timber harvest of native deciduous species like *Fagus sylvatica* continued until 1980s, although the twenty century saw a shifting to harvest non-native conifers from the commercial timber-producing plantations established by the State Forestry Administration intervention of public woodlands in Liébana. This shift favoured the co-occurrence of wild and synantrophic species (Ezquerra and Gil, 2004).

Second, our modelling methods show a negative association between distance to nearest town and species diversity, i.e. forest species diversity is higher in areas close to human settlements. This association might be due to the human pressure potentially associated to higher accessibility, as Guirado et al. (2007) also noticed for periurban oak forests of north-eastern Spain, where presence of recent human disturbances is associated to higher species richness, particularly synanthropic species. Other studies have also linked intensity of human disturbances to the presence of pioneer, non-native species in natural habitats (Battles et al., 2001; Gourlet-Fleury et al., 2013; Bauman et al., 2015). In our study, we observed *Castanea sativa* and *Ilex aquifolium*, species traditionally pruned for the provision of food (chestnuts) and winter fodder, in plots located less than 1000 m away from towns. We also recorded plantations of the exotic *Pinus radiata* D. Don inside or around accessible plots located less than 650 m away from human settlements. Inside these plots, we also recorded species untypical for the studied forest habitats, like *Arbutus unedo* L. and *Pyrus* sp. These results may provide further support to the idea that plot isolation does not necessarily result in an enriched forest habitat, as human intervention may sometimes increase the total number of species by introducing atypical species in a community (Helm et al., 2015).

Third, this research also shows a relevant association between species diversity and distance to nearest path, including unpaved roads or walking trails trampling by foot, animals, or wheeled vehicles, but in which vehicle traffic is limited to forest rangers and local inhabitants use. Species diversity bears a negative association distance to nearest path; in other words, forest species diversity is higher in better accessible areas, a finding in line with a recent study analysing the effects of human path, trails, and roads on plant species richness (Root-Bernstein and Svenning, 2018). Nevertheless, distance to path does not significantly differ between plots inside and outside Picos de Europa, for which this anthropogenic disturbance does not assist in explaining differences on species richness between protected and unprotected sites.

Fourth, our findings show that plots inside Picos de Europa tend to present more silvicultural treatments, particularly forest cover improvement treatments, than plots outside the park. Greater forestry operations inside the PA may be counterintuitive for practitioners considering the protection category of National Park as forest reserve where minimal intervention is applied to allow a continued succession and natural disturbances in the forests. In Spain, however, National Parks are actively managed by authorities who implement management strategies oriented to safeguard the natural systems that justified the PA designation (Law No. 30/2014 of Spanish Government). In Picos de Europa, these strategies center around silvicultural activities oriented to preserve the habitats that justified its designation, i.e. Atlantic Forest natural and semi-natural habitats such as *Fagus sylvatica* forest patches (Regional Forest Administration, pers. comm., September 2016). Although Picos de Europa National Park management plan was derogated in 2005 due to a legal action taken by local communities living in the buffer zone (Spanish Government, 2005), silvicultural operations are undertaken under the umbrella of national and regional legislation and European conservation initiatives like the LIFE+Cantabrian Capercaillie Project (LIFE09 NAT/ES/000513, 2016). Thus, weeding and brushing out of trails have been carried out to facilitate the access of visitors to the National Park (Park ranger, pers. comm., September 2016). In addition, the conservation strategy followed by the National Park Administration to promote the presence of the endangered bird subspecies Cantabrian capercaillie (*Tetrao urogallus cantabricus*), considered an umbrella species in montane forest ecosystems (Blanco-Fontao et al., 2011), includes silvicultural interventions on forest cover practices in an attempt to favour capercaillie's habitat. These silvicultural practices included brush cutting and thinning applied to reduce canopy closure and to facilitate *Vaccinium myrtillius* L. growth, an important food source to capercaillie's populations (Lakka and Kouki, 2009; Mikoláš et al., 2015). Accordingly, we found that plots inside Picos de Europa tend to present more silvicultural treatments, particularly forest cover improvement treatments, than plots outside the park. Overall, silviculture inside the reserve seems to affect tree composition by reducing woody species diversity.

Finally, when studying the influence of silvicultural systems on species' diversity, we find that the only variable with relative importance in our models is the presence of regeneration felling operations, which shows a negative association to the Shannon index. Our results mainly relate to the effects of selective felling, a low-intensity clear-cutting activity practice consistent in an individual-tree selection cutting 'that maintains or develops an uneven-aged forest structure over time' (Lexerød and Eid, 2006, p.503). Several reports have discussed the association of low intensity, close-to-nature silviculture to the presence of vegetation typical of late-successional stages (Battles et al., 2001; Saeki, 2007). On the one side, by benefitting the presence of late-successional ecosystems, selective felling contributes to a higher evenness of the first dominant tree species in the study area, *Fagus sylvatica* (Ministerio de Medio Ambiente, 1997–2007). On the other, when focusing on mixed broadleaf forests, our study brings deeper insights into the assumption that non-intensive silvicultural systems promotes uneven-aged stands that benefit species diversity by increasing vegetation composition associated to late-successional ecosystems of forest sites. Particularly, we observed *Corylus avellana*, as dominant tree species, and *Crataegus monogyna*, as accompanying tree species, in mixed broadleaf forests plots presenting selective felling practices. The latter finding is in line with published studies analysing the effects of selective felling in coniferous and deciduous species, which also consider selective felling as a management practice that favours biological diversity for those forest habitats (Atlegrim and Sjöberg, 2004; Martín-Alcón et al., 2015).

In sum, silviculture inside the reserve seems to affect tree composition by reducing woody species diversity. Among the silvicultural operations considered, presence of regeneration felling practices is the only treatment with relative importance for species' diversity, favouring the evenness of species associated to late-successional ecosystems. Remarkably, the anthropogenic variable distance to town is a key driver of species diversity in the study area, with a higher number of species in plots closer to human settlements.

3.4.3 Protection status and species diversity interaction in three forest habitats

When comparing species diversity of three different temperate deciduous forest habitats occurring in forests commons inside and outside Picos de Europa, we observe similar tree species pattern for mixed broadleaf forests and Pyrenean oak forests regardless the protection status. Our findings are in line with Gray et al. (2016) when comparing biodiversity between protected and unprotected areas with the same land use. We argue that the finding is explained by the fact that, despite differences in the protection status, both sites are managed by the Forest Administration of the regional government, who, in the absence of a valid management plan in the PA, applies similar forest management technical criteria inside and outside the protected area (Forest ranger, pers. comm., August 2016).

Interestingly, we found differences in the ecological features of one forest habitat type, *Fagus sylvatica* stands, which have a significant more heterogeneous species composition outside Picos de Europa. Specifically, we observed significant differences for the Shannon, Evenness and Richness indices and for the reciprocal form of the Berger-Parker index (Figure 3.2), similar to the ones observed by Bilek et al. (2011) when comparing managed and unmanaged beech forests in Central Europe. The low value of Shannon index in the beech forests inside Picos de Europa indicate a single-layered *Fagus sylvatica* composition of these stands, in line with observations in pure beech forests of the Basque Country, a neighbouring region (Peña et al., 2011). Nevertheless, although a high portion of accompanying species may not be considered a natural pattern of beech stands, studies analysing the temporal and spatial dynamics of near-natural deciduous forests dominated by *Fagus sylvatica* show a forest cycle based on 'gap-dynamics', in which species composition oscillate due to small scale disturbances such as canopy openings (Wissel, 1992; Emborg et al., 2000).

Briefly, by promoting the presence of monospecific beech stands without including species associated to the natural small-scale disturbances of these habitats, interventions carried out in beech forests inside Picos de Europa do not seem to contribute to higher stand diversification, plots inside the PA being poorer in terms of species than plots outside the PA.

3.5 Conclusion

Results presented herein show more heterogeneity of forest communities outside than inside Picos de Europa National Park, arguably as silvicultural systems applied inside the designated area benefit the presence of single-layered *Fagus sylvatica* forests. Aiming to provide some management guidelines to favour biodiversity maintenance in this protected area, we recommend the inclusion of practices emulating natural disturbances of beech forests, promoting the appearance of pioneer light-demanding specialist species associated to these forest habitats. This recommendation implies a strategy to enhance the resilience capacity associated to *Fagus sylvatica* ecosystems, particularly relevant in the context of climate change in the study area, which might increase vulnerability to pests and phenological changes of forest habitats (OECC, 2012). Further research under scenarios of climate change is required to attain more integrative management recommendations.

Within the range of management options for promoting tree species assemblages in this forest habitat type, our work brings into consideration the positive influence that human-induced disturbances hold for increasing tree species richness and evenness. This option might be especially relevant in natural landscapes with long histories of forest use, such as the study site. Our study also reinforces the social-ecological systems approach to biodiversity conservation by providing an example of how the persistence of some stewardship customs and traditional uses of local communities in forest commons of the Liébana valley seem to result in the long-term maintenance of diverse natural habitats and their associated species. We recommend further research to identify which particular forest-related practices performed in forest commons outside Picos de Europa enrich community composition through habitat-specific species.

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Chapter IV

Traditional wood carving of *albarcas* in Lamedo, Liébana (Cantabria, Spain). Photo credits: S.Guadilla-Sáez

The role of traditional management practices in shaping a species-rich habitat mosaic in a mountain region of north Spain

This chapter corresponds to the article: Guadilla-Sáez, S., Pardo-de-Santayana, M. & Reyes-García, V. The role of traditional management practices in shaping a species-rich habitat mosaic in a mountain region of north Spain. In review. *Land Use Policy*

Paper III

The role of traditional management practices in shaping a species-rich habitat mosaic in a mountain region of north Spain

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Abstract

Through traditional practices that typically impact the surrounding natural areas, rural communities worldwide have created and maintained landscapes with a diverse mosaic of species-rich habitats. In Europe, where a high portion of species is dependent on the persistence of traditional rural landscapes, the progressive abandonment of agricultural land use activities has been often accompanied by a biodiversity decline, although the precise implications of that landscape transformation for species and habitat conservation are not sufficiently well-known. This study examines the evolution of traditional management practices and the local perceptions of impacts on ecosystems diversity derived from the abandonment of traditional land uses in a mountain region in

Spain that preserved a complex farming system based on traditional agricultural land use activities until the mid-twentieth century. By exploring local communities' perception of rural landscape transformation, our results illustrate a set of traditional management practices favourable to habitat diversity, such as pastoralism of small ruminant livestock. Our results also suggest that local perception of landscape change in the area dovetails with literature, providing further understanding of the particular ecological implications of each underlying driver of land use change identified. We conclude that the combination of local knowledge and nature conservation science can help to develop effective regional conservation strategies based on management practices simultaneously favourable to biodiversity and economically profitable. Our study provides evidence that rural communities can be a valuable source of information to document landscape historical dynamics and to monitor environmental changes, which might be particularly relevant for landscape-orientated conservation policies aiming to prevent the biodiversity loss resulting from the abandonment of traditional land uses.

Keywords: Biodiversity conservation; oral history; rural landscape; traditional knowledge.

4.1 Introduction

Several studies report the geographical overlap between the world's biological hotspots and indigenous and local communities' homelands (Porter-Bolland et al., 2012; Guèze et al., 2015; Garnett et al., 2018). Through traditional practices closely connected to the surrounding natural areas, Indigenous peoples and local communities worldwide have created landscapes of high cultural and ecological value (Parrotta and Agnoletti, 2007). For example, historically low-intensity human disturbances have benefited stress-tolerant and habitat-specialist species over ecological competitors, often resulting in high biodiversity in traditional rural landscapes (Calvo-Iglesias et al., 2006; Rotherham, 2013). Indeed, almost all contemporary European landscapes are shaped by human intervention, resulting in a rich mosaic of habitats and species diversity closely dependant on the persistence of traditional practices (Biró et al., 2014; Babai et al., 2015; Molnár et al., 2016; Kis et al., 2017).

In Europe, the Industrial Revolution generally led to the breakdown of natural resources exploitation by rural societies and the progressive abandonment of traditional landscapes (Vidal-González, 2014), a process that became a major threat for the conservation of species linked to these ecosystems. Since the mid-twentieth century, further changes have taken place in Europe's rural landscape, resulting in additional land use changes, including the abandonment of agricultural lands. The literature associates agricultural lands' abandonment to a combination of drivers including demographic, economic, technical, and institutional (van Vliet et al., 2015). Overall, these changes have resulted in a simplification and homogenization of agricultural and woody landscapes (Agnoletti, 2013; Plieninger et al., 2016; Lasanta et al., 2017), as the natural vegetation succession that takes place in abandoned areas favours density of forest habitats and the reduction of the ecosystem mosaic shaped by traditional management practices (Rotherham, 2013; Viedma et al., 2015; Lavorel et al., 2017). However, the precise effect of each driver on European landscape transformation for species and habitat conservation are not sufficiently well-known (Corbelle-Rico et al., 2015).

This study examines landscape dynamics, as reported by local inhabitants, in a mountain region of north Spain that preserved a complex farming system based on traditional land use activities until the mid-twentieth century. The paper is divided into two sections. First, we explore the local perception of traditional management practices to determine which particular historical land uses may have contributed to the shaping of a set of high diverse ecosystems in the regional landscape. We further document the techniques and ecological knowledge associated to the historical land uses identified during the analysis. Second, we present the drivers of landscape change in the region to understand the environmental implications derived from the abandonment of traditional land uses that took place from the 1950s to the present day. For this purpose, we combine local inhabitants' discourses on historical landscape transformation with information provided by the literature, an approach that allows exploring the potential of local discourses to document landscape's land use change. We conclude with a discussion of the potential of traditional management practices to preserve rural landscape heterogeneity and the importance of using local discourses to document both landscape changes and the local ecological knowledge embedded in traditional management practices.

4.2 Case study

This research was conducted in the Liébana Valley, a region of 56,600 hectares located in the Cantabrian Mountains, north Spain (Figures 4.1 and 4.2). The area is in a tectonic basin consisting of steep reliefs, with altitudes ranging from 300 meters above sea level at the bottom of the valley to 2600 meters in the surrounding mountain system, with very limited communication routes to neighbouring regions. Liébana's orographic-driven isolation favoured the societal and economic differentiation of its inhabitants, which remained self-sufficient until the mid-twentieth century (ETSIM, 1978; Arbeo, 2012).

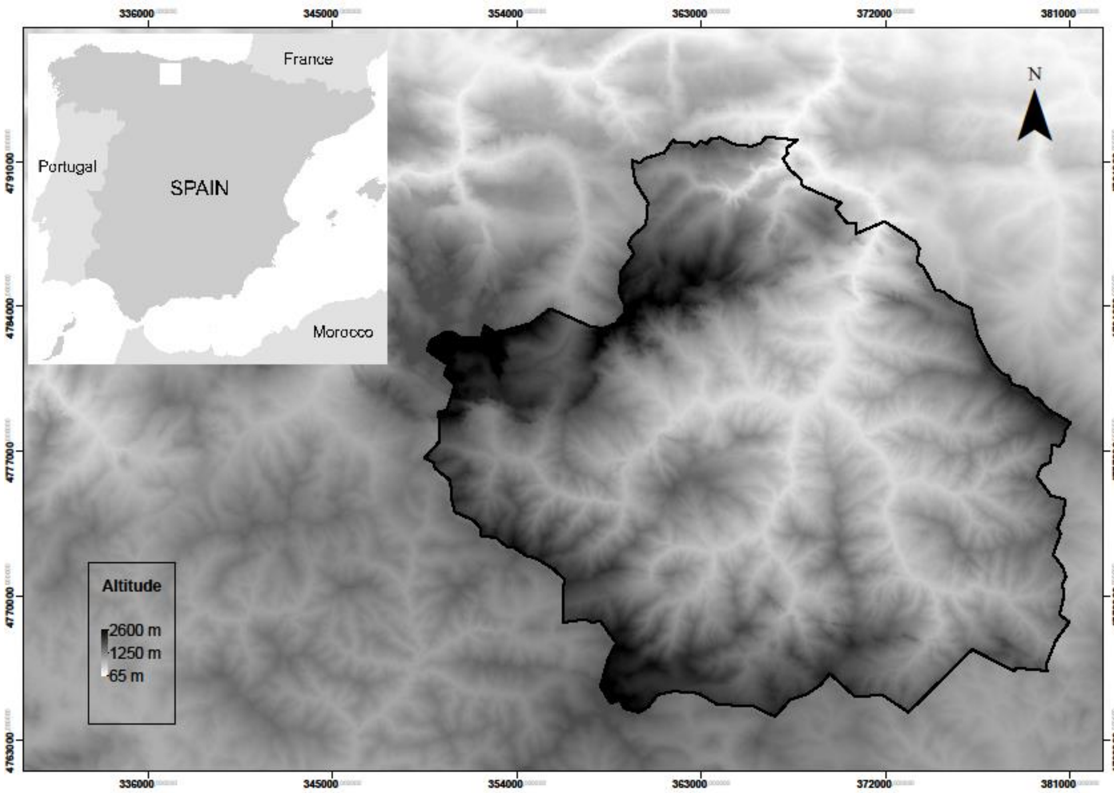

Figure 4.1. Geographical location and land surface elevation of the Liébana region (Cantabria, Spain).

Figure 4.2. Panoramic picture illustrates the steep relief of the Liébana region. Photo credits: S. Guadilla-Sáez.

Subsistence farming in Liébana was structured around the region's large variation of climatic conditions, which is influenced by both Atlantic and Mediterranean climates (Bertrand, 1964). Agriculture was traditionally based on the combination of cereal-legume cropping systems in the valley bottom and nearby hillslopes and livestock grazing in the steeper slopes and high altitudes. Topographic and climatic features also contributed to the presence of temperate deciduous forests from which Liébana's inhabitants obtained timber, firewood and fodder. Altogether, traditional land uses formed an altitudinal landscape mosaic structure consisting in dispersed human settlements surrounded by orchards or cereal crops in the valley, followed by a mixture of arable lands, meadows and forests in the lower slopes of the mountains, replaced by highland pastures and grasslands at higher altitudes (López, 1978; Castañón and Frochoso, 2007).

Between 1950 and 1980, the rural exodus associated to the region's integration into the market economy changed Liébana traditional landscape pattern. Migration, with the total population decreasing from 12,800 inhabitants in 1950 to 7,200 in 1981 –which continued to decline, but at a slower rate, in the following decades (ICANE, 2018)–, decreased the need of arable land supply. Additionally, the proximity to densely populated areas demanding meat and milk contributed to Liébana's specialization in livestock farming and its integration into supra-regional markets. As result, between 1955 and 1960, livestock farming became Liébana's economy main production focus (López, 1978;

González, 2001). From a landscape perspective, the conversion from arable land into pasture resulted in the county's landscape gradual simplification and homogenization.

Liébana's livestock farming operations, however, did not reach the level of mechanization that characterizes other rural areas of Spain. Uncertainty related to the shaky national market price for milk in the 1970s led to low levels of investment in the mechanization of Liébana's animal farms, as compared to other rural areas of the Cantabrian mountain range. In addition, milk production quotas established in 1986 with Spanish accession to the European Economic Community (EEC) contributed to invest in other economic activities, such as cheese production agri-businesses to channelize milk surplus, instead of in the mechanization of livestock farming operations (González, 2001).

The 1992 Common Agricultural Policy (CAP) reform, which addressed livestock farming innovation and encouraged intensive reproductive technologies, had important implications to extensive livestock farming in mountain areas such as Liébana. More intensive modes of livestock farming promoted by the CAP resulted in the progressive exclusion of local stock-breeders from the farming system (Corbera, 2006). From 1984 to 2003, Liébana experienced a decrease of a 40 per cent in the number of livestock farmers, although the number of cattle heads remained steady (Rescia et al., 2008).

Noteworthy, since 1960s the Spanish Administration has implemented incentives for tourism development in Liébana. Such incentives include communication infrastructures improvements and regional and private investments in ecotourism from the late 1980s onwards (González, 2001; Castañón and Frochoso, 2007). European initiatives in the 1980s further reinforced this trend, such as the 'Operating Programme for Development and Economic Diversification of Rural Zones' (PRODER) subsidies, a program promoting rural economies' diversification through the development of non-agricultural activities. These initiatives have resulted in Liébana regional economy moving towards the tertiary sector, particularly rural tourism.

In addition, the designation of national and international Protected Areas labels in the 1990s has been adopted as a strategy for regional development (Corbera, 2006; Voth, 2007). The outstanding natural and aesthetic values of Liébana landscape contributed to its inclusion in Picos de Europa National Park (hereafter Picos de Europa) in 1995. Picos de

Europa is part of the NATURA 2000 network of the European Union, and since 2003 designated UNESCO Biosphere Reserve. With more than 2,000,000 visitors in 2017 (MAPAMA, 2018), Picos de Europa has become a successful touristic destination, increasing the number of visitors in Liébana since its establishment (González, 2001). Nonetheless, as in other protected areas, the large number of visitors has also contributed to the intensification of land use (i.e., related to the touristic sector demands for transport and accommodation infrastructures and leisure activities), a demand that is incompatible with the presence of some extensive traditional practices key for the persistence of rural landscape (i.e., livestock grazing) (González, 2001; Bernués et al., 2005; Corbera, 2006). Moreover, park authorities' have regulated some traditional practices inside the protected area –such as extensive herding or firewood collection– in order to assure the compatibility of these uses with the national park conservation goals. Similarly to other European extensive land-use areas affected by a protection status (Riseth, 2007; Molnár et al., 2016), limitations to traditional uses have resulted in conflicts between park's managers and local population, particularly livestock herders (Rescia et al., 2008).

4.3 Methods

We used qualitative data collection methods to document traditional practices and perceptions of landscape change in the study area. Specifically, at the start of fieldwork, we used semi-structured interviews and participation in public meetings. We also reviewed written sources to identify i) traditional activities potentially important for preserving the Liébana's natural habitats heterogeneity and ii) factors that historically promoted or limited the use of forest-related resources in the region. We used this background information to prepare interview guides. Particularly, for all the relevant traditional practices identified, we conducted in-depth interviews to comprehensibly document the traditional ecological knowledge associated to them. For some local practices, such as timber harvesting, livestock breeding, and beekeeping, documentation also included the use of participant observation.

From April to September 2016 we conducted a total of 39 in-depth interviews with 42 residents of the study area. Participants ranged from 32 to 94 years of age, with most informants specialized in agriculture, livestock, or forestry private activities (Table 4.1). To choose key participants with experience in the local traditional system to manage forest-related resources, we used 'snowball sampling' (Gamborg et al., 2012). The selected participants were named as knowledgeable in one of the four activities identified as relevant during the first stages of fieldwork: livestock grazing, timber harvesting, non-timber forest products (hereafter NTFPs) collection, and hunting. Some respondents provided information on two or more knowledge domains. We collected information on each practice until we reached the saturation point, when no new information was coming from a new interview (Newing, 2011).

Table 4.1. Respondents' characteristics

	N	%
Gender		
Male	32	76
Female	10	24
Age category		
<60	18	44
>60	24	56
Activity group		
Agriculture, livestock or forestry private activities	26	62
Forest and National Park technicians	9	21
Political, educational, or clergy	7	17
Knowledge domain		
Livestock grazing	31	74
Timber harvesting	29	69
NTPFs	32	76
Hunting	14	33

All interviews were conducted in the locality where the informants lived or at their worksites. Before each interview, participants were informed about the purpose of the study and the eventual publication of the information provided; assurance of informants' anonymity was given and a verbal informed consent for participating in the research project was obtained (Bernard, 2006; Newing, 2011). Interviews were conducted in Spanish, lasted about an hour and followed an outline of 10 to 12 open questions (see Appendix III for full questionnaire). All interviews were recorded, transcribed, and imported to Microsoft Excel software for coding.

Data were analysed using qualitative and quantitative data analysis methods. To document traditional practices and their evolution to present-day farming and forestry systems, we applied a detailed qualitative content analysis (Albuquerque et al., 2014). The analysis consisted in an open coding of the knowledge domains that emerged from in-depth semi-structured interviews. To ensure consistency of the ethnography information reported, we organized the information provided by holders of traditional knowledge into a codebook proposed by Pardo-de-Santayana et al. (2014).

To understand local population's perception of the historical land use change in Liébana, we applied both qualitative and quantitative analyses. Since regional landscape have changed substantially over the past 60 years, analyses consisted in coding the drivers of change cited by respondents born between the 1920s and the mid-1950s (minimum 60 years old). Using as starting point for the discussion with respondents: "What main changes, if any, have occurred?" followed by "What most contributed this change?" the interviews covered expected topic of historical land use change. We applied qualitative content analyses to evaluate local perceptions on the consequences of each category of driver of change on landscape heterogeneity, classifying them depending on being unfavourable, indifferent, or favourable to landscape heterogeneity (Lopes-Fernandes et al., 2014). We conducted quantitative analyses to evaluate the perceived importance of a driver of change to regional land use change, measured as the frequency of total responses in the sample (Albuquerque et al., 2014). We distinguished between five main categories of drivers of change: demographic –e.g., rural exodus–, economic –e.g., integration to market economies–, technical –e.g., farming mechanization–, institutional –e.g., European

Union's PAC subsidies–, and ecological –e.g., ecological successional processes– (van Vliet et al, 2015; Plieninger et al., 2016; Colsaet et al., 2018).

4.4 Results

4.4.1 Perceptions of the potential of traditional management practices to biodiversity conservation

In this section, we comprehensively describe the traditional practices that according to the interviewees have contributed to the shaping of Liébana landscape through generations (Table 4.2). Our description includes informants' perceptions of how these practices support the maintenance of a more heterogeneous landscape mosaic.

Table 4.2. Traditional farming and forestry practices in Liébana.

Knowledge domain	Traditional practice	Cont.[a]	Transformation
Livestock farming	Transtermintance	No	Since late 1950s, transtermitance of collective herds to Liébana's summer upland pastures was mainly substituted by transhumance pasturing in private highland pastures of other regions, using trucks to move the animals.
Livestock farming	*Esmozar* (lopping)	No	Since 1980s, tree pruning for fodder was mainly abandoned due to the availability of commercial fodder.
Livestock farming	*Quemas* (prescribed burns)	No	Abandoned, although regional forest administration is considering to reintroduce controlled burning as a method to reduce woody biomass.
Timber harvesting	*Subastas* (logging)	No	In the 1960-1970s, the use of animals for transportation of logs was largely abandoned in favour of tractors and machines, and harvesting tools axes and saws were substituted by chainsaws. Currently, mainly exotic conifer species are professionally logged with sophisticated harvesting machinery such as tractor-mounted log loaders.

[a] Cont.: Continuity of the practice until present-day.

Table 4.2. (Cont.) Traditional farming and forestry practices in Liébana.			
Knowledge domain	Traditional practice	Cont.[a]	Transformation
NTFPs	*Suertes* (firewood collection)	Yes	To collect firewood, neighbours need to apply for a permit to the regional forest rangers. During the autumn, foresters mark the wooded lots applying a similar criteria to the one used in the past. For stands with holm oak (*Quercus ilex*) or Pyrenean oak (*Quercus pyrenaica*) as the dominant tree species, firewood collection is applied as a small thinning to an area. Whereas in stands with oak and beech as the dominant tree species, forest rangers specify the trees appropriate for firewood cutting, such as crooked-growing trees.
NTFPs	*Sudar el corcho* (cork stripping)	Yes	Abandoned since the 1970s, cork extraction has been recently reactivated in Liébana. Nowadays, a private company carries out the activity applying similar methods to the traditional barking, including the transportation of cork slabs by mules.
NTFPs	*Varear* (branch beating)	No	No longer practiced due to the injury risk of falling out of a tree.
NTFPs	Beekeeping	Yes	The traditional beekeeping system with bark hives changed forty years ago to the modern hive bee structure made of wood.
[a] Cont.: Continuity of the practice until present-day.			

4.4.1.1 Livestock farming

According to the informants, in the past livestock farming was characterized by a collective organizational system in which families in a village contributed with a limited number of domestic livestock, mainly goats and sheep, to a common herding flock that was gathered each morning in the village by a shepherd hired by the village council. Within this collaborative grazing system, known as *vecería*, villagers' domestic livestock –sheep, goats, cows, donkeys, horses– daily grazed together. Neighbours shifted turns to accompany the common shepherd, a system known as *cornuda*. Turns depended on the number of herding animals a family added to the common herd. For instance, "*Each twelve lambs, you went one day. If you had six, half day, that is, you went one turn but next time you didn't*" (male, 91-100 years old)[2]. This collaborative grazing system made livestock herding compatible with other farming activities, such as hay meadow, agriculture, or wood harvesting. As one informants described, the vecería system allowed "*conciliation, so that people could work*" (male, 71-80 years old).

In spring, villagers hired a shepherd or cowboy to take livestock to the higher pastures during the transtermitance (i.e., short transhumance from lowlands to highlands). Herds remained all summer in the highlands tended by a shepherd or cowboy, who lived in a hut with shepherds from other villages. During the transtermitance, villagers did not accompany the shepherd or cowboy but made turns to provide them with food. Informants pointed out that with the traditional herding practice, grazing animals were able to subsist with the resources available in the surrounding communal woodlands and pasturelands of Liébana. Informants also reported traditional practices that allowed to obtain fodder all year round. For instance, after the hay making (Figure 4.3), during the winter months community domestic livestock are able to freely graze in private fields, i.e., crops and hay meadows. Fields are closed to grazing from March onwards, to allow the growth of grass that is cut in summer and stored for winter, and opened again once the hay is cut, a system known as *derrotas*. During the months that private lands stay closed, livestock graze in the forest commons.

[2] To guarantee confidentiality of informants, their ages are grouped into 10-year intervals.

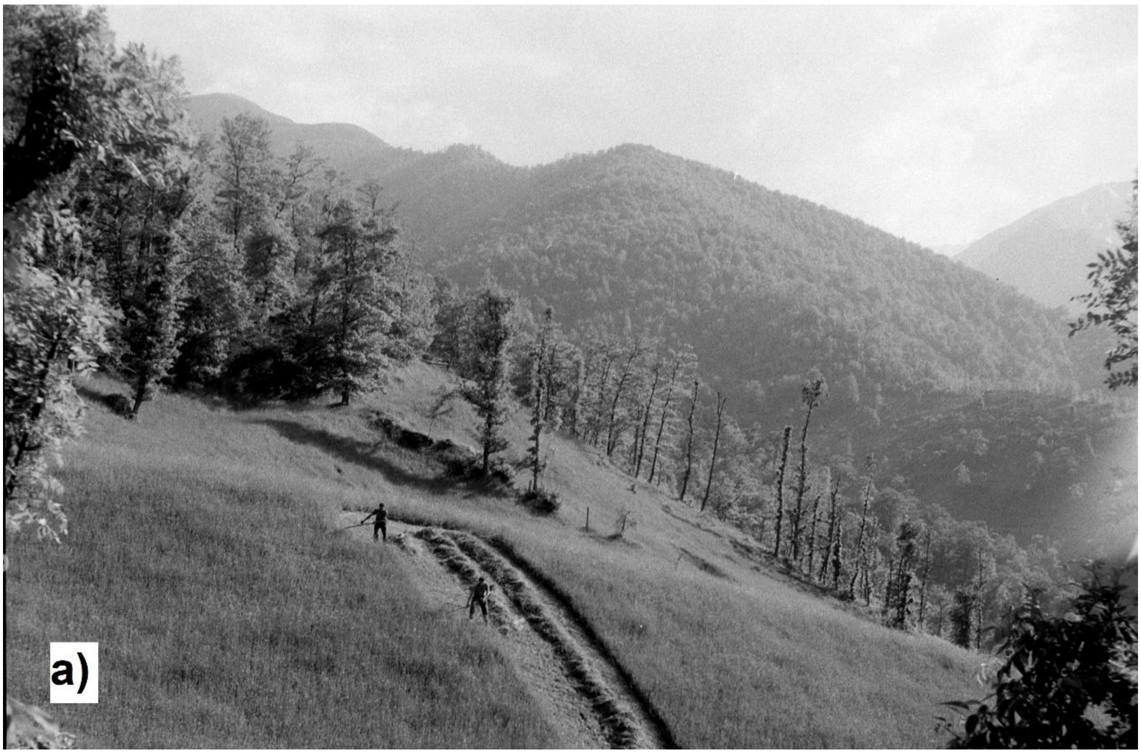

Figure 4.3. Past (a) and present (b) hay making practice in Liébana. Photo credits a) Eusebio Bustamante Miguel, and photo credits b) S. Guadilla-Sáez.

Another habitual practice to obtain winter fodder was lopping, or *esmozar*, described by informants as a type of pruning consisting in climbing to the tree crowns when branches bore acorns and leaves, and cut them to use as stock for wintertime. Individuals of *Fraxinus excelsior* L., *Ilex aquifolium* L., *Populus* and *Quercus* spp. located in private fields or communal lands were pruned (Figure 4.4). This technique was done on crescent moon, reportedly to enhance the growth of the branches during the following year. After the cut, twigs were collected in groups up to ten named *coloños* or *tarmaos*. Once dryed, the coloños were storaged in the housing farm and given to sheep, goats and cows when snow did not allow them to graze outdoors. Woody parts not eaten by the animals were used as firewood.

Figure 4.4. Lopped tree, or *esmozado*, in a private field in Lamedo, Liébana. Photo credits: S. Guadilla-Sáez.

Respondents also reported the use of prescribed burns, or *quemas*, to promote pastures for livestock and to reduce shrubland. It consisted of applying periodical burns as a clearing method to suppress the expansion of shrubland of *Pteridium*, *Erica* and *Ulex* spp.

Scrub burning did not affect the upper strata, i.e., tree canopy, and was followed by livestock grazing in springtime. For this controlled technique, a request to the forest ranger was granted before burning and it was compulsory for all neighbours to participate. A firewall of four to five meters was done on the opposite side of the field where the burn started. If there was not a firewall, neighbours located themselves in the opposite side of the burn to avoid the spread of the fire.

Informants argue that the recent transition to cattle and rural depopulation has fundamentally changed herding practices. Nowadays, animal husbandry mostly consists in herds grazing in privately owned meadows protected with electric fences to prevent attacks from wild animals. In winter months, livestock is housed on farms and fed with fodder acquired in the market; whereas in summertime, herds are moved to upland pastures in motorized transport. As one informant reported: "*Traditional extensive livestock systems is being replaced by a more intensive farming: meadows are fertilised by farmyard manure, grasslands are no longer harvested, there is not livestock herding nor transhumance…*" (female, 31-40 years old).

Depopulation is also considered an important determinant of the abandonment of traditional pastoralism, as farming activities are no longer distributed along family members. One informant explained changes in livestock farming in the area as follows:

> "*Small ruminant livestock has been abandoned because of the excessive labour necessary to take care of them. You need a farming house, being all day long with them, herding dogs; and, still, they are vulnerable to foxes, wolves… People are no longer used to it. Families were also bigger in the past with the father, grandfather, children… Instead, now a single person has to mow the meadows, attend the cattle and move it to upland pastures during summer… How can a person do everything? So, what does he leave aside? He leaves aside whatever gives him more work and less economic benefit. This is why sheep and goats have been abandoned and woodlands now look as they do*" (male, 41-50 years old).

Thus, informants considered the abandonment of extensive livestock farming as a key determinant of landscape variability, contributing to the expansion of tree and shrub encroachment on grassy areas. As another interviewee noticed, "*Nowadays livestock no* *longer goes into the forest, herders go to the farming house or to private meadows instead*" (male, 61-70 years old). In the same line, a retired shepherd observed, "*There are no longer shepherds or cowboys. Cows are moved from farmhouses to the meadows, with electric fences installed, and do not care about them. Is it possible more abandonment?*" (male, 81-90 years old). Moreover, informants argued that the resulting simplified mosaic landscape negatively affects the environment. In the words of one informant, "*The system of traditional or extensive livestock farming in Picos is essential… The disappearance of hay meadows, pastures, the abandonment of the communal land results in fire risks, problems in botanical diversity that is going to disappear*" (male, 51-60 years old).

Aware of the effects of pasture abandonment, the regional forest administration in collaboration with villages is promoting some initiatives to restore pasturelands. One village council representative explained: "*A mechanical clearing was made some years ago to recover part of our highland pasture… We increased the grassy area, positively affecting partridge birds, and now we are going to create a wetland for amphibians and for livestock to come and graze*" (male, 41-50 years old). Informants argue that livestock grazing after a mechanical clearing improves the long-term effectiveness of such interventions. Moreover, according to herders, small ruminant livestock that browse on woody plants, such as goats, are "*those who really clean the forest*" (female, 41-50 years old) (Figure 4.5), for which the decrease of browser range animals has been determinant on Liébana landscape transformation. As one informant recommended, "*There is a general idea that the Administration should give a payment to livestock farmers who accompany their flocks and have a minimum number of sheep or goats, so they would get an extra salary. It would be cheaper than forest fire prevention techniques*" (male, 31-40 years old).

Figure 4.5. Goat browsing woody plants in Dobarganes, Liébana. Photo credits: S. Guadilla-Sáez.

4.4.1.2 Timber harvesting

Timber harvesting, or *subastas*, were habitual in villages with large areas of forested common lands. In Liébana, stands of *Fagus sylvatica* L., *Quercus robur* L. and *Quercus petraea* (Matt.) Liebl. were the most abundant tree species felled. These tree native broadleaf species shed their leaves before winter, for which timber harvest was scheduled while the trees were leafless, i.e., from late October to late March or April, before sap flows. Local loggers also harvested according to the moon phase, cutting during the waning moon to prevent decay and termite hazard. In the subastas, groups of four to seven neighbours used to go into the woods with the harvesting equipment (i.e., hand axes, saws, wool rope and a carpenter's tape). They felled and limbed trees, bucking the tree to logs of prescribed lengths (e.g., into lengths of 1.40 or 2.80 meters for railroads ties), and transported them to a landing area, usually at the roadside, using draft animals such as cows, mules, oxen or horses. For remote areas, harvest operations included the construction of portable sawmills.

Income generated by the commercialization of trees used to be a key economic resource. Such income was reportedly used for the establishment and maintenance of villages' common services (i.e., electricity or piped water) or infrastructure (i.e., roads and bridges). When income was needed, the villages' councils requested to the regional forest administration an authorization to harvest trees. Regional foresters decided the volume and standing trees to be included in the woodlot, applying a criteria that is considered positive to forest conservation by most of the informants. As one interviewee pointed out, "*Foresters examined the forestland and defined the areas authorized for harvest. Areas with presence of young trees were excluded, whereas areas with old trees that were about to fall or that were impeding stand regeneration were included in the cutblock*" (female, 51-60 years old).

Cantabrian regional temporary banning of harvesting timber of native species in the late 1980s (Regional decree No. 64/1989, of 14 September) meant the end of traditional timber harvesting in Liébana[3]. Nowadays, professional timber harvesters operate in the region, using modern equipment for the logging operations, mostly harvesting plantations of *Pinus* spp. located in accessible private fields (Figure 4.6). Some informants argued that the 1980's regulation of tree harvesting ensured forest regeneration, but most of them considered that the traditional silvicultural management system implemented to remove the trees –i.e., shelterwood cutting, consisting in gradually replacing the mature trees of a stand through repeated cuts (Savill, 2015)– facilitated forest renewal. A respondent from a village with large area of beech forest stated: "*In my opinion, the banning of timber harvesting is a step backwards... Seedlings of beech benefit from the removal of old trees, because standing mature trees do not allow understorey trees to develop... In dense stands it seems necessary to cut by shelterwood cutting*" (male, 91-100 years old). Neighbours used to collectively carry out some forestry operations associated to traditional timber harvesting, like maintenance of logging unpaved roads or path clearing of forest commons

[3] Despite being a repealed decree, the harvest of native species stands has not been reactivated in Liébana. Notably, informants reported it as an active legislation.

through manual removal or swidden–fallow management. As one informant described, “*Each year some brushing or thinning was done to a small forest area. This consisted in the removal of thorny scrublands and shelterwood cutting*” (male, 81-90 years old).

Figure 4.6. Past (a) and present (b) timber harvesting practice. Photo credits a) Eusebio Bustamante Miguel, and photo credits b) S. Guadilla-Sáez.

There is a general local complain about the abandonment of forest management systems over the last few decades. Several informants argued that public administration should integrate into forest management planning actions to prevent the loss of traditional paths and to remove old or poorly formed trees from the stands.

4.4.1.3 Non-Timber Forest Products

The harvest of forest products other than timber is an age-old practice in Liébana that forms part of its complex farming subsistence system. Participants indicated that firewood, cork, fruits, medicinal plants, mushrooms, and beekeeping were the major NTFPs collected from Liébana's forest commons.

In the past, firewood collection, or *suertes*, was an important rural livelihood strategy because it provided household fuel needs for cooking and heating, as well as for the production of charcoal for forges and braziers. Great volumes of firewood obtained from poor quality stands of *Quercus ilex* L., *Quercus pyrenaica* Willd. and *Fagus sylvatica* L. were collected during the autumn, once the hay cut had finished. To that end, two or three family members went into the forest with hand axes, accompanied with draft animals to drag firewood out of the woods. As one neighbour described, "*We collected fuel wood from old trees or those damaged by wind or snow*" (male, 71-80 years old). Firewood collection persists in Liébana, although the total amount of wood collected is lower due to depopulation and to the use of other energy sources. As one stakeholder indicated, "*When butane gas arrived, the consumption of firewood reduced to less than 30 per cent of the volume previously extracted*" (male, 71-80 years old). Informants argued that the decrease of firewood collection had a negative effect in the maintenance of local landscapes, as they consider that this practice was essential to clear up the forest floor and promote the growth and fructification of the remaining trees. Commenting on the abundance of biomass on Liébana's forest commons, several interviewees signalled its potential for producing energy. As one respondent said, "*We can generate electricity with a biomass plant connected to power lines, or for public lighting... Also, in these close-packed towns, we can build a biomass central heating system for the whole village*" (male, 31-40 years old). Informants, however, were also cautious about the limits of this practice: "*If biomass starts* *being commercialized and everyone goes into the forest to collect firewood, it may cause ecological harm as a balance of forest biomass in the woodland seems necessary*" (female, 41-50 years old).

Barking, or *sudar el corcho*, was common in the 1950s, when cork was used in the wine-producing industry. The traditional technique consisted of removing the bark around the stem of *Quercus suber* L., using an axe to make three cuts through the outer bark. First, a vertical cut was done from the stump up to the first branch, followed by two horizontal cuts around the tree in both sides, aiming to produce rectangular slabs of cork as big as possible. Then, slabs were pulled off from the tree with special care for not damaging the slabs. Once removed, slabs were packed in piles and transported by draft animals. Barking was a seasonal activity limited to the summer months, "*when the cork tree most* suda" (male, 31-40 years old), i.e., during sap flow, which is considered to facilitate cork extraction. Because of its seasonality, barking used to involve up to fifteen neighbours of skilled labour on cork stripping. A tree could be barked every 14 years. Since the introduction of plastic stoppers, the demand decreased and the practice was abandoned in Liébana in the 1970s. Respondents argue that the abandonment of traditional barking has resulted in an excessive proximity between cork oak trunks with both negative economic and ecological impacts. On the one side, woody encroachment reduces the production of cork, as illustrates the comment of a village council representative, "*Scrubland up to the tree branches supresses the growing of the bark. Cork needs to be well-aired*" (male, 31-40 years old). On the other side, tree spacing is alluded as fire prevention mechanism of these stands, which one of them suffered from a severe fire ten years ago.

Local inhabitants also reported the occasional harvest of fruits, medicinal plants, and mushrooms for domestic use. They argued that fruit collection was more abundant in the past when acorns from *Quercus* spp. and *Fagus sylvatica* were mainly consumed by livestock, while nuts from *Castanea sativa* Mill. and *Juglans regia* L. were reserved for human consumption. The significant contribution of wild fruits to household economy is seen as the driver for the development of harvesting techniques as *varear*, a traditional fruit picking activity. Varear consisted of knocking down the fruits of a tree by beating the branches with a stick, named *caña*, large enough to reach the tree crown (Figure 4.7). At least two people were needed to complete the work: one person would climb on the tree and knock the crown and the other would assist during the climbing and collect the fallen fruits. According to the informants, this activity positively influenced fruit quality and production: "*Knocking down tree acorns favours sap flow*" (male, 81-90 years old). Furthermore, informants associate the abandonment of the activity of varear with the disappearance of some forest habitats: "*Since chestnuts and walnuts are no longer knocked down, trees have decayed*" (male, 90-99 years old). This practice was done in waxing moon and when trees were not wet. Acorn-bearing trees (*Quercus* spp.) were knocked in September and October, and their acorns loaded in bags for transportation to the town. Chestnuts (*Castanea sativa*) were harvested in November, and because their edible fruits are contained in a spiny husk, wooden tweezers to pick the chestnuts were used. To prevent injuries from spines, baskets were used for chestnuts transportation to the town.

Regarding medicinal plants collection, chamomile (*Chamaemelum nobile* L.) and té del puerto (*Sideritis hyssopifolia* L.) were reported to be picked nowadays by locals in mountain pastures during summertime. Frequently collected for self-consumption as an infusion, *té del puerto* has also commercial value in the local market, although the demand for the plant is low. In contrast, informants showed concerns about the need of regulating the gathering of mushrooms, occasionally done by locals, as there is a growing demand of mushrooms from visitors to Liébana who freely collect them.

Beekeeping for honey production has also a large tradition in Liébana, where wild beehives, or *setas*, used to be abundant in the surrounding forest commons. Informants reported that wild beehives located in tree branches were directly cut. When nesting was inside hollow trunks of old trees, beehives were difficult to find. To find the hives, harvesters waited in a fresh water stream the arrival of a bee and then followed bee's way back to the colony. Once in the tree, the harvester climbed with an axe to open the hollow trunk, remove the hive and carry it home in a container. In the following three days, the hive stayed also in the warmer place of the house, so that the heat melted the honey and easily separated the beeswax. Nevertheless, since the infestation of the parasitic mite *Varroa* in the 1980s, unmanaged wild honeybee colonies disappeared from forests.

Informants reported that in the past most families in Liébana used to have two to four beehives made of bark in the orchard. Every year, bark hives were crushed, mixing honey, beeswax and cork altogether, and filtered to obtain the honey by fine cloth. This activity was done in the warmest place of the house, the kitchen, to facilitate filtration. Nowadays, beekeeping of the honeybee *Apis mellifera* L. is done in beehives located in private or common lands outside the villages, placed at the low level of the valley in wintertime, and moved to the upland moors during the summer months. Over the last forty years, traditional bark hives have been substituted by wood hives, mainly built on *Pinus* spp (Figure 4.7). Informants refer to the practice of *freir* as a way to disinfect and enhance resistance of wooden beehives. This practice consists of dipping the boxes and foundations of the beehives in a beeswax reservoir melt for three minutes, "*and the beehive becomes eternal*" (female, 61-70 years old).

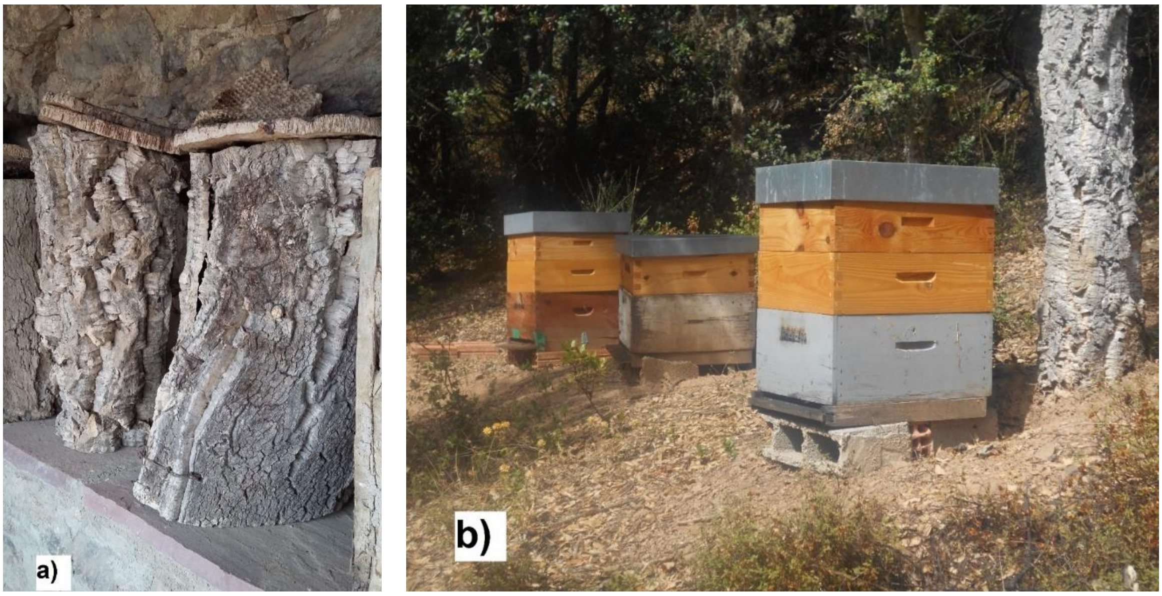

Figure 4.7. Traditional bark hives at *Casa de las Doñas* ethnographic museum in Enterrías, Liébana (a), and modern hives made of wood (b) in Valmeo, Liébana. Photo credits: S. Guadilla-Sáez.

Commenting on the decline of beekeepers in the region, informants were unanimous in the view that the maintenance of managed beehives is critical to avoid bee population decline, "*Without wild beehives in the woodland, the persistence of apiculture might safe the bees from extinction*" (male, 61-70 years old). Informants also pointed out the ecological threat of pollinator loss for wild plant reproduction if bee population declines: "*It is said that without the bees, wild plants will not persist more than four years*" (female, 61-70 years old).

4.4.1.4 Hunting

Hunting was an infrequent activity pursued by local inhabitants who went out alone or in groups to hunt for food and leisure. Hunting targeted small game species such as red squirrel (*Sciurus vulgaris*), hare (*Lepus* spp.), and partridge (*Perdix perdix*), and big game species including wild pig (*Sus scrofa*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), and chamois (*Rupicabra rupicabra*). Predators such as red fox (*Vulpes vulpes*), grey wolf (*Canis lupus*), and brown bear (*Ursus arctos*) were also hunted. Informants reported that when predator species such as foxes or wolves were hunted the hunter would show the dead animal door-to-door as a trophy to collect money.

Some informants reported the local disappearance of small game species like hares and partridges, which they associate to the abandonment of the cereal-legume cropping systems, "*Wheat, chickpea, and lentil, which are foods of the hare, are no longer cultivated*" (male, 80-89 years old). Habitat loss is also reported as a driver of small game species local extinction, "*Many wildlife species have disappeared as result of pastureland conversion to scrubland; there used to be a lot of hares and partridges*" (male, 31-40 years old). In contrast, landscape transformation to dense forests seem to have benefited other game species, particularly wild pigs, as they provide these species with food and cover. As one interviewee argued when asked about wild pig population significant increase, "*Nobody collects wild fruits from the forest, so [wild pigs] have cherries, hazelnuts, grapes, wild apples, chestnuts, beech nuts, oak acorns...*" (male, 31-40 years old).

Changes in game species are manifest in the current predominant hunting method in Liébana, the stand-hunting method or *batida* for wild pigs. Informants had different opinions regarding the ecological impacts of moving from a combined traditional hunting system to a predominant, almost unique, stand-hunting practice. Some participants felt that the use of the stand-hunting method helps to reduce wild pig population size, and even argue that the practice should be further promoted to regulate the species' stock, "*The criteria applied [by authorities] to stand-hunting method is not enough to respond to the ecological equilibrium needed between wild pigs and their natural habitat*" (male, 81-90 years old). However, other participants considered that the stand-hunting practice has harmful conservation outcomes and highly interferes with the dynamics of the natural ecosystems. As one informant describes,

> "*For wild pigs, stand-hunting has higher environmental impact than other hunting methods consisting on spot and stalk hunting. In this method, the hunter is accompanied by a forest ranger who has previously selected the trophy animals, so the method has little impact on the rest of the fauna. Contrarily, stand-hunting involves fifty people shouting and running in the forest with up to thirty trained dogs chasing all types of animals. Moreover, this is done every weekend during six months of the year, and the vulnerable fauna living in the forest suffers from it… If we constantly change wildlife equilibrium, we influence plant community, as forest regeneration depends on the number of ungulates*" (female, 31-40 years old).

Rental fees for big game hunting currently represents an important income for neighbouring councils, being regulated by the regional forestry administration through the concession of licenses. Informants reported that the economic revenue from recreational hunting is a great incentive for the local inhabitants to preserve wildlife, also commenting on further opportunities to obtain economic profit from game hunting, such as commercialising processed food products of game species to local restaurants.

4.4.2 Perceptions of landscape change

In this section we explore factors that, according to local informants over 60 years of age (N=24), have contributed to land use change in Liébana since the mid-twentieth century (Table 4.3).

Table 4.3. Drivers of regional landscape change reported by interviewees over 60 years of age (N=24).		
Drivers	n	%
Demographic drivers (population growth, migration, ageing, family status)	14	58
Economic drivers (market growth and commercialization, prices for agricultural/forestry products)	11	46
Technical drivers (agricultural mechanization, technological modernization)	3	12.5
Institutional drivers (agricultural policies, forestry policies, nature conservation policies)	23	96
Ecological drivers (successional dynamics)	18	75

The selected informants agreed that important land use changes have taken place in Liébana's landscape within the last decades. The most frequently cited driver of change corresponds to the institutional category, reported by 96% of the respondents (N=23). Demographic and economic drivers of change were mentioned in 58% (N=14) and 46% (N=11) of the interviews, with some of the informants associating demography with technological drivers. Three quarters of respondents (N=18) cited also ecological drivers (i.e., ecological successional processes) as a key factor of historical land use change in Liébana.

The most frequently stated landscape change due to institutional drivers referred to the prohibition of timber harvesting practices by the Cantabrian regional government in the late 1980s, cited by 54% of the informants (N=13). When asked about the consequences of this regulation, most interviewees considered that this regulation resulted in the encroachment of woody vegetation and landscape homogenization, increasing wildfire risks due to the accumulation of vegetation and the increase interface of forests and human settlements. Similar concerns regarding vulnerability to wildfire risks were also reported by some of the interviewees when discussing fire suppression policies. According to informants, repeated prescribed burns helped to reduce shrub biomass without affecting the canopy layer, "*Burning is forbidden nowadays, which is dangerous because in case of fire, it may easily become an uncontrolled wildfire*" (male, 81-90 years old). Four interviewees reported shifts on goat husbandry legislation, as this traditional activity was first forbidden and now is promoted by subsidies. National Park legislation was stated by 33% of informants (N=8) as the sole institutional driver for prohibitions to harvest timber, burn, hunt, fish, or collect NTFPs. Another informant noticed, "*National Parks consist in excluding all economic activities benefiting local populations in favor to some theoretical interests, such as nature conservation*" (male, 81-90 years old).

The second most cited driver of change, demography, was mentioned by 14 of informants (58%), who argued that the rapid depopulation of the area and farmer's ageing had resulted in the abandonment of agricultural activities such as livestock grazing and firewood collection. Most of the interviewees (N=11) also mentioned lifestyle changes, largely among the younger generation who have abandoned subsistence farming and migrated. Few informants (N=3) associated depopulation and technological changes, arguing that present-day farming activities require high monetary investments on mechanization to start running a business. Informants also reported lifestyle changes that arguably decreased pressure from Liébana's natural resources, such as the replacement of fuel wood by other types of fuel for cooking and heating, or the acquisition of fodder supply from market. According to respondents, uses of the forest were abundant in the past, "*because [local people] lived from the resources provided by the forest*" (female, 61-70 years old), and self-sufficiency of household economies implied daily livestock activities. Some participants (N=4) indicated that demographic changes had negative consequences for rural landscape conservation, as the abandoned meadows and cultivate fields have been gradually colonised by forest and shrub vegetation, decreasing habitat diversity.

Economic changes were identified as important drivers of landscape change by 46% (N=11) of respondents. As one informant described, "*Legumes, wheat, or potatoes crops were abundant in the past (...) When milk started to be commercialized, cattle number increased and arable lands progressively transformed to meadows for fodder*" (male, 80-89 years old). Some interviewees also mentioned the national and the international markets as drivers of change, as they made traditional activities like timber harvesting, livestock breeding, NTFPs collection or field cropping no longer profitable in Liébana. The abandonment of these activities is considered to favour the expansion of scrublands that cannot evolve to tree strata due to the lack of regular shrub clearings.

4.5 Discussion

Our results suggest a general concern among local inhabitants about the loss of managed landscapes in the absence of certain traditional farming and forestry uses in Liébana. Overall, informants stated that local forest commons provided them with relevant goods and services and that local management practices favoured the occurrence of high diverse forest ecosystems in an open patchy structure. Our results also suggest that institutional, demographic, and economic drivers are locally perceived to have greatly influenced the spatial and temporal transformation of Liébana since the mid-twentieth century, leading to the ecological succession to shrub and tree encroachment of the landscape associated to the abandonment of traditional farming management activities.

4.5.1 Potential of traditional management practices to biodiversity conservation

Based on our previous findings on the effects of human intervention in species distribution and long-term persistence of natural systems in Liébana (Guadilla-Sáez et al., 2019), this research sought to assess local inhabitants perceptions of which traditional management practices historically performed in Liébana might have enriched its landscape. Results presented here give valuable insights about the potential effect of the abandonment of certain traditional farming and forestry practices used in Liébana for the provision of fodder, domestic fuel and human consumption on species richness and landscape heterogeneity. Specifically, our study identifies seven traditional management practices with perceived positive effects for ecosystem diversity in Liébana.

First, the traditional mowing of meadows for hay making during the summer months and the autumn grazing might have helped to maintain biodiversity rich grassland patches thorough Liébana's mountain landscape. The gradual disappearance of grassland extension due to the abandonment of traditional hay making affects many European mountain areas and has been linked to the loss of plant species richness in these habitats (Doležal et al., 2011; Orlandi et al., 2016). Moreover, the decline in plant diversity affects the insect community richness and bird breeding, for which there are many conservation efforts being applied across Europe to restore semi-natural grassland habitats (Stoner and Joern, 2004; Jefferson, 2005; Graf et al., 2014; Valasiuk et al., 2018). For example, since 2016, a European Regional Development Fund is promoting the restoration and conservation of harvested meadows inside Picos de Europa National Park (UE-16-SOE1/P5/E0376). Still, most of these programmes consist on mechanical mowing to prevent grassland encroachment, not being followed by autumn livestock grazing, although scientific literature evidences the importance of accompanying hay cut with extensive grazing to avoid nutrient losses in meadows' soils and subsequent plant species reduction (Jefferson, 2005; Doležal et al., 2011). Indeed, local informants reported that grazing after hay cut was key for the long-term maintenance of grasslands. In contrast, hay meadows located around human settlements are over-exploited due to agricultural intensification, a finding that goes in line with recent studies documenting the evolution of farmland management systems across mountain areas of Europe (Lasanta et al., 2017; Burton and Riley, 2018). High livestock stocking rates negatively affect grassland conservation and its associated biodiversity, for which it has been argued that to reduce the on-going severe loss of semi-natural grassland areas, conservation efforts should combine mowing with extensive grazing practices near farms and extensive grazing stocking rates in under-grazed marginal lands (Graf et al., 2014; Orlandi et al., 2016).

The second traditional management practice with a perceived impact in ecosystem diversity in Liébana is pastoralism. According to our respondents, traditional pastoral systems helped to maintain the grassland-woodland habitat mosaic on Liébana's mountain landscape, as regular livestock herding prevented shrub encroachment of pastures. Moreover, because of this positive role, informants argued that agri-environmental subsidies should support properly managed livestock farming to make profitable pasturing practices, as these go in line with nature conservation. This recommendation has also been put forward by scholars, such as Z. Molnár et al. (2016) who suggested the inclusion of nature conservation approaches to the traditional herding profession to shape an effective management to restore and sustain the former dynamic open patch structure of rural landscapes. Our informants further consider that economic incentives should be oriented to small ruminant livestock herding practices, whose number had recently declined. Informants' argued that goat feeding behaviour prevents forest and shrub vegetation colonization over hay meadows and reduces the biomass load of woodland ecosystems. Interestingly; the potential of goat pastoral systems as a management tool to be included in wildfire prevention operations has also been reported in the literature (Landau, 2017; Marques et al., 2017).

Lopping is the third management practice with reportedly positive effects for natural ecosystems conservation in Liébana. The tree foliage harvest was an important fodder resource for livestock during the winter, but it also implied biomass removal. Although extensive biomass harvest might negatively impact biodiversity, the small-scale lopping of branches increases forest cover regeneration and enhances habitat diversity (Vetaas, 1997; Varguese et al., 2015). In our study, local people reported that the abandonment of lopping has resulted in a decline of tree species traditionally pruned for the provision of winter fodder. Informants partly attribute the abandonment of the practice to fodder availability in the market, for which isolated trees of *Populus* spp. and *Fraxinus* excelsior standing in private fields are not longer preserved for fodder production. Another explanation given by informants is the banning of the activity in relation to *Ilex aquifolium* L. by the regional forestry administration. The cessation of this traditional harvest practice is considered to be in detrimental of *Ilex aquifolium*, a finding consistent with studies linking anthropic activities in the past to the expansion of holly woods in central Spain (García, 2001; López et al., 2013).

Fire suppression policies on prescribed burns, the forth management practice considered to have positively influenced ecosystems conservation in Liébana, also seem to have delivered undesirable long-term conservation outcomes, contributing to current large wildfires (Seijo et al., 2015). Local respondents are aware of the use of burning as management tool to prevent scrubland encroachment of pastureland and to remove understory biomass in forested areas. This approach matches with recent studies recognizing the potential of low-intensity prescribed burning as a cheap wildfire prevention technique (Seijo et al., 2015; Fernandes, 2018). Moreover, from an ecological point of view, the literature evidences that seasonal prescribed fires can help control the expansion of exotic over native plant species in areas with fire-adapted natural ecosystems (Kral et al., 2018; Barefoot et al., 2019). Thus, in a region with historical fire regime like Liébana, the reintroduction of periodical low-intense burns may benefit certain stress-tolerant forest ecosystems such as forests dominated by *Quercus* species and chestnut woodlands (Hanberry et al., 2014, Seijo et al., 2015).

The fifth traditional practice locally perceived as having a positive impact on habitats conservation is the gathering of firewood, which also implied the removal of woody biomass from forests. Neighbours mostly selected standing dead trees, trees with evidence of disease, poor quality logs, dead branches, and twigs for firewood, for which firewood gathering acted as a clearing silvicultural treatment beneficial for the maintenance of healthy stands. Indeed, this local perception matches results from a recent study analysing the effect of selective logging for firewood in stands dominated by *Quercus* species in southern Mexican cloud forests (Ortiz-Colín et al., 2017). Moreover, the same study recommends a moderate extraction of biomass in *Quercus* spp. stands for their optimal regeneration. As previously discussed, the removal of fuel biomass due to firewood collection also prevents the establishment of dense forest ecosystems, reducing wildfire risks. Several of the informants expressed the potential for producing biomass energy from the harvest of firewood. This consideration is particularly relevant in the current context of climate change, as biomass burning may be a promising renewal source to substitute fossil fuels for electricity production or for combustion in residential heating (Sreevani, 2018). Still, informants advocating for the use of forest biomass for energy in Liébana recommended to provide general ecological information to local communities on a rational use of the resource, to prevent intensive firewood extraction. This recommendation is in line with recent studies warning of the ecological risks that large-scale deadwood stock extraction hold for the preservation of associated biodiversity (Stupak and Raulund-Rasmussen, 2016; Hof et al., 2018).

The sixth management activity identified as positive for forest habitats conservation is branch beating (varear). According to informants, branch beating positively influences fruit quality and tree production. Literature corroborates the idea that manual harvest methods, such as beating the branches and collecting the fallen fruits, have lower incidence of damage and cracks on the fruit than the mechanical harvest methods (Monarca et al., 2014), although the specific influence of harvesting in fruit production has not been documented. Informants also suggested that the cessation of this traditional harvesting method has had negative implications for the preservation of *Castanea sativa* and *Juglans regia* species. This outcome may be contrary to studies documenting the beating of trees with sticks as an epidemiological factor for disease incidence in some woody species (Panagopoulos, 1993). Still, as the abandonment of traditional agroforestry practices in *Castanea sativa* and *Juglans regia* stands seems to result in the long-term decline of these stands (Cantarello et al., 2014; Guadilla-Sáez et al., 2017) for which further studies describing the particular influence that branch beating may have for the persistence of these stands are recommended.

The last traditional management practice identified as environmentally friendly in the study area corresponds to beekeeping, an economic activity that provides pollination services to wild plants. Beyond the importance of bees for pollination of plant species, many studies across the globe have documented traditional beekeeping practices as an important part of the subsistence economy of rural communities (Oteng-Yeboah et al., 2012; Sight, 2014; Adal et al., 2015; Galbraith et al., 2017). Recent awareness about worldwide bee population loss is resulting in conservation initiatives promoting beekeeping, a management practice that does not disturb natural ecosystems nor compete for resources with other economic activities or conservation efforts in the landscape (Adal et al., 2015; Galbraith et al., 2017). Interviewees expressed concerns about the disappearance of unmanaged wild honeybee colonies from forests due to the parasitic mite Varroa, highlighting the need of the persistence of beekeeping in Liébana to avoid further bee populations decline.

Overall, our results illustrate a consensus between the arguments used by local informants and the scholarly literature to identify certain management practices that could contribute toward nature conservation. However, our results also reveal a set of practices assumed by local informants to enhance ecosystem diversity that have not been documented in the literature.

4.5.2 Perception of historical land use change and its implication to traditional landscape conservation

The second objective of this study was to explore the potential of local discourses to document the driving factors associated to the historical land use change in Liébana and to further describe driver's effects on landscape transformation. Overall, local inhabitants perceive the landscape change in Liébana since 1950-1960s as a progressive ecological succession to shrub and tree encroachment in areas that use to be pasture. These changes are largely associated to the abandonment of traditional agricultural land use activities due to institutional, demographic, and economic drivers.

Scholars and local population describe rural exodus and market integration as significant drivers for the abandonment of traditional uses, resulting in the simplification of the landscape mosaic, initially through the transformation of cultivated fields into pastureland, and later through the gradual encroachment of scrubland strata in abandoned pastures (López, 1978; González, 2001; Rescia et al., 2008). Institutional factors are also reported as key drivers of change in Liébana, both in the literature and in the interviews with local stakeholders (González, 2001; Castañón and Frochoso, 2007; Rescia et al., 2008). Nevertheless, local people and the literature differ in who they consider the main agent influencing local land use regulations. Whilst the literature emphasizes the effects of the European Union Common Agricultural Policy in shaping Liébana's current economic, environmental and social situation (Corbera, 2006; Rescia et al., 2008), informants associate land use regulations to the Cantabrian regional administration and to the National Park management. This finding matches with previous research on the area reporting local communities' unfavourable attitudes towards conservation initiatives (López, 1978; González, 2001; Rescia et al., 2008). Moreover, despite the lack of a valid management plan to regulate traditional uses implemented inside the reserve (repealed in 2005 due to a legal action taken by local communities living within Picos de Europa buffer zone), local communities perceive that the National Park's authorities exert regulations to land uses. Inhabitants' perception of limitations despite Picos de Europe is lacking of a successful management capacity in practice evidences that conservation initiatives are negatively perceived by communities even if the conservation outcomes have neutral impacts on local livelihoods (Bennet and Dearden, 2014; Babai et al., 2015; Lopes-Fernandes et al., 2018).

Ecological changes derived from the historical land cover change in Liébana described by local observations match those found in the literature. Both sources describe a successional trend to dense high forest coppices from abandoned lands. Concerns regarding an increase of fire-hazard associated to changes in forest fuels prevail in local discourses, a finding previously reported for *Quercus* spp. wood-pastures of south Spain (Garrido et al., 2017). This concern is also reflected in the recent literature associating fire-hazard in mountain rural areas of Spain with an increase in plant biomass and homogenized landscape resulting from the abandonment of traditional management practices (Viedma et al., 2015; Lasanta et al., 2017).

In sum, our results suggest that local understanding of the driving factors influencing land use change in Liébana dovetails with findings from the existing literature examining the spatial and temporal transformation of the study area.

4.6 Conclusions

The objective of this research was to understand how the abandonment of traditional land use activities in rural areas leads to a simplified habitat mosaic landscape, and to describe the ecological consequences of the driven forces associated to this process of abandonment.

In general, the local understanding is that the cessation of traditional practices adapted to the carrying capacity of Liébana's natural resources has had negative implications for preserving species and habitat diversity. Our results illustrate a set of traditional management practices reportedly favourable to biodiversity and economically profitable, which could be explored for developing effective regional conservation strategies. Moreover, the overlap of local perceptions and scientific reports go in line with conservation approaches encouraging the combination of local ecological knowledge and ecological science to assess effective biodiversity outcomes (Berkes and Turner, 2006; Hernández-Morcillo et al., 2014; Joa et al., 2018; Morales-Reyes et al., 2018).

Our results also suggest that local discourses identify the same drivers of historical land use change than those documented in the literature, providing a detailed understanding of the particular ecological significance of each driver on rural landscape transformation. This finding supports the idea that rural communities provide valuable information to document landscape historical dynamics and to monitor environmental changes (Calvo-Iglesias et al., 2006; Parrotta and Trosper, 2012). These findings could be particularly relevant for landscape-orientated conservation policies aiming to prevent the biodiversity loss resulting from the abandonment of traditional land uses. We recommend further research assessing the potential of local knowledge to monitor environmental landscape change.

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Chapter V

General discussion

This final chapter briefly summarizes the key findings of the study (Section 5.1), outlines its theoretical and methodological contributions (Section 5.2), discusses its policy implications (Section 5.3), and suggests areas for future research (5.4).

5.1 Summary of key findings

In the first chapter of this thesis, Chapter II, shows that the replacement of traditional community-based management systems in Spanish forest ecosystems had negative consequences for long-term biodiversity conservation. On the one side, the privatization of traditional community forests often resulted in the cut of the tree canopy by the new owner, who needed to compensate the cash value of the purchase. On the other side, public intervention on traditional community forests resulted in the progressive abandonment of local practices and the consequent mismanagement of ecological resources, favoring the homogenization of rural landscape, a process associated to a decrease of biodiversity and an increase of fire hazard risk. As an example of the positive effect of traditional management practices on biodiversity, this chapter also presents how several natural areas in which traditional community management was preserved or restored some decades ago currently overlap with biodiversity-rich areas.

Chapter III provides empirical data on temperate biomes of the Liébana valley, Northern Spain. Field data showed that there are no differences in biodiversity measures between traditional community forests under a restrictive protection category and the surrounding non-protected community forests. This finding might be explained by the lack of a valid management plan in the protected area, which in practice results in similar forest management actions being applied inside and outside the protected area. Additionally, this chapter also shows that, in the Liébana valley, variables related to human intervention are more important drivers of species distribution in forest ecosystems than the natural factors considered for the analysis.

The last chapter of the thesis, Chapter IV, shows how local perceptions can be a valid source of information to document landscape historical dynamics. According to perceptions of people in Liébana, institutional, demographic, and economic drivers have greatly influenced the spatial and temporal transformation of Liébana since the mid-twentieth century, leading to the ecological succession to shrub and tree encroachment of the rural landscape mosaic. It also identifies a set of traditional management practices that sustained local livelihoods while contributing towards biodiversity conservation.

5.2 Theoretical and methodological contributions

Results from this dissertation bring significant insights both at theoretical and methodological levels. At the theoretical level, this work contributes to the ongoing debate of the ability of protected areas to halt biodiversity loss by themselves, in which few studies have compared biodiversity levels between protected and unprotected areas matched by same land use type (e.g., between protected and unprotected forests) (Geldmann, 2013; Gray et al., 2016). This study shows a similar tree species composition in temperate deciduous forest commons inside and outside the reserve, suggesting that Picos de Europa National Park does not hold more biodiversity than surrounding areas. This finding contributes to the idea that certain areas managed by local communities can be effective in preserving species and ecosystems and should be considered as ‘other effective area-based conservation measures’ (OECMs) referred by the Convention on Biological Diversity’s Aichi target 11 (Jonas et al., 2017).

Second, this dissertation contributes to the existing literature suggesting the need to include anthropogenic factors when studying species distributions, particularly in human-dominated landscapes (Guèze et al., 2015; Mod et al., 2016). Humans can modify species composition in a particular habitat through management actions that include dispersal events, introduction of non-native species, favoring untypical native species, or the removal of species (Helm et al., 2015). However, few studies consider variables related to human activity in their modeling of species distributions (Mod et al., 2016). By exploring the influence of human disturbances to plant species richness, this study acknowledges the importance of anthropogenic variables, such as plot isolation, in explaining species distribution. This finding gives further support to research efforts towards delimiting suitable areas for conservation in semi-natural landscapes with historical human use (Guèze, 2011).

This work is also relevant to the literature addressing conflicts with local communities related to the establishment of protected areas in their territories. There is a general understanding that regulations limiting traditional uses without considering local livelihood needs can potentially result into a general opposition of local residents to conservation initiatives (West et al., 2006; Riseth, 2007; Hirschnitz-Garbers and Stoll-Kleemann, 2011). This was the case in the Liébana valley, where local opposition to Picos de Europa National Park resulted in a legal action in 2005 repealing the management plan of the park. Remarkably, although traditional uses inside the reserve are not being regulated due to the lack of a valid management plan, local communities perceive that the National Park's authorities exert regulations to traditional land uses. This finding, then, adds to research showing that conservation initiatives are negatively perceived by communities, signaling that this might be the case even if the conservation outcomes have neutral impacts on local livelihoods (Bennet and Dearden, 2014; Babai et al., 2015; Lopes-Fernandes et al., 2018).

The theoretical framework used in this thesis follows the assumption that certain practices applied in community-based resources management systems can be effective for the long-term conservation of natural resources (Agrawal and Gibson, 1999; Ostrom, 1999). However, some authors have raised questions of whether these systems are consistent with scientific biodiversity conservation goals (Berkes and Turner, 2006; Brooks et al., 2013). Although not all traditional practices are favorable to biodiversity conservation, this study identifies a set of management practices traditionally carried out in forest commons of the Liébana valley that seem to contribute to maintain species diversity and to foster sustainable use of biological resources. These findings give further support to the body of literature suggesting a better appreciation of the management practices carried out by Indigenous peoples and local communities to maintain the variety of life on Earth.

On a methodological level, this study adopts an interdisciplinary approach to combine social and natural sciences for evaluating the effectiveness of conservation initiatives. Despite the importance of applying inclusive frameworks to analyze the human dimensions of environmental problems (Bennet et al., 2017; Lele et al., 2018), few studies integrate social-ecological approaches on their methodology. Here, this research combines the most well-known ecological method for quantifying regional biodiversity (i.e., species richness and evenness indicator) and other ecological measures, such as forest structural indicators and model selection methods, with the most frequently data collection method used in ethnobiological studies (i.e., interviews) and other ethnobiological and historical data collection methods, such as participant observation and oral history techniques. This interdisciplinary methodology allowed me to combine non-academic knowledge with scientific literature to effectively construct an integrated understanding of the role of certain traditional management practices in preserving a species-rich habitat mosaic of the rural landscape.

5.3 Policy implications

This thesis identifies a set of traditional farming and forestry management practices beneficial for long-term ecosystems maintenance that might complement ecological science in the design and implementation of conservation strategies. Moreover, findings of this work dovetail with studies indicating that TEK practices may be a cost-effective, long-term sustainable tool for managing semi-natural habitats (Babai et al., 2015). Any regional conservation initiative in Liébana, and particularly Picos de Europa National Park, may benefit from the inclusion of the traditional practices identified in this work into its management planning.

This research also provides information on traditional management practices that were banned in the past by government technicians, and that are now being reconsidered for conservation objectives. These practices include prescribed burns or the use of small ruminant livestock herding to prevent shrub encroachment. This finding supports arguments considering that, although not all traditional practices might have positive ecological outcomes, some of them can be a useful source of information to site-specific conservation management strategies (Berkes et al., 2000; Joa et al., 2018). Moreover, in the present case, some traditional techniques, like techniques related to pastoralism, seem to be favorable to biodiversity, a finding that may be taken into account to develop effective integrative conservation initiatives (Berkes, 2007; Liu et al., 2007; Brooks et al., 2013).

Another important policy implication of this research is that it documents in a comprehensive way the community-based knowledge of resources management in an area where recent social and economic changes have resulted in the rapid loss of traditional knowledge. TEK-holders face significant threats in different parts of the world, particularly in developed societies where changes in traditional lifestyles, abandonment of rural areas, and policies neglecting the experience of local communities on ecosystem management, among other factors, have resulted in the declining interest on TEK (Parrotta and Agnoletti, 2007; Hernández-Morcillo et al., 2014). For this reason, a variety of policy and scientific forums increasingly encourage research on TEK-based practices relevant for the conservation and sustainable use of biodiversity, such as the Law No. 42/2007 of Spanish Government incentivizing the preservation of traditional knowledge related to biodiversity (Pardo-de-Santayana et al., 2014). The present study helps to document TEK in the Liébana valley, a region with an ageing population in which traditional farming and forestry practices are likely to disappear in the absence of incentives to encourage young generations to master farming and forestry skills under traditional systems.

5.4 Future research

The findings of this thesis provide insights that might guide future research aiming to examine how the inclusion of TEK in environmental policies and management plans might facilitate communication among local users and managers, particularly when referring to what should be preserved (Berkes, 2004). Results from this work suggest that the need to improve communication between stakeholders particularly applies to management rules, as regulation misunderstanding might be a major constraint for effective conservation and a potential source of conflict between local users and managers (Anderies and Janssen, 2016). For instance, there is a belief among many local informants that "*National Parks consist in excluding all economic activities benefiting local populations*" (Chapter IV:118), which contrast to some of the initiatives taken by the National Park administration, such as providing economic incentives to rebuilt shieling huts for livestock on the pasturelands inside the reserve. Therefore, further research is recommended to explore the role of TEK as a potential mechanism that might facilitate the dialogue between different stakeholders regarding biodiversity conservation. A research topic particularly worth exploring is how the inclusion of TEK-based practices into Picos de Europa National Park management system could be useful for the resolution of conflicts with local communities, especially livestock herders (Bennet and Dearden, 2014).

Further work also needs to be done towards designing interdisciplinary methodologies that combine biological and social data to study the role of TEK-based practices in resource and ecosystem management. Ideally, further research might address the methodological deficiencies of using biological indicators in which the relative abundance of species in a community is the only factor that determines its importance in a diversity measure, making no distinction between species that are exceptionally abundant and those that are extremely rare (Magurran, 2004). Although biological indicators are a firmly established measure to quantifying biological diversity, they seem limited in their ability to serve as indicators to evaluate positive conservation efforts. This is exemplified in the increase of forest species diversity observed in some of the studied plots, which was actually driven by the presence of non-native species, such as exotic (e.g., *Pinus radiata* D. Don) or untypical species (e.g., *Arbutus unedo* L. and *Pyrus* sp.). This research address this limitation by including structural indicators described in the literature for estimating spatial forest structure (Pommerening, 2002; Hui et al., 2011) and model selection. However, other approaches may include, for instance, the monitoring of successional species resilient to climate change (Rubio-Cuadrado et al., 2018), calculating biodiversity indicators that allow to distinguish between characteristics habitat-specific species pools and atypical species whose presence is driven by adverse human interventions (Helm et al., 2015), or using local knowledge indicators to monitor species population status and trends (Tomasini and Theilade, 2019).

Finally, this research proposes a novel methodological approach to understand the impact of conservation strategies: comparing the biodiversity status of protected and unprotected areas matched by same land use type. Similar studies measuring the differences in biodiversity outcomes in protected and unprotected broadleaf forests are recommended to provide additional evidence of the findings of this research.

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Appendix I

Besides the three manuscripts of this dissertation, during my PhD I have co-authored the following publications:

Guadilla-Sáez, S., Pardo-de-Santayana, M., Reyes-García, V., 2017. The dismantling of forest commons in Spain. Available at: Proceedings of the XVIth IASC Conference Practicing the commons. Self-governance, cooperation and institutional change. Utrecht, the Netherlands, July 10-14. http://hdl.handle.net/10535/10373.

Guadilla-Sáez, S., Pardo-de-Santayana, M.; Reyes-García, V.; 2017. Las prácticas tradicionales en el manejo de los bosques de castaño y las dehesas de quercíneas. Una visión histórica. Actas del 7º Congreso Forestal Español. 26-30th June 2017, Plasencia, Spain. Available at: http://7cfe.congresoforestal.es/sites/default/files/actas/7CFE01-214.pdf

Appendix I.a. The dismantling of forest commons in Spain

The dismantling of forest commons in Spain

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Abstract

In pre-industrial Spain, the use of local natural resources was essential in guaranteeing peasants' subsistence (Piqueras and Sanz, 2007), especially in high mountain areas, where forest resources were essential for everyday life (Parrotta and Trosper, 2012). Overtime, groups developed formal –e.g., local ordinances– and informal –e.g., cultural practices– rules to manage common resources and prevent common forests from uncontrolled exploitation (Ezquerra and Gil, 2004). Such rules and constraints were typically adapted to local conditions, allowing dynamic adaptations to changes, and largely legitimised and followed by local inhabitants (Moreno, 1998; Linares, 2000; Serrano, 2014), to the point that several of these management systems have survived the course of time.

The transition of Europe to capitalism, by the end of 18th century, gave rise to the establishment of a new liberal framework that initiated a process of privatising communal resources (Beltrán, 2015). These changes were also accompanied by the replacement of traditional uses by regulations, forest acts and forest management plans written down by the State (Parviainen, 2006). As soon as local communities were not allowed to manage their woodlands, illegal forest uses proliferated, no longer being locally sanctioned as a way of popular resistance. It is remarkable the fact that most of that resistance did not arise

because of property right losses; it appeared when local residents became not authorised to continue exerting their historical exploitation of forest resources. The consequences of the commons' dismantling process, which is detailed in this contribution, had devastating effects for the preservation of forests, being considered as one of the greatest environmental disaster for the Spanish forestry heritage (Fernández, 1992).

1. Introduction

In Spain, there is a large tradition of obtaining natural resources from forest commons, which had been documented since the Germanic tribes invasion to the Iberian Peninsula, in the fifth century. These tribes introduced the concept of woodlands collective property in the northwest areas of the peninsula. Thorough the collective property regime, forest resources were used by local communities, a management regime that –according to some authors– resulted in a supportive and sustainable use of forest (Aranda, 1996). Some centuries later, during the Christian Reconquest and due to the land concession strategy followed by medieval kings to promote the settlement of Christian population that displaced Muslims from the newly gained territories, the communal regime was generalized to other parts of the peninsula. The land privileges granted to the Christian populations consisted in common lands –including woodlands– that the new settlers, organized in village councils or *concejos*, could communally manage and exploit (Behar, 1983; Pardo and Gil, 2005). At that time, forest commons became the most habitual type of tenure of Spanish woodlands, a land tenure type that persisted until the nineteenth century.

Thanks to the communal land tenure, during the Old Regime rural communities were able to obtain freely goods from their surrounding woodlands. Forest-related resources guaranteed the economic persistence of many mountainous societies, and particularly, the subsistence of the poorest households in those peasant societies (Linares, 2000; Jiménez, 2002). Forest commons were used for pastures and for the collection of firewood, charcoal, wood, fruits, roots, medicinal plants and ice, among others. Forest commons were also a source of food as result of hunting, fishing and gathering activities and through the presence of crops in common lands. The versatility of the raw materials offered to peasants' economies enhanced the development of multiple local occupations such as woodcutter, sawyer, carpenter or charcoal burner, as well as industrial activities of great importance like the naval shipbuilding, highly consumer of wood (Rey, 2004).

The large variety of products obtained from forests was the consequence of the integral exploitation that rural communities made of all economic and ecological opportunities brought to them by the surrounding environment. This diversity provided to peasant communities a notable adaptation capacity towards changing conditions. For instance, forest commons diversity allowed households to confront market fluctuations in which Old Regime peasants depended on labour demand or for agricultural and forest products trade. Communities' use of diversifying income sources was a reasonable strategy to avoid risks linked to the changing dynamics of the market (Moreno, 1998). Another example of the strengthen derived from the communal lands comes from the easy adaptation of local communities to resource limitation due to changes in socio-environmental conditions. Thus, forest commons requirements were easily modified according to the volume of available resources and peasants' needs at different periods. For instance, in Cantabria, a region in northern Spain, farming cultivation or grazing pastures were enlarged at the expense of forest commons territories as a response to bad harvesting years or to increasing crops or cattle demand (Vázquez, 2016).

2. Commons regulations during the Old Regime

Authors argued that the clue for the persistence of the communal lands until the end of the eighteenth century was that commoners –i.e., users of the commons– individually exploited the available resources under a depletion threshold (Moreno, 1998). In addition, there was a social consensus for a conservative management of the communal resources, mainly oriented to safeguard resources availability for the future. From the consensus emerged a set of traditional collective norms oriented to guarantee the preservation of the economic and social activities carried out in common lands. Mostly, communal forests were regulated by local ordinances focused in the conservation and promotion of the profited goods. For instance, in Tudes, Cantabria, local ordinances reacted against the enclosing process occurred in the region during the sixteenth century driven by the increase of agricultural crops to respond to population growth. In 1591, the Tudes Concejo banned the 'tradition' of enclosing and ploughing forest commons, in order to amend the reduction of livestock activity in common pastures in favour to private farming (Vázquez, 2016). Close to Tudes, in the village of Potes, 1619 local ordinances forced each neighbour to plant in common lands two individuals of chestnut, walnut or ash species per year. Same neighbour was able to collect the fruits produced by the trees in later stages (Pérez-Bustamante and Baró, 1988). It is worth mentioning that the historiographical evidence suggest that there was also a group of informal rules that, being acknowledged by all members in the community, were not written down, but were mainly orally transmitted (Moreno, 1998; Piqueras, 2002; Rey, 2004).

Nevertheless, communities not always fully agreed on the local regulations to manage the commons. On the contrary, disagreements were habitual due to the presence of different social groups whose exploitation interests may not exactly corresponded or were even opposed (Moreno, 1998). As an example, in Spanish northern coastal areas like in the Basque Country, the growing pressure of naval shipbuilding and steel industries that occurred from the thirteenth century onwards provoked the subordination of *concejos* interests, such as the domestic use of firewood or timber and pastures for cattle, to the industrial activities (Aragón, 2003). This was also the case in La Rioja, an inner region limiting to the Basque Country in which common lands represented more than four-fifths of the mountainous area during the eighteenth century. The importance that the wool market used to have in the economy of this region resulted in the allocation of the communal areas to pastures for wool animals, which belonged to local influential families, but grazing displaced activities of other community members –that were traditionally performed in common lands– to private lands (Moreno, 1998).

This latter case also exemplifies how, along with the diverse interests between productive sectors, the access and use of communal lands varied according to the social framework of a given historical time. In this case, common lands and uses were part of an Old Regime social structure, which presented high social and economic inequalities. Access rights differed between the powerful members of the community, who could benefit from a higher portion of the common heritage, and the peasants, who obtained from the commons a reduced complementary goods, but which still represented an essential resource for their subsistence. Hence, in La Rioja communal lands, the powerful local actors –who ultimately were the owners of the wool animals–, influenced the regulation of common lands with the aim to guarantee the free access of their cattle to the collective pastures whilst other less influential commoners had to resort to small fenced private areas to carry out their agricultural practices (Moreno, 1998; Sanz, 2002).

With all, the existence of intra-community confrontations frequently was compatible with the persistence of solidarity mechanisms among neighbours and the existence of certain social cohesion within the community. It has been argued that this was so, on the one hand, because although influential members had an advantageous access to common lands, they were aware of the necessity to preserve those communal resources as a way to cushion social discontent; and on the other hand, because although social cohesion did not imply economic equality, it enforced members' integration to the system to the point that they defended it (Moreno, 1998).

Consensus over sustainable use of resources emerged from the economic role commons played in rural communities. Moreover, such consensus had a place in daily decisions oriented to maintain the use of collective areas, usually by trying to make traditional uses compatible with long-term preservation of the resources. The main source for the study of collective customs are local ordinances, which, despite their specificity to each local community and time period necessities, match with the general scope of conservation and promotion of communal goods. As result, although ordinances structure was very different among Spanish regions, as a whole, they generally regulate rights to access and use the commons, and the monitoring and punishment of transgressions to the code (Moreno, 1998).

Regarding the woodlands, most usual norms consisted in the banning of hurtful forest-related practices such as felling or debarking trees, extraction of resin, slash-and-burn agriculture, or the entrance of certain livestock species to wooded areas. For instance, in Leon, a northeast region of Spain where in the eighteenth century communal lands represented more than three quarters of the mountainous areas, local ordinances described in detail how to maintain forest commons' tree coverage. Woodcutting was only allowed in the areas assigned for timber extraction and it was compulsory to control and punish the non-compliance of this norms. Moving to southern regions, in the mountainous areas of Salamanca and north of Extremadura, local ordinances fixed the gap periods to be maintained between cork extraction of oak trees and forbade grazing in areas affected by fires (Rey, 2004). It is remarkable the regulation existing in these regions for chestnut forests (*Castanea sativa* Mill.), a species employed during the Christian Reconquest as a mean of claiming property for communal lands (Ríos-Mesa et al., 2011). Local ordinances of a village in Salamanca, named *Miranda del Castañar*, established in which period the cattle could graze in chestnut groves and banned chestnut felling except for domestic purposes and only after having previously obtained *concejo* permission. In some areas devoted to livestock refugee during the winter seasons, timber or firewood extraction was totally excluded (Rey, 2004).

The monitoring and punishment of common lands forest abuses, as it has been previously mentioned, was normally carried out by the commoners themselves. The key role that forest-related goods played for the subsistence of peasants' economies, particularly in mountainous regions, resulted in commoners' interest in supervising the accomplishment of local norms in their forest commons. As an example, the importance that the pastoral activity had for rural communities in Cantabria favoured the control by the commoners of foreigner cattle grazing within their concejo jurisdiction, which translated in commoners retaining the animals until their owner –frequently from boundary villages– paid a fine for release them (Vázquez, 2016). Thanks to this monitoring, forest commons did not present a level of depletion as greater as other types of forests, like royal ones, opened to everyone, and thus, less controlled and systematically more exploited (Rey, 2004).

However, when community needs were higher than the available resources, such as in inner areas with climatic limitations or when new ordinances were oriented only for the benefit of the influential members in the community and poorly accepted by the peasants, restrictions had to be enforced. This was done by *caballeros de sierra*, a sort of local forest rangers committed to control the use of forest commons (Parrotta and Trosper, 2012). This was the case, for example in Madrid, a central inner region of Spain, where the reduced number of trees and common lands usurpations by some local caciques led to the promulgation in 1567 of very restrictive ordinances –issued to prevent an increasing deterioration of the tree coverage– and the yearly establishment of caballeros de sierra (Rey, 2004).

3. Peasants' contestation to external threats before the nineteenth century

Although intra-community confrontations regarding the use of commons arose as a consequence of the social differentiation process occurring inside peasantry, most frequently social forces came together against external threats to their community (Moreno, 1998). This was the case, for instance, when nobility members exerted their influence for the usurpation of commons lands. Thus, in 1768 in a large forest common of Valencia, coastal region located in southeast Spain, the marquis Morella Antonio Belluga y Moncada divided the local common land and distributed it among the landowners of the area. Community members complained about the land distribution to the Council of Castile indicating that the forest was of common use and accusing the marquis and other landowners of common lands usurpation. In 1779, the Council urged to re-establish the communal regime of the area, so that neighbours could freely profit from grazing, and gathering fruits from the forest common.

Before the eighteenth century, main conflicts consisted in jurisdictional disputes regarding communal territories. Part of these problems came from the original nature of the commons, most of them created during the Christian Reconquest period of the Iberian Peninsula. In the Reconquest process, the crown, ultimately owner of the common lands, granted charters (*cartas-puebla*) to concejos, transferring them the management and use rights of common surfaces (Wing, 2015). Whereas these concessions detailed the parameters for an acceptable use of the commons, territorial boundaries were not clearly defined. These blurred limits later resulted in jurisdictional conflicts among concejos themselves, but also between concejos versus nobility, and concejos versus the crowd, provoking serious disputes due to usurpations, land abuses, and inequalities in resources' access and use rights (Aragón, 2003).

The first type of conflicts, among concejos themselves, comprised usurpations between neighbouring communities with boundary territories or in depopulated areas, without recognized owner. The second type of conflicts, among concejos and nobility, consisted on a social struggle starting with neighbours' opposition to pay canons for using the common lands, and habitually followed by the succession of litigation-resistance-judgment-agreement. The agreements, named *concordias*, were meant to indemnify the non-payments of canons when judgments sentenced in favour of nobility (Piqueras, 2002). The third type of conflicts, between concejos and crowd, will be further analysed below due to its similarities with the process of commons' dismantling followed later by the liberal politics that emerged in Spain after the Old Regime (nineteenth century).

Firstly, it should be remarked that, despite common lands were initially favoured and promoted by Spanish Christian crowd, over the centuries, royal policies regarding the communal lands management evolved towards a greater interventionism (Ramos, 2007). Interventionist policies grew particularly, from the beginning of the sixteenth century, when the king Philip II attributed the decay of Spanish naval shipbuilding empire to a negligence in woodlands conservation. Aiming to avoid further negligences, the crowd issued norms applicable to forests located nearby the coastal areas. These regulations contained a set of environmental rules with important social implication as they established monitoring systems among community members. Nonetheless, regulations did not achieve their purpose, as rural societies were not committed to the new code and they offered resistance to crowd efforts to privatize forest commons. Thus, peasants continued carrying out practices like the felling and debarking of trees or the ploughing of forested lands for its transformation into crops (Rey, 2004).

Regulations' unsuccessful result led to the promulgation of newer and more restrictive ordinances in 1748. Indeed, some authors have considered these ordinances as a first attempt of appropriation of the forests by the State (Valbuena-Carabaña et al., 2010). From the very beginning, there was a great opposition of the peasantry to the ordinances. Initially, resistance took legal forms, with peasants' complaints of common lands usurpations and the unclear legal concepts of ownership and use employed by the State during their intervention of forest commons. In northwest regions, like Galicia, regional governments claimed that woodlands provided not only pastures and agricultural areas to rural societies, but also firewood and timber for the construction of houses and crop fences, furniture, and raw material for multiple occupations. There were also accusations of forest abuses committed by the State agents, and errors in species choice. Additionally, community members also used resistance techniques such as the ruin of afforestation areas, bad practices of pruning in trees, no clearance of scrubs, or land usurpations (Rey, 2004).

The negative consequences resulting from forest commons intervention concluded with the reform of forest policies in 1790, which finally turn to a balance between State and rural communities interests' ones, with the government allowing traditional practices in forest commons again (Rey, 2004).

4. The dismantling of commons in the nineteenth and twentieth centuries

The feudalism crisis and the transition of Europe to capitalism gave rise to the establishment of a new liberal framework that initiated a process of privatisation of communal resources in the nineteenth century in Spain. This process provoked a social transformation due to the different interests of each social group. First, previous feudal middle class, foreshadow of the bourgeoisie, seek to move upward in the social ladder by emancipating from the feudal social order. Second, peasants defenced the resources they used for the subsistence of their household economies, in a period of time in which collective uses acquired a higher importance as support of the way of life of many rural families. Thirdly, the State and local entities, often in debt, look forward to revitalize municipal rents and enforce their management rights over local lands (Sanz, 2002).

All together contributed to, by the beginning of the twentieth century, traditional practices experienced a progressive decline, resulting in the disappearance by mid-century of the supportive role that commons had once played for the persistence of rural communities (Piqueras, 2002; Ortega, 2012).

4.1 The process for the commons' dismantling

Privatization strategies of the commons broke the social structure and cohesion between rural communities' members, ending up with the traditional solidarity among peasants (Sanz, 2002). Piqueras (2002) distinguishes among three ways of commons' dispossession: legal disposals, nobility appropriation, and land usurpations.

Legal disposals

Following the liberal movement spread through Europe at the end of the Old Regime, a set of legal codes were issued in Spain with important consequences for the Spanish communal system. The new legal framework enforced the usurpation of common lands and use rights accentuated since the seventeenth century. First, the new legal code written down in 1812 did not recognize the presence of communal property in Spain, despite the large tradition of exploitation of forest commons in the country. Second, the new legal framework enforced the usurpation of common lands thorough the disentailing policies approved in 1855, extended until 1924. And finally, the third way in which the new legal framework enforced the usurpation of common lands was by a disempowering process of the traditional forms of self-regulation historically performed by the concejos over their resources.

To put in practice the disentailing policies of concejos' properties, liberals based their ideas on the underutilization of communal resources. The original aim of this policy was to increase the number of rural small land-owners by releasing to the market land properties that were, according to liberal terms, lying stagnant. However, the law did not achieve the objective of creating a better land distribution, as vast quantities of property were acquired by an increasingly dominant bourgeoisie, particularly in the latifundist areas. Interestingly, the liberal political discourse adopted the opposite criteria for justifying years after the establishment of a state forest monitoring for the management of common forest-related resources survival from the disentailing process. By then, the overexploitation that rural communities exerted over public woodlands –property regime in which forest commons were included during the disentailing process– led to the creation of a technical administration to supervise forest uses. Thus, in 1863 the monitor of forest commons by a State Forestry Administration was implemented, with the aim to control custom uses by including them in annual forest management plans. However, as reported by many authors, those plans overly restrictive to traditional uses as their purpose was to avoid the use of woodlands by the local communities (Cobo et al., 1992; Moreno, 1998; GEHR, 1999; Linares, 2000; Piqueras, 2002).

Nobility appropriation

During the Old Regime, the privilege status of feudal lords allowed them to gain a dominant position over the rest of social classes, and thus enjoying of higher access and use of common lands than the rest of community members. In many occasions this privilege led to usurpations of collective uses. This is the case, for example, of Enguera village, in Valencia, which in 1846 sued to the Dukes of Cervellon for the usurpation and use of local woodlands. The court declared that those forests belonged to the Dukes, but that neighbours could use them from cattle grazing and extraction of timber and firewood without paying any tax. The final agreement between the Dukes and the community members was to give a portion of the former forest commons to the nobles, in exchange of free access to the rest of the forested area (Piqueras, 2002). The example shows how, during the establishment of the Liberal Regime, the occupation of local positions of power was an essential instrument for the bourgeois to continue with the enlargement of their personal domains at the expense of common lands (Sanz, 2002).

Their representation in local institutions guaranteed the nobles keeping relations of dependency and dominance over other members of the community. Besides, the control of the local governments enabled local elites to increase their properties on the detriment of common lands and uses. There were multiple patterns of appropriation: from the allocation of private uses in local lands, to the encroachment of common lands for private exploitation, or the encouraging of municipal debts in order to solve them later by transferring public lands in which common lands were included (Sanz, 2002).

This type of dispossession of the commons was justified by liberal politicians. Thus, in 1813, the seventh Count of Toreno –liberal deputy– affirmed that communal property, by being of everyone, was ultimately not preserve by anyone. In his opinion, wealthy people were the unique ones really using the commons, which, when transformed to private property had an owner interested on cultivated them; whilst in the opposite case, no one would take care of their preservation (Piqueras, 2002).

Encroachment and usurpation

Encroachment and distribution of common lands were already performed before disentailing policies; however, the phenomenon accelerated from nineteenth century, with the liberal policies. The commodification of land tenure, the inequality of land accumulation process, and the substitution of livestock activity for subsistence agriculture attending to new market demands resulted in the proliferation of privatising mechanisms among peasantry (Piqueras, 2002; Rey, 2004).

Some authors consider this tendency as a way to release workforce for landowners, as peasants, deprived of their traditional means of subsistence, were forced to sell their labour in the market. Statements such as the ones claimed by Fermín Caballero –liberal intellectual– reflect this idea. Thus, this author wrote, in 1863, that the commons were enhancers of laziness and bad practices, promoters of ideas against proprietorship, and producers of detestable customs from immature societies, among others. Caballero justified the enclosing of commons for a better care of moral and good habits in benefit of the agriculture (Moreno, 1998; Piqueras, 2002).

4.2 Peasants' contestation to external threats after the nineteenth century

The growing pressure exerted on the commons during the course of the eighteenth century led to a new period of rural conflict in response to the liberal policies emerged during the nineteenth century (Piqueras, 2002). Indeed, the issue was a kind of underlying conflict in the Spanish rural societies, that flourished when peasants tried to maintain their economical subsistence. But, in contrast to the resistance practiced during the Old Regime, consisted in a set of very heterogeneous typologies of conflicts which combined individual and collective actions (Sabio, 2002). Thus, whilst during the Old Regime peasantry habitually contested external threats to the commons with actions like illicitly felling trees or encroachment of common lands, their forms of contestation to the liberal policies included a set of individual, unplanned actions as the occupation of fields, illegal pastoral activities, moving of boundary makers or fires, with organized movements among community members such as the collective purchase of former forest commons put up for sale due to Disentailment (Gavira, 1998; Linares, 2000; Piqueras and Sanz, 2007; Arango, 2009; Valbuena-Carabaña et al., 2010).

Among the tactics for the preservation of traditional forest commons are remarkable the protests against the abolishment of communal property rights that arose in north-eastern regions of the peninsula, as well as the protests for the defence of historical rights to profit communal goods, which emerged over the rest of Spain.

In northeast areas of the peninsula, most forest commons were characterized by presenting collective ownership. This collective ownership belonged to the neighbour community members of a specific woodland area. In these forests, neighbours acquired use rights of forest commons through their residential condition (Caballero, 2015). Nonetheless, the liberal state –in his eagerness of privatizing whether commercially exploit forest-related resources– denied the presence of communal property in Spain and considered these forests as public property. To avoid state usurpation, many rural communities decided to privatize themselves the common areas by distributing them among the neighbours. With distributions, communities also aimed at maintaining the integral, multiple uses of former communal areas (Soto et al., 2007).

However, in many occasions, conflicts were generated by the defence of common goods' use rights against prohibitions issued by the liberal administration. The demands of commercial exploitation of common lands, mostly forests, were in confrontation with traditional uses, which were also considered incompatible with the scientific forestry approaches raised during the nineteenth century. Rural communities' access to land, pasture and common goods became thus threatened (Soto et al., 2007). Peasantry resistance against limitations to the profiting of common forests and their increasing privatization was expressed through diverse forms of protest, defending their collective interests from external attacks (Cobo et al., 1992; Sabio, 2002).

Notably, everyday forms of resistance of Spanish peasantry was the most abundant and efficient form of contestation against the usurpation of forest commons made by the State. Initially, disputes consisted in legal actions –as recourse to the courts–, frequently ending up in conflicts like frauds, hiding of lands, threats to forest administration officials, non-payment of taxes, moral discredit of elite members, non-cooperation, obstruction, coercion and violence with largest industrial timber producers... This various forms set up a tenacious resistance of peasant communities (Hervés et al., 1997; Sabio, 2002).

Within the legal framework, a remarkable form of contestation consisted in the collective purchase of forest commons affected by disentailing policies. This was done through the creation of neighbour societies that would pool capital for being able to collectively bid in the auctions and buy disentailed woodlands. Each commoner, often having to resort to loans, would pool money according to his possibilities. This collective action to avoid the loss of historical uses of the commons took place in many regions, particularly the north and inner Spain (Montiel, 2005; Medrano et al., 2013). This was the case, for example in Aragon, where the strategy followed by communities consisted in buying the most advantageous lots in order to avoid their acquisition by foreign buyers who habitually after the bid resold the lands a higher price. If lots could not be acquired, another strategy to prevent the acquisition of land by foreigners consisted on offering very high prices, and soon after, cancelling the sale. Sometimes, this way of obtaining common goods was accompanied by intimidations to foreign buyers and pressure to forest officials in order to reduce the price of the lots (Sabio, 2002).

This type of contestation also appeared in southern regions of Spain like Granada, where neighbours associated themselves to buy forest commons to local municipalities. That was the case of *Sierra de Güejar* mountainous area, which in 1864 created an association to acquire and regulate the access and management of their forest commons. The association drafted in 1866 a regulation specifying that neighbours were the legitimate owners of those common lands, and later, in 1907, included that the association had the capacity to take legal actions for the defence of communal uses and against usurpations (Ortega, 2012).

There were also episodes of resistance in not so legal ways. For instance, in El Bierzo, Leon, the contestation of local inhabitants to the state usurpation of their historical forests rights consisted on the damage of the natural resources in order to avoid their normal exploitation. This form of dispute, although it had long-term negative consequences for tree coverage, it impeded the exploitation of forest commons by the outsiders who bought them during disentailing process (Piqueras, 2002).

4.3 Adaptation

Although peasants' contestation had important successful results, rural community did not resists economic liberal policies assault nor the removal of social cohesion linkages existing during feudalism (Moreno, 1998). Factors that contributed to the dismantling of collective heritage in Spain include 1) the establishment of a monitoring ranger corps in 1876 as support to the constraint of traditional uses contained in the forest management plans written down by State Forest Administration; 2) the positioning of those plans to a higher influence of market forest-related products demand, reducing the multiple use of forest resources; 3) the transformation of small peasants into labourers for being able to participate in the market and obtain from it good previously acquired in the forest commons; 4) the substitution of concejos by local administrative entities conformed by most powerful community members; and 5) the aggressive and contradictory State intervention to communal lands.

The fragmentation of common goods in nineteenth century Spain meant the unbalance of agricultural activities and contributed to leave peasants without enough lands with complementary uses. In the medium term, the dismantling of commons implied a profound historical change of the way that human activities interact with natural resources, with the subsequent implications to landscape, social structures and economic activities. Unsurprisingly, at the beginning of the twentieth century, common lands survived to Disentailment became in many cases, with the exception of forest commons, residual for rural families' economic activity (Piqueras, 2002).

5. Conclusions

Although during the Old Regime there were horizontal and vertical conflicts due to common lands, the essential role that common goods had for rural household economies motivated a social resistance to the appropriation of communal lands. Strategies like the protectionism of local economy, resilience of products obtained from forest commons and complaints over the use and legal definition of common lands represent forms of contestation for the defence of community customs and rights to external threats such as a growing influence of the market and privatization attempts by State. These forms of contestation also reinforced the social cohesion of pre-nineteenth century rural communities (Moreno, 1998; Piqueras, 2002; Ortega, 2012).

However, the dismantling of communal lands during the liberal period in the nineteenth century resulted in a reduced volume of publicly owned property, mostly communal. This surface reduction, at the same time as the persistent limitation to traditional uses, favoured a shift of peasants' activities to less diverse exploitations in private properties.

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Appendix I.b. Las prácticas tradicionales en el manejo de los bosques de castaño y las dehesas de quercíneas. Una visión histórica

Las prácticas tradicionales en el manejo de los bosques de castaño y las dehesas de quercíneas. Una visión histórica

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Resumen

Esta comunicación proporciona una visión histórica de la importancia de las prácticas tradicionales para el mantenimiento de hábitats forestales en montes comunales españoles. Desde la Edad Media hasta principios del siglo XIX, el papel clave de los recursos forestales en las economías de subsistencia de la mayoría de comunidades campesinas favoreció el desarrollo de técnicas dirigidas a preservar el monte y fomentar su aprovechamiento múltiple. Pese a que en ocasiones dichas prácticas se hayan considerado obsoletas, la literatura científica actual estima que algunas de ellas son beneficiosas para el mantenimiento de la biodiversidad.

La llegada de las políticas forestales del siglo XIX supuso el traspaso de la gestión de numerosos montes comunales al Estado. Con ellas, usos tradicionales considerados incompatibles con el mantenimiento de la cubierta forestal fueron restringidos por distintos motivos. Sin embargo, estudios recientes sugieren que aunque el abandono de las prácticas tradicionales haya favorecido la extensión de la cubierta arbórea, la

homogeneización del paisaje forestal ha tenido consecuencias negativas para la diversidad de especies. Tomando como casos de estudio los castañares y las dehesas, esta comunicación describe prácticas tradicionales que favorecen la presencia de mosaicos de hábitats y cuyo progresivo abandono ha provocado una pérdida de biodiversidad.

Palabras clave: Multifuncionalidad; sostenibilidad; técnicas selvícolas para la biodiversidad.

1. Introducción

El monte constituyó un elemento indispensable para la economía pre-industrial española, ya que de él se extraían tanto madera y carbón para la industria naval y la siderúrgica como multitud de productos básicos para otros sectores como fertilizantes para la agricultura o pastos para la ganadería. También en el ámbito doméstico, especialmente en las zonas rurales de montaña, el monte supuso una fuente importante de combustible, alimento y materias primas. Además, el carácter mayoritario que la propiedad comunal presentaba en materia de montes hasta el siglo XIX permitía disponer los recursos forestales gratuitamente o por un pequeño canon, lo que garantizaba la subsistencia de los campesinos con menor poder adquisitivo (Casáis, 1988; Linares, 2000; Jiménez, 2002).

Dada la importancia del monte para la economía local, a lo largo de los siglos los habitantes fueron diseñando normativas para regular su uso. Generalmente, estas regulaciones se establecían de acuerdo a las necesidades vecinales y los recursos disponibles, tratando de combinar el aprovechamiento múltiple del monte con la sustentabilidad de sus recursos (Rey, 2004; Linares, 2007). La multifuncionalidad de los usos tradicionales permitía al campesinado diversificar sus actividades productivas, disminuyendo su vulnerabilidad frente a las fluctuaciones de precios de mercado (Moreno, 1998). A nivel ecológico, la combinación de diversas prácticas productivas favorecía el medio natural, creando un mosaico de hábitats que enriquecía las especies biológicas presentes en el paisaje forestal.

Sin embargo, a pesar de las ventajas que el mantenimiento de algunas de esas prácticas presentaba para compatibilizar el aprovechamiento combinado de distintos productos del monte, la política forestal implantada a principios del siglo XIX decidió no preservarlas. La severa deforestación que sufrían los montes españoles en esa época obligó a las autoridades a adoptar medidas para prevenir un deterioro mayor, prohibiéndose los usos tradicionales al considerarse muchos de ellos perjudiciales para la conservación del monte. Cabe decir, sin embargo, que la adopción de dichas políticas no tuvo en cuenta la complejidad de factores que habían llevado a la deforestación. Así, la deforestación de los montes españoles tenía múltiples causas incluyendo el fuerte crecimiento demográfico que experimentó la población desde finales del siglo XVIII, que se vio acompañado de una mayor demanda de productos agrícolas, lo cual –en ausencia de innovaciones técnicas que permitiesen mejorar la productividad del suelo– favoreció el fenómeno de roturación de los montes. Además, los altos requerimientos de madera de la industria naval y constructora; y las exigencias de carbón vegetal por las industrias energéticas, favorecieron también la deforestación (Casáis, 1988; Jiménez, 2002).

La decisión de excluir del manejo forestal actividades tradicionales consideradas entonces incompatibles con el mantenimiento del arbolado, como la saca de madera y leñas o el pastoreo, tuvo efectos contrarios a los deseados por las autoridades, ya que el impacto que la limitación a estas actividades tenía para la subsistencia de las economías campesinas dio lugar a un gran rechazo social de las políticas de manejo forestal. Ello trajo consecuencias ecológicas negativas inmediatas para los montes debidas a usos fraudulentos o incendios provocados como forma de protesta (Cobo et al., 1992; Linares, 2000). A largo plazo, estos cambios en las políticas provocaron que las economías rurales dejasen de emplear prácticas tradicionales (Seijo et al., 2015), con la consecuente pérdida de conocimiento ecológico local asociado a dichas prácticas. Esta contribución recupera parte de este conocimiento describiendo dos prácticas culturales que permitían compatibilizar actividades agrícolas y ganaderas con la presencia de arbolado. En concreto, se detallan dos labores llevadas a cabo en ecosistemas con eleva riqueza de especies como son los bosques de castaño (*Castanea sativa* Mill.) y las dehesas de quercíneas.

2. El castañar

Los castañares, bosques dominados por el castaño (*Castanea sativa*), son un claro ejemplo de masa forestal sometida a un manejo multifuncional con una incidencia positiva en la conservación de la biodiversidad. Sus productos –tanto el fruto como la madera– han sido aprovechados durante siglos por las comunidades locales. Además, estos bosques conforman un hábitat de gran importancia ecológica al albergar una amplia variedad de especies de flora y fauna, motivo por el cual fue incluido en el anexo I de la Directiva 92/43/CEE (Gondard et al., 2006; Guitián et al., 2012a).

Tratándose de una especie forestal nativa en las regiones atlánticas de la Península Ibérica, la extensión de los castañares aumentó notablemente durante los tiempos de poblamiento de griegos y romanos, los cuáles fomentaron su cultivo (Melicharová and Vizoso-Arribe, 2012). La influencia antrópica en la expansión de estos bosques se acentúo a partir de la Edad Media (Rubio, 2009). Las óptimas propiedades de conservación de su fruto favoreció su empleo durante el periodo medieval como fuente de alimento durante los meses de invierno, mientras que la madera se aprovechaba como materia prima para la elaboración de utensilios y muebles y su leña como combustible (Pereira-Lorenzo and Ramos-Cabrer, 2004). Asimismo, los procesos de Reconquista que experimentó España durante los siglos VIII-XV facilitaron la entrada del cultivo de castaño en regiones en las que previamente no existía, llegando incluso a utilizarse esta especie como símbolo reivindicativo de terrenos comunales, tipo de propiedad colectiva generalizada en los nuevos asentamientos (Gallardo, 2002; Ríos-Mesa et al., 2011).

La importancia del aprovechamiento de los castañares para las economías campesinas, especialmente en zonas de montaña del noroeste español, fue relevante para la persistencia de estos bosques, que solían ubicarse en los alrededores de los núcleos rurales (Guitián et al., 2012b). Aún ahora se pueden encontrar en algunas regiones de Galicia relictos de castañares con origen medieval (Pereira-Lorenzo and Ramos-Cabrer, 2004). No obstante, desde principios del siglo XX, las masas de *Castanea sativa* experimentan un importante retroceso atribuido tanto a su abandono causado por el despoblamiento rural como a la aparición de nuevas enfermedades (Gondard et al., 2007;

Guitián et al., 2012b). Dado que los cultivos de castaño requieren para su mantenimiento de una constante gestión humana, dicho abandono lleva consigo la desaparición de este hábitat y de su biodiversidad asociada (Gondard et al., 2006; Rubio, 2009).

Numerosas fuentes bibliográficas indican la relación entre una alta biodiversidad en los bosques de castaño con ciertas perturbaciones humanas, consideradas en esta comunicación como prácticas tradicionales (Gondard et al., 2006; Guitián et al., 2012a; Cevasco and Moreno, 2015). La poda, el pastoreo moderado, la quema, la selección de semillas o la formación de terrazas forestales, son algunos ejemplos de prácticas tradicionales que eran llevadas a cabo por las comunidades locales en los castañares (Gondard et al., 2006; Seijo et al., 2015) y que tienen un efecto positivo en la biodiversidad al favorecer la heterogeneidad del hábitat (Gondard et al., 2007). Cabe destacar la técnica del trasmocho, un tipo de poda que se realiza a una altura del árbol lo suficientemente elevada para que el ganado no pueda alcanzar los rebrotes (Figura A.1). Este tratamiento se ha llevado a cabo desde tiempos inmemoriales debido a las ventajas económicas que ofrecía a las comunidades rurales, ya que compatibilizaba el aprovechamiento maderable o de leñas con la actividad del pastoreo (Elorrieta, 1949; Viéitez et al., 1999). A nivel ecológico, el trasmoche favorece el mantenimiento de pies de elevada edad, que presentan huecos en su interior y madera muerta, necesarios para la supervivencia de organismos saprofitos (Siitonen and Ranius, 2015).

Figura A.1. La práctica tradicional del trasmocho permite la persistencia de árboles con el pastoreo, como ilustra esta fotografía obtenida en el castañar milenario El Habario de Cillorigo de Liébana (Cantabria). Créditos: S. Guadilla-Sáez.

La técnica de cultivo del castaño varía en función del producto que se espera obtener; madera, castaña, o ambos. Tradicionalmente en los montes del País Vasco y Navarra, el proceso se iniciaba enterrando las semillas en suelos de buena calidad durante los meses de invierno, particularmente durante el mes de noviembre, procurando situar de lado la castaña para facilitar la salida al exterior del brote. Entorno a dos o tres años después, los brinzales en disposición de ser trasplantados, denominados como *chírpias*, eran llevados a vivero entre los meses de noviembre y marzo, en fase creciente de luna para aprovechar la fuerza de la savia. A los dos años siguientes, se les sometía a una primera poda y se eliminaban los ejemplares más débiles. También se procedía a realizar un corte liso de la punta de la plántula, a una altura mínima de 2,5 metros, para evitar que el ganado alcanzase la copa de las que se estimulará la generación periódica de brotes. De nuevo, con un turno de espera de entre dos y tres años y una altura aproximada de 2,5 metros y un diámetro mínimo de 1 cm, los ejemplares eran trasladados al monte. Este segundo traslado también se llevaba a cabo entre noviembre y marzo dejando una separación de entre 5,5 a 7,5 metros en el marco de plantación de los futuros trasmochos. Habitualmente, dado que estas plantaciones se llevaban a cabo en monte abierto, había que proteger los arbolillos de los animales salvajes y del ganado mediante el empleo de matorrales espinosos locales como el de los géneros *Genista* y *Crataegus*. Esta protección vegetal también ayudaba frente a heladas tempranas. Finalmente, a los 16-17 años de edad, la guía se cortaba a una altura de entre 3 y 5 metros, proceso que se llevaba a cabo en luna menguante. La labor se realizaba a partir de finales de septiembre, en el caso de árboles desmochados por primera vez, y a partir del 20 de febrero en árboles ya trasmochados previamente, terminando el periodo de poda para ambos casos el 25 de marzo (Elorrieta, 1949; Pereira-Lorenzo and Ramos-Cabrer, 2004; Aragón, 2013).

La característica más importante del trasmocho es la repetición del corte en un intervalo de tiempo que varía según el destino final de la madera y la zona geográfica en la que se ubica el castañar. De este modo, el desmoche periódico puede realizarse desde los doce o quince años, hasta los cuarenta o cincuenta años (Rubio, 2009). Elorrieta (1949), por ejemplo, establece un turno de entre veinte y veinticinco años en castañares de la provincia de Lugo (Galicia) para el aprovechamiento de madera para vigas, postes y tablones. El cese de estos cortes periódicos tiene consecuencias para el arbolado y la capacidad productiva de la masa. Por un lado, los castaños reducen el tamaño de sus frutos y desarrollan chupones de las raíces, cuyo crecimiento echa a perder el cultivo (Pereira-Lorenzo and Ramos-Cabrer, 2004). Asimismo, el descontrol en el crecimiento de las nuevas ramas da lugar a un reparto heterogéneo de la biomasa en la copa, lo cual modifica el centro de gravedad de los árboles favoreciendo la aparición de roturas (de Francisco, 2013).

El declive de prácticas tradicionales en castañares ha tenido implicaciones negativas para la conservación de la biodiversidad y el paisaje forestal en estos ecosistemas. En el caso de la técnica descrita en este apartado de la contribución, el trasmocho, su aplicación favorecía la presencia de huecos en el interior de los troncos, elementos clave a nivel biológico al albergar numerosas especies de vertebrados e invertebrados, particularmente importantes hábitats para especies saprófitas (Siitonen and Ranius, 2015). La recogida de las ramas obtenidas a partir de su poda, así como su quema, también producía externalidades positivas en la cubierta herbácea. De hecho, se ha documentado una disminución de la biodiversidad al abandonarse la práctica del trasmoche (Cevasco and Moreno, 2015). Diversos estudios que han analizado la influencia de las perturbaciones humanas en la composición florística de los castañares también muestran una mayor riqueza biológica en los cultivos de castaño que en masas abandonadas (Gondard et al., 2006; Gondard et al., 2007; Guitián et al., 2012a). Ello se debe a que la ausencia de tratamientos silvícolas favorece la sustitución del castaño por masas del género *Quercus*, con menor número de especies acompañantes, y de naturaleza más cerrada, favoreciendo a su vez la vulnerabilidad del entorno natural a los incendios.

A nivel paisajístico, la actuación llevada a cabo por los organismos saprófitos en los troncos de los árboles trasmochos les confiere un aspecto singular, considerado de gran belleza por algunos autores (Viéitez et al., 1999). Asimismo, las particularidades con las que la técnica del trasmocho era aplicada a nivel local (herramientas, especies, usos finales, alturas de corte, época del año) le dotan de una gran riqueza cuyo legado no debería perderse (Cantero and Passola, 2013).

3. La dehesa de quercíneas

Las dehesas son praderas de tipo sabana con árboles del género *Quercus* dispersos en las que se aplican prácticas de manejo para maximizar la producción de su fruto, la bellota, empleado como alimento de ganado (Linares, 2007). Símbolo del paisaje mediterráneo, especialmente del suroeste de la Península Ibérica, las dehesas conforman un hábitat particular resultado de un uso histórico del ser humano que –al igual que sucedía en el bosque de castaño– alberga una alta diversidad biológica (Pulido et al., 2001; Carmona et al., 2013; García-Tejero et al., 2013; López-Sánchez et al., 2016). Precisamente debido a su importancia para la conservación de especies, las dehesas perennifolias de *Quercus* spp. están incluidas en el anexo I de la Directiva Hábitats 92/43/CE (Ramírez and Díaz, 2008).

Estudios polínicos indican la existencia de formas de manejo ancestrales en dehesas, como el pastoreo de ganado o el empleo de fuego (Alagona et al., 2013). El uso de estas actividades favoreció la evolución del bosque mediterráneo a paisajes adehesados similares a los actuales, en los que ejemplares de especies arbóreas nativas como la encina (*Quercus ilex* L.), el alcornoque (*Q. suber* L.), el quejigo (*Q. faginea* Lam.) o el melojo (*Q. pyrenaica* Willd.) se insertan en áreas de cultivo o pastizal natural (Tárrega et al., 2009; Acosta, 2014). Si bien, la conformación del paisaje característico de las dehesas debe su origen principalmente a la época de asentamientos cristianos, durante los siglos XI-XV (ALAGONA et al, 2013). Posteriormente, a mediados del siglo XV, tuvo lugar lo que en la literatura se denomina como el periodo de consolidación de la dehesa, momento en el que distinguió entre dos tipologías de propiedad: las dehesas privadas, pertenecientes a las clases altas y el clero, y las dehesas boyales, bajo control de los municipios o comunidades locales. No obstante, el sistema que conforma la dehesa se ha visto sometido a numerosos cambios tanto de propiedad, como de gestión y uso, conformando la variedad que observamos en la actualidad. Así, mientras que en el noroeste español las dehesas han persistido bajo sistemas tradicionales de manejo de pastos comunales, sus homólogas sureñas sufrieron las transformaciones liberales iniciadas en el siglo XIX, pasando en gran medida a manos privadas, proceso que alteró sustancialmente los patrones paisajísticos de estas masas (Tárrega et al., 2009; Alagona et al., 2013).

Las dehesas surgen ante la necesidad de aprovechar un espacio forestal de baja capacidad productiva, caracterizado por condiciones edafoclimáticas poco favorables para el desarrollo de actividades agrícolas, como son una marcada sequía estival y un suelo poco fértil. Mediante el pastoreo, las sociedades del Neolítico propiciaron la creación de claros en el bosque originario en los que poder establecer cultivos. La diversidad de aprovechamientos que las dehesas ofrecían, al conseguir compatibilizar usos agrícolas y ganaderos con la conservación de la vegetación arbórea, favoreció su expansión durante la Edad Media y su persistencia hasta épocas recientes (Olea and San Miguel-Ayanz, 2006; López et al., 2007). Sin embargo, la gestión tradicional que se venía desarrollando desde el Medievo sufre un severo abandono en la actualidad (Valbuena-Carabaña et al., 2008). Las políticas de desarrollo económico aplicadas en las últimas décadas han promocionado la intensificación de la agricultura y el aumento de cargas ganaderas, lo que ha supuesto una progresiva infrautilización de amplias zonas rurales, con importantes consecuencias socioeconómicas y ecológicas para las dehesas (Tárrega et al., 2009; López-Sánchez et al., 2016).

El cese de actividades tradicionales ha llevado a la matorralización de estos hábitats, lo que implica una mayor homogenización del paisaje y un incremento del riesgo de incendios, con las potenciales secuelas que un incendio tiene en la composición de especies y condiciones del suelo (Taboada et al., 2006; Tárrega et al., 2009; García-Tejero et al., 2013). Además, los sistemas de explotación ancestrales están siendo sustituidos por otros de carácter más intenso que promocionan la plantación de coníferas, afectando negativamente la pervivencia de numerosas especies silvestres asociadas a las dehesas de quercíneas (Taboada et al., 2006; Pulido et al., 2013). La alta riqueza biología que alberga este agroecosistema forestal hace necesaria la reactivación de su gestión tradicional, y en especial de las actividades de manejo que dan lugar a la estructura típica de su vegetación.

Prácticas como la poda de copas o la eliminación de matorral mediante el empleo del fuego, fueron habituales en estos bosques con el objetivo de favorecer el crecimiento de pasto y la producción de bellotas como fuente de alimento para el ganado (Valbuena-Carabaña et al., 2008; Fernández et al., 2014). El primer ejemplo, la poda de copas, es un tratamiento selvícola consistente en la eliminación de ramas centrales de la copa del árbol para alejar las hojas del tronco y así limitar su aporte de agua (Alejano et al., 2008). Dado que el desarrollo vegetativo y fructificación de un árbol está íntimamente relacionado con el desarrollo de las raíces y la cantidad de ramas y de hojas que constituyen las copas (Vela, 1959), no es extraño que la poda se haya considerado tradicionalmente como un método que incrementa la producción de bellota de los árboles. Aunque algunos estudios recientes cuestionan la efectividad real de la poda para aumentar la capacidad de fructificación de los árboles (Koening et al., 2013), esta práctica se empleaba también para favorecer el aprovechamiento multifuncional de la dehesa, ya que permitía compatibilizar la extracción de bellota con la obtención de leña y ramón, el desarrollo de pastos herbáceos y cultivos ocasionales, y la actividad de pastoreo (Fernández et al., 2014).

En esta contribución describimos la poda tradicional llevada a cabo en encinas, al ser ésta la especie forestal española asociada en mayor medida a las dehesas. En términos generales, la poda se hacía tradicionalmente para favorecer el buen desarrollo y formación de los árboles, procurando mantener un tronco sano que permitiese un aprovechamiento último maderero (Vela, 1959). La encina puede obtenerse a partir de la siembra de bellota o por pies elegidos durante la roza de matas de chaparro –es decir, matas de encina muy ramosas y de poca altura (SECF, 2005)–, comenzando su tratamiento cuando el árbol alcanza unas dimensiones adecuadas (2-3 metros de altura) (Pérez, 1917). Con anterioridad a ello, los cuidados culturales deben limitarse a eliminar ramas defectuosas o excesivas de cepas y chupones que salgan del tronco principal, con el fin de favorecer su crecimiento (Ximenez, 1948).

La primera poda, denominada como poda de formación, consiste en dejar un pequeño número de ramas principales (de dos a cuatro) distribuidas regularmente en planta para que sirvan de soporte del resto de ramas de la copa. Para ello, se realiza una primera operación de "abrir el chaparro" que consiste en retirar las ramas interiores. A continuación, se limpian de brotes y ramificaciones secundarias aquellas ramas que se han seleccionado como principales (Mateos, 2004). La altura a la que se realiza la poda de formación es variable, yendo desde los 1,30 metros (altura aproximada del pecho de una persona) para facilitar podas posteriores, hasta 2,50, 3 e incluso 4 metros de altura. Esta intervención se debe practicar en el periodo que transcurre desde el final de la cosecha de la bellota –finales de diciembre o en enero– hasta un poco antes de que comiencen a darse los renuevos –mediados de marzo–, para dar vigor a la nueva savia producida y enriquecer un suelo limpio y estercolado por el paso del ganado (Ximenez, 1948; Celorico, 1950; Acosta, 2014).

El cambio brusco de insolación, unido a la activación de yemas adventicias de la madera a consecuencia de la poda de formación, hacen necesario efectuar cuidados culturales posteriormente para la eliminación de brotes chupones y ramificaciones secundarias. Estas actuaciones, denominadas podas de mantenimiento, de conservación o simplemente limpias tienen una intensidad mayor y se deben aplicar una vez la encina llega a la fase adulta. Se llevan a cabo con el objetivo de reducir las ramas de menor fructificación y mantener la forma globosa de la copa. Para ello, se quitan las ramas secas, enfermas e interiores, así como las verticales, manteniéndose las horizontales y las colgantes para ralentizar el paso de la savia elaborada a través de las ramas, pretendiéndose fomentar con este método la producción de bellota. Se consigue así mismo la obtención de leña fina y ramón y se incrementa la radicación solar que llega al sustrato herbáceo, beneficiando a los posibles cultivos que pueda haber. También sirve de control de la altura del árbol en los casos en que las encinas son vareadas (Mateos, 2004; Acosta, 2014; Fernández et al., 2014).

A pesar de que la poda es una labor cultural recomendada en encinares –considerada por algunos autores la principal práctica de cuidado de la dehesa–, su coste económico hace que los turnos de corta se espacien cada vez más o simplemente no se lleven a cabo. Retrasar en exceso este tratamiento es perjudicial para los árboles, ya que favorece el desarrollo de mayores grosores de las ramas que, al cortarse, originan graves heridas (Ximenez, 1962; Acosta, 2014). No llevarlas a cabo, agudiza el envejecimiento generalizado que presentan las dehesas en la actualidad debido, principalmente, a la falta de regeneración que vienen padeciendo estas formaciones boscosas en las últimas décadas. La ausencia de la práctica de podar incrementa la problemática del envejecimiento de dehesas debido a que, por un lado, acelera el envejecimiento de los pies, ya que no se eliminan los procedentes de rebrotes de cepa o chupones, menos longevos que los pies procedentes de semilla. Por otro lado, el desuso de esta intervención tradicional produce un efecto similar al exceso de espesura, por el cual la masa paraliza su crecimiento, debilitándola frente a fenómenos como incendios o plagas (Montoya and Mesón, 1993).

La desaparición de la poda también tiene consecuencias paisajísticas y culturales, dado que esta intervención dota a las encinas en dehesas de su porte característico, estrechamente relacionado con el aprovechamiento ganadero (Acosta, 2014). Asimismo, se hace necesario reconocer el profundo conocimiento adquirido por aquellas personas que ejercían oficios tradicionales de dehesas –como el de podador–, y recuperar y mantener sus rutinas, ya que éstas conforman un patrimonio etnográfico de gran valor (San Miguel, 1998; Silva, 2010).

4. Conclusión

La recopilación histórica de las prácticas tradicionales de manejo, junto con los estudios ecológicos de los efectos de estas prácticas en la biodiversidad y la productividad local, avalan la idea de que las prácticas tradicionales de manejo han sido beneficiosas no solo para un aprovechamiento multifuncional del monte, sino también para la creación de sistemas ecológicos únicos y de alta biodiversidad (Guèze et al., 2015). Resulta por ello interesante tratar de recuperar el papel económico que los aprovechamientos forestales históricamente representaron en las comunidades rurales, procurando la inclusión en su gestión de aquellas formas de manejo tradicionales que contribuían a la presencia de paisajes más heterogéneos. No obstante, la preservación del conocimiento tradicional va más allá de las ventajas que ofrece para la conservación de la biodiversidad y para la economía forestal. Se trata de un legado cultural que nos transmite las relaciones históricas de nuestros antepasados con su entorno natural, resultado de sus esfuerzos y capacidades para subsistir con recursos limitados, y que urge documentar para prevenir su desaparición.

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Appendix II

Appendix II.a. Results for the ecological indices and explanatory variables in Chapter III

Ecological indices

Table A.1. Results for the Shapiro-Wilk normality test conducted through the species diversity and stand structure indices, for the mixed broadleaf woodlands (habitat 1), beech forest (habitat 2) and Pyrenean oak (habitat 3).

		Habitat 1			Habitat 2			Habitat 3		
Index	Code	Statistic	Df	p-value	Statistic	Df	p-value	Statistic	Df	p-value
Shannon	H'	.944	9	.631	.869	22	.007	.916	15	.165
Evenness	J'	.765	9	.008	.884	22	.014	.842	15	.013
Richness	D_{Mn}	.781	9	.012	.774	22	.000	.848	15	.016
Simpson complement	$1 - D$	.908	9	.301	.984	22	.967	.948	15	.495
Reciprocal Berger-Parker	$1/d$	.868	9	.117	.757	22	.000	.960	15	.699
Clark-Evans	R	.987	12	.998	.985	22	.975	.951	15	.540
Mingling complement	$1 - M_i$	.987	12	.998	.552	22	.000	.802	15	.004
Uniform angle	W_i	.991	12	.999	.984	22	.967	.995	15	1.00

Table A.2. Descriptive statistics for the ecological indices, and statistical test results (T= two sample t-test, W= two sample Wilconxon rank-sum test) between the plots inside and outside Picos de Europa, for the mixed broadleaf woodlands.

		Test of difference			Pool	Outside		Inside	
Index	Code		Df	p-value	Mean (±SD)	N	Mean (±SD)	N	Mean (±SD)
Shannon	H'	T	7	.197	.805 (±.529)	5	.594 (±.197)	4	1.07 (±.282)
Evenness	J'	W		.050	.659 (±.302)	5	.506 (±.337)	4	.849 (±.070)
Richness	D_{Mn}	W		.085	.348 (±.275)	5	.245 (±.186)	4	.477 (±.340)
Simpson complement	$1 - D$	T	7	.172	.451 (±.251)	5	.346 (±.266)	4	.584 (±.090)

Table A.2. (Cont.) Descriptive statistics for the ecological indices, and statistical test results (T= two sample t-test, W= two sample Wilconxon rank-sum test) between the plots inside and outside Picos de Europa, for the mixed broadleaf woodlands.									
		Test of difference			Pool	Outside		Inside	
Index	Code		Df	p-value	Mean (±SD)	N	Mean (±SD)	N	Mean (±SD)
Reciprocal Berger-Parker	$1/d$	T	7	.135	1.72 (±.653)	5	1.43 (±.167)	4	2.09 (±.394)
Clark-Evans	R	T	10	.498	1.49 (±.093)	8	1.50 (±.037)	4	1.46 (±.033)
Mingling complement	$1 - M_i$	T	10	.127	.646 (±.198)	8	.708 (±.070)	4	.521 (±.071)
Uniform angle	W_i	T	10	.812	.469 (±.0.99)	8	.463 (±.036)	4	.479 (±.052)

Table A.3. Descriptive statistics for the ecological indices, and statistical test results (T= two sample t-test, W= two sample Wilconxon rank-sum test) between the plots inside and outside Picos de Europa, for the beech forests (*Fagus sylvatica* L.). In bold, significant differences between outside and inside studied plots based on their *p-values* after Bonferroni correction (** *p-value* ≤ 0.005; * *p-value* ≤0.025).									
		Test of difference			Pool	Outside		Inside	
Index	Code		Df	p-value	Mean (±SD)	N	Mean (±SD)	N	Mean (±SD)
Shannon	H'	W		.004	.397 (±.410)	4	**.952**** (±.206)	18	**.274**** (±.335)
Evenness	J'	W		.009	.364 (±.346)	4	**.763*** (±.151)	18	**.275*** (±.313)
Richness	D_{Mn}	W		.011	.356 (±.288)	4	**.729*** (±.445)	18	**.273*** (±.165)
Simpson complement	$1 - D$	T	20	.990	.544 (±.364)	4	.546 (±.039)	18	.543 (±.095)
Reciprocal Berger-Parker	$1/d$	W		.009	1.28 (±.400)	4	**1.80*** (±.472)	18	**1.16*** (±.285)
Clark-Evans	R	T	20	.629	1.51 (±.100)	4	1.49 (±.042)	18	1.52 (±.025)
Mingling complement	$1 - M_i$	W		.064	.898 (±.185)	4	0.771 (±.258)	18	.926 (±.161)
Uniform angle	W_i	T	20	.236	.496 (±.107)	4	.437 (±.040)	18	.509 (±.026)

Table A.4. Descriptive statistics for the ecological indices, and statistical test results (T= two sample t-test, W= two sample Wilconxon rank-sum test) between the plots inside and outside Picos de Europa, for the Pyrenean oak stands.

		Test of difference			Pool	Outside		Inside	
Index	Code		Df	p-value	Mean (±SD)	N	Mean (±SD)	N	Mean (±SD)
Shannon	H'	T	13	.391	.897 (±.430)	12	.946 (±.104)	3	.698 (±.412)
Evenness	J'	W		.312	.653 (±.254)	12	.699 (±.198)	3	.468 (±.416)
Richness	D_{Mn}	W		.665	.459 (±.219)	12	.470 (±.239)	3	.413 (±.131)
Simpson complement	$1 - D$	T	13	.699	.559 (±.202)	12	.470 (±.413)	3	.412 (±.076)
Reciprocal Berger-Parker	$1/d$	T	13	.651	1.76 (±.521)	12	1.79 (±.135)	3	1.63 (±.472)
Clark-Evans	R	T	13	.259	1.46 (±.126)	12	1.44 (±.037)	3	1.54 (±.052)
Mingling complement	$1 - M_i$	W		.556	.811 (±.210)	12	.826 (±.144)	3	.75 (±.433)
Uniform angle	W_i	T	13	.878	.505 (±.066)	12	.507 (±.066)	3	.5 (±.083)

Explanatory variables

Table A.5. Results for the Shapiro-Wilk normality test conducted through the explanatory variables. See table 3.2 for codes of variables.

Variable	Statistic	Df	*p*-value
Plot topography			
UTMY	.967	50	.168
UTMX	.934	50	.008
SLO	.980	49	.569
Soil characteristics			
PH	.975	45	.423
TEX	.991	48	.971
OM	.982	50	.647
STO	.991	50	.968
Plot isolation			
DIST1	.725	50	.000
DIST2	.926	50	.004
POP	.903	48	.001
Anthropogenic disturbances			
GRA	.975	50	.013
SILV	.981	50	.354
FEL	.999	50	1.00
COV	.986	50	.809
GRO	.545	50	.000

Table A.6. Correlation matrix and contingency table for categorical variables conducted through the explanatory variables. See table 3.2 for codes of variables.

Variable	UTMY	UTMX	SLO	PH	TEX	OM	STO	DIST1	DIST2	POP	GRA	SILV	FEL	COV
UTMX	.234^S (.135)	-												
SLO	.095 (.517)	-.175^S (.267)	-											
PH	-.100 (.512)	.171^S (.279)	.211 (.168)	-										
TEX	.027 (.853)	.339^S (.028)*	-.165 (.266)	-.120 (.437)	-									
OM	.406 (.003)*	.102^S (.519)	.042 (.773)	.010 (.949)	-.048 (.745)	-								
STO	-.111 (.443)	.057 (.722)	.318 (.026)*	-.071 (.631)	3.63 (.726)	.152 (.290)	-							
DIST1	.018^S (.911)	.003^S (.987)	-.221^S (.160)	.201^S (.202)	-.306^S (.049)	.039^S (.808)	.156 (.324)	-						
DIST2	-.439^S (.004)*	.433^S (.004)*	-.109^S (.492)	.401^S (.008)*	-.074^S (.641)	-.131^S (.408)	.256 (.102)	.114^S (.743)	-					
POP	-.619^S (.000)*	-.099^S (.534)	-.086^S (.588)	.143^S (.366)	-.0124^S (.435)	-.557^S (.000)*	.258 (.099)	-.032^S (.842)	.324^S (.036)*	-				
GRA	-.050^S (.753)	.050^S (.753)	-.065^S (.684)	-.044^S (.783)	1.17 (.556)	-.206^S (.191)	1.52 (.678)	-.117^S (.461)	.075^S (.637)	.048^S (.762)	-			
SILV	-.287 (.100)	.083^S (.602)	-.159 (.276)	.081 (.598)	-.777 (.678)	-.181 (.208)	1.39 (.708)	.049^S (.758)	.274^S (.079)	-.045^S (.778)	.128 (.720)	-		
FEL	-.252 (.255)	.045^S (.775)	-.090 (.537)	.005 (.975)	-.183 (.402)	-.243 (.088)	1.44 (.695)	-.012^S (.941)	.108^S (.494)	-.057^S (.717)	.038 (.846)	21.95 (.000)*	-	
COV	-.373 (.008)*	.075 (.637)	-.266 (.065)	.001 (.995)	.504 (.777)	-.165 (.253)	2.29 (.514)	-.119 (.453)	.279 (.073)	.050 (.751)	.613 (.434)	37.76 (.000)*	18.84 (.000)*	-
GRO	-.092^S (.953)	-.018^S (.908)	.314^S (.043)*	-.037^S (.816)	1.33 (.515)	.023^S (.884)	2.09 (.554)	-.295^S (.057)	-.083^S (.601)	.056^S (.726)	.980 (.322)	1.07 (.300)	1.92 .166)	1.17 (.279)

Correlation coefficients are presented, Spearman rank correlation (indicated by S superscript) when the data were not normally distributed.

Table A.7. Summary of the ten best-ranked models of the ecological data for the dependent variables Shannon index (H') and Richness (D_{Mn}), including the total number of parameters in the model (K), followed by the values for both AIC, ΔAIC and w_i. Rankings based on Akaike's Information Criterion (AIC). See table 3.2 for codes of variables.					
Model no.	Model variables	K	AIC	ΔAIC$_i$	w_i (%)
Shannon					
1	$H' = (-0.002) \times \text{SLO} + (-0.003) \times \text{PH} + (-0.0002) \times \text{DIST2} + (-0.196) \times FEL + 1.287$	6	47.75	0	5.7
2	$H' = (-0.002) \times SLO + 0.015 \times PH + (-0.0003) \times DIST2 + 1.083$	5	48.08	0.3	4.8
3	$H' = (-0.0005) \times SLO + 0.112 \times \text{TEX} + (-0.0003) \times DIST2 + 0.904$	5	48.97	1.2	3.1
4	$H' = (-0.002) \times SLO + (-0.0003) \times DIST2 + (-0.184) \times FEL + 1.271$	5	49.15	1.4	2.8
5	$H' = (-0.001) \times SLO + 0.100 \times \text{TEX} + (-0.0003) \times DIST2 + (-0.162) \times FEL + 1.031$	6	49.16	1.4	2.8
6	$H' = (-0.002) \times SLO + 0.014 \times PH + (-0.0003) \times DIST2 + (-0.118) \times SILV + 1.180$	6	49.38	1.6	2.5
7	$H' = 0.0004 \times SLO + (-0.083) \times STO + (-0.0002) \times DIST2 + (-0.228) \times FEL + 1.310$	6	49.39	1.6	2.5
8	$H' = (-0.005) \times PH + (-0.0003) \times DIST2 + 1.084$	4	49.46	1.7	2.4
9	$H' = 0.129 \times TEX + (-0.0003) \times DIST2 + 0.823$	4	49.46	1.7	2.4
10	$H' = (-0.003) \times SLO + 0.011 \times PH + (-0.0003) \times DIST2 + (-0.104) \times COV + 1.212$	6	49.54	1.8	2.3
Richness					
1	$D_{Mn} = 0.028 \times OM + (-0.0004) \times DIST1 + 0.289$	4	5.22	0	2.3
2	$D_{Mn} = 0.024 \times OM + (-0.0001) \times DIST2 + 0.357$	4	5.48	0.3	2.0
3	$D_{Mn} = 0.029 \times OM + (-0.0001) \times DIST2 + 0.103 \times FEL + 0.282$	5	5.54	0.3	2.0
4	$D_{Mn} = 0.032 \times OM + (-0.0004) \times DIST1 + 0.077 \times FEL + 0.224$	5	6.13	0.9	1.5
5	$D_{Mn} = 0.026 \times OM + (-0.0001) \times DIST2 + 0.083 \times COV + 0.306$	5	6.26	1.0	1.4

Table A.7. (Cont.) Summary of the ten best-ranked models of the ecological data for the dependent variables Shannon index (H') and Richness (D_{Mn}), including the total number of parameters in the model (K), followed by the values for both AIC, ΔAIC and w_i. Rankings based on Akaike's Information Criterion (AIC). See table 3.2 for codes of variables.					
Model no.	Model variables	K	AIC	ΔAIC_i	w_i (%)
Richness					
6	$D_{Mn} = 0.002 \times SLO + 0.026 \times OM + (-0.0001) \times DIST2 + 0.278$	5	6.29	1.1	1.4
7	$D_{Mn} = 0.026 \times OM + (-0.0001) \times DIST2 + 0.081 \times SILV + 0.305$	5	6.34	1.1	1.3
8	$D_{Mn} = 0.023 \times OM + (-0.0001) \times DIST2 + 0.171 \times GRO + 0.348$	5	6.52	1.3	1.2
9	$D_{Mn} = 0.03 \times OM + (-0.0001) \times DIST1 + 0.062 \times SILV + 0.240$	5	6.53	1.3	1.2
10	$D_{Mn} = 0.028 \times OM + (-0.0004) \times DIST2 + 0.145 \times GRO + 0.282$	5	6.54	1.3	1.2

Appendix II.b. Field plot form used in Chapter III

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1. IDENTIFICATION

MUNICIPALITY ________________ TOPOGRAPHICAL MAP ________________ LATITUDE/LONGITUDE ________________

DATE ________________ SLOPE ________________% ELEVATION (m) __________ BEGINNING TIME ________

ENDING TIME ________ NAME OF THE PERSON FILLING OUT THIS FORM: ____________ ____________ ____________ ____________ ____________ ____________

2. CLASIFICATION OF THE PLOT

LAND USE CLASSIFICATION ________ TOTAL VEGETATION COVER ________ TREE COVER ________ FOREST TYPE ________

GROUND VEGETATION STRUCTURE COVER________ TREE CLASSES Nº ________ SPATIAL PATTERNING ________

0=Not wooded; 1=Uniform; 3= Scattered in clumps; 4= Scattered in strips; 5= Mosaic pattern; 6=Irregular; 7=Individuals; 8=Openings; 9=Others.

Vegetation structures	Code
Medium sized trees, bushes and woody plants (3 m < H.t.< 5 m)	
High shrubs (1,5 m < H.t.< 3 m)	
Medium shrubs (0,5 m < H.t.< 1,50 m)	
Low shrubs (0,05 m < H.t.< 0,5 m)	
Shrubs close to ground level (0,02 m < H.t.< 0,05 m)	
Forbland (Herbaceous, H.t. > 1m)	
Fern cover	
Herbaceous plant cover	

The sum of canopy covers could be more tan 100, considering overlapping. Code 0=no presence in the plot; 1=Rare species. Cover near zero; 2=Scattered plants. CC ≤1%; 3=Low number of plants. CC 1-5%; 4=Intermediate nº of plants. CC 5-10%; 5=High number of plants. CC 10-25%; 6=CC 25-50%; 7=CC 50-75%; 8=CC >75%.

SPECIES COMPOSITION ________ 1=Pure stands; 2=Mixed stand; 3=Stratified Mixture; 9=Others.

MAIN SPP ________ DENSITY ________ STAGE ________

________ ________ ________

________ ________ ________

1=Seedling or small sapling; 2=Thicket; 3=Pole wood (10-20 cm Ø); 4=High forest (>20 cm Ø); 9=Other.

3. STAND STRUCTURE MEASUREMENTS (R= 25m)

MAIN SPP ________ ________ ________

MAIN STRUCTURE ________ ________ ________

1=Even-aged; 2= Even-aged stand (same age class); 3=Two-aged stands; 4=Uneven-aged stand; 9=Other.

ORIGIN OF THE STAND ________ ________ ________

1=Natural; 2=Planted; 3=Second growth; 9=Other.

MANAGEMENT TYPE ________ ________ ________

1=High forest; 2=Coppice with standards; 3=Coppice forest; 9=Other.

WILDERNESS ________

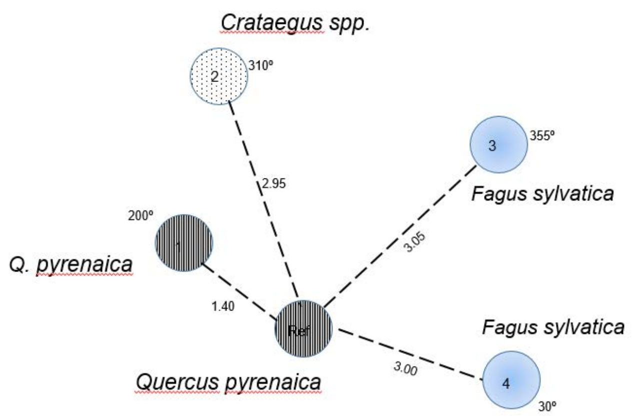
1=Primary forests; 2=Forest with assisted natural regeneration; 3=Forest plantation for production; 4=Forest plantation for conservation; 9=Others.

1=Seed; 2=Plantation; 3=Sprouts o suckers; 4=Combination of seed and sprouts; 5= Combination of seed and plantation; 6=Combination of plantation and strain stump (eucalyptus); 9=Other.

DISTRIBUTION 1

DISTRIBUTION 2

DISTRIBUTION 3



(example)

4. DENDROMETRY

Ratio ________

Nº	º	Dist. (m)	Spp.	Dbh (cm)	U	F	H.t.$_{base}$	H.t.$_{crown}$	H.t.$_{top}$	Dist.2	Crown	Partic	Health		
													Agent	I	E
1															
2															
3															
4															
5															
6															
7															
8															
9															
1 0															

Utility Class

1=Healthy tree, optimally shaped, without signs of old age, able to provide many valuable products, not dominated and with excellent long-term possibilities; 2= Healthy tree, vigorous, not dominated, without signs of old age, with some conformation defects and able to provide valuable products; 3= Tree not wholly healthy and vigorous, or a bit old or partly dominated, with many conformation defects, but still capable of providing some valuable products; 4= Diseased and weak tree or old, with many defects of conformation, only capable of providing secondary products, 5 = Tree very sick, weak or old, with poor conformation and scarce and of little value; 6 = Dead tree but not rotted and still able to provide some useful good; 9 = Other.

Form class

1=Fusiform stem, no branches, low taper, fine bark, round cross-section, more than 6 m long and dbh > 20 cm; 2= Fusiform stem, able for logging, no branches, longitude around 4 m; 3= Small fusiform stems, with dbh <75mm and lower than 4 m height, and belonging to one of the following species: 07, 12, 16, 23, 41-49, 55-57, 66, 67, 71, 72, 74, 75, 79 y 94; 5=Trees that their stem is either bent, damage or has too many branches. 6=Pruned trees with all their crown removed and belonging to one of the following species: 41-43, 55, 56, 71, 72 y 94; 9=Other.

Health-Agent: 100=No injuries observed; 200=Unknown causes; 300=Unknown biotic damage agents; 310=Fungus; 311=Insects; 312= Mistletoe and similar; 313=Epiphytes; 314=Wild animals; 315=Cattle; 316=Dominance; 320=Anthropic; 321=Logging; 322=Humans in general; 400= Unknown abiotic damage agents; 410=Snow; 411=Wind; 412=Drought; 413=Thunderbolt; 414=Frosts; 415=Hail; 421=Fire; 422=Rock fall; 423=Erosion; 900=Other.

Health-I: 1=Small; 2=Medium; 3=Big; 9=Other.

Health-E: 1=Bark; 2=Leafs; 3=Branches; 4=timber or stem; 5=Fruits; 6=Flowers; 7=Growing guide; 8=Crown; 9=All the tree; 900=Tree.

REGENERATION (R=10m)

Spp.	Type	Heights	Den.	Nº	H.m. (dm)

Type: 1=Sowing or seedling; 2=Plantation; 3=Basal shoots or root sprouts; 4=Unknown; 5=Uncertain; 6=Combined; 9=Other.
Heights: 1=Individuals<30 cm; 2=Between 30-130cm; 3=Height >130cm and $\emptyset_n$<2,5cm; 4=Height >130cm and $\emptyset_n$ between 2,5 and 7,5.
Den.: 1=Low. From 1 to 4 individuals in the plot; 2=Moderate. From 5 to 15 individuals; 3=High. More than 15 individuals; 9=Other.

5. SILVICULTURAL SYSTEM

REGENERATION FELLING ________ 0= Not observed; 1= Observed.

Type of regeneration felling ________

0=Not observed; 1=Clear cutting; 2=Group selection; 3=Shelterwood; 4=Uneven-aged system; 9=Others.

FOREST COVER IMPROVEMENT TREATMENTS ________

0=Not observed; 1=Weeding (grass-cutting, brush-cutting, brush-out, etc.); 2=Cleaning; 3=Thinning; 4=Pruning; 9=Others.

GROUND IMPROVEMENT TREATMENTS ________

0=Not observed; 1=Manual dibbling; 2=Mechanical dibbling; 3=Ripping; 4=Mounding; 5=Terracing; 9=Others.

6. DEAD WOOD (R=15m)

LD (log descomposition):
1 Bark intact, presence of twigs (<3 cm), intact texture.
2 Intact bark, absence of twigs, intact to partly soft texture.
3 Traces of bark, absence of twigs, hard texture, large pieces.
4 Bark absent, absence of twigs, small, soft blocky texture.
5 Bark absent, absence of twigs, soft and powdery texture..
6 Bark absent, absence of twigs and due to its descomposition level, hollows.
9 Still live, felled in a very short time.

	Species	LD	Diameter at breast height (cm)	Height (m)	F	Crown	Code
Standing dead wood							

	Species	LD	Diameter at breast height (cm)	Height (m)	F	Crown	Code
Downed dead wood							

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	Species	LD	Nº	Medium diameter (cm)	Average height (m)
Standing dead small trees					
Downed dead small tress					

	Species	LD	Maximum-diameter (cm)	Small diameter (cm)	Longitude (m)
Braches and logs					

	Species	LD	Diameter (cm)	Height (m)	Code
Stump d>7,5cm					

	Species	LD	Nº stumps	Diameter (cm)	Height (m)	Code
Strain stump d>7,5cm						

	Nº	Species	LD	Diameter (cm)	Average longitude (m)	Nº logs
Accumulations of branches and logs						

8. SOIL

STONINESS _______ 1=Without stones; 2=Low stony; 3=Stony; 4=Very stony; 5=Rubly; 9=Other.

TEXTURE _________ 1=Sandy. Impossible to make cylinders; 2=Loam. Possible to make thick cylinders; 3=Clay. Possible to make cylinders of 5 mm diameter; 9=Other.

WATER REGIME ______ 1=Dry; 2=Humid; 3=Permanently wet; 4=Waterlogged; 9=Other.

SOIL REACTION (pH) ________ Soil sample No: ________

Date of Collection ________ Time ________

SOIL TIPOLOGY ________

GROUND COVER PERCENTAGE (R=25m)

	Surface (%)		
Bedrock			
Stones			
Bare soil			
Organic matter			
Lichen and moss cover			
Fern cover			
Herbaceous plant cover			
Shrub			
Mulch			
Peat bog			
Seeds			
Waterlogged			
Pavement (human)			
Terraces			
Other infrastructures developed by humans			

The sum of the percentages must be = 100.

9. MICRO-SITES

	Observed
Accumulations of branches	
Hollowness (diameter>20cm)	
Anthill	
Mole burrow	
Burrows	
Caves	
Nests	
Others	

	Long (dm)				Wide (dm)			
Walls								
Bushes								
Roads								
Terraces								

Presence of cattle or apiculture ________

MH1= Horses; MH2=Cows; MH3=Sheeps; MH4=Goats; MH5=Pigs; MH6= Beehives/Bees; MH7=Others.

11. RISKS

SOIL EROSION

EROSION MANIFESTATIONS ________

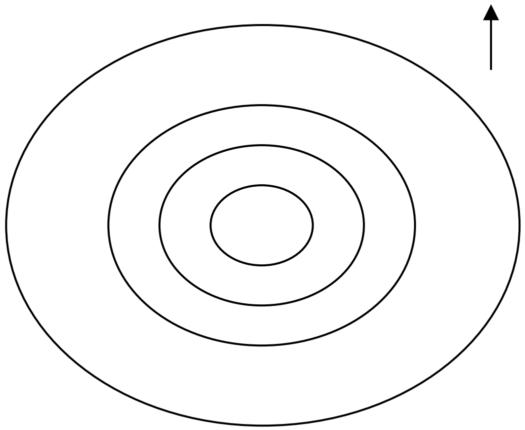
1=Not observed; 2=There is a small amount of erosion, exposed tree roots; 3=Presence of parallel gullies <20 cm depth; 4=V-Shaped gullies; 5=Stream bank erosion; 6=Mass movements; 9=Other.

FIRES

FUEL TYPE ________

LEAF LITTER, GRASSS, MOSS, AND LICHEN THICKNESS ________

12. PLOT OVERVIEW MAP



13. REFERENCE ITINERARY

DESCRIPTION --

LOCATION ________

ACCESS ________

ESTABLISHMENT ________

PRESENCE OF REGENERATION ________ 0= No regeneration observed; 1=Natural regeneration observed.

EFFECTIVENESS OF REGENERATION ________ 1=Low; 2=Normal; 3=High; 9=Other.

14. PLOT PHOTOGRAPHS

FOTO ID ________ PHOTO 1 ________ PHOTO 2 ________ PHOTO 3 ________ PHOTO 4 ________

15. OBSERVATIONS

Appendix III. Semi-structured questionnaire used in Chapter IV

Introduction

My name is Sara Guadilla and I am undertaken a research about the traditional management of forest commons inside and outside a protected area. My research project will culminate in the writing of a PhD thesis for the Institute for Environmental Sciences and Technology thesis programme of the Autonomous University of Barcelona. My research also collaborates with the Spanish National Inventory of Traditional Knowledge related to Biodiversity.

Within this study case, my aim is to understand the evolution of traditional uses that has taken place in forest commons of the Liébana valley. My aim is to collect information about local farming and forestry management practices such as timber harvesting and firewood collection, in particular how regulations to these practices have influenced on present use of natural resources from Liébana's forest commons.

All the information given during this interview will be managed confidentially. Only me will have access to the personal data, which will not be published, and the only reason for asking this information is for updates in the forthcoming years, if necessary. Information provided about the local practices will be used as part of the PhD thesis for understanding the impact of traditional management activities in current forest status, and conclusions will be sent to a scientific journal on the topic of study. This interview is expected to be one hour of duration.

General questions

Information about the informant:

1. Municipality and date of interview:
2. Name and age:
3. Educational level:
4. Occupation:

General script:

1. What is your opinion about the uses carried out by the neighbours of their surrounding forest commons?
2. Have these uses always been the same?
    a. Yes: What main changes, if any, have occurred? What most contributed to this change/ continuity? To whom/what do you think may benefit these changes?
    b. No: In which year did it change? How do you think it changed? How it has contributed to traditional uses loss? Would you like to keep any of the former uses?
3. Overall, what do you think are to most important ways to promote neighbours' use of their surrounding forest commons? Do you think it could engage more to local inhabitants in the preservation of the forests?
4. Do you make use of the products offered by your forest commons? Which ones?

Specific questions to livestock farmers:

Livestock farming

1. What type of livestock do you have? Which is the main breed? Which is the herd/flock size? Who is involved in livestock farming activities? How are they involved? Please describe. What is produced (e.g., milk, meat, cheese)? Does it graze in private fields or common meadows? Do you pay a fee to graze in common areas? Do you choose the grazing area? What criteria is applied? Please describe.

2. Does the livestock farming activities include pastoralism? Please provide as much detail as possible of the pastoral system: livestock density, grazing days, seasonal mobility, people involved, rangeland type. Does traditional pastoral system differ from nowadays livestock farming? If so, how has it change (range species, calendar, means of transportation to summer pasturelands)? Is transhumance still practiced? What about traditional lopping and pruning activities? Please describe (e.g., plant species, tools, customs, beliefs).

3. Are there any hindrances to traditional livestock farming practices (by whom, where, when, reason)? Please indicate a farming or forestry practice relate to livestock herding carried out in the past that is banned nowadays. Do you think illegal grazing arose because of the banning? Under what conditions do you consider that illegal grazing could be prevented (e.g., more licences, control)?

4. Is prescribed burning used to promote pastureland at the expense of shrub or forested lands? Is it used to prevent woody colonization of grasslands? Please describe traditional fire practices (e.g., number of people involved, suitable weather conditions, perimeter of the control line). How has the abandonment of traditional burning influenced on landscape change?

5. Does the proximity to Picos de Europa National Park affect livestock farming in this area (e.g., disturb animals grazing in upland summer pastures, food brand-new foundation, influence on species ecology of predators like wolves)? Do you think it brings visitors to this area? What is your opinion about the impact of visitors on the forest (e.g., have an adverse impact due to the presence of litter or a positive one by promoting the cleaning of paths)? Are visitors interested in buying agri-food products?

6. What is your opinion about the presence of livestock feeding on the forests? Would you say that livestock grazing is beneficial for preserving forest habitats? Why?

7. Finally, is there anything else you would like to include to help me to understand better how livestock farming in forest commons could be promoted?

Specific questions to loggers:

Timber harvesting

1. Which tree species do you harvest? When does the harvesting start and finish? What method of harvesting do you use? Who is involved? How are they involved? Please describe. What is the destination of the harvested timber (e.g., sawmill, local household construction)? Do you harvest from private fields or common areas? Do you pay a fee to harvest in forest commons? Do you choose the woodlot? What criteria is applied? How do you purchase a woodlot? Please describe.

2. Does timber harvesting technique applied in the past differ from nowadays? If so, how has it change (species, tools, means of transportation)? Please provide as much detail as possible of changes in any aspects of the practice. Are there hindrances at the time of harvesting (by whom, where, when, reason)? Can you indicate a practice involving the harvest of timber in the past that is banned nowadays? Is this area affected by illegal felling because of the banning? Under what conditions do you consider that illegal timber harvesting could be prevented (e.g., through more licences, control)?

3. Do you think timber harvesting has positively influenced on forest conservation? Does it impact on the landscape (e.g., in a decline/promotion of species diversity)? Is any silvicultural treatment (e.g., weeding, pruning, thinning) implemented in the stands? Who implements it? What is your opinion about the accessibility to the forests (e.g., good/poor trail layout)? Who participates in the maintenance of the unpaved paths? How are they maintained (e.g., mechanical clearing, livestock, burning)?

4. How does the proximity to Picos de Europa National Park affect timber harvesting in this area (e.g., considerations to reduce the visual impact of the cuttings, limitations to non-native species use)? Do you think it brings visitors to this area? What is your opinion about the impact of visitors on the forest (e.g., have an adverse impact due to the presence of litter or a positive one by promoting the cleaning of paths)? Are visitors interested in buying timber from any particular species?

5. Overall, would you say that timber harvesting is beneficial for preserving forest habitats? Why?

6. Finally, is there anything else you would like to include to help me to understand better how timber harvesting in forest commons could be promoted?

Specific questions to NTFP harvesters:

Non-Timber Forest Products

1. Tell me about the collection of plants for firewood, cork, fruits, or other non-timber forest products. Which species do you collect? When does the collection start and finish? How often do you gather these products? What method of harvesting do you use? Who is involved? How are they involved?

2. How much do you collect? What is the destination of the product collected (e.g., household use for cooking or heating, livestock fodder, human consumption, commercial uses)? Do you harvest from forest commons, common pastures, or private fields? Do you need a licence or pay a fee to collect these products (by/to whom, when)? Please describe.

3. Is there any regulation to the amount of product collected? For instance, in the gathering of firewood, what criteria (and by whom) is applied to choose the woody biomass to remove (e.g., dead wood, branches, twigs)? Does the technique applied in the past differ from nowadays? If so, how has it change (species, tools, limits to quantity gathered, location)? Please provide as much detail as possible of changes in any aspects of the practice.

4. Are there hindrances at the time of collecting NTFP (by whom, where, when, reason)? Can you indicate a practice involving NTFP collection in the past that is banned or no longer carry out nowadays? Is this area affected by illegal harvest of NTFP because of the banning? Under what conditions do you consider that illegal NTFP harvesting could be prevented (e.g., regulation)?

5. When going into the woods for collecting products, what is your opinion about the accessibility to these areas (e.g., easily to move around, well-maintained trail layout)? Who participates in the maintenance of the unpaved roads? Which method is applied to maintain those paths (e.g., mechanical clearing, livestock, burning)? Is any silvicultural treatment (e.g., weeding, pruning, thinning) implemented in the stands? Who implements it? Is any afforestation done afterwards to prevent the depletion of the resource?

6. How does the proximity to Picos de Europa National Park affect NTFP collection in this area (e.g., visual impact of biomass removal, gathering of products by visitors, promotion of certain plant species). What is your opinion about the impact of visitors on the forest (e.g., have an adverse impact due to the presence of litter or a positive one by promoting the cleaning of paths)? Are visitors interested in collecting or buying NTFP?

7. Overall, would you say that NTFP collection is beneficial for preserving forest habitats? Why?
8. Finally, is there anything else you would like to include to help me to understand better how NTFP collection in forest commons could be promoted?

Specific questions to hunters:

Hunting

1. What game species do you hunt? What method of hunting do you use? When does hunting season start and finish? Please describe further. What is the reason for hunting (e.g., recreational, food supply, wildlife population regulation)?

2. Who can hunt? Is it necessary to request a hunting license (by/to whom, when)? Where do you hunt (i.e., in private hunting reserves or forest commons)? Do you pay a fee to hunt in forest commons?

3. Does hunting system carried out in the past differ from nowadays? If so, how has it change (game species, hunting methods, habitat)? Please provide as much detail as possible of changes in any aspects of the practice. Is there any new regulation of the hunting practice (by whom, where, when, reason)? Is poaching frequent in the area? Under what conditions do you consider that poaching could be prevented (e.g., giving more hunting licences, control)?

4. When moving to the assigned location to stand hunting of wild boar or moving around the forest, what is your opinion about the accessibility to the forested areas (e.g., easily to move around, a well-maintained walking trail layout that progressively improves)? Who participates in the maintenance of the unpaved roads? Which method is applied to maintain those paths (e.g., mechanical clearing, livestock, burning)? What is your opinion about the impact of stand hunting (*batida*) in plant communities (e.g., brush cutting of unpaved paths and immediate surrounding areas of hunting locations)?

5. Does the proximity to Picos de Europa National Park affect hunting in this area (e.g., wildlife population number, bag limits on game, presence of predators like wolves)? What is your opinion about the impact of visitors on the forest (e.g., have an adverse impact due to the presence of litter or a positive one by promoting the cleaning of paths)? Are visitors interested in buying products obtained from hunting?

6. Overall, would you say that hunting is beneficial for preserving forest habitats? Why?

7. Finally, is there anything else you would like to include to help me to understand better how hunting in forest commons could be promoted?