UNIVERSIDADE DE LISBOA
FACULDADE DE CIÊNCIAS
DEPARTAMENTO DE BIOLOGIA ANIMAL

Galicio-Portuguese oak forest of *Quercus robur* and *Quercus pyrenaica*: biodiversity patterns and forest response to fire.

Vânia Andreia Malheiro Proença

Doutoramento em Biologia
Especialidade de Ecologia

2009

UNIVERSIDADE DE LISBOA
FACULDADE DE CIÊNCIAS
DEPARTAMENTO DE BIOLOGIA ANIMAL

Galicio-Portuguese oak forest of *Quercus robur* and *Quercus pyrenaica*: biodiversity patterns and forest response to fire.

Vânia Andreia Malheiro Proença

Tese orientada por:
Professor Doutor Luís Vicente
Doutor Henrique Miguel Pereira

Doutoramento em Biologia
Especialidade de Ecologia

2009

RESUMO

A história da floresta na Europa encontra-se intimamente ligada com a actividade humana. De um modo sumário, a floresta foi sendo sujeita a uma intensa desflorestação e transformação ao longo de milénios de ocupação humana. Portugal, não só não é excepção, como constitui um excelente exemplo da dinâmica entre a floresta e a sociedade. Originalmente, as florestas seriam dominadas por espécies de carvalhos (*Quercus* spp.) e cobririam grande parte do país. Com a ocupação humana, o território foi desflorestado para uso agrícola e para a obtenção de madeira. Actualmente, as florestas de carvalhos caducifólios representam 4% do território, enquanto que o pinheiro (*Pinus pinaster*) e o eucalipto (*Eucalyptus globulus*), plantados intensivamente durante o último século, dominam a floresta portuguesa. O êxodo rural que tem marcado as últimas seis décadas, tem conduzido à regeneração natural da floresta em campos abandonados, abrindo uma janela de oportunidade para o restablecimento dos carvalhais caducifólios, o que poderá vir a ser um contributo para o desenvolvimento de florestas multifuncionais. Considerando a longa história de perturbação a que as florestas naturais foram sujeitas, este trabalho teve por objectivo avaliar o seu valor para a conservação da biodiversidade e a sua resistência e resiliência à perturbação por fogo, que constitui um dos principais motores de alteração da floresta portuguesa. Estudaram-se os carvalhais Galaico-Portugueses de *Quercus robur* e *Quercus pyrenaica* que representam grande parte da floresta nativa a norte do Tejo. A contribuição destas florestas para a conservação de biodiversidade foi analisada em dois contextos: em comparação com plantações de pinhal e eucalipto e num contexto de paisagem rural. A resposta ao fogo foi analisada em comparação com florestas de pinhal após um incêndio de grandes dimensões. Relativamente à relevância para a conservação da biodiversidade os resultados mostram que os carvalhais suportam uma maior riqueza de espécies florestais, quer quando comparados com plantações, quer no contexto do mosaico de paisagem rural. Para além disso parecem ser o habitat preferencial, ou até único, para várias espécies. O valor de conservação dos carvalhais de maiores dimensões foi ainda detectado através da análise de relações espécies-área. Da análise da resposta ao fogo, os resultados sugerem que as florestas naturais de folhosas são mais resistentes e mais resilientes à pertubação pelo fogo do que as florestas de pinhal. Por fim, é discutido o papel que as florestas naturais poderão na gestão futura da floresta em Portugal.

Palavras-chave: Biodiversidade; Floresta natural; *Quercus pyrenaica*; *Quercus robur*; Resposta ao fogo.

ABSTRACT

The history of European forest is closely related to human activity in the territory. In brief, forests have been intensively exploited and transformed during millennia of human occupation. Portugal is no exception, representing an excelent case study of the dynamics between forest and society. In the beginning of the Holocene, oak forests (*Quercus* spp.) covered most of the country. Human settlements in the Neolithic marked the start of intense deforestation. Land was converted to agriculture and wood was overexploited. Presently, deciduous oak forests represent 4% of the Portuguese forest. On the other hand, pine (*Pinus pinaster)* and eucalypt (*Eucalytpus globulus*) are the dominant species after a century of intense forest plantation. The abandonement of agricultural areas, due to rural exodus in the last decades, has been promoting forest natural regeneration, and may represent a window of opportunity to restore natural forests and to support a transition to multifunctional forests. Considering the long history of pertubation that has affected natural forests, the objective of this dissertation was to assess the current value of natural forests for the conservation of biodiversity and their resistance and resilience to fire. This study was aimed at Galicio-Portuguese oak forests of *Quercus robur* and *Quercus pyrenaica* that are good representatives of natural forest in northern Portugal. The value of deciduous oak forest for biodiversity was investigated in two contexts: in comparison with pine and eucalypt plantations and in a countryside context. Response to fire was analysed in comparison with pine plantations after a large wildfire. With respect to the value of deciduous oak forests for biodiversity, results suggest that oak forests support more forest species than forest plantations and than other habitats in the multi-habitat framework. Moreover, oak forests seem to be the preferred habitat, or even the only habitat, for several species. The conservation value of large oak forest patches was also detected through the analysis of species-area relationships. Regarding forests response to fire, results suggested a lower vulnerability and higher resistance, in comparision to pine plantations, of these forests to fire disturbance. Finally, the potential role of deciduous oak forests in the future management of the Portuguese forest is discussed.

Key-words: Biodiversity; Forest response to fire; Natural forest; *Quercus pyrenaica*; *Quercus robur*.

AGRADECIMENTOS

Agradeço a todas as pessoas a seguir referidas por terem enriquecido o meu saber, sobre ciência, a vida, ou ambas, e ainda por:

Aos meus orientadores, Luís Vicente e Henrique Pereira por terem aceite a orientação da minha tese, por tudo quanto me ensinaram, pelos desafios, pelas oportunidades, e pela amizade.

Ao Henrique Pereira, pelo previlégio que é trabalhar e ir descobrindo a Peneda, e pela partilha e discussão de ideias em campo.

À Inês Gomes, Cibele Queiroz e João Guilherme, pela excelente colaboração ao longo da tese, em campo e em Lisboa, pela alegria e acima de tudo pela amizade.

À Ana Hasse, Filipa Filipe, Maria José Caramujo e Patrícia Rodrigues, pela companhia na sala de bolseiros e por toda a energia positiva.

A todos os colegas da sala de bolseiros, em especial ao José Pedro Amaral, Sofia Lourenço, Luís Costa, Filipe Ribeiro e José Pedro Granadeiro, por todo o ânimo ao longo da tese.

Ao Hélder Duarte, por ter animado o que teria sido um Verão interminável no laboratório.

À Cibele Queiroz, Inês Gomes, João Guilherme, Ana Hasse, José Pedro Amaral, Filipa Filipe, Maria José Caramujo, Méabh Boylan, Joaquim Sande Silva e Paulo Fernandes, pelas sugestões e comentários aos manuscritos que compõem esta tese.

Ao Carlos Teixeira, Catarina Gavinhos, Celina Pereira, Margarida Ferreira, Paulo Marques, colegas da EcoComp, da TBA e da ALTER-Net pela discussão de ideias.

Ao João Honardo pela disponibilidade e pela valiosa ajuda na identificação das plantas.

Ao Marcos Liberal, Duarte Silva, Miguel Pimenta, Armando Loureiro, Ana Fontes e Palhares do Parque Nacional da Peneda-Gerês pela ajuda em vários momentos da tese.

Ao Luís Crespo pela ajuda na identificação das aranhas.

À Yvone Cerqueira, Luisa Cardenete, Charo Garcia, Inma Jimenez e José Torres pela ajuda no trabalho de campo.

Aos meus amigos, à minha gente, por terem sido peças fundamentais neste processo, um abraço especial à Ana Rosendo e à Margarida Leal que estiveram sempre lá.

À minha família, por todo carinho e interesse constantes, em especial à Claúdia Tavares e ao Vasco Lopes que me apoiaram imenso.

Por fim, aos meus pais, Carlos e Ester, por tudo e por serem tudo.

TABLE OF CONTENTS

---

1 GENERAL INTRODUCTION

2 THE ROLE OF NATURAL FORESTS FOR BIODIVERSITY CONSERVATION IN THE NW OF THE IBERIAN PENINSULA

3 NATURAL OAK FORESTS AND BIODIVERSITY CONSERVATION IN A MULTI-HABITAT LANDSCAPE

4 FIRE SEVERITY AND POST-FIRE REGENERATION IN NATURAL BROADLEAVED FOREST AND PINE STANDS AFTER A WILDFIRE

5 ECOSYSTEM CHANGE, BIODIVERSITY LOSS AND HUMAN WELL BEING

GENERAL INTRODUCTION

The content of this section was partially based on Proença VM, Queiroz CF, Pereira HM, Araújo M. Biodiversidade. In: Pereira HM, Domingos T, Vicente L and V Proença (eds.) Ecossistemas e Bem-Estar Humano: Resultados da Avaliação do Milénio para Portugal. CELTA Editora. *In press*

1 General Introduction

1.1 Forests, Man and Biodiversity in Europe

Forests dynamics and the history of human societies have been linked since ancient times. This is particularly well illustrated by the relationship between forest and man in Europe. The history of forests in Europe has been shaped by periods of deforestation and periods of forest expansion, resultant from natural regeneration and forestation, which were associated with shifts in society dynamics and public attitude toward forests (McNeely 1994, Farrell et al. 2000). For example, by 900 BC in Ancient Greece forests were abundant and deforestation occurred without any regard towards forest sustainability, in fact forests were considered an obstacle to the expansion of agriculture and settlements (McNeely 1994, Farrell 2000). Five centuries later forest had become a scarce due to overexploitation and public attitudes changed: forest was no longer regarded as something inconvenient, it was strictly protected and deforestation was regretted (McNeely 1994, Farrell 2000). Later, the collapse of the Roman Empire by the fifth century gave way to a long period of forest expansion through regeneration and the experience of forest scarcity became lost from collective memory (Blondel and Aronson 1999, Farrel et al. 2000). With population growth and the expansion of farmland and settlements in the Middle Ages, forests were again subjected to a wave of deforestation (Blondel and Aronson 1999); 50% - 70% of forest cover in Europe was lost during this epoch (Shvidenko et al. 2005, McNeely 1994). The shortage of wood lead once more to a shift in social attitudes towards forest, forest benefits were valued and the importance of forest management acknowledged. By the eigthteenth century, forestry techniques were applied to optimize forests yield and new forests were intensively planted during the next centuries (Farrell et al. 2000, EEA 2008). This dynamic relation between forests condition and human societies was recurrent along the European history at different spatial and temporal scales (Bengtsson et al. 2000, Blondel and Aronson 1999), leading to generalized forest fragmentation (Wade et al. 2003).

Currently, only 5% of the European forests stay in an undisturbed state (more than a half in Sweden), the remaining 95% correspond to planted forests, including plantations of native and exotic species, and naturally regenerated forests (MCPFE 2007).

The dramatic loss of primary forest in Europe had inevitable consequences for biodiversity, including permanent changes in the composition of natural communities, such as the substitution of deciduous broadleaved forests by evergreen sclerophyllous forests and matorrals in the Mediterranean Basin (Naveh 1975, Blondel and Aronson 1999). In addition, most efforts spent in forest reestablishment after the eighteenth century were targeted to optimize forests yield and disregarded biodiversity conservation (EEA 2008). As the relevance of biodiversity and ecosystem services for human well-being becomes a consensual issue in governance and in society, the paradigm of productive forests is being replaced by the objective of developing sustainable forestry (Farrel et al. 2000, MCPFE 2007, EEA 2008). Sustainable forest management implies the use of forests as providers of multiple forest services, from wood products to soil protection, while maintaining biodiversity and ecosystem functions in the long term (MCPFE 2007). Primary and old-growth regenerated forests[1], besides being important reservoirs of biodiversity, may act as sources of forest species dispersion promoting the restoration of biodiversity in the landscape including in managed forests (Bengtsson et al. 2000, EEA 2006, Hermy and Verheyen 2007, Aubin et al. 2008).

---

[1] Primary forests and naturally regenerated forests may differ in structural features and in species composition (Aguiar and Pinto 2007, Hermy and Verheyen 2007). For example, naturally regenerated forests tend to lack decrepit trees, have a lower diversity in terms of trees age and a larger representation of pioneer species in their communities (Aguiar and Pinto 2007).

.

Definition of forest categories

The definition of what is a forest is not straightforward, varying with the geographic, social, economic and historic context (Shvidenko et al. 2005). According to FAO (2006) a forest ecosystem consists of an area dominated by trees higher than 5 m at maturity, with a canopy cover superior to 10% and occupying a minimum of 0.5 ha. This definition does not include forested areas which are predominantly under agricultural or urban land use, such as orchards or gardens (FAO 2006).

Forests can be classified in five classes (FAO 2006): primary forests ("forests of native species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed"), modified natural forests ("forests of naturally regenerated native species where there are clearly visible indications of human activities"), semi-natural forests ("forests comprising native species, established through planting, seeding or assisted natural regeneration"), protective forest plantations and productive forest plantations ("forests of introduced species, and in some cases native species, established through planting or seeding" for either the provision of ecosystem services or for productive purposes). Forest plantations also include forest dominated by native species, but characterized by a low species richness and low structural diversity.

For the purpose of this study two categories were used: natural forests and forest plantations, with natural forests comprising primary forests, modified natural forests and semi-natural forests. Primary oak forests are practically absent in Portugal, due to the long history of human activity in the Iberian Peninsula (ICN 2006), and semi-natural oak forests are not common in the study area. Therefore most of natural forests analysed were modified natural forests.

1.2 The Portuguese forest

Biogeographical and climatic context

The Portuguese continental territory is located in the Iberian Peninsula (South-western Europe) in the transition between two biogeographical regions, the Atlantic sub-region of the Eurosiberian region and the Mediterranean region (EEA 2003) (Figure 1.1a), featuring a diverse flora (ICN 1998). The climate is temperate and the country may be divided in two zones according to Knopp's classification (Cantelaube et al. 2002, INM 2009). An Oceanic climate zone characterized by rainy winters and dry and mild summers, and Mediterranean climate zone characterized by rainy winters and dry and hot summers (Figure 1.1b).

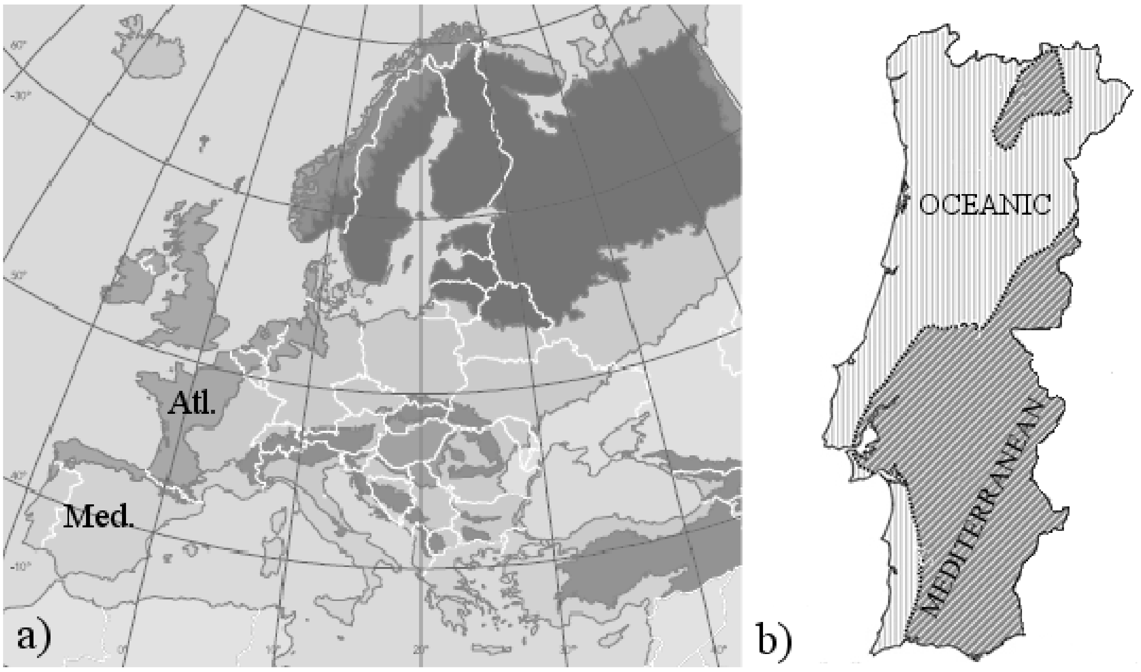


Figure 1.1 – Biogeographical and climatic context of Portugal. Biogeographic regions (a): Atlantic (Atl.) and Mediterranean (Med.) (adapted from EEA 2003). Climatic zones (b): Oceanic climate and Mediterranean climate (adapted from INM 2009).

A brief historical report of the Portuguese forest

After the Holocene (ca. 10.000 year ago) vegetation cover in the Portuguese territory suffered a gradual shift from boreal forests and steppe habitats, which dominated during the last glacial period (Wurm), to more temperate forests of broadleaved species (Ramil-Rego 1998, Castro et al. 2001, Aguiar et al. in press). Broadleaved trees in western Iberia had probably persisted during glacial times in refugia in the Cantabric and Atlantic coasts (Galicia, Asturias, Beira Litoral) (Castro et al. 2001).

Two millennia after the beginning of Holocene, broadleaved forest was the main type of land cover in Portugal occurring from the lower coastal regions up to mountain areas. Deciduous oaks predominated to the north of Tagus river, the common oak (*Quercus robur*) in areas of oceanic influence at lower altitudes and up to 1000 m – 1200 m and the Pyrenean oak (*Quercus pyrenaica*) in more elevated areas up to 1600 m and in the interior mountains; the Portuguese oak (*Quercus faginea*) occurred in the transition between the Atlantic and the Mediterranean zones; in the south of Tagus where the climate was Mediterranean the forest was dominated by perennial oaks, the cork oak (*Quercus suber*) and the holm oak (*Quercus rotundifolia*) (Caldas 1998). Forest cover was nevertheless absent in higher altitudes, where shrublands and pastures dominated (Aguiar et al. in press, Ribeiro et al. 1988), in sandy soils of costal areas and in other locations where the microclimatic conditions or the soil composition did not allow the development of a tree canopy (Blondel and Aronson 1999).

Palynologic profiles from the northwest of Portugal suggest that forest dominance lasted from 8000 BP to 3000 BP, after that period there was a progressive decline in forest cover probably due to the intensification of human activity (Ramil-Rego et al. 1998). However, as in the rest of Europe, human activity in the Neolithic (ca. 6000 BP) marked the start of intensive forest loss and degradation that conduced to dramatic changes in land cover. The recurrent use of fire to create and maintain pasturelands and agricultural fields, as well as forest exploitation for timber and fuel, were the main causes of forest loss (Aguiar and Maravalhas 2003; Aguiar et al. in press).

Forest statistics from the late nineteenth century reveal that agriculture and uncultivated land were by that time the principal categories of land cover, occupying about 4 600 000 ha and 4 200 000 ha respectively, forests covered only 7% of the territory (670 000 ha) (National Plan for Forest Defense Against Fire (PNDFCI) – Annex D, Resolution of the Ministers Council nº 65/2006). Perennial oak forests (montados) and pine plantations were the principal types of forest, occupying respectively 370 000 ha and 210 000 ha, in contrast deciduous oaks and chestnuts occupied, as single category, 50 000 ha (PNDFCI – Annex D, Resolution of the Ministers Council nº 65/2006).

From early to mid twentieth century (1900 - 1950), the area covered by agriculture and forests kept increasing, as uncultivated land that was fit for agriculture and forestry was actively occupied (Box 1.1). Rural exodus after the 1950s caused a shift in land cover composition, agricultural fields decreased due to abandonment and shrublands increased after natural regeneration in abandoned areas (PNDFCI – Annex D, Resolution of the Ministers Council nº 65/2006). Forests cover has also been increasing in the last sixty years, due to natural regeneration in unmanaged land but most of all due to intensive forestation of pine and eucalypt (*Eucalyptus globulus)* (EC 2004, Radich and Alves 2000, Mendes and Dias 2002) (Box 1.1).

Box 1.1 Major initiatives that shaped Portuguese forest during the last century:

Campanha do Trigo (1930 to 1950)
Extensive areas of montado were deforested and transformed in agricultural fields for cereals production.

Plano de Povoamento Florestal (1935 to 1972)
Intensive afforestation (318 000 ha) of communal lands, mostly with pine.

Projecto Florestal Português (1981 to 1987)
Funded by the World Bank this forestry plan was designed to meet the increasing demand for pulp and timber. More than 60 000 ha of private and communal lands were planted with pine and eucalypt.

Programa de Acção Florestal (1986 to 1997) and *Plano de Desenvolvimento Florestal* (1994-1999)
Both programmes were co-funded by EU, and promoted the plantation of new areas of cork oak but also of pine. These programmes funded the plantation or stand improvement of about 200 000 ha of pine forest.

Current composition

The current distribution of forest in native habitat is greatly fragmented to the north of Tagus (Figure 1.2a). Most broadleaved forest patches occur in this region, particularly in Oceanic climate zones (Pereira et al. 2002). The forests of *Querci* and/or *Betula* of the Galicio-Portuguese mountains and of the western Beira-Duriense mountains constitute good examples of natural forests, as well as the forests of *Quercus pyrenaica* and *Quercus rotundifolia* in the more steeper slopes of the remaining northern mountains (Aguiar et al. in press). The southern half of the country is still dominated by perennial oaks, cork oak and holm oak, which constitute the principal type of natural forest in this region (Figure 1.2a). The remaining forest is mainly constituted by plantations of pine and eucalypt (Figure 1.2b). Pine forest was until a decade ago the forest species with the largest area of occupation in Portugal (971 000 ha, DGF 2001). According to the most recent forest inventory, cork oak forests have now the largest cover area (737 000 ha), being followed by pine (711 000 ha) and eucalypt

(647 000 ha) (DGRF 2007)[2]. It should be noted that these three species have a direct economic value for timber and cork production, which may have been the principal reason explaining the efforts on their plantation and maintenance (Radich and Alves 2000). Deciduous oak forest represent less than 4% of the Portuguese forest, with a total cover area of 118 000 ha (DGRF 2007).

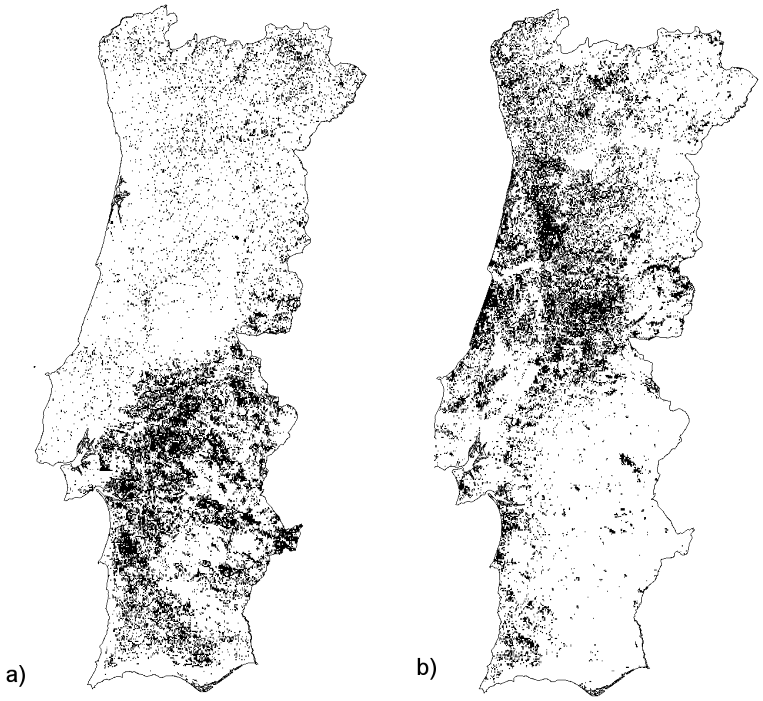


Figure 1.2 – Forest cover in Portugal. (a) Forest in native habitat - cork oak, holm oak, other oak species, chestnut and other broadleaved species. (b) Forest plantations - maritime pine, stone pine (*Pinus pinea*), other conifers and eucalypt. Although maritime pine and stone pine are native species their present distribution results from plantation and does not correspond to native distribution (ICN 2006). The categories used in these maps follow the Third National Forest Inventory (DGF 2001).

---

[2] These values respect to adult populations. However, if data on young populations are included the eucalypt raises to the top position as the species with the largest forest cover (829 600 ha), followed by pine (784 800 ha) and cork oak (751 600 ha) (Silva et al. 2008).

1.3 The Galicio-Portuguese oak forests

Distribution and abundance

The Galicio-Portuguese oak forests are dominated by *Quercus robur* and/or *Quercus pyrenaica* and constitute climacic habitats in their entire area of distribution which includes France (Eurosiberian region), Spain and Portugal (Eurosiberian and Mediterranean region) (ICN 2006). In Portugal they are distributed in the north of Mondego in the Cantabrio-Atlantic Province and above the 600 m in the Carpetano-Iberico-Leonesa Province, in Alto Alentejo in areas above 450 m in Toledano-Tagano sector and in the rainy areas (e.g., in the north of Sintra Mountains) of Gaditano-Onubo-Algarvia Province (ICN 2006).

As discussed above the Galicio-Portuguese forests were formerly abundant in the Portuguese territory but their area of occupation was severely reduced by human activity. Presently their abundance is slowly increasing due to natural regeneration in abandoned agricultural fields (see section 1.4) (ICN 2006).

Legal protection

The Galicio-Portuguese oak forest is under the protection of the Habitats Directive (Council Directive 92/43/CEE), being listed in Annex I under the designation of ("Galicio-Portuguese oak woods with *Quercus robur* and *Quercus pyrenaica*", habitat code 9230). This Annex I consists is a list of natural habitats acknowledged by their ecologic importance at the European scale and whose conservation requires the designation of special areas of conservation.

The Habitats Directive also protects two other types of deciduous oak forests found in Portugal, the Iberian oak forests of *Quercus faginea* and *Quercus canariensis* (habitat code 9240) and the riparian mixed forests of *Quercus robur, Ulmus laevis* and *Ulmus minor* (habitat code 91F0).

All these forests are protected at the national level. Portugal transposed the Habitats Directive to the national legislation in the Decree of Law nº 140/99 which has the objective of conserving biodiversity through the conservation and reestablishment of natural habitats and wild species of flora and fauna, while considering the economic, social and cultural needs at national, regional and local levels.

Forest ecology and floristic composition

Deciduous oak forests are characteristic of the coline and montane belts of the Iberian Peninsula (Castro et al. 2001). Because these forests were object of severe deforestation, due to their presence in areas with special interest for farming and pastures and to the high quality of the wood, the current knowledge about their original structure and natural communities is still incomplete (Castro et al. 2001). The Galicio-Portuguese oak forests are often located in oligotrophic soils in mountain slopes, and their communities are usually composed of acidophilic plants (Castro et al. 2001, ICN 2006).

Mature forests are characterized by shade tolerant trees with a slow growth rate and dense wood tissues (ICN 2006). The canopy presents a closed structure, with nearly 100% cover, creating a shadowy and wet interior environment with small temperature variations (ICN 2006). Although the canopy is dominated by *Quercus robur* and/ or *Quercus pyrenaica* (Box 1.2), the tree layer is composed by several species (Castro et al. 2001, ICN 2006).

Floristic communities are conditioned by soil's oligotrophy, and contain a large representation of siliceous species, such as *Melampyrum pratense, Teucrium scorodonia, Lathyrus montanus, Holcus mollis* and also the ferns *Blechunm spicant* and *Pteridium aquilinum* (Castro et al. 2001, Honrado 2003). Moreover, there is also an important presence of nemoral species, such as *Euphorbia dulcis, Anemone trifolia albida, Stellaria holostea*, and ferns from the genus *Dryopteris* (Castro et al. 2001, Honrado 2003). Finally, these forests also feature some endemic taxa of the northwest of the Iberian Peninsula, such as *Omphalodes nitida, Saxifraga spathularis* and *Anemone trifolia albida* (Castro et al. 2001, Honrado 2003). Phytosociologically these communities are part of the *Quercenion pyrenaicae* alliance (Honrado 2003, ICN 2006).

Box 1.2 Characterization of *Quercus robur* and *Q. pyrenaica* (Franco 1971, Carvalho et al. 2007a,b).

The genus *Quercus* includes about 450 species distributed throughout the temperate regions of the northern hemisphere and reaching the tropical montane forests of Central America and of Indomalaya, in Africa occurs in the north in the Mediterranean Basin (Jones 1959).

Quercus robur

Common oaks can growth up to 45m. The canopy is more or less regular with a round shape. The leaves are deciduous with round lobes. Both leaves and twigs are hairless. Trees are monoecious with unisexual flowers. Male flowers are disposed in linear catkins and female flowers are disposed in groups of 1 to 5 along a peduncle. Flowering is between April and May, but male and female flowers in the same tree flower at different times to avoid self-fertilization. Pollination is anemophilous. The fruit is an acorn and it matures in autumn. For distribution see Figure 1.3

Quercus pyrenaica

Pyrenean oaks can growth up to 20m and up to 30m in managed forests. The canopy is irregular and ovoid with ascendant branching. The leaves are deciduous, with deep narrow lobes and a woolly beneath. Twigs are also woolly. Trees are monoecious with unisexual flowers. Flower and fruit morphology similar to *Quercus robur*. Flowering of male and female flowers is simultaneous and pollination is anemophilous. Fruit matures in autumn. For distribution see Figure 1.3

Figure 1.3 – Distribution of *Quercus robur* and *Quercus pyrenaica* (adapted from Castro et al. 2001)

Biodiversity and ecosystem services

Deciduous oak forests provide an important habitat for many species of flora, fauna and fungi (Carvalho et al. 2007a,b). Some examples include the roe deer (*Capreolus capreolus*) that shows a strong preference for oak forests and has been increasing in density due to natural forest regeneration, the great spotted woodpecker (*Dendrocopus major*) that depends of oak forests for feeding nesting and shelter, the pine marten (*Martes martes*) that shows a preference for oak forests, probably due to the availability of preys, and the iberian wolf (*Canis lupus signatus*) that use oak forests for shelter and reproduction (Castro et al. 2001, Carvalho et al. 2007a). Moreover there are species with a high conservation value (e.g., *Lucanus cervus*, a xilophagus beetle) whose presence depends of snags and decrepit trees found in old growth oak forests (ICN 2006, Carvalho et al. 2007b).

At the landscape level, and particularly in the past, mature forests were shaped by the action of natural disturbance agents, namely herbivores and extreme weather events, contributing for the maintenance of a diverse landscape mosaic composed by habitats that were associated and/or dependent of oak forests, such as meadows and floristic communities of fringes and clearings in forests (ICN 2006).

Regarding ecosystem services, deciduous oak forests are particularly important due to their multifunctionality (Box 1.3). However, it should be noted, that because most oak deciduous forests are located in protected areas, their benefits for human society are mainly related with conservation and protection values, and their contribution for the most profitable provisioning service, timber production, is currently marginal. Nevertheless oak wood is acknowledged by its high quality and has a high commercial value (Carvalho et al. 2007b,c).

Box 1.3 - Ecosystem services provided by deciduous oak forests (ICN 2006; Carvalho et al. 2007b,c). The classification follows the categories proposed in the MA (2005).

Provisioning services

- Timber, fuelwood (wood products)
- Medicinal and aromatic herbs, mushrooms, pastures and fodder (non-wood products)
- Clean water supply
- Genetic resources

Regulating services

- CO2 sequestration
- Climatic regulation
- Water cycle regulation
- Natural hazard regulation (e-g- fire prevention)
- Soil protection from erosion

Cultural services

- Recreation (ecotourism, outdoor activities)
- Aesthetical and spiritual experience (scenic landscapes)
- Cultural heritage
- Science and education activities

Supporting services

- Nutrient and water cycling
- Soil formation
- Decomposition
- Primary production
- Wildlife habitat

1.4 Main drivers affecting the distribution and condition of deciduous oak forest.

Forest exploitation and forest conversion to agricultural land were the main factors affecting the condition of deciduous oak forests until mid twentieth century. Presently, although land use change continues to be an important driver its effects are now related with forest plantations and rural abandonment. Fire is also an important driver affecting both the distribution and condition of deciduous oak forests.

Land use change and fire are closely related. The abandonment of agricultural fields and the plantation of forests using fire prone species have been pointed as the principal causes contributing to increased fire risk in the last decades (Moreira et al. 2001a, Pausas et al. 2004).

Forest plantations

Portugal is one of the countries in the world with the largest annual gain of forest (FAO 2006). This was due to an increase in forest plantations. For example, between 1974 and 2001, eucalypt forests increased in 174%, representing 21% of the Portuguese forest in 2001 (DGF 2001).

The environmental impacts of plantations have been a matter of discussion, in particular issues related with ecosystem functioning and biodiversity (Onofre 1990, Abelho and Graça 1996, Madeira et al. 2002). Those impacts, such as the reduction of local diversity (Abelho and Graça 1996), are promoted by current options of forest management. Forest management is still focused on wood production encouraging the plantation of large and dense monospecific forests (Alves et al. 2007). The inadequate planning and management of plantations has also contributed to increase fire risk. Pine and eucalypt forests are highly flammable being a cause for the occurrence of large and severe fires that among other impacts lead to the degradation of soils (Doerr et al. 2000). The increase in the area covered with plantations of pine and eucalypt may affect the distribution and condition of deciduous oak forest, either because plantations replace oak forests or occupy land where deciduous oak forest could regenerate or be planted, or because fire compromises the possibility of future forest regeneration or causes the degradation of existing forests.

Rural abandonment

Rural abandonment is an increasing phenomenon in Europe, with mountain areas being the most affected (Prieler et al. 1998, MacDonald 2000). In Portugal, for example, 80% of the municipalities under the risk of severe rural abandonment are located in mountain areas

(Alves et al. 2003). Concurrently, the majority of deciduous oak forest patches are also found in these areas.

While rural abandonment may have negative consequences for the maintenance of species that use open areas and benefit from traditional farming habitats (Bignal and McCracken 1996, Moreira et al. 2001b), it also opens the way to natural regeneration and the reestablishment of native forest and associated communities (Bernaldez 1991, Green et al. 2005). However, the development of shrubs in abandoned land increases fires risk through the accumulation of flammable fuels and the occurrence of fires may inhibit natural succession by keeping communities in early succession stages (Blondel and Aronson 1999). Therefore, the successful reestablishment of native forest by natural regeneration will depend on the control of fire severity during early succession stages and the accompaniment of communities' succession to later stages, eventually reaching the climacic condition.

Fire

Fire is one of the main causes of deciduous oak forest degradation (ICN 2006). Although adult trees tend to resist to fire disturbance, understory communities may suffer compositional changes causing the loss of biodiversity (ICN 2006). Moreover, fire also affects other components of the ecosystem, such as soil structure and nutrients balance (Honrado 2003, Carvalho et al. 2007c).

1.5 The Peneda-Gerês National Park

Some of the best examples, in terms of size and condition, of natural deciduous oak forest in Portugal are found in the Peneda-Gerês National Park (Figure 1.4)

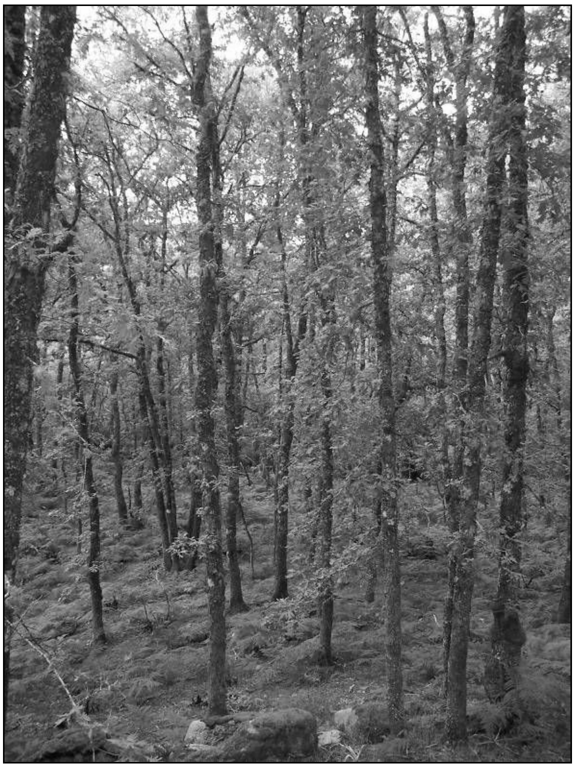


Figure 1.4 Deciduous oak forest in the Peneda-Gerês National Park

The Peneda-Gerês National Park (hereafter National Park) is located in the north of Portugal (between 41º41'N and 42º 05´ N and 7º 53'W and 8º 25' W) on the western limit of the Cantabric mountains and in the proximity of the Atlantic coast (Figure 1.5). The National Park was created in 1971 (Decree of Law nº 187/71, 8th of May) due to the natural and cultural assets found in this region. The main objective was to enhance cultural, educational and scientific values through the conservation of soils, water, flora, fauna and landscapes. In 1997 it was included in the "Natura2000" network (Site code PTCON0001 - "Serras da Peneda e Gerês") and in 1999 was designated as a Special Protection Area for Wild Birds. Moreover it also encompasses an important area of natural forest, "Mata de Palheiros – Albergaria" that integrates the European Network of Biogenetic Reserves, and is recognized as a National Park by the International Union for Conservation of Nature. In 2007 was accepted in the PAN Parks network (http://www.panparks.org) which certifies the quality of protected areas according to rigorous criteria of nature conservation, cultural services and sustainability.

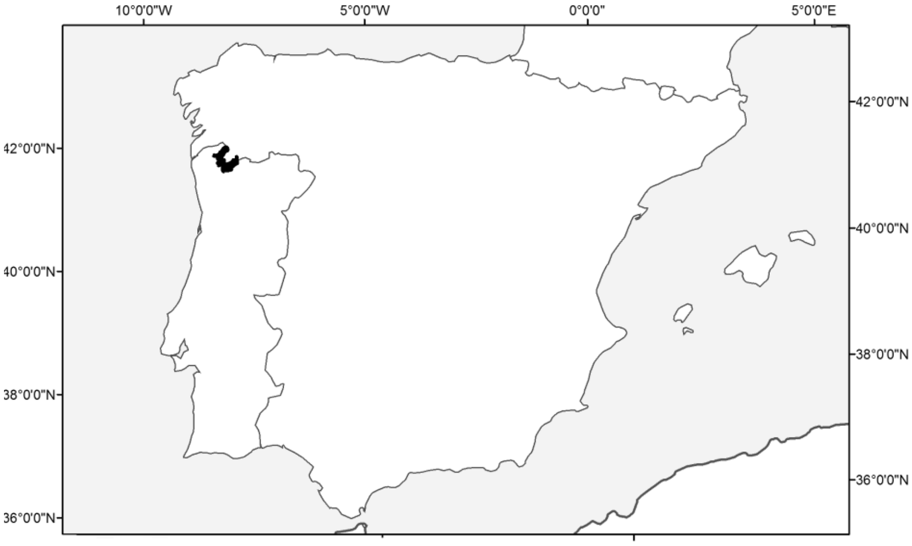


Figure 1.5 - Geographic location of the Peneda-Gerês National Park

Physiography and hydrography

The National Park covers an area of approximately 70 000 ha. The topography is complex being composed by the mountains of Peneda (1340 m), Soajo (1430 m), Amarela (1350 m) and Gerês (1545 m), the plateaux of Castro Laboreiro (1340m) on the northwest and Mourela (1380) on the northeast. Amarela and Soajo mountains are separated by the valey of river Lima and the south border of the national Park is determined by the valey of river Cavado (Figure 1.6).

The hydrographic network of the National Park is composed by several rivers that integrate three main river basins, the Minho basin in the north (2% of the National Park area), the Lima basin in the centre (47.8%) and the Cavado basin in the south (50.2%) (Honrado

2003). Geologically, this region is mainly composed of Hercynian granite outcrops and small strips of shale (Honrado 2003).

Figure 1.6 - Physiography of the Peneda-Gerês National Park (Honrado 2003).

Soils

In general, the soils in the National Park are permeable, have a light to medium texture and present a degraded superior horizon due to long history of agricultural practices and climatic conditions (PNPG 1995). Soils in mountain slopes are currently in an immature condition after centuries of soil erosion and lixiviation due to vegetation destruction (PNPG 1995). The majority of deep soils are now found in flat areas where soil particles have sedimented (PNPG 1995).

Climate

The climate in the National Park is greatly influenced by its topography. The mountains exert a barrier effect to the passage of hot and wet air masses coming from the Atlantic Ocean, which is the reason for much precipitation throughout the year (PNPG 1995). Moreover, the complex topography, with different slope aspects and altitudes, also contributes to a diversity of microclimatic conditions (PNPG 1995).

The mean annual precipitation ranges between 1600 mm in the Mourela plateau and 3000 mm in some areas of the Amarela and Gerês mountains (Honrado 2003). The mean temperature in the region varies between 10 ºC and 16 ºC, reaching absolute values of -14 ºC in the winter (data collected in Lamas de Mouro) and 40 ºC in the summer (data collected in Arcos de Valdevez) (PNPG 1995, Honrado 2003). The mean humidity varies between 75% and 85% (PNPG 1995).

Natural values

The biogeographic setting of the National Park, in the transition between the Eurosiberian and the Mediterranean regions, the complex topography and the diversity of microclimates have all contributed to a large diversity of species and natural communities (Honrado 2003).

The floristic diversity of the National Park includes 823 vascular taxa that occur in 128 types of natural vegetation (Honrado 2003). This large diversity of natural vegetation types may be summarized in seven major groups: (1) forests and matorral pre-forests, (2) grassland vegetation, not meadow, of fringes and clearings in forests, (3) shrubland and helophytic matorrals, (4) meadows and pioneer vegetation of leptosoils, (5) herbaceous hygrophilous vegetation, (6) riparian and epiphytic vegetation, (7) synanthropic nitrophilous vegetation.

Shrublands are the main type of vegetation cover in the National Park, covering 74% of the territory (Gomes 2008). Oak forests cover less than 10% of the territory, and present a fragmented distribution (Gomes 2008).

The National Park is also characterized by a diverse fauna, which encompasses 235 species of vertebrate, 204 of them under national or international protection and 71 being included in the Portuguese Red List of threatened species (ICNB 2009).

Human activity and land-use changes in the National Park

The first archaeological signs of human settlement in the National Park date back to 6000 BP and are found in the plateau of Castro Laboreiro (PNPG 1995, Honrado 2003). Human activities at that time consisted of animal husbandry and incipient agriculture (Honrado 2003). Archaeological signs are supported by palynologic profiles that show a strong decrease in forest cover (Zapata et al. 1995, Ramil-Rego 1998). The deforestation of the plateaux prior to the valleys is not a common pattern. In Central Europe deforestation of mountain areas followed an inverse pattern, with valleys being occupied first (Blondel and Aronson 1999, Aguiar and Pinto 2007).

The use of fire in plateaux and mountain slopes caused the flux of soil and nutrients to valleys increasing the fertility of these areas while leaving the soils in upslope degraded (Aguiar and Pinto 2007). This change in the distribution of soil fertility in the landscape, combined with the development of agricultural practices and agricultural implements made people move to valleys 3000 years ago, triggering the deforestation of valleys (Aguiar and Pinto 2007). The occupation of valleys not only conduced to the deforestation of lowlands but also to the continuous burning of mountain slopes, which were sacrificed to produce wood ashes for soil fertilization (Aguiar and Pinto 2007). During the Roman occupation agriculture suffered a great expansion, causing the loss of more forest (Honrado 2003).

The reoccupation of mountain areas started in the twelfth century and it was intensified in the sixteenth century with the introduction of maize, bean and potato from the Americas (Honrado 2003). Agricultural fields occupied former pastures, and these were displaced to more elevated areas (Honrado 2003). Additionally, forest patches were kept next to villages and agricultural fields for collection of fuel and non-wood products (e.g., medicinal herbs). This pattern of land use shaped the landscape in a mosaic of fields, pastures and forest, and it was maintained until the beginning of twentieth century (Honrado 2003).

The forestation of uncultivated lands, imposed by the government in 1935, had negative consequences for rural livelihood, namely by reducing available pastures, and contributed to rural exodus and to an intense depopulation after the 1950s (Lima 1996, Honrado 2003). Land abandonment has opened the way to natural forest regeneration and to the reestablishment of oak forests in their former land.

1.6 Objectives and outline of the dissertation

Dissertation objectives

After millennia of deforestation and land-use change and more recently of intensive forest plantation, the current pattern of natural regeneration of oak forests opens a window of opportunity to the reestablishment of natural forests and may also promote the transition to multifunctional forest ecosystems. However, natural regenerated forests, even if old growth, are distinct from primary forests because secondary forests may lack structural features or species important to maintain ecosystem stability and their biotic communities may be changed. Therefore it is important to assess the current performance of these forests in terms of biodiversity and resistance and resilience to disturbance.

Considering this context and the current main drivers, some questions have been formulated:

1) Does the diversity of forest species in forest habitats responds to forest naturalness[3]?

2) Does the relative diversity of forest species in natural forests and planted forests varies with the scale of analysis?

3) Does the diversity of forest species in oak forest responds to patch size and shape?

4) What is the contribution of oak forest patches for species diversity at the landscape scale?

5) How resistant are natural forests to fire disturbance?

6) How resilient to fire disturbance are the communities in natural forest?

The study was conducted in the Peneda-Gerês National Park that combines a long history of land cover change with some of the best examples, in terms of size and condition, of natural deciduous oak forest.

Dissertation outline

The present chapter, General Introduction, consists in a brief introduction to the history and dynamics of forests at three scales, European, national and local (The Peneda-Gerês National Park). It also provides information on the main characteristics of Galicio-Portuguese oak forests and of the study area. This chapter was partially based on a chapter accepted for publication that will integrate the book with the results of the Portuguese Assessment of the Millennium Ecosystem Assessment. Book chapter reference: Proença VM, Queiroz CF, Pereira HM, Araújo M. Biodiversidade. In: Pereira HM, Domingos T, Vicente L. and V. Proença (eds.) Ecossistemas e Bem-Estar Humano: Resultados da Avaliação do Milénio para Portugal. CELTA Editora. In press.

---

[3] Naturalness is defined as the degree to which an area is free of human influence, including the introduction of exotic species (Boteva et al. 2004).

The second chapter consists in a research manuscript that investigates the role of natural oak forests for biodiversity conservation. It compares local and regional patterns of diversity of forest and non-forest species of plants and birds in natural oak forest patches and in plantations of pine and eucalypt, addressing questions 1, 2 and 3. This manuscript was submitted for publication in Acta Oecologica. Manuscript reference: Proença VM, Pereira HM, Vicente L. The role of natural forests for biodiversity conservation in the NW of the Iberian Peninsula. Submitted to Acta Oecologica (in review).

The third chapter consists in a research manuscript on biodiversity patterns in a countryside landscape, composed of oak forest, agricultural land and shrubland. It compares the relevance of each habitat for the maintenance of biodiversity at the landscape scale, and tests the countryside species-area relationship (Pereira and Daily 2006), addressing question 4. This manuscript is under preparation. Manuscript reference: Proença VM, Pereira HM, Vicente L. Natural oak forest patches and biodiversity conservation in a multi-habitat landscape (in prep.)

The fourth chapter consists in a research manuscript that analyses fire severity and post-fire regeneration in natural broadleaved forests and pine plantations after a large summer wildfire that burned more than 4000ha in the National Park in 2006. This fire event created a unique opportunity to assess the effects of wildfire in these two types of forest. Because it was a single fire event and the study sites were geographically close, differences in forest responses were expected to be mainly due to forest features. Moreover, this study responds to a lack of empirical data about the effects of wildfires, as most studies are conducted in the context of prescribed burning. This study addresses questions 5 and 6. The manuscript was submitted for publication in Landscape Ecology. Manuscript reference: Proença VM, Pereira

HM, Vicente L. Response of natural broadleaved forest and pine plantations to a wildfire: fire severity and post-fire regeneration. Submitted to Landscape Ecology (in review).

The fifht chapter integrates the concepts of biodiversity, ecosystem services and human well-being and provides a discussion about the feedback loop between human well-being and biodiversity using the evolution of the Portuguese forest as a case study. This manuscript will integrate the Encyclopedia of Environmental Health to be published in 2010. Manuscript reference: Proença VM, Pereira HM. Ecosystem changes, biodiversity loss and human well-being. In: Encyclopedia of Environmental Health. Elsevier Press. In press.

The last chapter presents the main results of the study and integrates them in a discussion about the present and future condition of the Portuguese forest.

References

Abelho M, Graça MAS (1996) Effects of eucalyptus afforestation on leaf litter dynamics and macroinvertebrates community structure of streams in central Portugal. Hydrobiologia (Historical Archive) 324:195-204.

Aguiar C, Pinto B (2007) Paleo-história e história antiga das florestas de Portugal continental até à Idade Média. In: Silva JS (ed.). Floresta e Sociedade. Uma história em comum. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Aguiar C, Rodrigues O, Azevedo J, Domingos T (in press) Montanha. In: Pereira HM, Domingos T, Vicente L and V Proença (eds.) Ecossistemas e Bem-Estar Humano: Resultados da Avaliação do Milénio para Portugal. CELTA Editora.

Aguiar C, Maravalhas E (2003) A ecologia das borboletas. In: Maravalhas E (ed) As Borboletas de Portugal. Porto.

Alves AL, Carvalho NS, Silveira SCB, Marques JP, Costa Z, Horta ALL (2003) O abandono da actividade agrícola. Ministério da Agricultura do Desenvolvimento Rural e das Pescas, Grupo de trabalho agro-ambiental, Lisboa.

Alves AM, Pereira JS, e Silva JMN (eds.) (2007) O Eucaliptal em Portugal. Impactes Ambientais e Investigação Científica. ISAPress, Lisboa.

Aubin I, Messier C, Bouchard A (2008) Can plantations develop understory biological and physical attributes of naturally regenerated forests? Biological Conservation 141:2461-2476.

Bengtsson J, Nilsson SG, Franc A, Menozzi P (2000) Biodiversity, disturbances, ecosystem function and management of European forests. Forest Ecology and Management 132:39-50.

Bernaldez FG (1991) Ecological consequences of the abandonment of traditional land use systems in central Spain. Options Méditerranéenes - Série Séminaires 15: 23-29.

Bignal EM, McCracken DI (1996) Low-intensity farming systems in the conservation of the countryside. Journal of Applied Ecology 33: 413-424.

Blondel J, Aronson J (1999) Biology and wildlife in the Mediterranean region. Oxford University Press, Oxford.

Boteva D, Griffiths G, Dimopoulos P (2004) Evaluation and mapping of the conservation significance of habitats using GIS: an example from Crete, Greece. Journal for Nature Conservation 12:237-250.

Caldas EC (1998) Agricultura na História de Portugal. EPN - Empresa de Publicações Nacionais, Lda., Lisboa.

Cantelaube P, San-Miguel-Ayanz J, Schmuck G, Biard F (2002) Towards forecasting forest fire risk:Comparison of Meteorological forecast data from MeteoFrance with measured meteorological data from the JRC-MARS database. JRC, IES, Land Management Unit. European Communities, Italy.

Carvalho J, Alves PC, Grosso-Silva JM, Santos TM (2007a) Biologia e ecologia das florestas de carvalho-negral. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Alves PC, Grosso-Silva JM, Santos TM (2007b) Biologia e ecologia do carvalho-roble. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Santos JA, Reimão D, Santos J (2007c) A valorização dos carvalhais. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Santos JA, Reimão D (2007d) Utilizações do carvalho roble. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Castro EB, Casado MA, Costa M, Escribano R, García M, Génova M, Manzaneque AG, Moreno JC, Morla C, Regato P, Sáinz H (2001) Los Bosques Ibéricos: Una Interpretación Geobotánica. Editorial Planeta, Barcelona.

DGF – Direcção Geral das Florestas (2001) Inventário Florestal Nacional. 3ª Revisão. Direcção Geral de Florestas, Lisboa.

DGRF – Direcção Geral dos Recursos Florestais (2007) Resultados do IFN 2005/06. Direcção Geral dos Recursos Florestais. Available in http://www.dgrf.min-agricultura.pt.

Doerr SH, Shakesby RA, Walsh RPD (2000) Soil water repellency: its causes, characteristics and hydro-geomorphological significance. Earth-Science Reviews 51:33-65.

EC – European Community (2004) Evaluating Financing of Forestry in Europe, Projecto EFFE, Consórcio European Forest Institute (Finlândia), University of Joensuu

(Finlândia), University of Ghent (Bélgica), University of Freiburg (Alemanha), Polish Forest Research Institute (Polónia), Aristoteles University of Thessaloniki (Grécia) e Universidade Católica Portuguesa-CRP (Portugal).

EEA – European Environmental Agency (2003) Europe's environment: the third assessment. Office for Official Publications of the European Communities.

EEA – European Environmental Agency (2006) Progress towards halting the loss of biodiversity by 2010. EEA report 5/2006.

EEA – European Environmental Agency (2008) European forests - ecosystem conditions and sustainable use. EEA report 3/2008.

FAO (2006) Global Forests Resources Assessment 2005: Progress towards sustainable forest management. FAO, Rome.

Farrell EP, Führer E, Ryan D, Andersson F, Hüttl R, Piussi P (2000) European forest ecosystems: building the future on the legacy of the past. Forest Ecology and Management 132:5-20.

Franco JA (1971) Nova Flora de Portugal. Volume I. Lisboa.

Gomes I (2008) Valorização da Biodiversidade na definição de prioridades de ordenamento no Parque Nacional da Peneda-Gerês. Dissertação para obtenção do Grau de Mestre. Instituto Superior Técnico, Lisboa.

Green RE, Cornell SJ, Scharlemann JPW, Balmford A (2005) Farming and the Fate of Wild Nature. Science 307: 550-555.

Hermy M, Verheyen K (2007) Legacies of the past in the present-day forest biodiversity: a review of past land-use effects on forest plant species composition and diversity. Ecological Research 22:361-371.

Honrado JJP (2003) Flora e Vegetação do Parque Nacional da Peneda-Gerês. Dissertação de candidatura ao grau de Doutor em Biologia, Faculdade de Ciências da Universidade do Porto, Porto.

ICN – Instituto para a Conservação da Natureza (1998) Primeiro Relatório de Portugal a submeter à Conferência das Partes da Convenção sobre a Diversidade Biológica. Instituto de Conservação da Natureza, Ministério do Ambiente, Lisboa.

ICN - Instituto da Conservação da Natureza (2006) Plano Sectorial da Rede Natura 2000. Vol. II. Valores naturais. Fichas de caracterização ecológica e de gestão. ICN.

ICNB - Instituto da Conservação da Natureza e da Biodiversidade (2009) Parque Nacional da Peneda-Gerês. Available from http://portal.icnb.pt/ICNPortal/vPT2007-AP-Geres (accessed March 2009).

INM - Instituto de Meterorologia (2009) Clima de Portugal Continental Available from http://www.meteo.pt/pt/areaeducativa/otempo.eoclima/clima.pt/index.html (accessed March 2009).

Jones EW (1959) Biological flora of the British Isles: Quercus L. The Journal of Ecology 47: 169-222.

Lima ACPS (1996) Castro Laboreiro, povoamento e organização de um território serrano. Instituto da Conservação da Natureza, Parque Nacional da Peneda-Gerês e Câmara Municipal de Melgaço, Braga.

MA - Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

MacDonald D, Crabtree JR, Wiesinger G., Dax T, Stamou N, Fleury P, Lazpita JG,. Gibon A (2000) Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. Journal of Environmental Management 59: 47-69.

Madeira MV, Fabião A, Pereira JS, Araújo MC, Ribeiro C (2002) Changes in carbon stocks in Eucalyptus globulus Labill. plantations induced by different water and nutrient availability. Forest Ecology and Management 171:75-85.

McNeely JA (1994) Lessons from the past: forests and biodiversity. Biodiversity and Conservation 3:3-20.

MCPFE - Ministerial Conference on the Protection of Forests in Europe (2007) State of Europe's Forests 2007: The MCPFE report on sustainable forest management in Europe. Ministerial Conference on the Protection of Forests in Europe Liaison Unit Warsaw.

MENDES A, Dias R (2002) Financial instruments of forest policy in Portugal in the 80s and 90s. EFI Forest Policy Research Forum: Cross-Sectoral Policy Impacts on Forests, 4-6 April, Sanvonlinna.

Moreira F, Rego FC, Ferreira PG (2001a) Temporal (1958-1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. Landscape Ecology 16: 557-567.

Moreira F, Ferreira PG, Rego FC, Bunting S (2001b) Landscape changes and breeding bird assemblages in northwestern Portugal: the role of fire. Landscape Ecology 16: 175-187.

Naveh Z (1975) The evolutionary significance of fire in the Mediterranean region. Vegetatio 29: 199–208.

Onofre N (1990) Impactes do eucaliptal sobre a fauna dos eucaliptais. In: M. Alves M, Pereira JS (eds) Impactes ambientais e sócio-económicos do eucaliptal em Portugal. DEF, UTL-ISA, Lisboa.

Pausas J, Bladé C, Valdecantos A, Seva J, Fuentes D, Alloza J, Vilagrosa A, Bautista S, Cortina J, Vallejo R (2004) Pines and oaks in the restoration of Mediterranean landscapes of Spain: New perspectives for an old practice — a review. Plant Ecology 171: 209-220.

Pereira HM, Daily GC (2006) Modelling biodiversity dynamics in countryside landscapes. Ecology 87:1877-1885.

Pereira JS, Correia AV, Correia AP, Branco M, Bugalho M, Caldeira MC, Souto-Cruz C, Freitas H, Oliveira AC, Pereira JCM, Reis RM, Vasconcelos MJ(2002) Forest and biodiversity. In: Santos FD, Forbes K, Moita R. (eds.). Climate Change in Portugal: Scenarios, Impacts and Adaptation Measures – SIAM Project, Gradiva. Lisboa.

PNPG – Parque Nacional da Peneda-Gerês (1995) Plano de Ordenamento do Parque Nacional da Peneda-Gerês. Relatório de síntese. ICN, Braga.

Prieler S, Lesk'o A, Andberg S (1998) Three Scenarios for Land-Use Change: A Case of Study in Central Europe, International Institute for Applied Systems Analysis.

Radich MC, Alves AAM (2000) Dois Séculos da Floresta em Portugal, CELPA, Lisboa.

Ramil-Rego P, Muñoz-Sobrino C, Rodríguez-Guitián M, Gómez-Orellana L (1998) Differences in the vegetation of the North Iberian Peninsula during the last 16,000 years. Plant Ecology 138:41-62.

Ribeiro O, Daveau S, Lautensach H (1988) Geografia de Portugal: II – O ritmo climático e a paisagem. Ed. J. Sá da Costa, Lisboa.

Shvidenko A, Barber CV, Persson R (2005) Forest and woodland systems. In: Hassan R., Scholes R., Ash N. (eds) Ecosystems and human well-being: Current state and trends. Island Press, Washington D. C., pp. 585-621.

Silva JS, Deus E, Saldanha L (2008) Evolução dos incêndios florestais em Portugal, antes e depois de 2003. In: Silva JS, Deus E, Saldanha L (eds.) Incêndios Florestais, 5 anos após 2003. LPN, AFN.

Wade TG, Riitters KH, Wickham JD, Jones KB (2003) Distribution and causes of global forest fragmentation. Conservation Ecology 7: 7. [online] http://www.consecol.org/vol7/iss2/art7.

Zapata BR, Correia AI, García MJG (1995) Evolución del paisaje en la Sierra de Peneda (NO Portugal) durante los últimos 3500 BP, Geogaceta 18: 105-108.

THE ROLE OF NATURAL FOREST FOR BIODIVERSITY CONSERVATION IN THE NW OF THE IBERIAN PENINSULA

Proença VM, Pereira HM, Vicente L. The role of natural forests for biodiversity conservation in the NW of the Iberian Peninsula. Submitted to Acta Oecologica *(in review)*.

2 The role of natural forests for biodiversity conservation in the NW of the Iberian Peninsula.

Abstract

Forest ecosystems have been subjected to a continuous dynamic between deforestation and forestation. Assessing biodiversity responses to these processes could be essential for conservation planning. We analysed patterns of species richness of plants and birds in patches of natural oak forest and in stands of pine (native species) and eucalypt (exotic species) in NW Portugal. We analysed data of forest and non-forest species separately. Data were analysed at the local (intra-patch) and the regional (inter-patch) scales. Oak forest patches were the richest in forest species at the local scale and in forest plants at the regional scale. Pine forest was associated to more forest birds at the regional scale. Eucalypt stands had the lowest values of forest species richness at both scales. The species-area relationships of forest species in fragments of oak forest had consistently a higher slope, at both the local and the regional scales, than the species-area relationships of forest species in pine and eucalypt stands. These findings stress the relevance of oak forest for the conservation of forest species diversity, also pointing the need to conserve large areas of oak forest due to the apparent vulnerability of forest species to area loss. Finally, pine forest presented intermediate results between oak forest and eucalypt forest, suggesting that forest species patterns are affected by forest naturalness.

Keywords

Eucalypt; Forest species; Natural forest; Oak; Planted forests; Pine; Species-area relationship

2.1 Introduction

Forests provide several ecosystem services (Shvidenko 2005). Due to their direct use and market value, forest goods, in particular wood, have been object of more demand than other benefits provided by forests, such as soil fixation (Liu and Diamond 2005), climate regulation (Malhi et al. 2008) or biodiversity (Barlow et al. 2007). This bias has contributed to two processes of land use change: the loss and fragmentation of native forests and the establishment and maturation of new plantation forests (Ciancio and Nocentini 1997, FAO 2006). Production targets instead of conservation targets have often shaped these new forests (Ciancio and Nocentini 1997, Koch and Skovsgaard 1999, FAO 2006), which are frequently monocultures often composed of introduced species and even-aged trees, a design that generally does not promote biodiversity (Hartley 2002). Because biodiversity has a key role in sustaining ecosystem services and promoting ecosystem resilience (MA 2005), production forests with low biodiversity tend to be more vulnerable to disturbance and environmental change than natural forests (Lugo 1997, Carnus et al. 2006, Bassi et al. 2008).

In a world facing fast environmental changes, the role of forest ecosystems for conservation, as reservoirs of biodiversity and providers of multiple ecosystem services, is of unquestionable importance (Shvidenko et al. 2005). According to recent previsions, land-use change will be the main cause of biodiversity loss in the next decades (van Vuren et al. 2006). Therefore, understanding how forest biodiversity responds to remaining patches of native forest and to new planted forests is fundamental for conservation planning.

In Portugal, production forests are mainly composed of pine (*Pinus pinaster*) and eucalypt (*Eucalyptus globulus*). These two species currently account for more than 40% of the Portuguese forest and now dominate the northern half of the country (DGRF 2007). *Pinus pinaster* is native to the country, but its distribution has been modified by plantation development. *Eucalyptus globulus* is an exotic species, originally from south-eastern Australia.

On the other hand, deciduous oaks, such as the common oak (*Quercus robur*), were once the dominant species in this region but now represent 4% of the country forest and their distribution is very patchy (Ramil-Rego 1998, DGRF 2007). Moreover, the natural forests of *Quercus robur* and *Quercus pyrenaica* are acknowledged by their value as providers of several ecosystem services being listed in the Annex I of the Habitats Directive (ICN 2006).

Here, we investigate the role of deciduous oak forest (natural forest, native species), pine forest (planted forest, native species) and eucalypt forest (planted forest, exotic species) for the conservation of forest biodiversity. We study the diversity patterns of two taxa, plants and birds, according to their habitat affinity, forest and non-forest species.

We hypothesize that richness of forest plant and bird species will be affected by forest naturalness and that these forest species will be particularly vulnerable to the loss of natural forest. More specifically, we predict that: (i) forest species richness will be higher in oak forest patches, while pine stands will support fewer species than oak patches but more than eucalypt stands; (ii) forest species richness will respond more strongly to patch size in natural forest than in planted forests; (iii) forest species richness will respond more strongly to patch size in oak forest than non-forest species richness. To test our predictions we analyse plant and birds species richness, according to species habitat affinity, and determine species-area relationships at local (intra-patch) and regional (inter-patch) scales. We also test the effect of perimeter-area ratio on the richness of forest species and non-forests species.

2.2 Methods

This study was conducted in the Alto Minho region in NW Portugal (41ºN 8ºW) (Figure 1). The landscape is mainly composed of shrubland, agricultural fields and forest, mainly pine, eucalypt and oak (DGRF 2007).

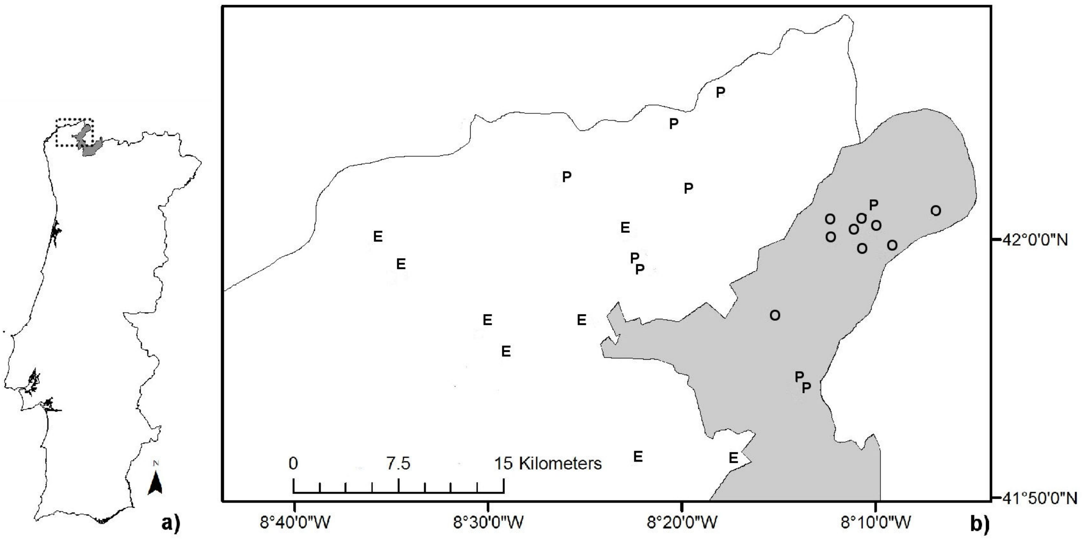


Figure 2.1 - Location of the study area in Portugal (ca. 40km x 25km, limited by the dotted frame) (a); Distribution of forest patches within the study area (O - oak, P - pine, E - eucalypt) (b). The region in grey represents the National Park of Peneda-Gerês.

We selected fragments of oak forest (*Quercus robur* and *Quercus pyrenaica*) and stands of pine and eucalypt according to the following criteria: mature forest and isolated from other forests by a matrix of shrubland or agriculture. The size of forest patches ranged from 0.22 ha to 36.52 ha (Table 1). The area of the patches was calculated from ortophotomaps using ArcGIS 9.0 software.

We defined as sample unit a square of 10 m x 10 m (100 m^2). Plants were sampled in five sub-units of 1mx1m (1 m^2). The sampling effort was proportional to a species-area relationship, $U= A^{0.33}$, where U is the number of sub-units and A the area (m^2) of the patch. The number of sample units was the integer part of U/5. We chose the value 0.33 as exponent because it was within the proposed range of values for isolated areas ($0.25 < z < 0.35$) (Rosenzweig 1995), and resulted in a feasible sampling effort. Moreover, this exponent agrees with the mean value of z found for isolated areas by van Vuuren et al. (2006). Sample units were arranged in a way that maximized the distance between them, while maintaining a buffer of at least 15 m from the edge. Data were collected during July and August 2005.

Plants were identified at the species level whenever possible, and were classified as either forest or non-forest plants. The classification followed the species ecological information in Honrado (2003). We opted for an inclusive classification of forest plants, accepting every species that was reported to appear in forest environments.

Birds were surveyed through 15-minute point counts at the centre of each sample unit. Bird data were collected during the first three hours of the day and never under rainy conditions. Birds were visually and acoustically identified in a range of up to 25 m. Birds were classified either as forest or non-forest (generalist) species according to their habitat specificity: forest species nest within forests, whereas non-forest species can nest outside forests. Species classification followed Pimenta and Santarém (1996).

All subsequent analyses were performed considering each of the four groups described above: forest and non-forest plants, forest and non-forest birds. The separate analysis of groups of species was recommended in other species-area studies due to possible differences in the response of the groups (Godefroid and Koedam 2003, Magura et al. 2008). Moreover, this approach may contribute to a more accurate assessment of the effect of land use change on native species in countryside landscapes where native and non-native habitats coexist (Pereira et al. 2004, Pereira and Daily 2006).

Species were classified as shared (species observed in at least two types of forest), exclusive (species observed in only one type of forest, but in at least two patches), or singleton (species observed in only one patch). We opted to distinguish between exclusive and singleton species because although singleton species contributed to the total number of species, their role in biotic communities was less certain. They could be low-abundance species with a widespread distribution (i.e., they could be found in other forest types as well) or even transient species, in the case of birds, or truly rare species only found in that type of habitat (Novotny and Basset 2000, Longino et al. 2002).

Mean species richness of sample units was compared between forest types using the Welch test, an alternative to ANOVA recommended when homoscedasticity is not verified (Grissom 2000, Quinn and Keough 2002). Posterior pairwise comparisons were performed using the t test (p value adjusted by the Holm's method).

We studied species-area relationships (SARs) using the power model (Arrhenius, 1921), $S = c\ A^z$, and the exponential model (Gleason 1922), $S = c + z \log (A)$, where S is the number of species, A is the area and c and z are parameters of the model. Some studies suggested that the power model is a better choice for intermediate and larger scales of analysis, whereas the exponential model, or semi-log, would perform better at small scales of analysis (Tjorve 2003, Rosenzweig 2005). Moreover, Ugland et al. (2003) also suggested that the semi-log model should be preferred when dealing with species-accumulation data, i.e., species increase due to accumulation of equal sized samples within a habitat.

We evaluated the adjustment of species-area data to the power model (on its log-log form, $\log S = c + z \log A$) and to the exponential model through the comparison of residuals distribution and the value of r^2. The power model was overall better adjusted to species-area data at the regional level, with the exponential model being only slightly better for eucalypt stands data. At the local level all intra-patch SARs were significant ($p>0.05$) and better adjusted to the semi-log model. Species data at the local level were log(x+1) transformed (power model) and (x+1) transformed (exponential model) before testing the models.

Intra-patch SARs were calculated for plants using data from three sample scales: 1 m^2, 100 m^2 and the patch area. A two-way ANOVA and the post-hoc TukeyHSD were used to compare the z' values of intra-patch SARs (the ANOVA assumptions were verified). Forest type and species group were set as factors.

The influence of perimeter-area ratio ($perimeter/\sqrt{area}$) on species richness was analysed through simple linear regression. We used the perimeter-area ratio as a measure of the extent of forest edge. The independence between area and perimeter-area ratio was tested through Pearson's correlation coefficient. Statistical analyses were performed using the R statistical software system (www.cran.r-project.org).

2.3 Results

During this study we recorded 106 plant taxa and 47 bird species (see Appendix 2.1). Some plants could not be identified at the species level. Because all records in the species list correspond to different taxonomic units, they were used as different species in data analysis, and hereafter will be referred to as species.

At the regional level, more species of forest and non-forest plants were found in oak forest than in planted forests (Table 1). Although the number of shared species was similar among the three types of forest, differences in species richness derive from the greater number of exclusive and singleton species found in oak forest (Figure 2). With respect to birds, pine forest was the richest for both species groups (Table 1). The number of shared species was similar among forests types, with a larger number of exclusive and singleton species observed in pine forest (Figure 2).

Table 2.1 - Patches data and species richness by taxa: area, perimeter-area ratio (PAr), number of sample units (SU), total number of species, forest species and non-forest species. The total number of species also includes taxa whose habitat affinity (forest or non-forest species) was not determined.

				Plants			Birds		
Patch	Area(ha)	PAr	SU	Total	Forest	Non-forest	Total	Forest	Non-forest
Oak Forest				88	52	32	28	17	11
O1	0.39	4.103	3	30	18	11	9	5	4
O2	1.22	4.333	4	39	27	10	10	4	6
O3	2.2 0	5.942	5	43	26	16	16	7	9
O4	2.85	4.104	5	31	24	6	16	9	7
O5	3.36	4.836	6	37	25	10	14	9	5
O6	5.19	7.565	7	45	28	16	19	10	9
O7	6.00	4.754	7	43	32	8	20	12	8
O8	10.30	5.748	9	52	37	14	25	16	9
O9	21.66	6.449	11	40	26	13	18	10	8
Pine forest				64	33	28	40	20	20
P1	0.22	4.037	2	16	8	8	11	3	8
P2	0.53	4.255	3	20	11	9	12	4	8
P3	1.01	5.863	4	17	10	7	16	10	6
P4	2.83	4.671	5	24	14	10	15	8	7
P5	4.76	5.755	6	19	11	8	17	8	9
P6	8.62	5.157	8	42	20	21	16	8	8
P7	11.91	5.815	9	27	14	11	19	12	7
P8	15.98	5.863	10	16	10	5	23	12	11
P9	31.09	4.303	12	18	13	5	15	9	6
Eucalypt forest				51	28	21	30	13	17
E1	0.62	3.806	3	8	4	4	6	4	2
E2	1.19	4.771	4	17	8	9	7	2	5
E3	1.89	4.516	5	24	14	9	12	5	7
E4	2.76	5.286	5	17	8	9	9	4	5
E5	5.62	5.246	7	16	7	9	11	6	5
E6	7.92	8.294	8	14	5	9	11	6	5
E7	18.81	5.288	11	25	12	13	11	5	6
E8	36.52	4.408	13	28	15	12	16	6	10

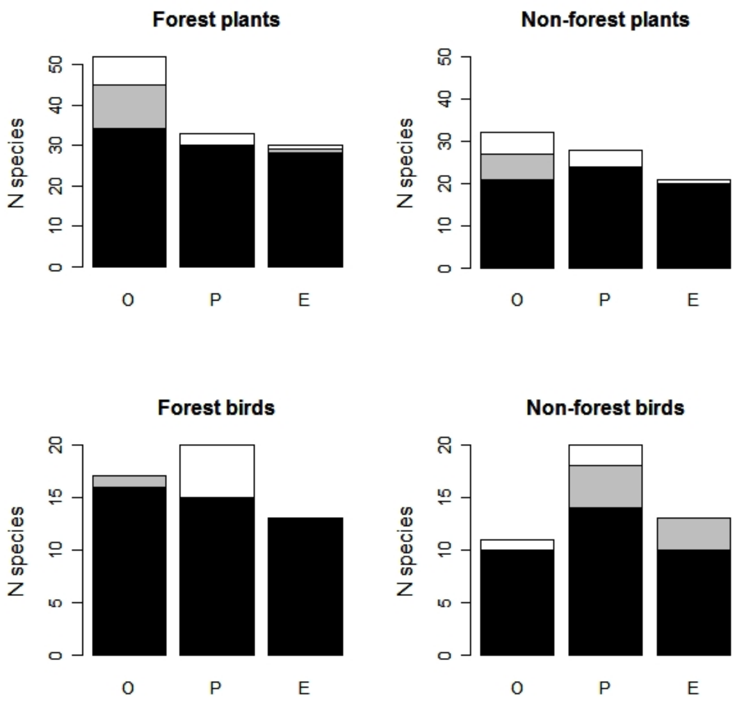


Figure 2.2 - Number of plant and bird species in each forest type divided by their occurrence: shared species (species observed in at least two types of forest - black), exclusive species (species observed in only one type of forest, but in at least two patches - grey) and singleton species (species observed in just one patch - white). Horizontal axis: oak (O), pine (P) and eucalypt (E).

At the local level, the mean species richness in sample units (100 m^2) differed among forest types for all species groups: forest plants ($F_{2,106} = 180.21$, $p < 0.001$), non-forest plants ($F_{2,108} = 6.37$, $p < 0.01$), forest birds ($F_{2,101} = 51.16$, $p < 0.001$) and non-forest birds ($F_{2,110} = 10$, $p < 0.001$). Results from pairwise comparisons indicated that the mean richness of forest plants was highest in oak patches, but not significantly different between pine and eucalypt patches. The mean number of forest bird species was highest in oak patches, followed by pine patches, whereas eucalypt patches were the least rich. The mean richness of non-forest plants was similar in oak and pine patches, and both forests were richer than eucalypt forest. Finally, the mean richness of non-forest birds was highest in oak patches, whereas richness in pine and eucalypt patches did not differ significantly.

A two-way ANOVA tested for the effect of forest type ($F_{2,46}$= 50.4, p < 0.001), taxon ($F_{1,46}$= 44.37, p < 0.001) and also an interaction between these factors ($F_{2,46}$= 30.33, p < 0.001) on the z' values of intra-patch SARs. The mean z' value of forest plant species in oak patches was higher than the mean z' value of each of the other pairs "forest type: species group" (Figure 3).

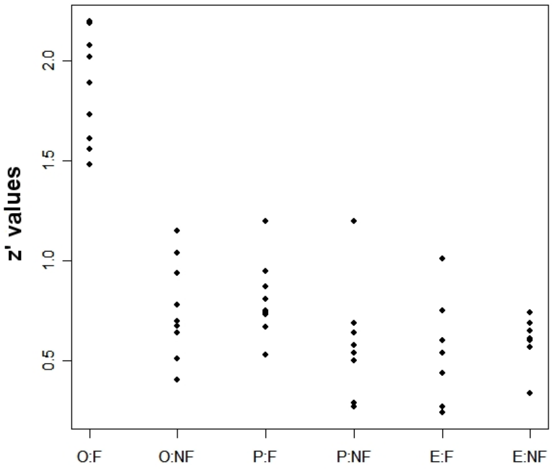


Figure 2.3 - Magnitude of z' values of intra-patch species-area relationships in function of forest type and plants habitat specificity. Horizontal axis: oak (O), pine (P), eucalypt (E), forest species (F), non-forest species (NF).

At the regional level, we found significant species-area relationships for forest plants, forest birds and non-forest birds in oak forest, for forest birds at pine forest and for non-forest plants and non-forest birds in eucalypt forest (Table 2, Figure 4). The extent of the forest edge was related with the richness of non-forest plants ($r^2 = 0.67$, $p < 0.01$) and non-forest birds ($r^2 = 0.52$, $p < 0.05$) in oak forest fragments, and with forest bird richness ($r^2 = 0.64$, $p = 0.01$) in pine stands. Perimeter-area ratio and area varied independently in all forest types.

Table 2.2 - Inter-patch species-area relationships (log S = c + z log A). Significant regressions in bold ($p<0.05$).

		Plants				Birds			
	n	z	c	r^2	p	z	c	r^2	p
Oak : Total	9	0.09	2.7	0.40	0.07	**0.24**	**0.26**	**0.73**	**0.003**
Oak : Forest	9	**0.12**	**2.0**	**0.49**	**0.04**	**0.29**	**-0.9**	**0.65**	**0.008**
Oak : Non-forest	9	0.05	1.8	0.04	0.62	**0.18**	**0.08**	**0.52**	**0.029**
Pine : Total	9	0.06	2.5	0.09	0.45	**0.1**	**1.7**	**0.57**	**0.02**
Pine : Forest	9	0.09	1.5	0.32	0.11	**0.23**	**-0.4**	**0.65**	**0.009**
Pine : Non-forest	9	-0.03	2.4	0.01	0.80	0.01	2.0	0.002	0.91
Eucalypt : Total	8	0.20	0.74	0.48	0.06	**0.19**	**0.26**	**0.72**	**0.007**
Eucalypt : Forest	8	0.18	0.13	0.30	0.16	0.17	-0.3	0.40	0.095
Eucalypt : Non-forest	8	**0.2 0**	**0.02**	**0.64**	**0.02**	**0.24**	**-0.9**	**0.54**	**0.038**

2.4 Discussion

We compared patterns of species richness of forest and non-forest plants and birds in three types of forest: oak forest (natural forest, native species), pine forest (planted forest, native species) and eucalypt forest (planted forest, exotic species).

As predicted oak forest patches were the richest in forest species, pine stands appeared to be also important for forest birds and eucalypt stands had the lowest species richness. The higher number of forest species in natural forest is in agreement with other studies that also compared species assemblages in natural forest and in forest plantations (Kwok and Corlett 2000, Lindenmayer et al. 2003, Zurita et al. 2006). The higher number of exclusive forest species, mainly plants, found in oak forest strengthens the value of natural forest for conservation, as these species may be particularly vulnerable to the loss of natural forest. Pine and eucalypt forests on the other hand, lacked exclusive forest species. In fact, the higher richness of forest birds in pine forest was entirely due to five singletons and the only exclusive species found in eucalypt forest was the eucalypt itself.

Differences between natural and planted forests may be explained through forests naturalness and forest dynamics. Oak forest patches resulted from the fragmentation of continuous native forest and also from later natural regeneration. Forest fragmentation has probably started around 5500 BP, according to pollinic studies that detected the first signs of human intervention in this area, and continued until the nineteenth century (Ramil-Rego et al. 1998, Sobrino et al. 2001, Honrado 2003). Forest regeneration has been the dominant trend in the last five decades as rural population declined and migrated to urban centres (Moreira et al. 2001). Despite past and present (e.g., fire) anthropic impacts, that drove most of the remaining oak forest to a pre-climacic state, oak forest patches still provide a suitable environment for the occurrence and persistence of forest species (ICN 2006). Pine and eucalypt forests on the other hand were planted and their communities are the assemblage of formerly present species and colonizers from neighbouring areas (Christian et al. 1998, Michelsen et al. 1996). Because planted forests are managed to meet production targets they may fail to provide the adequate conditions for the establishment or persistence of more sensitive species (Endels et al. 2004, Fraterrigo et al. 2005).

For example, Moreira et al. (2003) working in the same region of our study, found a negative correlation between abundance of eucalypt stands and the abundance of bird tree nesters. They suggested that the presence of these species could be affected by the characteristics of eucalypt stands, such as vertical vegetation structure, food availability and microclimatic conditions. Kwok and Corlett (2000), Zurita et al. (2006) and Magura et al. (2008) also point the simplified structure of monoculture forest stands as a reason for their lower richness in forest birds.

Results from species-area analyses provided evidence on the vulnerability of forest species to the loss of natural forest. Intra-patch SARs of forest plants in oak patches had consistently higher z’ values than the other tested SARs. This result has implications for conservation because the higher the slope, the stronger the relation between species richness and area. This may imply that forests species benefit from larger areas of natural forest and are particularly vulnerable to area loss, which assigns a higher value of conservation to larger fragments of oak forest. Moreover, results from intra-patch SARs were supported by results at the regional scale. Inter-patch SARs of forest plants and birds in oak forest were both significant and stronger than SARs in planted forests confirming the effect of oak patch size on species richness. This finding is relevant because while intra-patch SARs necessarily express a non decreasing relation between area and species richness, inter-patch SARs curves are not under such constraint (Scheiner 2003). Our results are in agreement with Magura et al. (2008) that have also reported a significant SAR of forest specialist birds in native forest patches but not in exotic stands in Hungary.

We found insular z values for forest birds in oak patches but not for forest plants. Forest plants had a lower z, typical of mainland areas ($0.13 < z < 0.18$ (Rosenzweig 1995)). Lawesson et al. (1998) note that the effects of fragmentation may not be detected for plants, because the rates of local extinction and immigration of some species may be very slow and some species may disperse along mid-distances. This might explain the mainland z, as some oak patches probably result from the past fragmentation of a continuous area of forest. Moreover, the composition of the surrounding matrix, shrubland and pasture fields, may allow the flux of species between patches. On the other hand, the effects of forest fragmentation on forest birds have been reported in previous studies (Brotons and Herrando 2001, Santos et al. 2002). The presence of some forest species may be limited by the existence of a minimum core area or a minimum diversity of habitats within forest patches (Herrando and Brotons 2002), which would explain the stronger relation between bird richness and patch size.

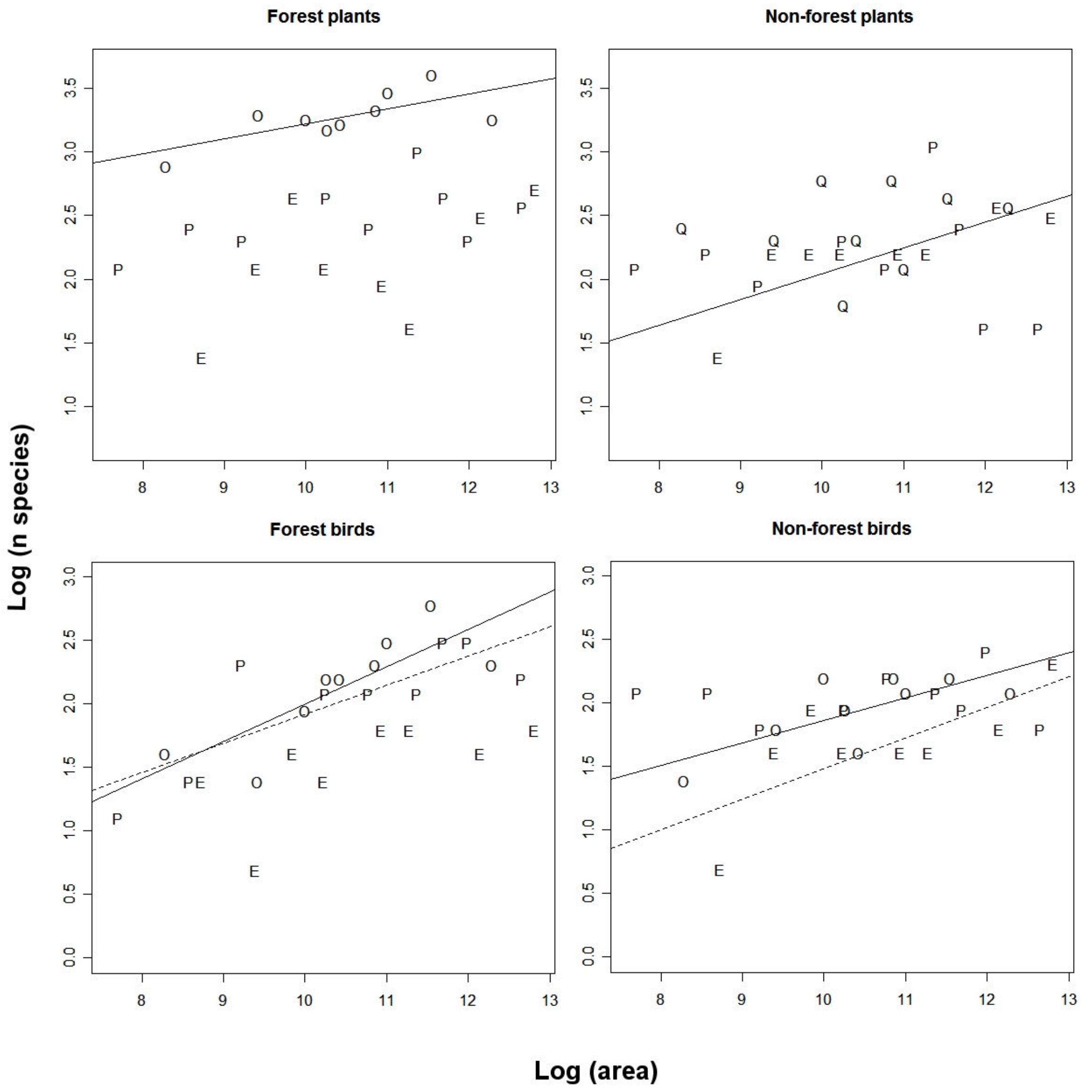


Figure 2.4 - Species-area relationships of forest species and non-forest species at the regional level. Points: oak patches (O), pine patches (P) and eucalypt patches (E). The regression line is presented for significant relationships. Forest plants: oak (solid line). Non-forest plants: eucalypt (solid line). Forest birds: oak (solid line), pine (dashed line). Non-forest birds: oak (solid line), eucalypt (dashed line). Species-area parameters are presented in Table 2.

Although the number of forest species was only related to area in oak fragments, the number of non-forest plants and birds was related with the extent of the fragment edge, suggesting the occurrence of edge effects (Saunders et al. 1991, Murcia 1995), corresponding to the flux of species from the matrix. The richness of non-forest birds was also related with area, probably due to their mobility that enables the dispersion throughout the fragment. In eucalypt stands, the number of non-forest species increased with patch size but not with edge extent, possibly because these species do not sense the border of the stand as a true edge. The increase in species richness was probably due to an increase in the sampling effort because these species do not depend of forest ecosystems. In pine stands the richness of forest birds was related with forest area and edge extent, suggesting that while forest birds use pine forest their presence is on the one hand limited by stands size and on the other hand by the extension of forest edge, which allows species colonization from outside areas.

Forest species presented an intermediate response in pine forest, in relation to oak and eucalypt forests, in all analyses performed in this study. This suggests that the underlying gradient of forest naturalness, from natural oak forest to exotic eucalypt plantations, with native pine plantations in-between, has affected forest species patterns.

2.5 Conclusion

Our results evidenced the importance of oak forest for the conservation of forest biodiversity: there was a higher number of forest species in oak forest patches, including species not observed in pine or eucalypt forests, and forest species seemed to be particularly affected by the loss of oak forest. However, despite the relevance of oak forest for biodiversity, its present distribution in the country is very reduced and patchy, which might put at risk the conservation of forest species diversity. We reinforce the need to acknowledge the priority value for conservation of natural oak forests and to develop management practices that support the natural regeneration and succession in disturbed forest fragments.

Acknowledgments We thank João Guilherme for doing the bird census and to Cibele Queiroz and Inês Gomes for their valuable assistance during field work and plant sampling. João Honrado, Helena Tiago (PNPG) and Duarte Silva (PNPG) helped with plant identification and Miguel Pimenta (PNPG) with bird classification. Rodríguez Riguero kindly provided an important background reference. Cibele Queiroz, Ana Ceia Hasse and José Pedro Amaral contributed with useful comments on a first version of the manuscript. Vânia M. Proença was supported by a fellowship from Fundação para a Ciência e a Tecnologia (BD/12395/2003), and the field work was supported by the Ecoforsite project (POCI-AMB-55701/2004) and the ABAFOBIO project (PTDC/AMB/73901/2006).

References

Arrhenius O (1921) Species and Area. The Journal of Ecology 9:95-99.

Barlow J, Gardner TA, Araujo IS, Ávila-Pires TC, Bonaldo AB, Costa JE, Esposito MC, Ferreira LV, Hawes J, Hernandez MIM, Hoogmoed MS, Leite RN, Lo-Man-Hung NF, Malcolm JR, Martins MB, Mestre LAM, Miranda-Santos R, Nunes-Gutjahr AL, Overal WL, Parry L, Peters SL, Ribeiro-Junior MA, da Silva MNF, da Silva Motta C, Peres CA (2007) Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proceedings of the National Academy of Sciences 104:18555-18560.

Bassi S, Kettunen M, Kampa E, Cavalieri S (2008) Forest Fires: Causes and Contributing Factors to Forest Fire Events in Europe Study for the European Parliament Committee on Environment, Public Health and Food Safety under contract IP/A/ENVI/FWC/2006-172/LOT1/C1/SC10.

Brotons L, Herrando S (2001) Factors affecting bird communities in fragments of secondary pine forests in the north-western Mediterranean basin. Acta Oecologica. 22: 21-31.

Carnus JM, Parrotta J, Brockerhoff E, Arbez M, Jactel H, Kremer A, Lamb D, O'Hara K, Walters B (2006) Planted forests and biodiversity. Journal of Forestry 104: 65-77.

Christian DP, Hoffman W, Hanowski JM, Niemi GJ, Beyea J (1998) Bird and mammal diversity on woody biomass plantations in North America. Biomass and Bioenergy 14: 395-402.

Ciancio O, Nocentini S (1997) The forest and man: the evolution of forestry thought from modern humanism to the culture of complexity. Systemic silviculture and management on natural bases. In: Ciancio O (Ed) The forest and man, Accademia Italiana di Scienze Forestali, Firenze, Italy, pp. 21-114.

DGRF (2007) Resultados do IFN 2005/06. Direcção Geral dos Recursos Florestais. Available from http://www.dgrf.min-agricultura.pt.

Endels P, Adriaens D, Verheyen K, Hermy M (2004) Population structure and adult plant performance of forest herbs in three contrasting habitats. Ecography 27: 225- 241.

FAO (2006) Global Forests Resources Assessment 2005: Progress towards sustainable forest management. FAO, Rome.

Fraterrigo JM, Turner MG, Pearson SM (2005) Previous land use alters plant allocation and growth in forest herbs. Journal of Ecology 94: 548-557.

Gleason HA (1922) On the relation between species and area. Ecology 3: 158-162.

Godefroid S, Koedam N (2003) How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? Global Ecology and Biogeography 12: 287-298.

Graham CH, Blake JG (2001) Influence of patch- and landscape-level factors on bird assemblages in a fragmented tropical landscape. Ecological Applications 11: 1709-1721

Grissom RJ (2000) Heterogeneity of Variance in Clinical Data. Journal of Consulting and Clinical Psychology. 68: 155-165.

Hartley MJ (2002) Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management 155: 81-95.

Herrando S, Brotons L (2002) Forest bird diversity in Mediterranean areas affected by wildfires: a multi-scale approach. Ecography 25: 161-172.

Honrado JJP (2003) Flora e Vegetação do Parque Nacional da Peneda-Gerês. Dissertação de candidatura ao grau de Doutor em Biologia, Faculdade de Ciências da Universidade do Porto, Porto.

ICN (2006) Plano Sectorial da Rede Natura 2000. Vol. II. Valores naturais. Fichas de caracterização ecológica e de gestão. ICN. Available from http://www.icn.pt/psrn2000/caracterizacao_valores_naturais/habitats/9230.pdf.

Keeley JE, Fotheringham CJ (2003) Species area relationships in Mediterranean climate plant communities. Journal of Biogeography 30: 1629-1657.

Koch NE, Skovsgaard JP (1999) Sustainable management of planted forests: some comparisons between Central Europe and the United States. New Forests 17: 11-22.

Kwok HK, Corlett RT (2000) The bird communities of a natural secondary forest and a Lophostemon confertus plantation in Hong Kong. Forest Ecology and Management 130: 227-234.

Lawesson JE, de Blust G, Grashof-Bokdam C, Firbank L, Honnay O, Hermy M, Hobitz P, Jensen LM (1998) Species diversity and area-relationships in Danish beech forests. Forest Ecology and Management 106: 235-245.

Lindenmayer DB, McIntyre S, Fischer J (2003) Birds in eucalypt and pine forests: landscape alteration and its implications for research models of faunal habitat use. Biological Conservation 110: 45-53.

Liu J, Diamond J (2005) China's environment in a globalizing world. Nature 435: 1179-1186.

Longino J, Colwell RK, Coddington JA (2002) The ant fauna of a tropical rainforest: estimating species richness three different ways. Ecology 83: 689-702.

Lugo AE (1997) The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. Forest Ecology and Management 99: 9 -19.

MA - Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Magura T, Baldí A, Horvath R (2008) Breakdown of the species-area relationship in exotic but not in native forest patches. Acta Oecologica 33: 272 -279.

Malhi Y, Roberts TJ, Betts RA, Killeen TJ., Li W, Nobre CA (2008) Climate Change, Deforestation, and the Fate of the Amazon. Science 319: 169 -172.

Michelsen A, Lisanework N, Friis I, Holst N (1996) Comparisons of Understorey Vegetation and Soil Fertility in Plantations and Adjacent Natural Forests in the Ethiopian Highlands. Journal of Applied Ecology 33: 627-642.

Moreira F, Delgado A, Ferreira S, Borralho B, Oliveira N, Inácio M, Silva JS, Rego F, (2003) Effects of prescribed fire on vegetation structure and breeding birds in young Pinus pinaster stands of northern Portugal. Forest Ecology and Management 184: 225–237.

Moreira F, Rego FC, Ferreira PG (2001) Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. Landscape Ecology 16: 557-567.

Murcia C (1995) Edge effects in fragmented forests: implications for conservation. Trends Ecological Evolution 10: 58–62.

Novotny V, Basset Y (2000) Rare species in communities of tropical insect herbivores: pondering the mystery of singletons. Oikos 89: 564 - 572.

Pereira H, Daily GC, Roughgarden J (2004) A framework for assessing the relative vulnerability of species to land-use change. Ecological Applications 14: 730-742.

Pereira HM, Daily GC (2006) Modeling biodiversity dynamics in countryside landscapes. Ecology 87: 1877-1885.

Pimenta M, Santarém ML (1996) Atlas das Aves do Parque Nacional da Peneda-Gerês. ICN/NPPG, Porto.

Quinn GP, Keough MJ (2002) Experimental Design and Data Analysis for Biologists. Cambridge University Press, Cambridge.

Ramil-Rego P, Muñoz-Sobrino C, Rodríguez-Guitián M, Gómez-Orellana L (1998) Differences in the vegetation of the North Iberian Peninsula during the last 16 000 years. Plant Ecology 138: 41–62.

Rosenzweig ML (1995) Species diversity in space and time. Cambridge University Press, Cambridge, UK.

Santos T, Tellería JL, Carbonell R (2002) Bird conservation in fragmented Mediterranean forests of Spain: effects of geographical location, habitat and landscape degradation. Biological Conservation 105: 113-125.

Saunders DA, Hobbs RJ, Margules CR (1991) Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18-32.

Scheiner SM (2003) Six types of species-area curves. Global Ecology and Biogeography 12: 441-447.

Shvidenko A, Barber CV, Persson R (2005) Forest and woodland systems. In: Hassan R, Scholes R, Ash N (Eds.) Ecosystems and human well-being: Current state and trends. Island Press, Washington D. C., pp. 585-621.

Sobrino C, Ramil-Rego P, Rodriguez Guitián MA, (2001) Vegetation in the mountains of northwest Iberia during the last glacial-interglacial transition. Vegetation History and Archaeobotany 10: 7-21.

Tjorve E (2003) Shapes and functions of species–area curves: a review of possible models. Journal of Biogeography 30: 827-835.

Ugland KI, Gray JS, Ellingsen KE (2003) The species accumulation curve and estimation of species richness. Journal of Animal Ecology 72: 888-897.

van Vuuren DP, Sala OE, Pereira HM (2006) The future of vascular plant diversity under four global scenarios. Ecol. Soc. 11: 25. Available from http://www.ecologyandsociety.org/vol11/iss2/art25/.

Zurita GA, Rey N, Varela DM, Villagra M, Bellocq MI (2006) Conversion of the Atlantic Forest into native and exotic tree plantations: Effects on bird communities from the local and regional perspectives. Forest Ecology and Management 235: 164-173.

Appendix 2.1

List of observed species and number of patches in which they were observed by forest type: oak (O), pine (P) and eucalypt (E). Species habitat affinity: forest species (F), non-forest species (NF), not determined (nd).

Taxon	Family	Species	O (N=9)	P (N=9)	E (N=8)
Plants	Aquifoliaceae	*Ilex aquifolium* (F)	1	0	1
Plants	Araliaceae	*Hedera hibernica* (F)	6	4	1
Plants	Blechnaceae	*Blechnum spicant* (F)	4	1	1
Plants	Boraginaceae	*Lithodora prostata* (F)	8	5	1
Plants	Boraginaceae	*Omphalodes nitida* (F)	4	0	0
Plants	Campanulaceae	*Jasione montana* (F)	2	1	0
Plants	Caprifoliaceae	*Lonicera periclymenum* (F)	5	2	0
Plants	Caryophyllaceae	*Arenaria montana* (F)	8	4	4
Plants	Caryophyllaceae	*Silene nutans* (F)	5	4	2
Plants	Caryophyllaceae	*Silene vulgaris* (F)	3	0	1
Plants	Cistaceae	*Halimium lasianthum* (NF)	6	5	7
Plants	Cistaceae	*Xolantha globulariifolia* (NF)	2	1	0
Plants	Compositae	*Centaurea limbata* (NF)	2	0	0
Plants	Compositae	*Centaurea nigra* (NF)	2	1	0
Plants	Compositae	*Cirsium filipendulum* (F)	8	1	0
Plants	Compositae	*Conyza* sp1 (NF)	0	1	0
Plants	Compositae	*Crepis lampsanoides* (F)	8	0	0
Plants	Compositae	*Hieracium* sp1 (F)	4	0	0
Plants	Compositae	*Hieracium* sp2 (F)	0	1	0
Plants	Compositae	*Hypochoeris glabra* (NF)	1	0	0
Plants	Compositae	*Hypochoeris radicata* (NF)	8	2	1
Plants	Compositae	*Picris hieracioides* (NF)	5	0	0
Plants	Convolvulaceae	*Cuscuta epithymum* (NF)	0	1	1
Plants	Crassulaceae	*Sedum spp.* (NF)	1	0	0
Plants	Ericaceae	*Arbutus unedo* (F)	0	0	1
Plants	Ericaceae	*Calluna vulgaris* (NF)	2	6	5
Plants	Ericaceae	*Daboecia cantabrica* (NF)	2	4	6
Plants	Ericaceae	*Erica arborea* (F)	7	4	3
Plants	Ericaceae	*Erica ciliaris* (NF)	1	3	3
Plants	Ericaceae	*Erica cinerea* (NF)	5	6	8
Plants	Ericaceae	*Erica umbelata* (NF)	0	5	7
Plants	Ericaceae	*Vaccinium myrtillus* (F)	5	0	3
Plants	Euphorbiaceae	*Euphorbia dulcis* (F)	8	0	0
Plants	Fagacea	*Quercus ilex* (F)	1	3	1
Plants	Fagacea	*Quercus pyrenaica* (F)	8	2	1
Plants	Fagacea	*Quercus robur* (F)	6	8	2
Plants	Gramineae	*Agrostis curtisii* (NF)	4	9	7
Plants	Gramineae	*Agrostis spp.* (NF)	9	8	8
Plants	Gramineae	*Agrostis truncatula* (NF)	2	1	3
Plants	Gramineae	*Anthoxanthum odoratum* (F)	8	2	1
Plants	Gramineae	*Arrhenatherum elatius* (F)	9	8	4
Plants	Gramineae	*Briza maxima* (NF)	0	1	0
Plants	Gramineae	*Dactylis glomerata* (F)	4	1	0
Plants	Gramineae	*Festuca spp.* (nd)	1	0	0
Plants	Gramineae	Gramineae 1 (nd)	0	1	1
Plants	Gramineae	*Holcus lanatus* (NF)	3	0	0
Plants	Gramineae	*Lolium spp.* (NF)	3	1	0
Plants	Gramineae	*Vulpia bromoides* (NF)	1	0	1
Plants	Guttiferae	*Hypericum humifusum* (NF)	1	0	0
Plants	Guttiferae	*Hypericum linarifolium* (NF)	0	4	3

Plants	Guttiferae	*Hypericum perforatum* (F)	1	0	0
Plants	Hypolepidaceae	*Pteridium aquilinum* (F)	9	9	5
Plants	Juncaceae	*Luzula spp.* (nd)	2	0	0
Plants	Labiatae	*Clinopodium vulgare* (F)	4	0	1
Plants	Labiatae	*Melittis mellissophylum* (F)	0	1	0
Plants	Labiatae	*Prunella vulgaris* (F)	3	0	0
Plants	Labiatae	*Teucrium scorodornia* (F)	9	5	3
Plants	Lauraceae	*Laurus nobilis* (F)	0	1	0
Plants	Leguminosae	*Acacia dealbata* (NF)	0	0	1
Plants	Leguminosae	*Cytisus spp.* (F)	9	5	6
Plants	Leguminosae	*Genista falcata* (F)	0	1	1
Plants	Leguminosae	*Genista micrantha* (NF)	0	1	1
Plants	Leguminosae	*Lathyrus linifolius* (F)	1	0	0
Plants	Leguminosae	*Lotus corniculatus* (NF)	5	1	1
Plants	Leguminosae	*Pterospartum tridentatum* (NF)	5	2	1
Plants	Leguminosae	*Ulex europaeus* (NF)	1	8	5
Plants	Leguminosae	*Ulex minor* (F)	6	6	4
Plants	Liliaceae	*Asphodelus lusitanicus* (F)	9	0	0
Plants	Liliaceae	*Simethis mattiazzi* (F)	6	9	7
Plants	Malvaceae	*Malva tournefortiana* (F)	1	0	0
Plants	Myrtaceae	*Eucalyptus globulus* (F)	0	0	8
Plants	Pinaceae	*Pinus pinaster* (F)	1	7	3
Plants	Pinaceae	*Pinus pinea* (F)	1	0	1
Plants	Plantaginaceae	*Plantago lanceolata* (NF)	3	0	0
Plants	Polygalaceae	*Polygala serpyllifolia* (NF)	4	1	0
Plants	Polygalaceae	*Polygala vulgaris* (NF)	5	2	3
Plants	Polygonaceae	*Rumex acetosa* (F)	3	1	0
Plants	Polygonaceae	*Rumex acetosella* (NF)	2	1	1
Plants	Ranunculaceae	*Anemone trifolia* (F)	7	1	0
Plants	Ranunculaceae	*Aquileguia vulgaris* (F)	2	0	0
Plants	Ranunculaceae	*Ranunculus bulbosus* (NF)	3	0	0
Plants	Ranunculaceae	*Thalictrum speciosissimum* (F)	1	0	0
Plants	Rhamnaceae	*Frangula alnus* (F)	6	4	2
Plants	Rosaceae	*Potentilla erecta* (NF)	9	4	1
Plants	Rosaceae	*Pyrus cordata* (F)	4	1	0
Plants	Rosaceae	*Rubus spp.* (F)	9	6	3
Plants	Rubiaceae	*Galium broteroanum* (F)	1	0	0
Plants	Rubiaceae	*Galium lucidum* (NF)	1	0	0
Plants	Rubiaceae	*Galium rotundifolium* (F)	2	0	0
Plants	Rubiaceae	*Sherardia arvensis* (NF)	1	0	0
Plants	Santalaceae	*Thesium pyrenaicum* (NF)	0	1	0
Plants	Saxifragaceae	*Saxifraga spathularis* (F)	1	0	0
Plants	Scrophulariaceae	*Digitalis purpurea* (F)	2	1	2
Plants	Scrophulariaceae	*Melampyrum pratense* (F)	7	0	0
Plants	Scrophulariaceae	*Rhinanthus minor* (NF)	2	0	0
Plants	Scrophulariaceae	*Veronica officinalis* (F)	3	0	0
Plants	Scrophulariaceae	*Veronica* sp1 (nd)	0	1	0
Plants	Umbelliferae	*Carum verticillatum* (NF)	0	1	0
Plants	Umbelliferae	*Conopodium pyrenaeum* (F)	1	0	0
Plants	Umbelliferae	*Eryngium juresianum* (F)	3	0	0
Plants	Umbelliferae	*Physospermum cornubiense* (F)	8	1	0
Plants	Violaceae	*Viola lactea* (NF)	3	3	0
Plants	Violaceae	*Viola riviniana* (F)	1	1	0
Plants	Violaceae	*Viola spp.* (nd)	9	2	0
Plants	--	Plant 1 (nd)	0	0	1
Plants	--	Plant 2 (nd)	1	0	0
Birds	Accipitridae	*Accipiter nisus* (F)	0	1	0
Birds	Accipitridae	*Buteo buteo* (NF)	0	2	0
Birds	Aegithalidae	*Aegithalos caudatus* (F)	2	1	0
Birds	Alaudidae	*Alauda arvensis* (NF)	1	0	0

Birds	Caprimulgidae	*Caprimulgus europaeus* (F)	0	1	1
Birds	Certhiidae	*Certhia brachydactyla* (F)	8	7	0
Birds	Columbidae	*Columba livia* (NF)	0	1	0
Birds	Columbidae	*Columba palumbus* (F)	0	1	0
Birds	Columbidae	*Streptopelia decaocto* (NF)	0	1	2
Birds	Columbidae	*Streptopelia turtur* (NF)	0	1	3
Birds	Corvidae	*Corvus corone* (NF)	0	2	0
Birds	Corvidae	*Garrulus glandarius* (F)	7	5	4
Birds	Emberizidae	*Emberiza cia* (NF)	7	3	3
Birds	Falconidae	*Falco peregrinus* (NF)	0	1	0
Birds	Falconidae	*Falco tinnunculus* (NF)	0	0	1
Birds	Fringillidae	*Carduelis cannabina* (NF)	3	5	2
Birds	Fringillidae	*Carduelis chloris* (F)	4	8	6
Birds	Fringillidae	*Fringilla coelebs* (F)	7	8	1
Birds	Fringillidae	*Pyrrhula pyrrhula* (F)	7	2	5
Birds	Fringillidae	*Serinus serinus* (NF)	1	4	1
Birds	Hirundinidae	*Hirundo rustica* (NF)	0	4	2
Birds	Muscicapidae	*Erithacus rubecula* (NF)	8	6	4
Birds	Muscicapidae	*Ficedula hypoleuca* (F)	3	1	0
Birds	Muscicapidae	*Saxicola torquata* (NF)	0	1	1
Birds	Oriolidae	*Oriolus oriolus* (F)	0	1	0
Birds	Paridae	*Parus ater* (F)	9	8	7
Birds	Paridae	*Parus caeruleus* (F)	7	5	2
Birds	Paridae	*Parus cristatus* (F)	9	8	7
Birds	Paridae	*Parus major* (F)	3	3	0
Birds	Passeridae	*Passer domesticus* (NF)	0	4	0
Birds	Picidae	*Dendrocopos major* (F)	1	5	1
Birds	Picidae	*Picus viridis* (NF)	8	7	4
Birds	Prunellidae	*Prunella modularis* (NF)	6	2	3
Birds	Prunellidae	*Regulus ignicapillus* (F)	5	6	2
Birds	Sittidae	*Sitta europaea* (F)	5	0	0
Birds	Strigidae	*Athene noctua* (NF)	0	0	1
Birds	Strigidae	*Otus scops* (NF)	0	0	1
Birds	Strigidae	*Strix aluco* (F)	2	0	1
Birds	Sylviidae	*Cisticola juncidis* (NF)	0	1	0
Birds	Sylviidae	*Phylloscopus collybita* (F)	0	1	0
Birds	Sylviidae	*Phylloscopus ibericus* (F)	3	0	1
Birds	Sylviidae	*Sylvia atricapilla* (NF)	9	7	4
Birds	Sylviidae	*Sylvia undata* (NF)	5	2	4
Birds	Troglodytidae	*Troglodytes troglodytes* (NF)	9	9	7
Birds	Turdidae	*Turdus merula* (NF)	8	6	2
Birds	Turdidae	*Turdus philomelos* (F)	0	1	0
Birds	Turdidae	*Turdus viscivorus* (F)	1	1	1

NATURAL OAK FORESTS AND BIODIVERSITY CONSERVATION IN A MULTI-HABITAT LANDSCAPE.

Proença VM, Pereira HM, Vicente L. Natural oak forest patches and biodiversity conservation in a multi-habitat landscape. *In preparation*

3 Natural oak forest patches and biodiversity conservation in a multi-habitat landscape.

Abstract

European temperate broadleaved forests are severely fragmented due to a long history of human activity. Nowadays, remaining forest patches occur in a matrix of human modified habitats and forest is expanding due to natural regeneration in former agricultural land. While deforestation is no longer a threat, the effects of forest regeneration for biodiversity conservation in countryside landscapes are a debated issue. This study was conducted in a countryside landscape where land abandonment is occurring. The main objective was to analyse the current contribution of forest patches for biodiversity at the landscape level. Results suggest that although fragmentation has promoted changes in the composition of plant communities in forest patches, they still maintain a distinct composition and support several exclusive species that are not found in agricultural or shrublands habitats, thus providing an important contribute for species diversity in the landscape.

Keywords

Countryside biogeography, Landscape ecology, Land use change, Oak forest, Plant diversity, Species-area relationship

3.1 Introduction

Landscapes in temperate Europe have been shaped by millennia of human activities, in particular through deforestation and conversion of forests into open habitats (Blondel and Aronson 1999, Farrell et al. 2003). Today, broadleaved and mixed forests in Europe are the biome most affected by anthropogenic fragmentation (Wade 2003).

The fragmentation of natural habitats can cause the decline, or even extinction, of populations unable to persist in fragmented patches or to adapt to modified environments, thus leading to biodiversity loss at local level, or even at larger spatial scales (Daily et al. 2001, Saunders et al. 2001). On the other hand, some species may endure habitat loss and colonize modified habitats, while other species may even benefit from the new conditions (e.g., landscape heterogeneity) and increase in abundance (Davies et al. 2000, Atauri and Lucio 2001, Hughes et al. 2002, Pereira and Daily 2006). Therefore, remnants of natural habitats and human modified habitats should be equally considered in the analysis of the effects of land use change on biodiversity, in order to better understand species dynamics in the landscape (Rickets et al. 2001, Bennett et al. 2006, Kupfer et al. 2006).

While deforestation and fragmentation are a growing threat in tropical countries (FAO 2006, Shvidenko et al. 2005), in Europe 50%-70% of forest cover was lost during the Middle Ages (Shvidenko et al. 2005) and deforestation does no longer represents a threat for conservation (FAO 2006, EEA 2008). On the contrary, the current regeneration of forests in former agricultural land after millennia of land use is now a matter under discussion (MacDonald et al. 2000, Chazdon 2008) in particular in what regards the potential effects on biodiversity in countryside landscapes. Given this context, the main objective of this study was to assess the current contribution of oak forest patches for biodiversity in a countryside landscape in the NW of Portugal where land abandonment is currently occurring. We approach this issue through the analysis of communities' composition and species-area relationships in a multi-habitat landscape and in continuous forest.

3.2 Methods

Study sites

The main bulk of this study was conducted in the Castro Laboreiro parish (42º N and 8º 10' W; elevation 1000 m) in the Peneda-Gerês National Park (PGNP), NW Portugal (Figure 3.1). The long history of land use in this region makes it particularly interesting for the study of biodiversity in a countryside context. The first signs of human occupation date from 6000 BP (Honrado 2003). Farming and animal husbandry caused a severe loss of forest cover, through deforestation and burning, and shaped the landscape in a mosaic of agricultural fields (small-scale farming), pastures and shrublands (maintained for grazing) and natural forest patches. Presently, and due to the combination of economic growth and intense rural exodus, the agricultural fields still in use are mostly maintained as pastures, for animal grazing and /or for the production of fodder, and are not used for food production. We have also collected data on an area of continuous forest, the "Mata de Palheiros" (41º 48' N and 8º 9' W; elevation 750 m), which integrates the European Network of Biogenetic Reserves (Council of Europe) and distances 25 km from Castro Laboreiro. Fire is the principal cause of disturbance in this area with human activities having a lower impact.

The climate in PGNP is temperate, with an annual precipitation ranging from 1500 mm in lower regions up to 3000 mm in mountainous areas. Mean temperatures range from 4 ºC to 9 ºC in January and from 15 ºC to 21 ºC in July (Honrado 2003).

Sampling methods

We sampled five habitat mosaics in Castro Laboreiro, with the following land cover composition (Figure 3.1): 50% oak forest and 50% shrubland (1); 50% oak forest and 50% agriculture (1), 50% shrubland and 50% agriculture (1); 33% oak forest, 33% shrubland and 33% agriculture (2).

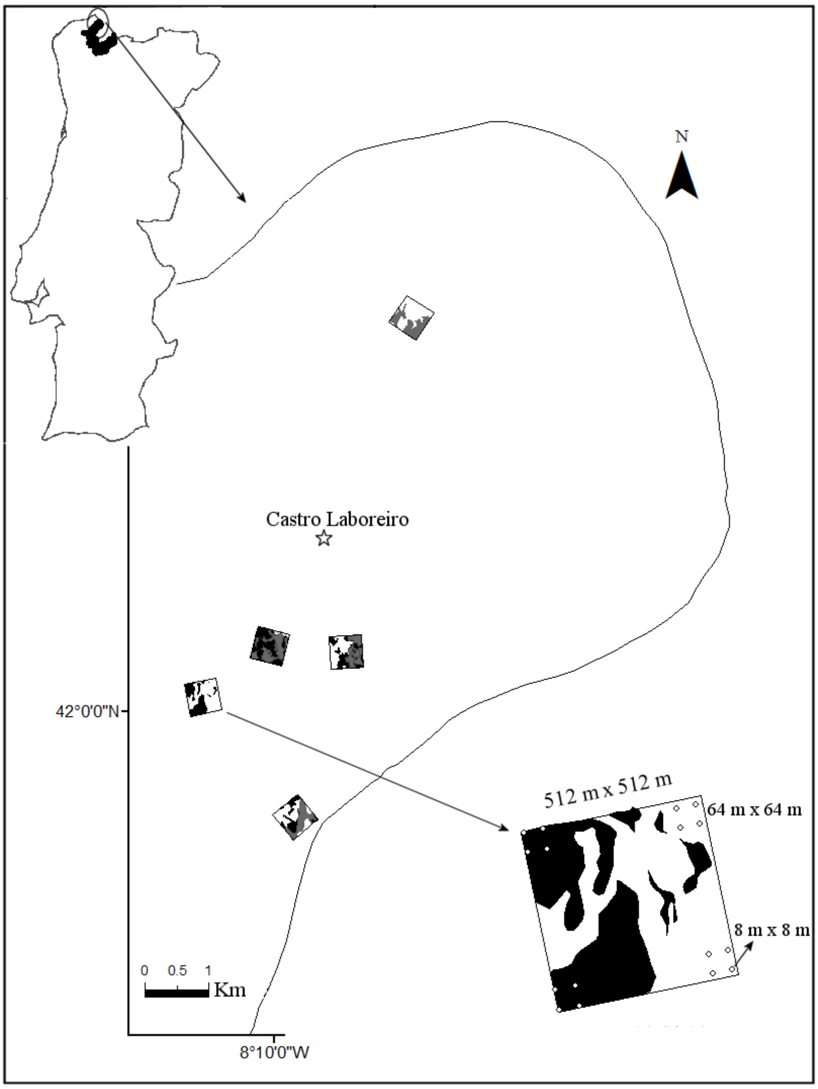


Figure 3.1 - Study area location in Peneda-Gerês National Park, NW Portugal (top left map) and distribution of habitat mosaics in the study area (black - oak forest, grey - agricultural fields, white - shrubland) (central map). Nested sampling scheme (see text for description; bottom right figure).

In each mosaic, we recorded the presence and the percentage cover of understory plant species in 64 plots of 1 m^2 (1 m x 1 m) using the Braun Blanquet's cover scale (Kent and Coker 1994). The disposition of sampling plots (1 m^2) in each mosaic followed a nested scheme (Figure 3.1): 1 m^2 plots were aggregated in groups of four, each group forming a 8 m x 8m square (64 m^2), then 8 m x 8 m squares were aggregated to form 64 m x 64 m (4096 m^2) squares and these were finally aggregated in one 512 m x 512 m square (26.2 ha; mosaic size). Most 64 m x 64 m squares were composed by only one type of habitat (forest, agriculture or shrubland), the exception were two squares, each one located in a three-habitat mosaic, that integrated both forest and agricultural habitats. This sampling scheme was repeated in Mata de Palheiros in an area of 512 m x 512 m with 100% forest cover. Percentage cover of each habitat in each mosaic was calculated using the ArcGIS 9.0 software and a land cover map produced by the technical staff of the PGNP. Data were collected in June and July 2006.

Data analysis

To assess plants' habitat affinity we performed principal components analysis (PCA) using taxa cover data in 1 m^2 plots as variables. Plant taxa were divided in three habitat affinity groups (oak forest, agriculture and shrubland) according to their loadings on the first and the second components of PCA (PC1, 8.94%; PC2, 4.32%) (Appendix 3.1). Only taxa that occurred in at least 5 sampling plots were used in this analysis, in a total of 91 taxa. The remaining taxa were associated to the habitat in which they occurred most and eventual ties were resolved using ecologic information in Honrado (2003). Data from Mata de Palheiros were not used in the determination of species groups, because the floristic combinations at oak forest patches in Castro Laboreiro and in Mata de Palheiros were distinct. In Mata de Palheiros we found *Ruscus aculeatus* which is an indicator of *Rusco-Quercetum roboris* communities (more thermophile) and in Castro Laboreiro we found oak forests of *Myrtillo-Quercetum roboris* (Honrado 2003). The determination of the habitat affinity of taxa observed in Mata de Palheiros followed the results of the ordination of Castro Laboreiro taxa. In the case of species that were just observed in Mata de Palheiros, we followed the ecological information in Honrado (2003), and associated all species that were reported to occur in forests to this type of habitat.

The similarity in species composition between forest, agricultural and shrubland habitats in Castro Laboreiro was calculated using the Sørensen similarity index applied to presence-absence data from 64 m x 64 m plots.

To study species diversity in the multi-habitat landscape we analysed the following species-area relationships (SARs):

i) SARs of species groups in each type of habitat (only areas with a single habitat were used: 1 m^2, 64 m^2 and 4096 m^2);

ii) SARs of each species group in the multi-habitat landscape;

iii) SARs of all species in the multi-habitat landscape;

In i) we used the classic SAR ($S = cA^z$, where S is the number of species, A is the area, and c and z are parameters (Arrhenius 1921)), in its log-log form ($\log S = c + z \log A$, species data were (x+1) transformed).

In ii) and iii) we used two approaches to determine multi-habitat SARs: the classical SAR and the countryside SAR (Pereira and Daily 2006). The countryside SAR, accounts for a differential use of habitats by different species groups (Pereira and Daily 2006):

$$S_i = c_i \left(\sum_j h_{ij} A_j \right)^z$$

S_i is the number of species in group i, h_{ij} is the affinity of group i to habitat j and A_j is the area covered by the habitat j. The total number of species in the landscape, where m is the number of species groups, is:

$$S = \sum_{i=1}^{m} S_i \ ,$$

The models were fit to the dataset choosing the parameters that minimized the Sum of Squares error (SSE) and were compared using the Akaike's Information Criterion (AIC). The AIC is calculated as, AIC = -2 ln (maximum likelihood of the model) + 2 (n parameters), and model selection is based on the comparison of AICs (models with a lower AIC have better fit). In the case of the countryside SAR, the habitat affinity of each group in its preferred habitat was assumed as $h=1$ (maximum affinity, $0<h<1$), thus for the countryside SAR of each species group four parameters were found: c, z, h_i and h_j ($h_{i,j,}$ $i \neq j$,represents the affinity to "non preferred" habitats).

We also compared the SARs of forest species in continuous forest (Mata de Palheiros) and in fragmented forest (Castro Laboreiro). The SAR in fragmented forest was determined using data from two habitat mosaics in Castro Laboreiro (both with 50% forest) complemented with data on the total number of forest species found in forest patches partially included in these mosaics. Data on total species richness in forest patches were collected in a previous study (Proença et al. in review), and were added to the dataset because the largest area sampled in Castro Laboreiro with 100% forest cover was of 4096 m^2 (64 m x 64 m). The log-log form of the classic SAR was used in this analysis.

The Sørensen similarity index was calculated using the EstimateS software (Colwell 2005). Statistical analyses were performed in the R statistical software system (www.cran.r-project.org) and Mathematica 7.0 was used to fit and evaluate multi-habitat SARs.

3.3 Results

We observed a total of 143 plant taxa in Castro Laboreiro and 41 in Mata de Palheiros (Table 3.1, Annex 3.1). Some records were not identified up to the species level but, because all records correspond to different taxonomic units, they were considered as separate species in data analysis and hereafter will be referred to as species. Moreover, species groups will be designated by their habitat affinity: forest species, agricultural species and shrubland species.

Table 3.1 – Number of species and the number of families they represent (in brackets) observed in each habitat and divided by habitat affinity.

	Mata de Palheiros	Castro Laboreiro		
Habitat affinity:	Forest (continuous)	Forest (fragmented)	Agriculture	Shrubland
Forest	33(23)	44 (23)	19 (14)	14 (11)
Agriculture	3 (3)	17 (12)	68 (25)	16 (12)
Shrubland	5 (3)	18 (10)	13 (5)	28 (13)
Total	41 (25)	79 (29)	100 (29)	58 (24)

Three-habitat mosaics were the most species rich, while the mosaic in continuous forest was the least rich of all mosaics (Table 3.2).

Regarding plant communities' composition, forest and shrubland were the most similar (mean Sorensen index = 0.34), followed by shrubland and agricultural fields (mean Sorensen index = 0.24) and lastly, forest and agricultural fields (mean Sorensen index = 0.19).

Table 3.2 – Total species richness and richness of species groups in each habitat mosaic.

		Species habitat affinity		
Mosaic composition	Total	Forest	Agric.	Shrubl.
50% Forest : 50% Shrubland	57	35	7	15
50% Forest : 50% Agriculture	79	31	35	13
50% Shrubland : 50% Agriculture	75	13	43	19
33% Forest : 33% Shrubland : 33% Agriculture	90	29	41	20
33% Forest : 33% Shrubland : 33% Agriculture	86	28	37	21
100 % Forest	41	33	3	3

Although species richness of all groups was significantly related with area in every habitat type, the number of species was better determined by area when groups were tested in the preferred habitat (Table 3.3).

Table 3.3 – Species area relationships of each species group in each type of habitat ($p < 0.0001$ for all regressions).

	Forest				Agriculture				Shrubland			
Species habitat affinity	c	z	r^2	n	c	z	r^2	n	c	z	r^2	n
Forest	6.96	0.16	0.64	146	1.55	0.16	0.34	146	1.90	0.14	0.28	126
Agriculture	1.34	0.13	0.28	146	9.12	0.15	0.41	146	1.26	0.1	0.19	126
Shrubland	2.20	0.14	0.25	146	3.0	0.12	0.30	146	4.85	0.12	0.42	126

When the three habitats were simultaneously included in species-area relationships, the countryside SAR provided models that were better fit than the ones obtained using the classic approach (Table 3.4).

Table 3.4 – Species-area relationships (classic: SAR and countryside: C. SAR) for each species groups in the multi-habitat context. Habitat affinity: affinity for oak forest (h_F), agricultural habitats (h_A) and shrubland(h_S). For each pair of models, the best model fit has the lower AIC.

Species groups	c	z	h_F	h_A	h_S	adj r^2	AIC
Forest							
SAR	2.85	0.18	--	--	--	0.64	2397
C. SAR	6.5	0.14	1	0.001	0.001	0.89	1882
Agriculture							
SAR	3.18	0.18	--	--	--	0.51	2737
C. SAR	9.05	0.13	0.001	1	0.001	0.79	2373
Shrubland							
SAR	2.57	0.15	--	--	--	0.79	1819
C. SAR	4.12	0.13	0.002	0.016	1	0.88	1593

For all species in the landscape the countryside SAR also provided a better model than the classic SAR:

C. SAR $_{(SSE = 10539)}$: $S_{total} = 8.59 + Area^{0.17}$

Cs. SAR $_{(SSE = 8890)}$: $S_{total} = S_{forest} + S_{agriculture} + S_{shrubland}$ (=)

$S_{tota} =(6.5\ (A_f + 0.001A_a + 0.001A_s)^{0.14})+(9.05\ (0.001A_f + A_a + 0.001\ A_s)^{0.13})+(4.12\ (0.002A_f + 0.016A_a + A_s)^{0.13})$

The SAR of forest species in continuous forest had a lower intercept (*c*) and a higher slope (*z*) than SARs in fragmented forest (Figure 3.2).

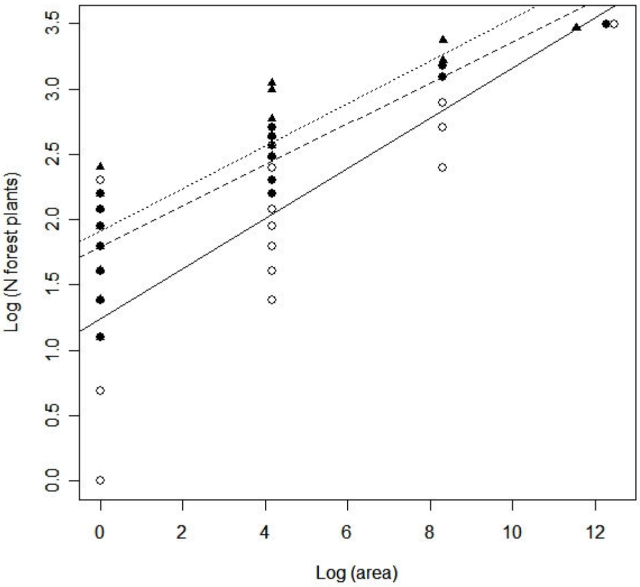


Figure 3.2 – Species area relationships of forest species in fragmented and continuous oak forest. Continuous forest: open circles, solid line (c = 3.46, z = 0.19, r^2 = 0.45). Fragmented forest: closed circles, dashed line (c = 6.0, z = 0.16, r^2 = 0.78) and closed triangles, dotted line (c = 6.75, z = 0.16, r^2 = 0.68) (p < 0.0001 in all regressions).

3.4 Discussion

Our results suggest that plant communities in each habitat maintain a distinct core of species while also comprising species characteristic from other habitats. In the case of forest communities, more than half of species were classified as forest species (communities include species from all groups) and half of forest species were restricted to forest habitats (22 in 44 species). Moreover, values of similarity were low, reinforcing the distinct character of communities. Forest and shrubland were the most similar, which may be due to the higher naturalness (i.e., the degree to which an area is free of human influence (Boteva et al. 2004)) of these habitats when compared to agricultural habitats.

Regarding species diversity in habitat mosaics, our results suggest that species diversity is more related with the number of habitats than with the proportion of the richest habitat in the mosaic. The predictive value of habitat diversity in determining species richness is usually tested against the effect of area, because both variables tend to contribute to species accumulation (Triantis et al. 2003). In this case, species richness in habitat mosaics was determined by habitat diversity because mosaics were equally sized. The relationship between habitat diversity and species diversity is not always straightforward, total species richness will also depend on the level of overlap between the communities present (Tjorve 2002). When plant communities are the object of study, as in our study, the relationship between habitat diversity and species diversity tends to be more direct because habitats are characterized by dominant plant formations (Rickleffs and Miller 1999).

Because we did not know the composition of pristine communities, our perception about compositional changes, including species loss, was limited. A possible approach to deal with this limitation was to compare data collected in continuous forest with data from fragmented forest.

The proportion of non-forest species found in fragmented forest was much higher than in continuous forest, suggesting that forest communities in the multi-habitat mosaic suffered relevant compositional changes. The higher number of agricultural and shrubland species in fragmented forest may result from the greater extension of edges in forest patches, in relation to continuous forest, that provide the conditions for the establishment of species from the matrix (Tabarelli et al. 1999, Davies et al. 2001).

The larger number of forest species associated with forest patches in the multi-habitat context, should be analysed cautiously because the determination of species groups was performed using data obtained in the multi-habitat context, therefore, species groups correspond to the species that are nowadays associated with conditions in forest patches and not with conditions in undisturbed continuous forest. This includes for example, the species in forest edges that may contribute for the higher richness in fragmented forest patches (Fagan et al. 1999, Davies-Colley et al. 2000).

Parameters from SARs suggest that plant communities in forest patches present a high alfa diversity (local species richness) but a low beta diversity (species turnover) when compared to the plant community in continuous forest (single habitat). Moreover, all plant communities in the multi-habitat landscape presented a species turnover, inferior to the one found in the single-habitat community.

It has been suggested that countryside landscapes as the one we studied, are usually characterized by low values of alfa diversity and by high values of beta diversity when compared to unmodified habitats (Blondel and Aronson 1999). Lower values of alfa diversity would be expected as a result of the limited size of habitat patches, which in the case of native habitats would also cause the decline of populations or even species loss. However, we have found high values of local richness that may be due to species dynamics in the landscape, with forest species using matrix habitats and vice versa. Regarding beta diversity, species turnover would be higher in landscape mosaics than in continuous habitat, due to the transitions between different communities. Although species turnover was high in habitat mosaics, SARs at the landscape scale revealed smooth slopes, suggesting a more or less homogenous distribution of species at the landscape scale.

These results suggest that we may be observing a stabilization effect on landscape composition, caused by species transition between habitats, which would cause the decrease of species turnover while increasing local richness (Davis et al. 2001). Moreover, species natural regeneration, promoted by reduced human activity, may also contribute for this effect.

On the whole and despite compositional changes, oak forest patches maintain a distinct composition and support several exclusive species that are not found in agricultural or shrublands habitats, thus providing an important contribute for species diversity in the landscape. The investigation of other taxa, in particular vagile fauna (e.g., birds) that select actively which habitats to use, may provide more information on the relevance of oak forest patches for biodiversity conservation. For example, Rickets et al. (2001) studied moths in a countryside landscape in Costa Rica and found that although moths used agricultural habitats, their richness varied negatively with distance to native forest patches. Hughes et al. (2002) working in that same region found that although 46% of forest birds used human modified habitats the disappearance of tall trees from the landscape could result in the decline 40% of forest bird richness.

Although we have not focused our study on the significance of fragment size, the higher SAR slope in continuous forest is supported by the results from a previous study which show that fragment size has an important effect on the diversity of forest species (Proença et al. in review). Therefore, large forest fragments are particularly important for biodiversity conservation. Moreover, oak forest patches also provide a wide range of ecosystem services, including important services for the maintenance of the other habitats, such as regulation of the water cycle or pollination.

While forest regeneration may benefit species that use or even depend of forest habitats, such as the Iberian wolf and several forest birds, it may also cause the decline of other species, in particular farmland birds (Blondel and Aronson 1999, Moreira et al. 2001).

Therefore, land abandonment and forest regeneration pose a huge challenge to landscape management. Future management options should target solutions that preserve biodiversity but also consider the ecosystem services and human well-being.

Acknowledgments We thank Cibele Queiroz for assistance in field work, Ana Ceia Hasse for company and discussions during field work and João Honrado for helping in plant identification. Vânia M. Proença was supported by a fellowship from Fundação para a Ciência e a Tecnologia (BD/12395/2003). Field work was supported by the Ecoforsite project (POCI-AMB-55701/2004) and the ABAFOBIO project (PTDC/AMB/73901/2006).

References

Atauri JA, de Lucio JV (2001) The role of landscape structure in species richness distribution of birds, amphibians, reptiles and lepidopterans in Mediterranean landscapes. Landscape Ecology 16:147-159.

Blondel J, Aronson J (1999) Biology and wildlife in the Mediterranean region. Oxford University Press, Oxford

Bennett AF, Radford JQ, Haslem A (2006) Properties of land mosaics: Implications for nature conservation in agricultural environments. Biological Conservation 133:250-264.

Boteva D, Griffiths G, Dimopoulos P (2004) Evaluation and mapping of the conservation significance of habitats using GIS: an example from Crete, Greece. Journal for Nature Conservation 12:237-250.

Chazdon RL (2008) Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands. Science 320:1458-1460.

Colwell RK (2005) EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.5. User's Guide and application published at: http://purl.oclc.org/estimates

Daily GC, Ehrlich PR, Sanchez-Azofeifa GA (2001) Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11:1-13.

Davies KF, Margules CR, Lawrence JF (2000) Which traits of species predict population declines in experimental forest fragments? Ecology 81:1450-1461.

Davies KF, Melbourne BA, Margules CR (2001) Effects of within- and between-patch processes on community dynamics in a fragmentation experiment. Ecology 82:1830-1846.

Davies-Colley RJ, Payne GW, van Elswijk M (2000) Microclimate gradients across a forest edge

New Zealand Journal of Ecology 24: 111-121.

EEA – European Environmental Agency (2008) European forests - ecosystem conditions and sustainable use. EEA report 3/2008

Fagan WF, Cantrell RS, Cosner C (1999) How Habitat Edges Change Species Interactions. The American Naturalist 153:165-182.

FAO (2006) Global Forests Resources Assessment 2005: Progress towards sustainable forest management. FAO, Rome

Farrell EP, Führer E, Ryan D, Andersson F, Hüttl R, Piussi P (2000) European forest ecosystems: building the future on the legacy of the past. Forest Ecology and Management 132:5-20

Hughes JB, Daily GC, Ehrlich PR (2002) Conservation of tropical forest birds in countryside habitats. Ecology Letters 5:121-129.

Honrado JJP (2003) Flora e Vegetação do Parque Nacional da Peneda-Gerês. Dissertação de candidatura ao grau de Doutor em Biologia, Faculdade de Ciências da Universidade do Porto, Porto

Kent M, Coker P (1994) Vegetation description and analysis: a pratical approach. John Wiley and Sons, Chichester

Kupfer JA, Malanson GP[, Franklin SB[(2006) Not seeing the ocean for the islands: the mediating influence of matrix-based processes on forest fragmentation effects. Global Ecology & Biogeography 15:8-20.

MacDonald D, Crabtree JR, Wiesinger G., Dax T, Stamou N, Fleury P, Lazpita JG,. Gibon A (2000) Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. Journal of Environmental Management 59: 47-69

Moreira F, Ferreira PG, Rego FC, Bunting S (2001) Landscape changes and breeding bird assemblages in northwestern Portugal: the role of fire. Landscape Ecology 16: 175-187

Pereira HM, Daily GC (2006) Modelling biodiversity dynamics in countryside landscapes. Ecology 87:1877-1885

Proença VM, Pereira HM, Vicente L The role of natural forests for biodiversity conservation in the NW of the Iberian Peninsula. Submitted to Acta Oecologica (in review).

Ricketts TH, Daily GC, Ehrlich PR, Fay JP (2001) Countryside Biogeography of Moths in a Fragmented Landscape: Biodiversity in Native and Agricultural Habitats. Conservation Biology 15:378-388a

Ricklefs RE, Miller G (1999) Ecology. W. H. Freeman

Saunders DA, Hobbs RJ, Margules CR (1991) Biological Consequences of Ecosystem Fragmentation: A Review. Conservation Biology 5:18-32

Shvidenko A, Barber CV, Persson R (2005) Forest and woodland systems. In: Hassan R., Scholes R., Ash N. (eds) Ecosystems and human well-being: Current state and trends. Island Press, Washington D. C., pp. 585-621

Tabarelli M, Mantovani W, Peres CA (1999) Effects of habitat fragmentation on plant guild structure in the montane Atlantic forest of southeastern Brazil. Biological Conservation 91:119-127.

Tjorve E (2002) Habitat size and number in multi-habitat landscapes: a model approach based on species–area curves. Ecography 25:17–24.

Triantis KA, Mylonas M, Lika K, Vardinoyannis K (2003) A model for the species–area-habitat relationship. Journal of Biogeography 30: 19–27

Wade TG, Riitters KH, Wickham JD, Jones KB (2003) Distribution and causes of global forest fragmentation. Conservation Ecology 7: 7. [online] http://www.consecol.org/vol7/iss2/art7

Appendix 3.1

Plant taxa observed in this study, their habitat affinity (F - oak forest; A - agricultural habitats; S - shrubland), and presence at each study site (CL - Castro Laboreiro; MP - Mata de Palheiros). Habitat affinity was determined through a Principal Components analysis. Taxa were assigned to an affinity group according to their loadings on the first (PC1) and the second (PC2) components of PCA. Only taxa that occurred in at least five sampling plots were considered in this analysis (see Methods for criteria in the classification of the remaining taxa).

Study site	Family	Species	Habitat affinity	PC1	PC2
CL; MP	Aquifoliaceae	*Ilex aquifolium*	F	--	--
CL; MP	Araliaceae	*Hedera hibernica*	F	0.1492	-0.2744
MP	Aspidiaceae	*Dryopteris* sp1	F	--	--
MP	Aspleniaceae	*Asplenium* sp1	F	--	--
CL	Boraginaceae	*Echium lusitanicum*	A	-0.2388	-0.0135
CL; MP	Boraginaceae	*Lithodora prostrata*	F	0.2498	-0.0293
CL	Boraginaceae	*Myosotis laxa*	A	--	--
CL	Boraginaceae	*Omphalodes nitida*	F	--	--
CL	Campanulaceae	*Campanula lusitanica*	A	-0.2627	0.0133
CL	Campanulaceae	*Jasione montana*	S	0.0250	0.1077
CL; MP	Caprifoliaceae	*Lonicera periclymenum*	F	--	--
CL; MP	Caryophyllaceae	*Arenaria montana*	F	0.1274	-0.2663
CL	Caryophyllaceae	*Cerastium glomeratum*	A	-0.5699	-0.0461
CL	Caryophyllaceae	*Polycarpon tetraphyllum*	A	-0.2069	0.0423
CL	Caryophyllaceae	*Silene nutans*	F	0.0174	-0.0627
CL	Caryophyllaceae	*Silene vulgaris*	A	-0.0259	-0.1137
CL	Caryophyllaceae	*Spergularia capillacea*	A	--	--
CL	Caryophyllaceae	*Stellaria graminea*	A	--	--
CL	Cistaceae	*Cistus psilosepalus*	S	0.0612	0.0014
CL	Cistaceae	*Halimium lasianthum*	S	0.1507	0.3470
CL	Cistaceae	*Xolantha globulariifolia*	S	0.2342	0.3760
CL	Compositae	*Achillea millefolium*	A	-0.5743	-0.0371
CL	Compositae	*Arnica montana*	S	--	--
CL	Compositae	*Carduus platypus*	A	--	--
CL	Compositae	*Centaurea nigra*	A	-0.5405	-0.1018
CL	Compositae	*Centaurea sp1*	S	0.0553	0.2483
CL	Compositae	*Chamaemelum nobile*	A	-0.4555	-0.0168
CL	Compositae	*Cirsium filipendulum*	A	-0.2376	-0.1546
CL	Compositae	*Cirsium* spp.	F	--	--
CL	Compositae	*Crepis capillaris*	A	-0.2328	0.0161
CL; MP	Compositae	*Crepis lampsanoides*	F	0.1522	-0.3393
CL	Compositae	*Hieracium* sp1	F	--	--
CL	Compositae	*Hieracium* sp2	F	--	--
CL	Compositae	*Hypochoeris glabra*	A	-0.3339	-0.0027
CL; MP	Compositae	*Hypochoeris radicata*	A	-0.5733	0.0517
CL	Compositae	*Picris hieracioides*	F	0.1586	-0.2249
CL	Compositae	*Senecio sylvaticus*	A	--	--
CL	Compositae	*Senecio vulgaris*	A	--	--
CL	Compositae	*Serratula tinctoria*	S	--	--
CL	Compositae	*Solidago virgaurea*	S	--	--
CL	Crassulaceae	*Sedum arenarium*	S	0.0006	0.1996
CL	Crassulaceae	*Umbilicus rupestris*	A	--	--

CL	Cruciferae	*Capsella bursa-pastoris*	A	-0.2379	0.1428
CL; MP	Cruciferae	*Coincya monensis*	A	--	--
CL	Cyperaceae	*Carex binervis*	A	-0.3638	0.0047
CL	Cyperaceae	*Carex leporina*	A	-0.0685	0.0184
CL	Cyperaceae	*Carex* sp1	A	--	--
MP	Ericaceae	*Arbutus unedo*	F	--	--
CL; MP	Ericaceae	*Calluna vulgaris*	S	0.1276	0.2523
CL; MP	Ericaceae	*Erica arborea*	F	0.2019	-0.1296
CL	Ericaceae	*Erica ciliaris*	S	--	--
CL; MP	Ericaceae	*Erica cinerea*	S	0.2616	0.4706
CL	Ericaceae	*Erica umbellata*	S	0.1991	0.4188
MP	Ericaceaa	*Vaccinium myrtilus*	F	--	--
CL	Euforbiaceae	*Euforbia amygdaloides*	F	--	--
CL	Euforbiaceae	*Euforbia dulcis*	F	0.1466	-0.2898
CL	Fagaceae	*Fagus sylvatica*	A	--	--
CL	Fagaceae	*Quercus pyrenaica*	F	0.1681	-0.1310
CL; MP	Fagaceae	*Quercus robur*	F	0.0115	-0.1085
CL	Geraniaceae	*Geranium molle*	A	--	--
CL	Geraniaceae	*Geranium robertianum*	A	--	--
CL	Gramineae	*Agrostis curtisii*	S	0.3022	0.3516
CL; MP	Gramineae	*Agrostis spp.*	A	-0.5218	-0.0007
CL	Gramineae	*Agrostis truncatula*	S	0.0634	0.2549
CL	Gramineae	*Anthoxanthum odoratum*	S	-0.5180	-0.0280
CL; MP	Gramineae	*Arrhenatherum elatius*	S	-0.3523	-0.1144
CL	Gramineae	*Avenula sulcata*	S	-0.0830	0.0107
CL	Gramineae	*Briza maxima*	S	-0.2733	-0.0125
CL	Gramineae	*Bromus hordeaceus*	S	-0.4613	-0.0636
CL	Gramineae	*Dactylis glomerata*	S	-0.5309	-0.0093
CL	Gramineae	*Festuca* spp.	S	-0.0131	0.0705
CL	Gramineae	*Gramineae* spp1	F	0.1434	-0.4373
CL	Gramineae	*Gramineae* spp2	F	0.2390	-0.3170
CL	Gramineae	*Holcus lanatus*	A	-0.6058	-0.0629
CL	Gramineae	*Holcus mollis*	A	--	--
CL	Gramineae	*Secale cereale*	A	-0.0563	0.0340
CL	Guttiferae	*Hypericum humifusum*	A	--	--
CL	Guttiferae	*Hypericum linarifolium*	A	--	--
MP	Hypericaceae	*Hypericum perforatum*	F	--	--
CL; MP	Hypolepidaceae	*Pteridium aquilinum*	F	0.1949	-0.3692
CL	Juncaceae	*Juncus effusus*	A	-0.1521	0.0138
CL	Juncaceae	*Luzula* spp.	A	-0.2131	-0.0138
CL; MP	Labiatae	*Clinopodium vulgare*	F	0.1186	-0.2140
CL	Labiatae	*Mentha suaveolens*	A	--	--
MP	Labiatae	*Melittis melissophyllum*	F	--	--
CL	Labiatae	*Prunella grandiflora*	F	--	--
CL	Labiatae	*Prunella vulgaris*	A	--	--
CL	Labiatae	*Scutellaria minor*	A	--	--
CL; MP	Labiatae	*Teucrium scorodonia*	F	0.1195	-0.2107
CL	Labiatae	*Thymus caespititus*	S	--	--
CL; MP	Leguminosae	*Cytisus* spp.	F	0.0247	-0.0521
CL	Leguminosae	*Lathyrus linifolius*	F	--	--
CL	Leguminosae	*Lotus corniculatus*	A	--	--
CL	Leguminosae	*Lotus hispidus*	A	--	--
CL	Leguminosae	*Lotus pedunculatus*	A	-0.2700	-0.0390
CL	Leguminosae	*Ornithopus perpusillus*	A	-0.3110	0.0174

CL; MP	Leguminosae	*Pterospartum tridentatum*	S	0.3132	0.5816
CL	Leguminosae	*Trifolium campestre*	A	-0.1569	-0.0215
CL	Leguminosae	*Trifolium dubium*	A	--	--
CL	Leguminosae	*Trifolium pratense*	A	-0.1640	0.0115
CL	Leguminosae	*Trifolium* sp1	A	-0.3742	-0.0101
CL; MP	Leguminosae	*Ulex minor*	S	0.2634	0.0934
CL	Liliaceae	*Allium scorzonerifolium*	A	-0.2629	-0.0355
CL; MP	Liliaceae	*Asphodelus lusitanicus*	F	0.3116	-0.2986
CL	Liliaceae	*Hyacinthoides hispanica*	A	-0.3770	-0.1677
CL	Liliaceae	Liliacea sp1	A	-0.0005	-0.0034
MP	Liliaceae	Liliacea sp2	F[1]	--	--
MP	Liliaceae	*Polygonatum odoratum*	F	--	--
MP	Liliaceae	*Ruscus aculeatus*	F	--	--
CL	Liliaceae	*Simethis mattiazzi*	S	0.4259	0.1850
CL	Malvaceae	*Malva neglecta*	A	--	--
CL	Papaveraceae	*Ceratocapnos claviculata*	A	--	--
CL	Plantaginaceae	*Plantago lanceolata*	A	-0.7479	-0.0719
CL	Plantaginaceae	*Plantago radicata*	S	0.0449	0.1715
CL	Polygalaceae	*Polygala* spp.	S	0.0273	0.0334
CL	Polygonaceae	*Rumex acetosa*	A	-0.4736	-0.1117
CL	Polygonaceae	*Rumex acetosella*	A	-0.5999	0.0384
CL	Polypodiaceae	*Polypodium* spp.	F	0.0982	-0.1900
CL	Portulacaceae	*Montia fontana*	A	--	--
MP	Primulaceae	*Primula acaulis*	F	--	--
CL; MP	Ranunculaceae	*Anemone trifolia*	F	0.2620	-0.4521
CL	Ranunculaceae	*Caltha palustris*	A	-0.2041	0.0026
CL	Ranunculaceae	*Ranunculus bulbosus*	A	-0.7703	-0.0909
CL	Ranunculaceae	*Ranunculus repens*	A	-0.2217	-0.0276
CL; MP	Rhamnaceae	*Frangula alnus*	F	0.2063	-0.3606
CL	Rosaceae	*Potentilla erecta*	A	-0.0487	-0.1815
CL; MP	Rosaceae	*Pyrus cordata*	F	0.1313	-0.2764
CL; MP	Rosaceae	*Rubus* spp.	F	0.2884	-0.4328
CL	Rubiaceae	*Galium broterianum*	F	--	--
MP	Rubiaceae	*Galium rotundifolium*	F	--	--
CL	Rubiaceae	*Galium saxatile*	S	--	--
CL	Rubiaceae	*Galium* spp.	A	-0.1258	-0.0042
CL; MP	Saxifragaceae	*Saxifraga spathularis*	F	--	--
CL	Scrophulariaceae	*Digitalis purpurea*	F	--	--
CL	Scrophulariaceae	*Linaria saxatilis*	A	--	--
CL	Scrophulariaceae	*Melampyrum pratense*	F	0.2467	-0.4245
CL	Scrophulariaceae	*Rhinanthus minor*	A	-0.4241	-0.0931
CL	Scrophulariaceae	*Veronica arvensis*	A	-0.3649	-0.0153
CL	Scrophulariaceae	*Veronica officinalis*	A	-0.2228	-0.0231
CL	Umbelliferae	*Carum verticillatum*	A	-0.1389	-0.0171
CL	Umbelliferae	*Conopodium majus*	F	0.0317	-0.1965
CL; MP	Umbelliferae	*Eryngium juresianum*	F	0.1380	-0.2534
CL; MP	Umbelliferae	*Laserpitium eliasii*	F	0.1210	-0.2185
CL; MP	Umbelliferae	*Physospermum cornubiense*	F	0.3633	-0.4048
CL	Umbelliferae	Umbelliferae sp1	A	--	--
CL	Violaceae	*Viola lactea*	S	--	--
CL	Violaceae	*Viola palustris*	F	--	--
CL; MP	Violaceae	*Viola riviniana*	F	0.0194	-0.3119
CL	--	Plant 1	A	--	--
CL	--	Plant 2	A	--	--

CL	--	Seedling 1	A	--	--
CL	--	Seedling 2	A	--	--
CL	--	Seedling 3	F	--	--
CL	--	Seedling 4	F	--	--

1 – Liliaceae sp2 was classified as a forest species because it appeared in three of the four 64 m x 64 m plots sampled in Mata de Palheiros.

RESPONSE OF NATURAL BROADLEAVED FOREST AND PINE PLANTATIONS TO A WILDFIRE: FIRE SEVERITY AND POST-FIRE REGENERATION.

Proença VM, Pereira HM, Vicente L. Response of natural broadleaved forest and pine plantations to a wildfire: fire severity and post-fire regeneration. Submitted to Landscape Ecology (*in review*).

4 Response of natural broadleaved forest and pine plantations to a wildfire: fire severity and post-fire regeneration.

Abstract

The response of an ecosystem to disturbance reflects the ecosystem stability, which is determined by two components: resistance and resilience. We addressed both components in order to analyse the response of natural broadleaved forest (*Quercus robur* and *Ilex aquifolium*) and pine plantations (*Pinus pinaster* and *Pinus sylvestris*) to a single fire event that burned more than 4000 ha of land in the Peneda-Gerês National Park (Portugal). Forest resistance was assessed using descriptors of fire severity, including tree mortality, and sapling persistence. Forest resilience was assessed through the comparison of floristic composition, diversity measures and seedling abundance in burned and reference plots. Fire severity in broadleaved transects was in general low and there were no differences in mean tree mortality between burned and reference transects. Fire severity in pine transects was heterogeneous and mean tree mortality was higher in burned transects. Saplings were equally affected in both types of forest. Plant communities in burned broadleaved forest presented a larger overlap with reference communities than plant communities in burned pine forest. Species richness, evenness and Shannon-Wiener diversity were equivalent in burned and reference plots in broadleaved forest while burned plots in pine forest had less species and were less diverse than reference plots. Seedling abundance was similar in burned and reference plots in both forest types. Results suggest a higher resistance and resilience of broadleaved woods that renders them more reliable when it comes to the maintenance of multiple forest services.

Keywords

Fire; Holly; Natural forest; Oak; Pine; Plantations; Resilience; Resistance

4.1 Introduction

Fire is a major disturbance affecting ecosystems. Fire can impact an ecosystem directly, by damaging plant tissues or by causing destruction to the habitat, rendering it unsuitable for fauna. Indirect impacts from fire may include soil erosion, deregulation of local climate or changes in the composition of natural communities (Neary et al. 1999, Brown and Smith 2000).

The ability of an ecosystem to return to the pre-disturbance condition, i.e., ecosystem's stability, is mainly determined by two components: resistance and resilience (Halpern 1988, McCann 2000). Resistance can be defined as the degree to which an ecosystem variable (e.g., canopy cover, vegetation composition or species richness) remains unchanged in face of disturbance and resilience measures the rate of recovery to the pre-disturbance value (Halpern 1988, Lavorel 1999, McCann 2000, Diaz-Delgado et al. 2002). Regarding fire disturbance, forest resistance is particularly associated with susceptibility of dominant trees and forest structure to fire (Brown et al. 2004, González et al. 2006, Fernandes in press). Forest resilience on the other hand is associated with species ability to recolonize the post-fire environment through seed germination or bud sprouting (Pausas and Vallejo 1999).

The time required for post-fire reestablishment and the dynamics of post-fire succession differ amongst communities and are also dependent on the fire regime (Lavorel 1999, Brown and Smith 2000, Domínguez et al. 2002). Plant communities in fire prone ecosystems, such as the Mediterranean-type habitats, encompass species with traits that either provide protection to fire or enhance the post-fire recovery of populations, therefore increasing ecosystem stability (Pausas and Vallejo 1999, Calvo et. al. 2003, Bond and Keely 2005, Verdú and Pausas 2007). Despite the role of fire in shaping these communities, ecosystem stability may be affected by changes in the fire regime or in the composition of communities (Pausas et al. 2004, Bond and Keely 2005). For example, an increase in fire frequency could compromise the total recovery of a plant community between two fires by preventing the reestablishment of slow recovering species while promoting the dominance of fast recovering ones (Lloret et al. 2002, Mouillot et al. 2002, Baeza et al. 2007). These shifts in composition will affect ecosystem stability and increase its vulnerability to disturbance (Folke et al. 2004). In some cases, a new state of stability may be achieved when the former community, unable to recover, is eventually replaced by a simpler one, as in the case of forests giving way to shrubland (Naveh 1975, Lloret et al. 2002, Mouillot et al. 2002, Baeza et al. 2007).

Forest susceptibility to fire has been assessed through different approaches in various studies, from empirical observations (Rigolot 2004, Ordoñez et al. 2005, Moreira et al. in press) to larger scale analyses, using published data and GIS tools (González et al. 2006, Fernandes et al. 2008, Fernandes in press, Silva in press). Post-fire regeneration has been also studied before, usual approaches include analysing tree recovery (Gracia et al. 2002, Brocano et al. 2005, Calvo et al. 2008, Moreira et al. in press) or understory vegetation reestablishment (Domínguez et al. 2002, Calvo et al. 2003, Ladd et al. 2005, Buhk 2006, Baeza et al. 2007). These studies provided important, but nevertheless partial approaches to the understanding of forests response to fire, as most focused either on the resistance or on the resilience of forest systems or were restricted to a single forest type. Moreover, most empirical data on forest vulnerability were collected in the context of prescribed fires (Rigolot 2004, Fernandes 2008).

Here we study and connect both components of ecosystem stability, resistance and resilience, in natural broadleaved forest (*Quercus robur* and *Ilex aquifolium*) and pine plantations (*Pinus pinaster* and *P. sylvestris*) after a single fire, which burned more than 4000 ha in the Peneda-Gerês National Park (NW Portugal) in August 2006. This event created a singular opportunity to study responses to wildfire in two key forest types of the Iberian Peninsula.

Resistance is assessed using fire severity descriptors. Resilience, at an early stage of post-fire reestablishment, is evaluated through the analysis of seedling abundance, species diversity and specific and functional composition of communities. Because forests were geographically close (ca. 5 km) and were affected by the same fire event we expect differences in forest responses to be mainly due to forest features and less dependent, or not dependent at all, of factors such as regional climate or fire season.

4.2 Methods

Study area

This study was conducted in the Soajo Mountains (max. alt. 1416 m) in the Peneda-Gerês National Park (41º N 8º W), NW Portugal. The climate is temperate, with an annual precipitation ranging from 1500mm in lower regions up to 3000 mm in mountainous areas. Mean temperatures range from 4 ºC to 9 ºC in January and from 15 ºC to 21 ºC in July (Honrado 2003).

In August 2006, a fire burned more than 4000 ha in the National Park, in a total area of 6000 ha. The fire occurred under unusual dry conditions after a long period of drought that started in the end of 2004 and lasted until the spring of 2006 (INM 2009). The fire lasted for more than a week producing a mixed-severity mosaic. Two major areas were affected during the fire: an important natural area, Ramiscal, and a forest commons, Mezio (Figure 1).

The dominant species in Mezio are *Pinus pinaster*, *Betula celtiberica* and *Pinus sylvestris* (Sequeira 1995). In Ramiscal the dominant trees are *Quercus robur* and *Ilex aquifolium*. This area is under total protection due to its high natural value, which results from an important floristic diversity, including some rare remnants of climacic communities and unique examples of ancient oaks and hollies (PNPG 1995, Torres et al. 2001).

Sampling methods

We sampled 80 transects: burned broadleaved forest (n=20), burned pine forest (n=20), reference broadleaved forest (n=20) and reference pine forest (n=20). Reference sites were located outside the limits of the burned region (Figure 1, Appendix 1). Broadleaved transects were dominated by oak, *Quercus robur*, and holly, *Ilex aquifolium*, and pine transects by *Pinus pinaster* and *P. sylvestris*. The transects location was determined prior to field sampling using land cover maps, produced by the technical staff of the National Park, and without any prior knowledge on fire severity patterns within the burned region. Data were collected during the first growing season in April and May 2007.

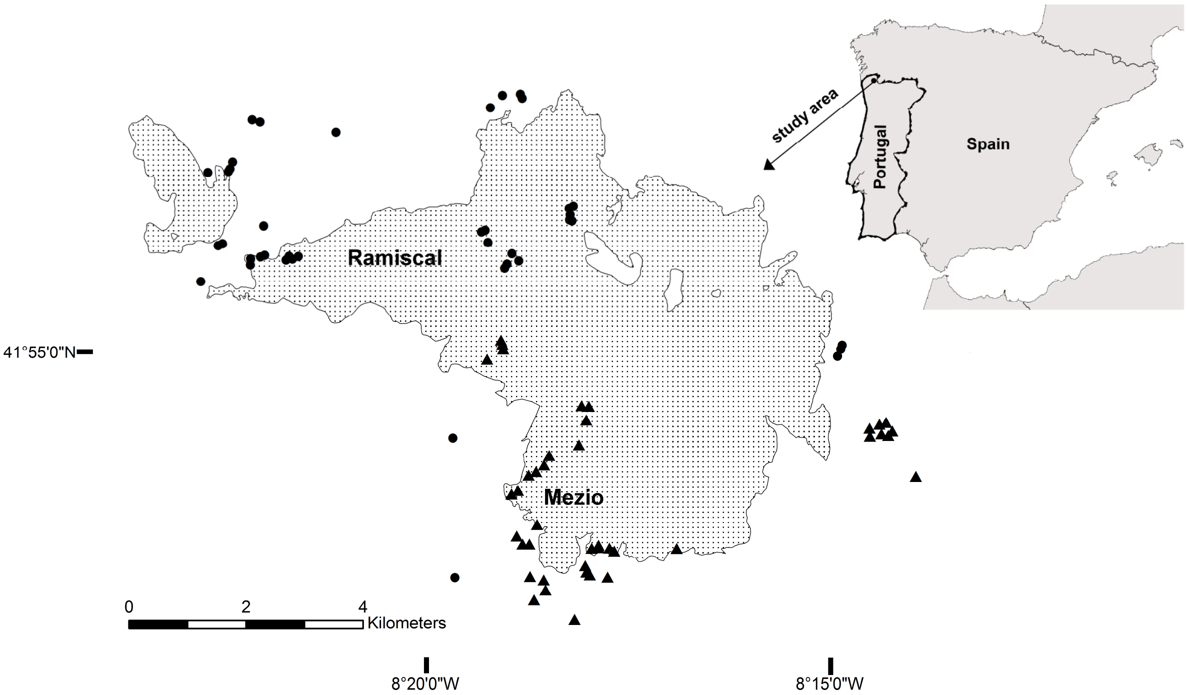


Figure 4.1 - Study area location in NW Portugal (black circle, top right map) and distribution of transects. Transects in broadleaved forest are represented by circles and transects in pine forest by triangles, the shaded region represents the burned area (ca. 4000 ha).

Sampling at each transect (50 m x 10 m) followed the scheme presented in Figure 2. We collected data on tree species, height, DBH, identified dead trees (trees without any green foliage or buds) and counted live saplings. *Frangula alnus* and *Pyrus cordata* were sampled as trees whenever they presented tree morphology. Young seedlings (height < 50 cm) were counted. Cover percentage of vascular plant species (excluding trees) was determined in five squares of 1 m^2 (1 m x 1 m), using the Braun-Blanquet's cover scale (Kent and Coker 1994): + (<1%); 1 (1-5%); 2 (6-25%); 3 (26-50%); 4 (51-75%), 5 (76-100%).

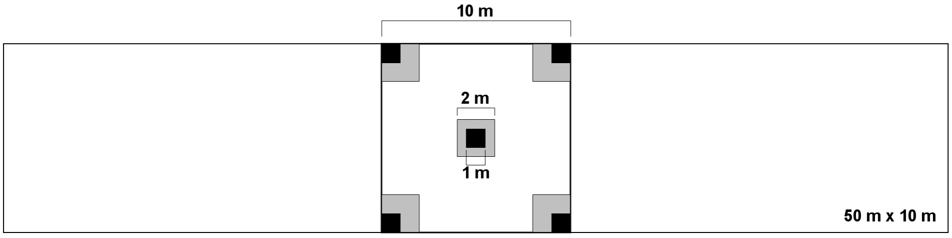


Figure 4.2 - Scheme of sampling at each transect (50 m x 10 m). Trees (height ≥ 1.3 m (breast height) and DBH (diameter at breast height) ≥ 10 cm) and live saplings (height ≥ 1.3 m and DBH < 10 cm) were sampled along each transect (500 m^2), seedlings in five 4 m^2 (2 m x 2 m) sub-plots (grey) and vascular plants in five 1 m^2 (1 m x 1 m) sub-plots (black).

At each transect we evaluated the level of fire severity through a qualitative scale of fire damage (Turner et al. 1999, USDI 2001) (Table 1).

Table 4.1 - Qualitative scale of fire severity.

Fire severity class	Vegetation description
Unburned	Not burned.
Scorched	Tree canopy retains green foliage. Some leaves and small branches scorched, leaves still attached to supporting twigs.
Low severity	Some leaves and twigs consumed in the canopy. Larger branches are undamaged.
Moderate severity	Partial mortality at the canopy level. Leaves and small branches consumed, larger branches persist.
High severity	Total mortality at the canopy level. Some larger branches and trunk may persist but severely burned.

Data analysis

Fire severity, tree mortality (percentage of dead trees per transect) and number of live saplings were used as measures of forest resistance to fire. A chi-square analysis was performed to test the influence of forest type on fire severity. Apart from this analysis, broadleaved and pine forests were tested separately along the study. We tested the effect of slope, aspect, elevation and mean DBH on fire severity using the Spearman's rank correlation. Aspect is a circular variable and was transformed in a measure of northness = cosine (aspect) (Blake and Schuette 2000). Variables independence was also tested using the Spearman's rank correlation. We compared tree mortality and number of live saplings between burned and reference transects using the Welsh test for two samples (the test does not assume homocedasticity).

Because the impact of fire on vegetation will determine the starting point of post-fire regeneration (Halpern 1988, Turner 1999), we first tested if the mean similarity between understory communities in burned plots and communities in reference plots varied with fire severity. The similarity between communities in burned and reference plots was determined using the Sørensen similarity index and differences in mean similarity were tested using the Welch test and the t-test for posterior pairwise comparisons, with the p value adjusted by the Holm's method The mean similarity of communities in plots affected by different levels of fire severity and communities in reference plots was not significantly different ($p>0.05$). Scorched plots in pine forest were the exception as they were more similar to reference plots than plots affected by severer fire.

The resilience of forest communities was assessed using four different approaches: overlap in specific composition, diversity measures, analysis of the functional composition and of seedling abundance. We used three measures to compare diversity between burned and reference plots: plant species richness (S), the Shannon-Wiener diversity index ($H = -\sum p_i \ln p_i$) and evenness ($H/\ln(S)$). Values were compared using two-tailed t-tests. To test differences in functional composition we determined the mean percentage cover of each Raunkaier's life forms (1937) at each plot. Percentage cover values were compared using the Welch test for two samples. For all quantitative analysis percentage cover classes were transformed to midpoint values: + (0.1%); 1 (2.5%); 2 (15%); 3 (37.5%); 4 (62.5%); 5 (87.5%). Mean values of seedling abundance per plot were compared using a two-tailed t-test.

The Shannon-Wiener index and Sørensen similarity index were calculated using the EstimateS software (Colwell 2005). Statistical analyses were performed using the R statistical software system (www.cran.r-project.org).

4.3 Results

Most broadleaved transects presented a closed canopy (>30% cover; cover data was assessed a posteriori using aerial photographs), the exception were two transects that presented an open structure (one was lightly burned and the other moderately burned). All pine transects presented a closed canopy. The mean canopy height varied between 6 and 10 metres in broadleaved transects and between 8 and 14 metres in pine transects.

Broadleaved forest was less damaged by fire than pine forest, most transects being lightly burned and none presenting signs of heavy burning (χ^2 = 8.83, df = 3, p = 0.03; Figure 3).

Fire severity was not related (p>0.05) with any variable (slope, northness, elevation and mean DBH) in burned broadleaved forest but was positively related with slope (rho=0.67, p=0.001) in burned pine transects. All variables varied independently, except for elevation and DBH in broadleaved transects (rho=0.65, p=0.002) and elevation and northness (rho=0.71, p=0.0004) in pine transects.

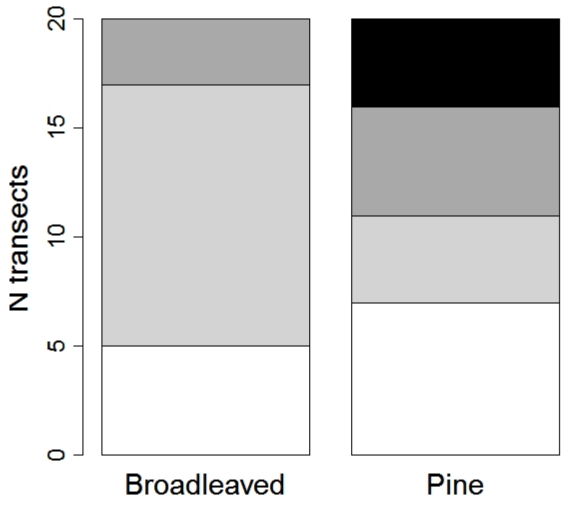


Figure 4.3 - Fire severity in broadleaved and pine forest transects: scorched (white), low severity (light grey), moderate severity (dark grey) and high severity (black).

No differences were found between mean tree mortality (percentage of dead trees) in burned and reference transects in broadleaved forest (Table 2). Mortality was null in thirteen transects and reached 44% in two transects dominated by holly. Fire severity was probably higher in the surrounding matrix, as evidenced by the extensive destruction of the shrub canopy and dispersed trees (V. M. Proença, personal observation). Tree mortality was significantly higher in burned pine transects than in reference transects (Table 2). Total mortality (100%) at the canopy level was recorded in four transects and none presented null mortality. Long term tree mortality could be worse than the reported because many living trees were seriously injured, which could compromise the maintenance of their physiological processes and increase the vulnerability to fungi or insect infestations (Amman and Ryan 1991, Lentile et al. 2005).

Live sapling abundance was significantly lower in burned transects than in reference transects in both forest types (Table 2).

Table 4.2 - Tree mortality (% dead trees per transect), number of live saplings (500 m^2) and number of seedlings (4 m^2) (n = 20 for all groups). Significant results with ($p<0.05$).

	% Dead trees		Saplings		Seedlings	
	Mean	SD	Mean	SD	Mean	SD
Broadleaved forest						
Burned	7.1	13.7	0.85	1.42	1.22	2.14
Reference	4.1	7.5	3.65	3.62	1.93	2.24
t value (df), p	-0.85 (29.6), 0.4		3.22 (24.8), 0.004		1.03 (38), 0.31	
Pine forest						
Burned	35.7	37.5	0.30	0.66	1.10	1.88
Reference	2.8	5.8	3.70	5.13	1.82	3.22
t value (df), p	-3.87 (19.9), 0.001		2.94 (19.6), 0.008		0.86 (38), 0.39	

A total of 54 and 68 plant species were found respectively in reference and in burned plots in broadleaved forest and 31 species were found in both reference and burned plots in pine forest (Appendix 2). From the species found in reference plots in broadleaved forest, 42 (78%) were also found in burned plots. In the case of pine forest, nearly half of the species (48%) found in reference plots were also found in burned plots (Figure 4). We note that part of the discrepancies in the floristic composition of reference and burned plots probably resulted from their different location. Species overlap may have been larger if data were collected before and after fire in the same plots.

With respect to functional composition (Figure 4), phanerophytes (trees not included) were the group most affected by fire in both forest types, few species were recorded in burned plots and there was a significant decrease in mean percentage cover (broadleaved: t=2.34, df=38, p=0.03; pine: t=3.65, df=20, p=0.002). With respect to the other life forms species richness was in general higher in burned plots and mean percentage cover did not vary significantly in pine plots (p>0.05), but there was a significant increase in the mean percentage cover of hemicryptophytes (t=-3.34, df=29, p=0.002), geophytes (t=-2.20, df=27, p=0.04) and therophytes (t=-2.48, df=19, p=0.02) in burned broadleaved plots.

Diversity measures in burned and reference plots in broadleaved forest were not significantly different. In the case of pine forest, burned plots were less rich and diverse than reference plots (Table 3). Seedling abundance was similar in burned and in reference plots in both forests (p>0.05) (Table 2).

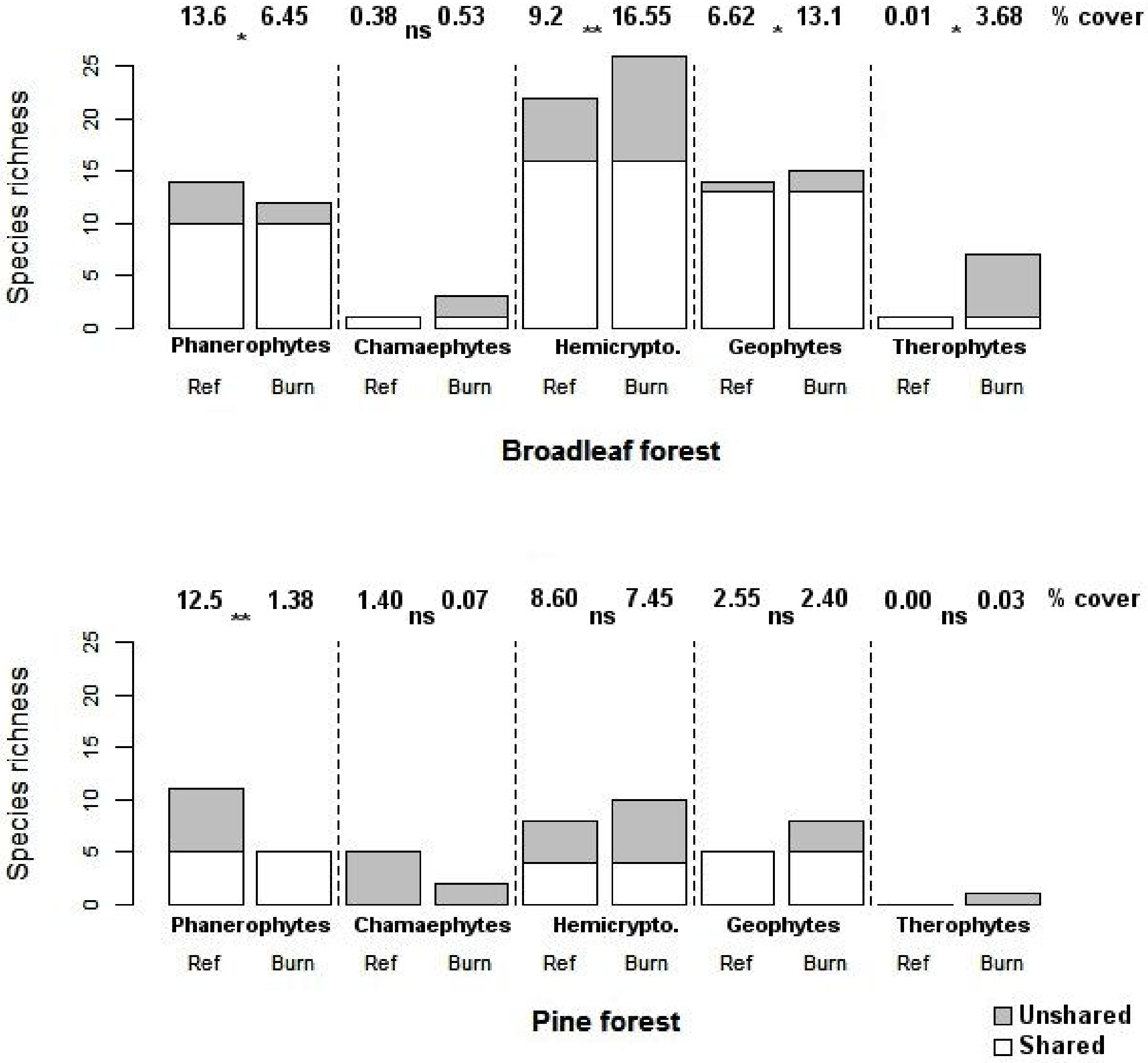


Figure 4.4 - Plant communities in reference (Ref) and burned (Burn) plots: total number of species and mean percent cover of plant life forms. Shared species were observed in reference and burned forests. Unshared species were observed in only one of the treatments. Mean percentage cover of each life form is indicated above each column (statistically significant differences: * p<0.05, ** p<0.005, ns – not significant).

Table 4.3 - Comparison of plant species richness, evenness and diversity in burned and reference plots (n = 20 for all groups). Significant results with $p<0.05$.

	Species richness		Evenness		Diversity	
	Mean	SD	Mean	SD	Mean	SD
Broadleaved forest						
Burned	12.85	3.34	0.74	0.12	1.85	0.35
Reference	11.90	3.91	0.80	0.08	1.94	0.39
t value (df=38), p	-0.83, 0.41		1.91, 0.06		0.73, 0.47	
Pine forest						
Burned	4.45	2.37	0.72	0.26	0.97	0.46
Reference	6.65	2.54	0.71	0.20	1.33	0.48
t value (df=38), p	2.83, 0.01		-0.07, 0.94		2.45, 0.02	

4.4 Discussion

Fire severity

Fire severity measures the magnitude of ecosystem change after fire and depends on the interaction of several factors, including fire intensity, fire patchiness, time since the last fire and season of burning (Whelan 1995). In the context of the present study, fire season did not vary, and former fires were probably of little importance to explain differences in fire severity, because all transects presented a developed understory and canopy before the fire. Therefore, differences may be discussed in view of three major factors affecting fire intensity and fire patchiness: fuels, local meteorology and topography (Rothermel 1983, Whelan 1995).

The extreme dry climate that preceded this fire may have had an important influence in fire behaviour. Due to fuel characteristics (amount, shape, arrangement, water content and chemistry) pine stands become very vulnerable to fire in a circumstance of drought (Schoennagel et al. 2004). Fire intensity is potentiated by the large amount and piled arrangement of fine fuels, in particular in closed and tall pine stands, and also by the chemical constitution of fuels (e.g., resins) and low water content. Fire extent is further potentiated by the occurrence of a continuous litter layer and ladder fuels (e.g., lower dead branches, suspended needles) that promote the vertical development of fire (Bond and van Wilgen 1996, Fernandes and Rigolot 2007, Ormeño et al. 2009, Fernandes in press).

In contrast, in broadleaved forest the development of crown fires was probably restrained by lower combustion temperatures, due to the higher moisture content of fuels (Bond and Midgley 2001, Castro et al. 2001, Dimitrakopoulos and Papaioannou 2001, Fernandes in press). This feature may also explain the lack of response of fire severity in broadleaved forest to topographic variables, as the lower combustion temperature of fuels may have limited the effect of other variables on fire intensity (Turner and Romme 1994).

Fire severity did not vary significantly with the mean diameter of adult trees in both forest types. However the considerable sapling mortality (inferred from sapling persistence) found in both forests suggests that below a certain size tree diameter has an effect on tree resistance. Tree diameter has been pointed as important variable affecting tree survival (Turner et al. 1999, González et al. 2006, Fernandes and Rigolot 2007), as larger trees are more resistant to fire damage, in particular cambium injuries, due to a thicker bark (Fernandes and Rigolot 2007, Fernandes et al. 2008). Nevertheless, it should be noted that oak saplings may preserve the ability to regenerate within one year after the fire, even if all tissues above ground have been killed (Tietje et al. 2001). This regenerative capacity has also been observed in burned oak forest in other areas of the National Park (V. M. Proença, personal observation).

Overall, differences in fire severity suggest a higher resistance of natural broadleaved forest to fire. This result is in agreement with other studies in the Mediterranean Basin that also report a lower resistance of pine species in relation to broadleaved species (Pausas 2008).

Community reestablishment

We found a higher percentage of species common to burned and reference communities in broadleaved forest than in pine forest which suggests a more advanced stage of recovery of burned plots in broadleaved forest. This large species overlap is consistent with the process of autosuccession (i.e., community reestablishment through resprouting of surviving plants or germination of surviving seeds) described in Mediterranean communities (Hanes 1971, Buhk et al. 2006, Núñez et al. 2008). Post-fire resilience is associated with the pre-fire composition of communities, in particular the presence of species able to promptly recolonize post-fire environments. Broadleaved communities presented a predominance of herbaceous perennial species (hemicryptophytes and geophytes). These plants have the ability to quickly regenerate from stem bases or storage organs located belowground (Trabaud 1982, Domínguez et al. 2002, Calvo 2003). Pine communities were dominated by woody species (phanerophytes and chamaephytes) in reference plots. Although some of these plants (e.g., *Erica arborea*) are also able of vegetative regeneration they tend to recover slower than herbaceous species (Domínguez et al. 2002, Calvo 2003), which may explain the less advanced state of understory recovery in this forest type.

The increase in the richness of herbaceous species (hemicryptophytes, geophytes and therophytes) observed in burned plots has been described as an opportunistic and temporary process promoted by the reduced competitive pressure from woody species and by sudden availability of nutrients resultant from the incineration of organic matter (Trabaud 1982, Naveh 1994, Calvo et al. 2003). As plant cover is re-established the species that first benefited from post-fire conditions will tend to regress and species richness is expected to return to pre-fire values (Trabaud 1982, Guo 2001, Keely et al. 2005).

The different representation of life forms in broadleaved and pine understories may also explain the maintenance of local diversity in burned plots in broadleaved forest and its decline in burned plots in pine forest.

Finally, seedling abundance did not differ between burned and reference transects in both types of forest. In broadleaved forest, we did not expect to find differences in seedling abundance because *Quercus robur* and *Ilex aquifolium* are resprouter species (Whelan 1995). In the case of pine forest we expected to find more seedlings in burned transects, because both *Pinus pinaster* and *Pinus sylvestris* rely on seeds to recolonize post-fire environments, and start recruitment immediately after fire (Valbuena et al. 2001, Fernandes et al. 2005, Calvo et al. 2008). The reason for the low abundance of seedlings may be explained by a depleted stock of viable seeds, particularly in *P. sylvestris* plots (60% of burned plots). *P. sylvestris* contrary to *P. pinaster* does not retain seeds in the canopy and the seeds are released after maturation and germinate during the spring (Reyes and Casal 1995, Núñez et al. 2003, Tapias et al. 2004). If a summer fire occurs, it will kill the seedlings and reduce the viability of the remaining seeds (Núñez et al. 2008). Accordingly, we found seedlings of *P. sylvestris* in scorched and low severity plots but not in moderate and high severity plots.

On the whole, plots in broadleaved forest were in a more advanced stage of recovery than plots in pine forest, suggesting a higher resilience of broadleaved forest. This finding is in agreement with Domínguez et al. (2002) and Calvo et al. (2003) that also reported a faster post-fire regeneration of oak communities in relation to pine communities. A later visit to the study area, in November 2008, confirmed the ongoing regeneration in broadleaved forest. It was not possible to assess the state of recovery of pine plantations because logging started just after field sampling.

Implications for conservation

Fire is the principal threat to forest ecosystems in Southern Europe (Bassi et al. 2008). Costs for society go beyond forest loss and include, among other impacts, the loss of human lives and destruction of public infrastructures (DGRF 2007, Bassi et al. 2008). Although fire has long been an important factor in Mediterranean ecosystems (Blondel and Aronson 1999), fire frequency and burned area have increased in the last decades (Pausas et al. 2004, DGRF 2007). The abandonment of rural areas, which promoted the increase of shrublands, the expanded distribution of fire prone plantations and the higher frequency of extreme climatic events have been pointed as key reasons for changes in the fire regime (Moreira et al. 2001, Lloret et al. 2002, Pausas et al. 2004).

Governments reaction to severer fire regimes has mainly consisted in measures of prevention and combat, such as investing in fire combat equipments. Although these measures are important, they are costly (over €475 million were spent between 2000 and 2004 in Portugal (DGRF, 2007)) and fail to deal with two important causes of fire severity: inadequate land planning and management. Rethink forest composition aiming at forests stability is a way of addressing these issues.

Broadleaved species are less vulnerable to fire. Increasing their representation in forest composition will contribute to reduce the costs associated with fire impacts and fire combat. Measures to achieve this transition should include the plantation of broadleaved species and the support of secondary forest regeneration in abandoned fields (Fabio et al. 2003, Gomes 2006). Moreover, these measures will also contribute to the development of stable and multifunctional forests meeting the aims of the Ministerial Conferences on the Protection of Forests in Europe (http://www.mcpfe.org/www-mcpfe/). Broadleaved forests provide a wide range of ecosystem services, such as carbon sequestration, soil protection, water purification, non-wood forest products, recreation and biodiversity (Fabbio et al. 2003). In addition, although these forests are not presently valued for their wood, some species, such as deciduous oaks, produce high quality timber (Carvalho 1997), therefore presenting a direct economic value.

Acknowledgments We thank Luisa Cardenete, Charo Perez Garcia, Inmaculada Santos Jimenez and José Torres Ruiz for assistance in field work, João Honrado for helping in plant identification and Leonor Calvo for providing bibliography. Paulo Fernandes, Joaquim Sande Silva and Meabh Boyle commented a first version of the manuscript. PNPG and ADERE provided logistic support. Vânia M. Proença was supported by a fellowship from Fundação para a Ciência e a Tecnologia (BD/12395/2003). Field work was supported by the Ecoforsite project (POCI-AMB-55701/2004) and the ABAFOBIO project (PTDC/AMB/73901/2006).

References

Amman GD, Ryan KC (1991) Insect infestations of fire-injured trees in the greater Yellowstone area. USDA Forest Service, Intermountain Forest and Range Experiment Station. Research Note INT-398. Ogden.

Baeza M, Valdecantos A, Alloza J, Vallejo V (2007) Human disturbance and environmental factors as drivers of long-term post-fire regeneration patterns in Mediterranean forests. Journal of Vegetation Science 18: 243–252.

Bassi S, Kettunen M, Kampa E, Cavalieri S (2008) Forest Fires: Causes and Contributing Factors to Forest Fire Events in Europe Study for the European Parliament Committee on Environment, Public Health and Food Safety under contract IP/A/ENVI/FWC/2006-172/LOT1/C1/SC10.

Blake JG, Schuette B (2000) Restoration of an oak forest in east-central Missouri: Early effects of prescribed burning on woody vegetation. Forest Ecology and Management 139: 109-126.

Blondel J, Aronson J (1999) Biology and wildlife in the Mediterranean region. Oxford University Press, Oxford.

Bond JW, Keeley JE (2005) Fire as a global ‘herbivore’: the ecology and evolution of flammable ecosystems. Trends in Ecology and Evolution 20: 387–394.

Bond WJ, Midgley JJ (2001) Ecology of sprouting in woody plants: the persistence niche. Trends in Ecology and Evolution 16: 45–51.

Bond WJ, van Wilgen BW (1996) Fire and Plants. Chapman & Hall, London.

Broncano MJ, Retana J, Rodrigo A (2005) Predicting the Recovery of Pinus halepensis and Quercus ilex Forests after a Large Wildfire in Northeastern Spain. Plant Ecology 180: 47-56.

Brown JK, Smith JK (2000) Wildland fire in ecosystems: effects of fire on flora. USDA Forest Service, Rocky Mountain Research Station. Gen.Tech. Rep. RMRS-GTR-42-vol.2. Ogden.

Brown RT, Agee JK, Franklin JF (2004) Forest Restoration and Fire: Principles in the Context of Place. Conservation Biology 18: 903–912.

Buhk C, Gotzenberger L, Wesche K, Sánchez Gómez P, Hensen I (2006) Post-fire regeneration in a Mediterranean pine forest with historically low fire frequency. Acta Oecologica 30: 288–298.

Calvo L, Santalla S, Marcos E, Valbuena L, Tárrega R, Luis-Calabuig E (2003) Regeneration after wildfire in communities dominated by *Pinus pinaster*, an obligate seeder, and in others dominated by *Quercus pyrenaica*, a typical resprouter. Forest Ecology and Management 184: 209–223.

Calvo L, Santalla S, Valbuena L, Marcos E, Tárrega R, Luis-Calabuig E (2008) Post-fire natural regeneration of *Pinus pinaster* forest in NW Spain. Plant Ecology 197: 81–90.

Carvalho A (1997) Madeiras portuguesas: estrutura anatómica, propriedades e utilização. Instituto Florestal, Lisboa.

Castro EB, Casado MA, Costa M, Escribano R, García M, Génova M, Manzaneque AG, Moreno JC, Morla C, Regato P, Sáinz H (2001) Los Bosques Ibéricos: Una Interpretación Geobotánica. Editorial Planeta, Barcelona.

Colwell RK (2005) EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.5. User's Guide and application published at: http://purl.oclc.org/estimates.

DGRF – Direcção Geral dos Recursos Florestais (2007) Estratégia Nacional para as Florestas. Imprensa Nacional da Casa da Moeda, Lisboa.

Diaz-Delgado R, Lloret F, Pons X, Terradas J (2002) Satellite Evidence of Decreasing Resilience in Mediterranean Plant Communities after Recurrent Wildfires. Ecology 83: 2293–2303.

Dimitrakopoulos A, Papaioannou KK (2001) Flammability Assessment of Mediterranean Forest Fuels. Fire Technology 37: 143–152.

Domínguez L, Calvo L, Luis E (2002) Impact of wildfire season on regeneration of *Quercus pyrenaica* forest and *Pinus sp.* stands. Journal of Mediterranean Ecology 3: 47–54.

Fabbio G, Merlo M, Tosi V (2003) Silvicultural management in maintaining biodiversity and resistance of forests in Europe--the Mediterranean region. Journal of Environmental Management 67:67-76.

Fernandes P (in press) Combining forest structure data and fuel modelling to classify fire hazard in Portugal. Annals of Forest Science.

Fernandes P, Botelho H, Rego F (2005) A Piroecologia do Pinheiro Bravo. Silva Lusitana 13: 233–248.

Fernandes PM, Rigolot E (2007) The fire ecology and management of maritime pine (*Pinus pinaster* Ait.). Forest Ecology and Management 241: 1–13.

Fernandes PM, Vega JA, Jiménez E, Rigolot E (2008) Fire resistance of European pines. Forest Ecology and Management 256: 246-255.

Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS (2004) Regime shifts, resilience and biodiversity in ecosystem management. Annual Review of Ecology and Systematics 35: 557–581.

Gomes JFP (2006) Forest fires in Portugal: how they happen and why they happen. International Journal of Environmental Studies 63: 109-119.

González JR, Palahí M, Trasobares A, Pukkala T (2006) A fire probability model for forest stands in Catalonia (north-east Spain). Annals of Forest Science 63: 169-176.

Gracia M, Retana J, Roig P (2002) Mid-term successional patterns after fire of mixed pine–oak forests in NE Spain. Acta Oecologica 23: 405–411.

Guo Q (2001) Early post-fire succession in California chaparral: Changes in diversity, density, cover and biomass. Ecological Research 16: 471–485.

Halpern CB (1988) Early Successional Pathways and the Resistance and Resilience of Forest Communities. Ecology 69: 1703–1715.

Hanes TL (1971) Succession after Fire in the Chaparral of Southern California. Ecological Monographs 41: 27–52.

Honrado JJP (2003) Flora e Vegetação do Parque Nacional da Peneda-Gerês. Dissertação de candidatura ao grau de Doutor em Biologia, Faculdade de Ciências da Universidade do Porto, Porto.

INM - Instituto Nacional de Meterorologia (2009) Caracterização climática: Ano 2006. Instituto de Meterologia, I.P. Available from http://www.meteo.pt (accessed January 2009).

Keeley JE, Fotheringham CJ, Baer-Keeley M (2005) Factors affecting plant diversity during post-fire recovery and succession of mediterranean-climate shrublands in California, USA. Diversity and Distributions 11: 525–537.

Kent M, Coker P (1994) Vegetation description and analysis: a pratical approach. John Wiley and Sons, Chichester.

Ladd PG, Crosti R, Pignatti S (2005) Vegetative and seedling regeneration after fire in planted Sardinian pinewood compared with that in other areas of Mediterranean-type climate. Journal of Biogeography 32: 85-98.

Lavorel S (1999) Ecological diversity and resilience of Mediterranean vegetation to disturbance. Diversity and Distributions 5: 3–13.

Lentile LB, Smith FW, Shepperd WD (2005) Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forest Research 35: 2875–2885.

Lloret F, Calvo E, Pons X, Díaz-Delgado R (2002) Wildfires and landscape patterns in the Eastern Iberian Peninsula. Landscape Ecology 17: 745–759.

McCann KS (2000) The diversity-stability debate. Nature 405: 228–233.

Moreira F, Catry F, Duarte I, Acácio V, Silva J (in press) A conceptual model of sprouting responses in relation to fire damage: an example with cork oak (Quercus suber L.) trees in Southern Portugal. Plant Ecology.

Moreira F, Rego FC, Ferreira PG (2001) Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. Landscape Ecology 16:557-567.

Mouillot F, Rambal S, Joffre R (2002) Simulating climate change impacts on fire frequency and vegetation dynamics in a Mediterranean-type ecosystem. Global Change Biology 8: 423–437.

Naveh Z (1975) The evolutionary significance of fire in the Mediterranean region. Vegetatio 29: 199–208.

Naveh Z (1994) The role of fire and its management in the conservation of Mediterranean ecosystems and landscapes. In: Moreno, J.M., Oechel, W.C. (eds) The Role of Fire in Mediterranean-Type Ecosystems, vol. 107 Ecological Studies. Springer-Verlag, pp. 163–186.

Neary DG, Klopatek CC, DeBano LF, Folliott PF (1999) Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management 122: 51–71.

Núñez MR, Bravo F, Calvo L (2003) Predicting the probability of seed germination in *Pinus sylvestris* L. and four competition shrub species after fire. Annals of Forest Science 60: 327–334.

Núñez MR, Calvo L, Pando V, Bravo F (2008) Floristic changes induced by fire on *Pinus sylvestris* plantations in northwest of Spain. Investigación Agraria: Sistemas y Recursos Forestales 17: 168–177.

Ordóñez JL, Retana J, Espelta JM (2005) Effects of tree size, crown damage, and tree location on post-fire survival and cone production of Pinus nigra trees. Forest Ecology and Management 206: 109-117.

Ormeño E, Céspedes B, Sánchez IA, Velasco-García A, Moreno JM, Fernandez C, Baldy V (2009) The relationship between terpenes and flammability of leaf litter. Forest Ecology and Management 257: 471-482.

Pausas J, Bladé C, Valdecantos A, Seva J, Fuentes D, Alloza J, Vilagrosa A, Bautista S, Cortina J, Vallejo R (2004) Pines and oaks in the restoration of Mediterranean landscapes of Spain: New perspectives for an old practice — a review. Plant Ecology 171: 209-220.

Pausas J, Vallejo VR (1999) The role of fire in European Mediterranean ecosystems. In: Chuvieco, E. (ed) Remote Sensing of Large Fires in the European Mediterranean Basin. Springer, New York, pp. 3-16.

Pausas JG, Llovet J, Rodrigo A, Vallejo R (2008) Are wildfires a disaster in the Mediterranean basin? – A review. Int. J. Wildland Fire 17: 713-723.

PNPG – Parque Nacional da Peneda-Gerês (1995) Plano de Ordenamento do Parque Nacional da Peneda-Gerês. Relatório de síntese. ICN, Braga.

Raunkaier C (1937) Plant life forms. Clarendon Press, Oxford.

Reyes O, Casal M (1995) Germination behaviour of 3 species of the genus Pinus in relation to high temperatures suffered during forest fires. Annals of Forest Science 52: 385–392.

Rigolot E (2004) Predicting postfire mortality of Pinus halepensis Mill. and Pinus pinea L. Plant Ecology 171: 139-151.

Rothermel RC (1983) How to predict the spread and intensity of forest and range fires. U.S. Forest Service General Technical Report INT-143.

Schoennagel T, Veblen TT, Romme WH (2004) The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54: 661–676.

Sequeira N (1995) Plano de gestão florestal da mancha florestal do Mezio. Parque Nacional da Peneda-Gerês, Arcos de Valdevez.

Shvidenko A, Barber CV, Persson R (2005) Forest and woodland systems. In: Hassan R., Scholes R., Ash N. (eds) Ecosystems and human well-being: Current state and trends. Island Press, Washington D. C., pp. 585-621.

Silva JS, Moreira F, Vaz P, Catry F, Godinho-Ferreira P (in press) Assessing the relative fire proneness of different forest types in Portugal. Plant Biosystems.

Tapias R, Climent J, Pardos J, Gil L (2004) Life histories of Mediterranean pines. Plant Ecology 171: 53–68.

Tietje W (2001) Live oak saplings survive prescribed fire and sprout. California agriculture 55: 18-22.

Torres J, Honrado J, Caldas FB (2001) A vegetação da "Mata do Ramiscal" (Serra do Soajo, Parque Nacional da Peneda-Gerês). UGEV-ICETA, Universidade do Porto.

Trabaud L (1982) Effects of past and present fire on the vegetation of the French Mediterranean region. In: Symp. dynamics and management of Mediterranean-type ecosystems, USDA.

Turner MG, Romme WH (1994) Landscape dynamics in crown fire ecosystems. Landscape Ecology 9: 59–77.

Turner MG, Romme WH, Gardner RH (1999) Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire 9: 21–36.

USDI, National Park Service (2001) Fire Monitoring Handbook. Nation Interagency Fire Center, Boise.

Valbuena L, Nuñez R, Calvo L (2001) The seed bank in Pinus stand regeneration in NW Spain after fire. Web Ecology 2: 22–31.

Verdú M, Pausas J (2007) Fire drives phylogenetic clustering in Mediterranean Basin woody plant communities. Journal of Ecology 95: 1316-1323.

Whelan R (1995) The Ecology of fire. Cambridge University Press. Cambridge, England.

Appendix 4.1

Summary of transect features and fire severity. Dominant species (>50% canopy dominance): Ia - *Ilex aquifolium*, Qr - *Quercus robur*, Pp - *Pinus pinaster*, Ps - *Pinus sylvestris;* fire severity (FS): 1- unburned, 2 - scorched, 3 - low severity, 4 - moderate severity, 5 - high severity; elevation; dominant aspect: N - north, S - south, E - east, W - west; slope; number of trees in the transect (including snags); mean DBH and DBH standard deviation.

Broadleaved forest							
Dominant	FS	Elev. (a.s.l.) (m)	Aspect	Slope	N Trees	Mean DBH (cm)	DBH SD (cm)
Qr	1	400	N	0.5	12	37.88	18.02
Qr	1	400	N	0.5	18	32.82	10.52
Qr	1	523	W	0.25	11	30.04	6.75
Qr	1	525	N	0.2	17	25.95	15.05
Qr	1	620	E	0.25	21	21.01	5.90
Qr	1	620	N	0.3	20	17.83	4.83
Qr	1	620	E	0.25	20	26.32	10.60
Qr	1	643	N	0.3	10	33.17	14.86
Qr	1	644	E	0.3	25	19.14	4.78
Qr	1	655	W	0.3	21	32.80	10.47
Qr	1	658	W	0.4	24	26.79	11.26
Qr	1	670	W	0.3	22	26.09	4.94
Qr	1	710	W	0.3	29	24.77	11.59
Qr	1	720	W	0.3	11	36.14	15.67
Qr	1	843	W	0.4	12	36.21	6.43
Qr	1	904	N	0.5	27	21.43	6.35
Ia	1	1170	-	0	10	66.24	29.65
Ia	1	1200	-	0	9	43.36	23.54
Ia	1	1239	W	0.3	13	45.10	13.31
Ia	1	1239	W	0.2	10	53.32	20.65
Qr	2	453	W	0.45	21	21.71	8.96
Qr	2	454	N	0.15	9	30.03	16.71
Ia	2	1080	N	0.4	5	55.96	25.95
Ia	2	1147	N	0.4	5	71.17	12.84
Ia	2	1167	W	0.4	13	43.51	15.50
Qr	3	400	N	0.55	17	21.76	10.15
Qr	3	432	S	0.35	11	35.53	9.73
Qr	3	435	W	0.5	11	27.75	7.17
Qr	3	447	N	0.5	11	36.35	19.14
Qr	3	448	N	0.55	19	20.42	6.68
Qr	3	457	N	0.5	8	28.29	9.94
Qr	3	1034	N	0.5	9	39.61	39.97
Qr	3	1060	W	0.5	11	29.81	10.70
Qr	3	1103	N	0.6	11	48.18	13.13
Qr	3	1107	W	0.3	16	30.24	13.05
Qr	3	1140	N	0.7	5	64.23	15.66
Qr	3	1167	W	0.2	13	36.51	9.11
Ia	4	1150	N	0.55	4	35.25	9.01
Ia	4	1167	N	0.25	9	49.87	18.50
Ia	4	1180	N	0.35	9	40.04	39.09

Pine Forest							
Dominant	FS	Elev. (a.s.l.) (m)	Aspect	Slope	N Trees	Mean DBH (cm)	DBH SD (cm)
Pp	1	439	S	0.15	27	31.34	12.25
Pp	1	484	S	0.2	40	28.26	10.16
Pp	1	550	N	0.1	41	22.51	6.71
Pp	1	555	S	0.1	35	25.95	9.56
Pp	1	560	S	0.15	27	30.40	7.62
Pp	1	632	N	0.1	16	30.40	6.74
Pp	1	647	-	0	19	31.31	4.17
Pp	1	648	E	0.1	16	31.67	10.31
Pp	1	652	E	0.1	17	28.35	7.00
Pp	1	684	-	0	34	30.33	8.77
Pp	1	700	S	0.05	23	32.34	9.30
Pp	1	709	-	0	36	29.05	6.53
Pp	1	720	W	0.2	85	20.18	4.99
Pp	1	730	W	0.3	67	20.47	5.14
Pp	1	733	W	0.05	60	19.07	4.74
Pp	1	737	W	0.3	69	19.92	4.28
Pp	1	758	S	0.05	66	20.99	4.96
Pp	1	776	W	0.2	82	19.82	4.81
Pp	1	789	W	0.2	71	19.79	4.89
Pp	1	833	W	0.3	61	22.28	5.91
Ps	2	701	S	0.1	17	27.96	8.48
Pp	2	728	W	0.1	34	30.06	6.82
Pp	2	758	W	0.05	16	32.49	11.33
Ps	2	998	W	0.4	24	26.70	6.77
Ps	2	1030	S	0.2	28	29.02	5.44
Ps	2	1072	W	0.1	41	24.87	6.99
Ps	2	1078	W	0.1	34	27.33	6.26
Ps	3	619	S	0.3	31	22.87	8.51
Ps	3	626	S	0.15	34	26.31	7.67
Ps	3	646	S	0.2	41	27.34	8.29
Pp	3	828	W	0.2	23	28.51	13.31
Pp	4	620	S	0.25	54	26.93	7.76
Pp	4	641	S	0.4	44	26.54	8.35
Ps	4	1030	W	0.2	62	23.94	7.33
Ps	4	1070	W	0.2	54	22.99	6.71
Ps	4	1083	W	0.3	49	23.98	5.66
Pp	5	888	W	0.45	9	36.36	9.75
Pp	5	928	W	0.25	25	20.83	9.80
Ps	5	932	W	0.4	30	27.15	8.95
Ps	5	1118	W	0.25	50	21.83	5.95

Appendix 4.2

List of species observed in the study and their occurrence (number of transects) by forest type. Raunkaier’s plant life forms (LF): P – phanerophytes (species with aerial renewing buds), C – chamephytes (species with renewing buds slightly above soil level), H – hemicryptophytes (species with renewing buds at soil level), G – geophytes (species with subterranean organs from which renewing buds emerge), T – therophytes (annual species that remain dormant as seed during unfavourable periods), nd – not determined. All the entries in the list were treated as different species in data analysis, even if not identified until the species level.

			Broadl. forest		Pine forest	
Family	Species	LF	Refer.	Burned	Refer.	Burned
Aceraceae	*Acer pseudoplatanus*	P	1			
Amaryllidaceae	*Narcissus bulbocodium*	G		1		2
Aquifoliaceae	*Ilex aquifolium*	P	9	6		
Araliaceae	*Hedera hibernica*	P	11	5		
Aspleniaceae	*Asplenium* sp.	H	1			
Blechnaceae	*Blechnum spicant*	H	2	1	1	
Campanulaceae	*Campanula lusitanica*	T		4		
Campanulaceae	*Campanula rapunculus*	H		1		
Caprifoliaceae	*Lonicera periclymenum*	P	5	1		
Caryophyllaceae	*Arenaria montana*	C	7	8		2
Caryophyllaceae	*Silene acutifolia*	H		1		1
Caryophyllaceae	*Silene gallica*	T		3		
Caryophyllaceae	*Silene latifolia*	H		1		
Caryophyllaceae	*Silene nutans*	H	6			
Caryophyllaceae	*Silene vulgaris*	H	2	1		
Caryophyllaceae	*Stellaria graminea*	H		2		
Caryophyllaceae	*Stellaria media*	T		3		
Compositae	*Conyza sumatrensis*	H		1		1
Compositae	*Crepis lampsanoides*	G	3	2		
Compositae	*Hypochoeris radicata*	H	1			
Compositae	*Picris hieracioides*	H	1			
Compositae	*Senecio* spp.	T				1
Crassulaceae	*Sedum arenarium*	T		3		
Crassulaceae	*Sedum brevifolium*	C			1	
Crassulaceae	*Umbilicus rupestris*	H	4	4		1
Cruciferae	*Capsella bursa-pastoris*	T		1		
Cruciferae	*Raphanus raphanistrum*	T	1	1		
Cyperaceae	*Carex* sp1	H	1		8	
Ericaceae	*Calluna vulgaris*	P			2	
Ericaceae	*Daboecia cantabrica*	C			2	
Ericaceae	*Erica arborea*	P	12	4	8	1
Ericaceae	*Erica australis*	P	1			
Ericaceae	*Erica cinerea*	C			2	
Ericaceae	*Erica umbellata*	P			6	
Euphorbiaceae	*Euphorbia dulcis*	H	1			
Fagaceae	*Castanea sativa*	P	1	1		
Fagaceae	*Quercus robur*	P	12	5	1	
Gramineae	*Agrostis curtisii*	H	11	10	15	5

Gramineae	*Agrostis* sp1	H		1	1	
Gramineae	*Anthoxanthum odoratum*	H	1	6		
Gramineae	*Arrhenatherum elatius*	H	20	18	14	11
Gramineae	Gramineae1	nd				4
Gramineae	Gramineae2	nd	2			
Guttiferae	*Hypericum humifusum*	C		2		1
Hemionitidaceae	*Anogramma leptophylla*	G	1	1		1
Hypoleppidaceae	*Pteridium aquilinum*	G	19	14	14	10
Juncaceae	*Luzula campestris*	H		1		
Juncaceae	*Luzula multiflora*	H	2	2		1
Labiatae	*Lamium maculatum*	H	1	2		
Labiatae	*Teucrium scorodonia*	H	7	1		1
Labiatae	*Thymus caespititius*	C			1	
Leguminosae	*Cytisus* spp.	P	5	6	1	9
Leguminosae	*Genista florida*	P		1		
Leguminosae	*Lotus corniculatus*	H		5		1
Leguminosae	*Pterospartum tridentatum*	P			1	
Leguminosae	*Ulex europeus*	P	1		1	
Leguminosae	*Ulex minor*	P			14	
Liliaceae	*Asphodelus lusitanicus*	G	7	4	1	1
Liliaceae	*Erythronium dens-canis*	G	2	5		
Liliaceae	*Polygonatum odoratum*	G	1			
Liliaceae	*Ruscus aculeatus*	G	3	2		
Liliaceae	*Scilla monophyllos*	G	10	13	2	4
Liliaceae	*Simethis mattiazzi*	G	2	1		
Liliaceae	Liliaceae 1	G		4	1	1
Liliaceae	Liliaceae 2	G	2	3		1
Liliaceae	Liliaceae 3	G	4	3	13	12
Papaveraceae	*Ceratocapnos claviculata*	T		9		
Pinaceae	*Pinus pinaster*	P	1		16	5
Pinaceae	*Pinus sylvestris*	P			1	5
Polygonaceae	*Rumex acetosa*	H	2	4		
Polygonaceae	*Rumex acetosella*	H	4	1	1	3
Primulaceae	*Primula acaulis*	H	1	7		
Ranunculaceae	*Anemone trifolia*	G	2	14		
Ranunculaceae	*Ranunculus bolbosus sl.*	G	4	9		
Rhamnaceae	*Frangula alnus*	P	12	2		
Rosaceae	*Amelanchier ovalis*	P		1		
Rosaceae	*Potentilla erecta*	H	2	1	2	
Rosaceae	*Pyrus cordata*	P	9	3		
Rosaceae	*Rubus* spp.	P	9	14	1	1
Rubiaceae	*Galium saxatile*	G	2	1		
Saxifragaceae	*Saxifraga spathularis*	H	2	7		
Scrophulariaceae	*Sibthorpia europaea*	C		1		
Scrophulariaceae	*Veronica officinalis*	C			1	
Umbelliferae	*Peucedanum lancifolium*	H		1		
Umbelliferae	*Physospermum cornubiense*	H	4	1	2	1
Violaceae	*Viola palustris*	H		3		
Violaceae	*Viola riviniana*	H	2	4		
…	Plant 1	nd			1	

…	Seedling 1	nd	1	5		4
…	Seedling 2	nd		1		1
…	Seedling 3	nd			2	1
…	Seedling 4	nd		3		1
…	Seedling 5	nd		1		
…	Seedling 6	nd		1		

ECOSYSTEM CHANGES, BIODIVERSITY LOSS AND HUMAN WELL-BEING

Proença VM, Pereira HM. Ecosystem changes, biodiversity loss and human well-being. In: Encyclopedia of Environmental Health. Elsevier Press. *In press*.

5 Ecosystem changes, biodiversity loss and human well-being.

Synopsis

Biodiversity is the diversity of life on earth at various organization levels, from gene to species and ecosystems. Biodiversity comprises the dynamic web of organisms and the interactions between them and with the environment. Biodiversity is fundamental for the maintenance of ecosystem functioning and the delivery of ecosystem services. Ecosystems services are direct benefits that humans obtain from ecosystems such as food and clean water, or indirect benefits, such as climate regulation and pollination. Human well-being is dependent on ecosystem services and thus on the condition of ecosystems. However, humans are placing increasing pressure on ecosystems, due to the exponential growth of world population over the last decades and increasing consumption patterns. As a result ecosystems are being degraded and destroyed, resources are collapsing and the loss of biodiversity has reached unprecedented levels. Ecosystems condition is seriously threatened along with the maintenance of the benefits provided by ecosystems. The effects on human well-being are felt at several scales from local communities to the global population. Human well-being is being affected worldwide by the consequences of ecosystem changes and biodiversity loss, which include natural disasters, health problems, and poverty. Environmental sustainability is a key concept for the future, being fundamental to find solutions that preserve biodiversity and ecosystems, without disregarding people's needs for ecosystem services and economical development.

Keywords

2010 target, Biodiversity loss, Biodiversity, Ecosystem change, Ecosystem services, Environmental sustainability, Fire, Forests, Human well-being, Millennium Development Goals, Millennium Ecosystem Assessment

5.1 Introduction

Human well-being is deeply connected with biodiversity. From subsistence communities to highly developed urban communities, everyone needs food, clean water and air, fibers, fuel, medicines and environmental stability. Ecosystems provide these services and biodiversity sustains ecosystems and their processes.

As the world population and consumption patterns per capita increase so does the demand for natural resources (e.g., wood, fish) and the impacts of human activities on natural habitats. Impacts might be direct (e.g., habitat destruction for urbanization) or indirect (e.g., carbon emissions which cause global warming), but they all lead to biodiversity loss and consequently threat ecosystems balance and human well-being. Human well-being is an inclusive concept that embraces the physical and mental components of human health, but also social well-being and freedom of choice.

There is a feedback loop between human well-being and biodiversity: human well-being is dependent on biodiversity, biodiversity and ecosystems condition are affected by human options towards environment and these options are influenced by the level of well being and the socio-economic choices of communities This cycle will be analyzed throughout the article. The article starts with a brief overview of what is biodiversity and its distribution around the globe. Next we discuss the value of biodiversity and ecosystem services. The link between ecosystems services and human well-being is analyzed. The following section focus on biodiversity loss and drivers of environmental change and the consequences for human well-being. Finally, a case study is analyzed integrating these concepts and providing a more concrete view of the feedback loop between biodiversity and human well-being. We conclude with some remarks about the need to find solutions that promote human well-being and also prevent biodiversity loss.

5.2 What is Biodiversity

Biodiversity is the variety of life on Earth. The Convention on Biological Diversity (article 2) defines biodiversity as “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.”

In other words, biodiversity includes genetic diversity, species diversity, and ecosystem diversity. Genetic diversity is the simplest level of diversity, including the different varieties of crops and the variation between individuals. Species diversity is basically composed by all the different species in the world, from mushrooms to mammals. Ecosystem diversity comprises the different species assemblages of each ecosystem and their relations to the environment, such as deserts, temperate forests and coral reefs.

The diversity of species is vast and still counting with new species being described every year. So far scientists have described about 1.75 million species with more than half of those being invertebrates. Estimates of global species richness range from 3 million to 100 million species. This lack of precision expresses how much is still unknown about the living planet.

5.3 Biodiversity around the globe

The distribution of species around the Earth is not homogeneous. Some world regions are more diverse than others. Some regions are not only very diverse but also support a large number of endemic species (i.e., species that occur exclusively in that region). This uniqueness confers them a high level of irreplaceability making them priority areas for conservation.

The biodiversity hotspots are examples of such areas (Figure 1). These hotspots support a high level of plant endemism and face a severe threat of habitat loss, with at least 70% of the original vegetation already lost.

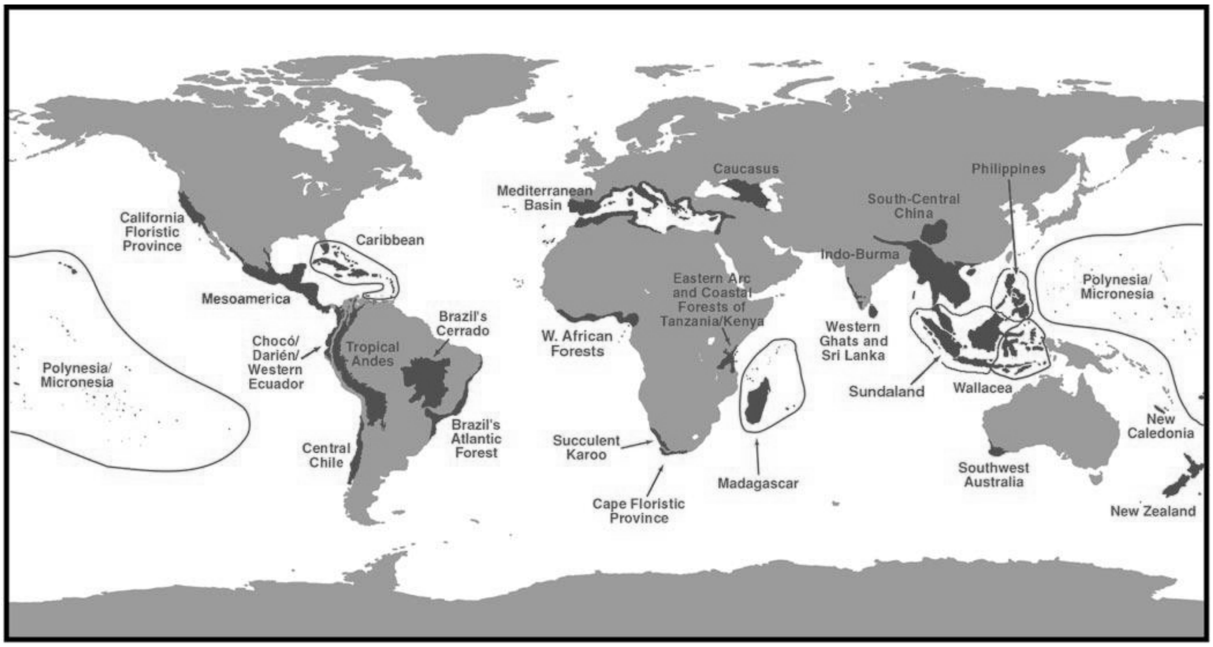


Figure 5.1 - Biodiversity hotspots (from Myers N. et al. (2000) Biodiversity hotspots for conservation priorities. Nature 403, 853–858).

In 2000, 25 hotspots were identified around the world. Four years later the evaluation was reviewed, with the redefinition of hotspots limits and the classification of additional areas. In total 34 regions are now classified as hotspots, containing at least 150 000 endemic plants, about 50% of world plant diversity and 77% of all vertebrates. Originally these regions occupied 15.7% of earth surface, but 86% of their area was altered by human activities and now only 2.3% remain undisturbed. 38% of these areas are located in the Asia-Pacific region, 24% in Africa, 15% in South America, 12% in Europe and Central Asia and 12% in North and Central America.

While biodiversity hotspots are highly threatened and irreplaceable regions, another category of important biodiversity regions includes irreplaceable areas that are still pristine and have low anthropogenic influence. These regions are as known by high biodiversity wilderness areas and comprise five world regions: the North American deserts and Amazonia in the American continent, the Congo forest and the Miombo-Mopane woodlands, which include the Okavango Delta, in Africa and the New Guinea in Australasia. Endemism in these areas comprise about 17% of global plant diversity and 8% of global vertebrate diversity, and although these values are lower than the values found in biodiversity hotspots these regions are still important due to the pristine condition of their ecosystems.

5.4 Biodiversity and ecosystem services

Ecosystems provide many services to humans, which range from commodities like food, fibers or medical substances, to indirect benefits like carbon retention, pollination or water filtering. Ecosystem services can be classified in four categories: provisioning services, regulating services, cultural services and supporting services. The existence and maintenance of ecosystem services is sustained by biodiversity (Figure 2). Provision services correspond to the goods directly obtained from ecosystems. Cultural services are non-material benefits obtained from ecosystems such as high quality spaces for leisure or the feeling of satisfaction derived from observing a rare butterfly. Regulating services are the indirect benefits obtained from the regulation of ecological processes such as climate regulation or soil protection from erosion. Finally, supporting services provide the basis for the production of all the other ecosystem services, and include services as oxygen production by photosynthesis, nutrient cycling and habitat provisioning.

Each component of biodiversity, such as species richness, species composition or species interactions, plays a role in ecosystem services. Ecosystem functioning depends on the presence of organisms from different functional groups (i.e., that perform different roles in ecosystem processes). For example, the process of litter decomposition depends on organisms specialized on breaking down particles of different size, from earthworms to microbes. Therefore, species composition, with elements from different functional groups, is a key factor to assure the maintenance of supporting services. Species richness is central to the stability of ecosystems, a regulating service. Ecosystems with a rich and complex web of species interactions are more protected from the negative effects of environmental changes than species-poor systems. Environmental changes may affect the function of certain species on ecosystem processes or eventually lead to species extinctions. Thus, if a large number of native species exist in a given area, it is more probable that some species persist and assure the maintenance of ecosystem services. Also, there is evidence that habitats maintaining the original species composition are more resistant to invasion of non-native species.

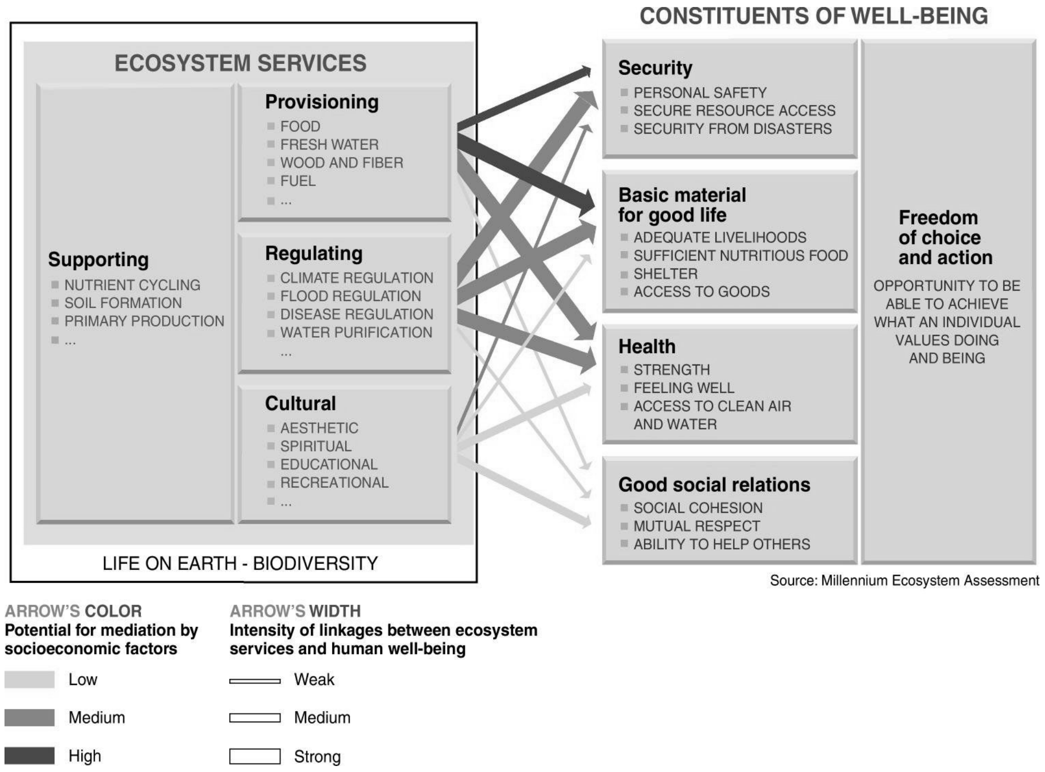


Figure 5.2 - Linkages between biodiversity, ecosystem services and human well-being (from Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: synthesis. Washington, DC: Island Press).

The services provided by biodiversity and ecosystems might be valued according to a utilitarian approach (Table 1). Use values are assigned to services that have a concrete utility to humans, either providing direct use benefits or indirect use benefits, or a potential utility in the future, either for the individual or for future generations (option values). Existence value is the value that people assign to a species or ecosystem even if they do not obtain any benefit besides the satisfaction of knowing that the species or ecosystem exists. For example, people in Europe might contribute to a conservation program to save pandas in China, only because they have a philanthropic interest in assuring the species survival.

Table 5.1 - Utilitarian value of ecosystem services, examples of ecosystem services and general correspondence with categories of ecosystem services.

	Examples	**Category**
Use value		
Direct use value		
Material benefit	Food, fuel, medicines	Provisioning services
Non-material benefit	Recreational areas Sacred forests (spiritual benefit)	Cultural services
Indirect use value	Climate regulation Water purification Soil formation Oxygen production	Regulating and supporting services
Option value	Vaccines, medicines Genetic resources for investigation Key species for ecosystem functioning	Provisioning, cultural and regulating services
Non use value		
Existence value	Satisfaction of knowing that a species or ecosystem exists	Cultural services

5.5 Ecosystem services and human well-being

Ecosystem services, along with other factors such as education, political systems or available technology, contribute to human well-being. The concept of human well-being is inevitably dependent on cultural and socio-economical settings, which influence what people consider to be most imperative for a comfortable life. Nevertheless, the elements necessary to human well-being can be classified into five components: sense of security, basic materials for a good life, health, good social relationships and freedom of choice (Figure 2).

Sense of security exists when people feel safe about the availability of resources and the protection from eventual natural disasters and feel that their physical integrity and economical independence are safeguarded. If provisioning services fail and limit the access of people to food, water or fuels, this will affect their sense of security. Also if regulating or supporting services suffer changes, communities will be in greater risk of natural disasters or diseases, and their sense of security will be weakened.

The basic materials for a good life comprise food, water, fuel, and also the earning of an income. When provisioning services are affected, access to basic materials is also affected. For example, access to food, forest materials and clean water is seriously compromised when crops are destroyed by plagues or climate disasters, when wildfires occur and when rivers are polluted.

Health is a central component of human well-being. Imbalances in regulating and provisioning services are the principal causes of public health problems and deficits. Contaminated water is in the origin of diseases as diarrhea, cholera and typhoid fever and is responsible for the death of thousands of people every year. Air pollution is a problem in urban areas where it causes lung and heart diseases. Climate change is promoting the expansion of the area affected by several diseases, such as malaria. Failure in provision services affects the access to basic materials and consequently the access to an adequate diet, to potable water or to medicines, which are essential for good health conditions. Cultural services also contribute to human health, in particular mental health.

Good social relationships are dependent on the other well-being components. When basic materials or security are not assured, communities are under stress and their social relations deteriorate. The failure in provisioning or regulation services might conduce to famine or climate disasters, leading to unstable social environments. When communities are culturally connected with the environment, by faith or ancient traditions, landscape changes may affect their social stability and their emotional health.

When one of the other well-being components fails, freedom of choice and action is affected. For example, if people have to walk several kilometers to get water, if their properties are destroyed by fire or if they need to compete for food or shelter, their range of life options will be much reduced. Freedom of choice is transversal to the achievement of the other components of well-being. When people live a good life they are in condition to make better options about ecosystem use and management. This influences the state of ecosystem services and consequently the condition of the other components of well-being.

Poorer communities are more vulnerable to the degradation of ecosystems and to the effects of changes in ecosystems services, in particular if they depend directly on local ecosystems. Wealthy societies, on the other hand, are on a safer position because they have the economical power to minimize the consequences of natural disasters, can afford medicines to combat diseases and if local ecosystems fail, products may always be imported from other locations. This economical advantage of wealthy societies is sometimes mischievous, because it allows the transfer of production demands to poorer regions, causing the exploitation of ecosystems in those regions with little benefits to local people.

5.6 Human activity, biodiversity loss and implications for human well-being

World population has increased exponentially over the last decades: 2.5 billion people in 1950, 6.5 billion in 2005 and projections say 9 billion by 2050. Furthermore, per capita consumption has also been increasing A direct consequence of this trend is the increase of the demand for natural resources, often above sustainable levels. Native forests are being logged and replaced by agricultural fields or production forests, world fisheries are in imminent risk of collapsing, and about two thirds of the world's available fresh water is polluted. During the last centuries human activity has raised species extinction rates up to 1000 times the values found in the fossil record. According to "The IUCN Red List of Threatened Species", a world report on species conservation status, there are presently more than 5000 endangered species of vertebrates and 8000 of vascular plants.

Biodiversity loss encompasses loss at the genetic, species and ecosystem levels. The loss of genetic diversity increases species vulnerability to ecosystem changes. This is especially alarming in the case of crops. The intensification of agricultural practices has led to a decline of the genetic diversity of cultivated species. The decline of agrobiodiversity reduces resilience of our crops to ecosystem changes, threatening the stability of food production. Losses of species diversity comprise either the extinction of species and populations (at local scales), but also changes in community composition. A current trend is the simplification of biotic communities due to the increasing dominance of species better adapted to human modified ecosystems (species that are more tolerant to perturbation, that benefit from nutrient loadings, etc.). An identical result is observed in the case of invasive species that, in the absence of predators, pathogens or competitors, become dominant, leading to the reduction or even extinction of native populations. As a consequence, biotic communities around the world are becoming less distinct, and there is a loss of diversity from local to global levels. Finally, biodiversity loss at the ecosystem level is transversal to most terrestrial biomes (e.g., temperate forests, grasslands, tropical forests), mainly due to conversion to cultivated land.

The main drivers of ecosystem change and biodiversity loss are land-use changes, pollution, overexploitation of resources, spread of invasive species and climate change. These drivers have a direct effect on ecosystems, but their dynamics are influenced by indirect drivers such as sociopolitical context, economic activity, demographic changes, cultural practices and scientific and technologic advances. For example, the adoption of environmental practices that conduce to sustainable use of resources is more likely in sociopolitical regimes that encourage the dialogue between different sectors of the society.

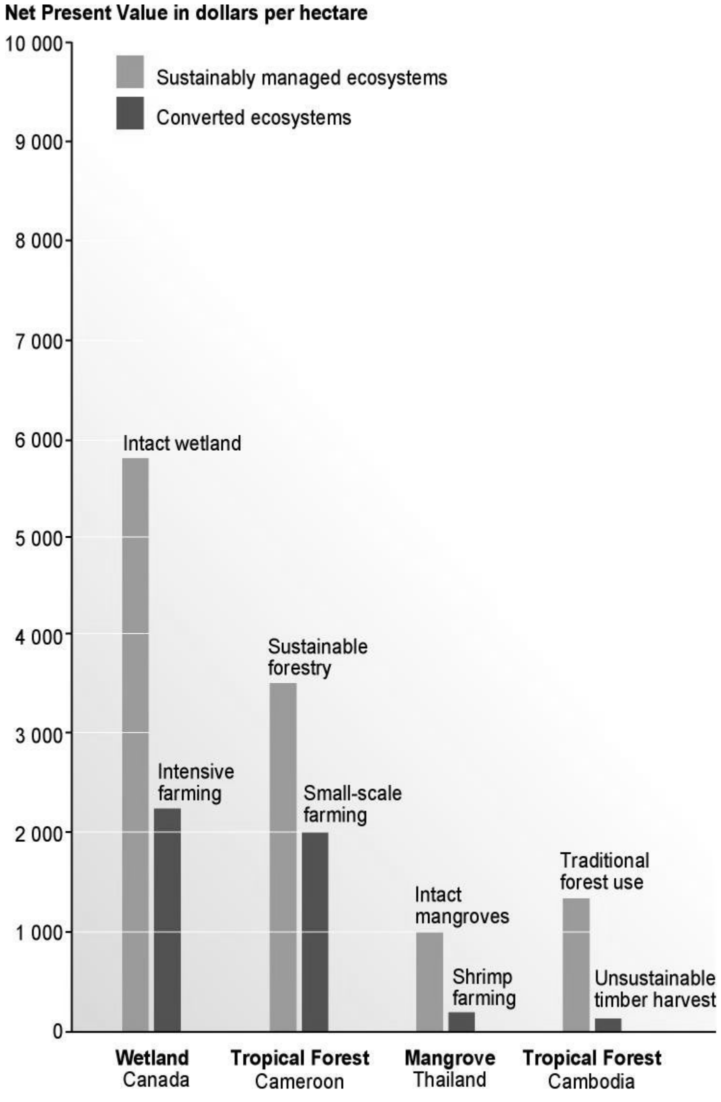
The importance of each driver is not the same across ecosystems. Terrestrial ecosystems (e.g., forests, grasslands) are especially affected by land use changes, particularly the conversion of natural habitat to agricultural land. The main driver affecting marine ecosystems is overexploitation of fish stocks, whereas pollution and invasive species are currently the major threats to freshwater ecosystems.

Ecosystem changes are the result of synergistic combinations of the interactions between drivers. Moreover, drivers also interact across spatial and time scales and ecosystem changes might be caused by events that occurred somewhere in the past. For example, isolated events as deforestation of tropical forests, fires in Mediterranean ecosystems and global emissions of greenhouse gases from fossil fuel combustion will all contribute to climate change. Climate change affects local communities worldwide through the occurrence of storms, floods, sea rise and droughts.

Reports of natural catastrophes (e.g., floods, storms, tornados) costs are quite demonstrative of the effects of ecosystem changes and biodiversity loss in human well-being. Global costs of natural disasters between 1980 and 2004 reached values superior to $1800 billion. In 2002 alone economic losses were evaluated in $70 billion. The effects of natural disasters go beyond economic losses: more than a million deaths between 1980 and 2004, destruction of public infrastructures and social instability. Part of these economic and human losses can be attributed to the deterioration of regulating services and poor land planning. Poorer communities, unable to react to disasters are more affected by these events and might face subsequent epidemics, famine and social conflicts.

Human pressure on ecosystems usually intends to intensify the production of ecosystem goods, but frequently disregards the degradation of other services, often regulating services. For instance the use of pesticides and fertilizers in agriculture enhances production levels but negatively affects the quality of groundwater. Commodities have a market value and their economic benefits are easily accessed, therefore they are considered in management options. In contrast, there are no markets for regulating and supporting ecosystem services, and as a consequence those services lack economic value and are often disregarded. However, the costs of losing these services are sometimes higher than the economical benefits obtained from marketed goods, and the final balance can be critical to human well-being.

Some studies have compared the economic benefits from preserving natural ecosystems versus the economic profits obtained from converted land (Figure 3). In Canada, freshwater marshes are drained and used for agriculture due to their high fertility. Preserved habitats offer high quality areas for outdoor activities, as hunting and fishing, and provide higher economic benefits than converted wetlands. In Cameroon forests are also converted to farming land. Benefits from maintaining forests include soil protection against erosion, carbon retention but also biodiversity option values and existence values. In Thailand mangroves are converted into aquacultures for shrimp farming. Non-converted mangroves supply several goods as timber, charcoal and fish and provide coastal protection from storms. Traditional forest use in Cambodia includes the practice of swidden agriculture (agriculture made in short-term fields created from cutting and burning forest patches) and extraction of forest products (timber, food, medicines). Provisioning services from traditional use provide fewer profits than unsustainable logging. However if other ecosystem services are considered, as carbon retention, water retention and biodiversity, unsustainable use of forests becomes less profitable than traditional use.



Source: Millennium Ecosystem Assessment

Figure 5.3 - Economic benefits from preserving natural ecosystems versus the profits obtained from converted land (from Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: synthesis. Washington, DC: Island Press).

In all these cases, the gains from the production of goods are large for private owners but for the country economy the final balance is not lucrative, either due to the costs of converting land (e.g., cost of draining marshes) or due to the loss of the services obtained from sustainable managed ecosystems. A last example (not represented in the graphic) comes from New York City. The city watershed had been under pressure for development with negative consequences for water quality. The city faced two options: build water treatment facilities to deal with decreasing water quality or protect the watershed ecosystems. The cost of building water treatment facilities was estimated at $8 billion, plus $300 million per year for maintenance. The cost of having that service provided by ecosystems was $1 billion, corresponding to the ecological restoration of the watershed that supplies the city with water and to economic compensations to land owners in order to maintain the habitat preserved. As a result of this valuation, New York City has decided to protect the watershed ecosystem.

5.7 Forest ecosystem services and human well-being

Forests constitute ubiquitous ecosystems vital for the biosphere equilibrium. Forests are central to the biogeochemical cycles (e.g., carbon cycle), support much biodiversity and provide many ecosystem services (Figure 4). Humans benefit from forest services at all spatial scales, for example fuelwood at the local scale, water purification at the regional scale and climate regulation at the global scale. Due to historical human action about 40% of the original world’s forests have been destroyed and much of the remaining forest is fragmented or perturbed.

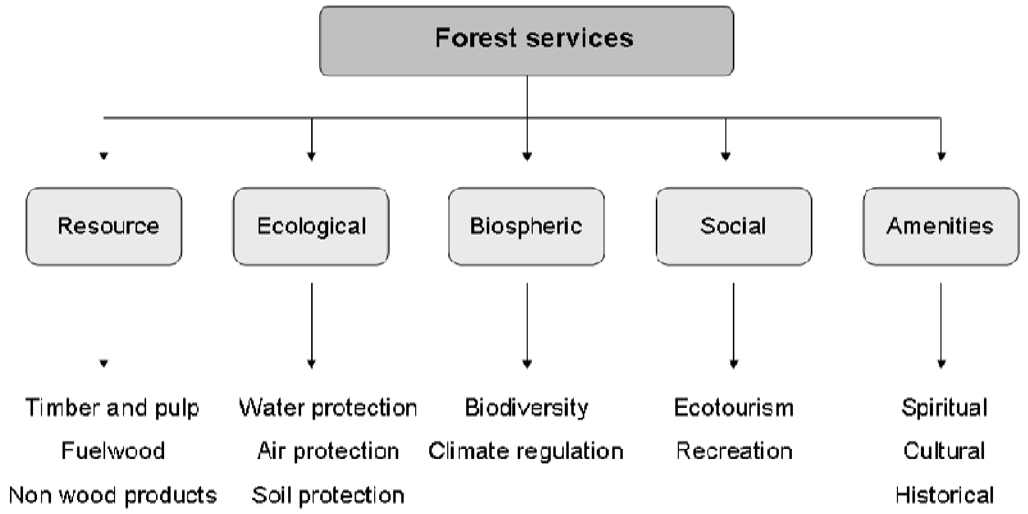


Figure 5.4 - Forest ecosystem services (adapted from Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: Current state and trends. Washington, DC: Island Press with permission).

This section examines in more detail the feedback loop between forest biodiversity, ecosystems condition and human well-being. The evolution of the Portuguese forest is used as a case study for this analysis.

The Portuguese landscape was dominated by oak forests (*Quercus spp.*) in the beginning of the Holocenic. Evergreen oaks, such as the cork oak (*Quercus suber*) and holm oak (*Quercus ilex*), were dominant in the south and deciduous oaks, such as the common oak (*Quercus robur*) and pyrenean oak (*Quercus pyrenaica*), in the north. Land use changes started with human settlement in the Neolithic and gradually continued until today. Forests were first cleared due to the need of land for pastures and agriculture and then due to the increasing need for fuelwood and industrial wood, in particular for naval construction during the maritime discoveries (fifteenth to sisteenth centuries). Records dating from the seventeenth century report the occurrence of severe floods. The deforestation of mountains lead to soil erosion and to the sedimentation of soil particles into riverbeds. The changes in the riverbeds and the increased surface run-off caused the floods. Agriculture was by that time the main land use in the country and existing forest had already changed in composition, with pine becoming the dominant species.

Since the late nineteenth century several major forestation programs took place and forest is nowadays the main land-cover class. Forestation programs were first developed by the government and later by privates. More recently, since Portugal became a member in 1986, there were also subsidies from the European Community. The first motivation for these programs was the need of restoring forested area and forest ecosystem services including soil protection and flood regulation. Later there was commercial demand for wood and wood pulp due to the growth of forest industry. The species used for forestation were mainly pine (*Pinus pinaster*) and eucalypt (*Eucalyptus globulus*), both fast growing species and with a high commercial value.

The state of degradation of ecosystems after centuries of unsustainable use conditioned the selection of trees species to those that were able to cope with poor environmental conditions and simultaneously halt land erosion. Pine was chosen due to its ecologic properties and commercial value. It is a fast growing species with pioneer abilities (i.e., it is able to colonize poor environments and create the conditions for the establishment of richer and complex ecosystems) and also provides important economic benefits from timber, resin and other forest products. Pine was intensively planted in the first half of the twentieth century, and its area of distribution reached a maximum in the 1980s. Although it is an autochthonous species its actual distribution is unnatural and much expanded.

The plantation of eucalypt was motivated only by interests in direct economic benefits (wood pulp). Plantation started after 1950 and was more intense in the 1980s. The growth of the wood pulp industry was the principal driver of eucalypt forest plantation.

Pine and eucalypt are now the dominant species in the northern half of the country. Broadleaf deciduous oaks have a very limited distribution, being dominant only in some mountain areas. In the southern half of the country, perennial oaks are still dominant, due to their commercial value for the production of cork and their production of acorns for cattle.

Recent statistics indicate that Portugal is on the top ten countries with the largest annual gain of forest (production forest plantations), an annual net gain of 40.000 ha y-1 (2000-2005). The Portuguese forestry sector is a source of income to approximately half million people. This shows the importance of planted forests for human well-being. Pine and eucalypt forests are the principal sources of wood products, namely timber and pulp. Planted forests also provide non-timber forest products, such as pine nuts and resin, and if properly managed they also provide other ecosystem services such as soil fixation (some pine forests were planted to protect coastal dunes) and carbon retention. Although planted forests generally support less biodiversity than native forests, they may promote biodiversity when placed in formerly degraded habitats.

Nevertheless, failures in the planning and management of planted forests facilitated the occurrence of wildfires, which is now one of main environmental and economic problems affecting Portugal (in 2003 about 300 000ha of forest was burnt, which corresponds to about 3% of the country area). Indeed, most of the planted forests were not planned to be structurally diverse. Instead they are monospecific and occupy large continuous areas. Moreover the intense migration of people from rural areas to urban centers during the last decades, left forests abandoned and without management. Therefore fuel loads have accumulated and the landscape is currently much less compartmentalized, leading to large-scale fires. This situation is especially difficult to control because most forest belongs to small private owners and only their organized action would have effective results on forests condition.

The problem is aggravated by the fact that both pine and eucalypt trees contain flammable substances that intensify fire and even the litter, either fallen needles or leafs, is very flammable, thus contributing to fire progression. In fact, both species have evolved in fire prone ecosystems and have traits to cope with fire damage and to persist after fire occurrence. The consequence of all these factors (bad planning, lack of management and planted species flammability) is the occurrence of severe fires every year, in particular during summer months, when temperatures are high and precipitation low.

Forest fires have severe effects on human well-being. First, there are the direct effects on the economy, affecting several sectors of the society, from industries to small owners and also public administration. Besides the destruction of forests, fires also affect public infrastructures and destroy private property, as houses, cars, and cattle. There are also health costs, such as injuries and deaths caused by fires and respiratory problems caused by particle emissions. Furthermore, fire combat operations are very expensive, with a single hour of air operations costing over €2000.

Indirect effects on human well-being are mediated by the consequences of wildfires for ecosystem services. Wildfires have negative impacts in air purification services (during the process of photosynthesis, oxygen is produced and carbon dioxide and polluting particles removed from the atmosphere) and water cycle regulation. Soil quality is affected after fire as well. During the combustion of forest materials, minerals are released and stay on the soil surface. However, due to the absence of vegetation, the soil erodes by losing minerals and the remaining organic material. The lack of vegetation also increases the probability of burnt areas to suffer landslides and floods. Fires affect climate regulation as well, both at the local and global scale. Climate is affected at a local scale when forest cover disappears, since forests have a vital role in the water cycle, which regulates air humidity and temperatures.

Fires also affect climate at the global scales, since the release of carbon to the atmosphere will contribute to global warming.

After a long history of forest destruction and the reversion of the pattern with the plantation of new forests, the Portuguese forest now faces new threats. Forested area is increasing but forest plantations present several barriers to attain sustainability. They support less biodiversity than native forests and are wildfire prone. As a consequence forest habitats are facing a progressive degradation and are being replaced by scrublands, an early successional habitat. This tendency has serious implications for people that lose the benefits of forest ecosystems.

The need for new forest management strategies is evident. Future actions have to consider the promotion of biodiversity and ecosystem services. Oak forests are climax forests, i.e., they represent the last stage of succession. Oak forest communities are biodiverse and adapted to local environmental conditions, which confers them resistance and resilience to perturbation. In the case of fire, and considering the present fire regimes, oak forests are more resistant to fire perturbation than pine or eucalypt forests. This is partially due to their high water content (water is retained in soil, litter and vegetation), which raises moisture levels and diminishes flammability. This resistance to fire also assures the maintenance of forest processes and forest services, reducing the risks of soil erosion, soil nutrient loss, floods, etc.

Oak forest provides important regulating services, such as water protection, climate regulation and soil formation. The water protection service is especially important because the remaining oak forests are mostly found in mountain environments where precipitation is usually greater. Because oak forests usually have a highly permeable soil, the amount of rain water lost by runoff is reduced. Some water stays retained in the superficial soil layers and the remaining is percolates deeper and deeper being purified in the process and finally reaching groundwater and in due course joining streams and rivers. With respect to climate regulation, oak forests have an important role as carbon sinks owing to their stability.

Due to their past destruction and actual distribution the larger stands of oak are located in protected areas and their management is focused in nature conservation targets. Therefore the most profitable provisioning service, timber provisioning, is currently marginal. Nevertheless, oak wood has a high quality, being resistant and durable and with a significant commercial value. The extant forests might have a good economical potential as productive forests if correctly managed, and investment in new plantations of oak for timber production can be economically attractive. Finally, native forests also provide important cultural services. For example, their natural spaces present a high potential for ecotourism and outdoor activities, which also has a use value for local communities as an important source of income.

All summed, and although there are not economic estimations of the value of services provided by native forests in Portugal, it is quite probable that their total benefits might surpass the actual profits of pine and eucalypt forest, when the cost of wildfires is accounted for. Moreover, native forests support much biodiversity and should also be promoted by their option and existence values.

5.8 Finding the way to sustainability

The overexploitation of resources has led to the present biodiversity crisis. The need to halt biodiversity loss is unquestionable. The Convention for Biological Diversity (CBD) was opened for signature in 1992. Since then, 195 countries committed to the challenge of halting biodiversity loss. More recently, the Conference of the Parties, the governing body of the Convention, has decided “to achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional and national level as a contribution to poverty alleviation and to the benefit of all life on earth”. The large number of countries involved in the “2010 target” is a sign of the international community awareness about the urgent need of actions towards the protection of ecosystems and their services. On the other hand, making progress towards the 2010 target will also contribute to the Millennium Development Goals (MDG) of the United Nations. The MDG determine, among other things, the reduction of poverty and child mortality, the investment in education and the integration of the principles of environmental sustainability in political decisions.

Human well-being and ecosystem protection have been sometimes at odds at the local scale because of the spatial disconnection between where ecosystems services are provided and where people benefit from them. This is further aggravated because many of these services are nonmarket services, being often disregarded in economic decisions. Nevertheless the value of the benefits of nonmarket services is considerable and should be included in management strategies and development policies. For instance much of the remaining natural areas are located in developing regions, where essential components of well-being have not yet been achieved. If nonmarket ecosystem services are valued, these populations will be in possession of tradable goods. The populations in these regions can be paid to maintain their natural habitats and use this new source of income for development. The commerce of carbon credits is an example. Polluters, from countries to industries or common citizens, compensate their carbon emissions by paying for the maintenance of forests that will sequester and retain carbon. Some states have also recognized the importance of ecosystem services of certain habitats by passing legislation directed at protecting those habitats. For instance wetlands, which are important for water purification and flood regulation, are currently protected by laws in many countries.

An alternative is to find win-win solutions at the local scale, that is, options that favor both ecosystem condition and socio-economic development locally. This approach has been followed in some projects aiming at the sustainable management of forests by commercializing a wider range of forest products, including timber, edible products such as mushrooms, fodder for domestic animals, medicines, and exudates such as gums.

Probably, a combination of both local and global approaches will be the key to the best sustainability strategies. And, at the consumer end, the adoption of a behavioral conduct of saving energy and other resources, as food and water, is also essential. Ecosystems would be preserved from unnecessary pressure demands and as a consequence human well-being would be improved.

Finally, it is necessary to promote studies, from local to global scales, to obtain information about the state of ecosystems and their linkages with human well-being, and make the results available to the broadest specter of people, from general public, to politicians, teachers and scientists. Recently one such study was conducted, the Millennium Ecosystem Assessment (MA). The MA involved more than 1300 scientists from 95 countries. The result was a comprehensive set of reports, freely available, on the state of ecosystems, their services and the linkages to human-well being, at different spatial scales. An MA2 is now being planned to start somewhere around 2010, but other studies looking at successful management strategies using local and global approaches are needed.

In summary, human well-being and biodiversity are intimately related. The consequences of biodiversity loss to human well-being are diverse and difficult to predict with exactitude. They might reach unexpected severity and affect people anywhere in the world. To assure the long-term improvement of human well-being worldwide we must develop sustainable strategies for managing ecosystems services.

Further reading

Brooks TM, Mittermeier RA, da Fonseca GAB, et al. (2006) Global biodiversity conservation priorities. Science 313: 58-61

Balmford A, Bruner A, Cooper P, et al. (2002) Economic reasons for conserving wild nature. Science 297 : 950-953

Costanza R, d'Arge R, de Groot R, et al. (1997) The value of the world's ecosystem services and natural capital. Nature 387: 253 – 260

Daily GC, Söderqvist T, Aniyar S, et al. (2000) The value of nature and the nature of value. Science 289: 395–396

Díaz S, Fargione J, Chapin FS, Timan D (2006) Biodiversity loss threatens human well-being. PLoS Biology 4: 1300–1305

FAO (2006) Global forests resources assessment 2005: progress towards sustainable forest management. FAO, Rome

Loreau MS, Naeem S, Inchausti P, et al. (2001) Biodiversity and ecosystem functioning: current knowledge and future challenges. Nature 294: 804-808

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington, DC

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press,Washington, DC

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: current state and trends. Island Press, Washington, DC

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: health synthesis. World Health Organization, Geneva

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403: 853–858

Pereira HM, Cooper DH (2006) Towards the global monitoring of biodiversity change. Trends in Ecology and Evolution 21: 123-129

Web based resources

http://www.biodiv.org

http://www.conservation.org

http://www.millenniumassessment.org

http://www.unep.org

CONCLUDING REMARKS

6 Concluding Remarks

The main purpose of this dissertation was to assess the performance of modified natural forest in terms of biodiversity and response to fire. This study is pertinent because natural regeneration is currently occurring after agricultural land abandonment and also because the Portuguese forest has been much affected by human activity along history, presenting today an altered composition that promotes the occurrence of severe fires.

The study focused the Galicio-Portuguese oak forests of *Quercus robur* and *Quercus pyrenaica*. The contribution of these forests for biodiversity conservation was assessed in two contexts: in comparison with pine and eucalypt plantations, the most widespread forest species in Portugal, and in a countryside landscape, which represents the current framework of deciduous oak forests in Portugal. Resistance and resilience to fire were assessed in comparison to pine plantations after a wildfire that affected both forest types.

Natural forests patches are important for the conservation of forest species

Deciduous oak forest patches supported more forest species than pine and eucalypt plantations and several species showed a preference for oak forest habitats. Moreover, the richness of forest species responded more to area in natural forest patches than in plantations, suggesting that larger patches of oak forest are particularly important for the maintenance of forest species.

Natural forest patches are important for species diversity in countryside landscapes

Forest patches in the multi-habitat landscape supported a floristic community distinct from communities in agricultural and shrubland habitats, thus contributing to overall richness in the landscape. Moreover, the fact that some species only occurred in forest patches stressed the relevance of these habitats for biodiversity conservation.

Natural forest patches are more resistant and resilient to fire than pine plantations

Signs of fire severity after a wildfire suggested that natural forest patches of broadleaved species (*Quercus robur* and *Ilex aquifolium*) are more resistant to fire than pine plantations. Tree mortality in burned natural forest did not differ from mortality in unburned natural forest, while it was significantly higher in burned pine stands relative to unburned stands. With regard to post-fire reestablishment, floristic communities in burned natural forest were in a more advance stage of recovery at the time of the study, thus suggesting a higher resilience of natural broadleaved forests.

A vision for the Portuguese forest in 21st century

Despite the extensive forest loss and degradation caused by millennia of human activity, deciduous oak forest patches continue to support rich communities of species, being of critical importance for several species. Moreover, when compared to pine plantations, this forest is better adapted to fire disturbance, due to the higher resistance of dominant trees and the higher resilience of floristic communities. Considering all that, it is urgent to acknowledge the priority value of deciduous oak forests for conservation and their potential role in future forest management.

The Portuguese forest is presently dominated by eucalypt and pine plantations, in result of forestation programmes carried out during the last century (DGRF 2007, Silva et al. 2008). While the restoration of ecosystem services was one of the reasons motivating the first forestation initiatives, forest plantations during the second half of the twentieth century were mainly set due to economical reasons, for timber and pulp production (Radich and Alves 2000, Mendes and Fernandes 2007). The lack of an adequate forest planning and management lead the Portuguese forest to its present condition: a forest dominated by fire prone species, extending continuously through large areas and often lacking adequate management (Fernandes 2007). This setting increases fire risk compromising not only wood production but also other services (e.g., erosion control) that these forests may also provide.

The current distribution of deciduous oak forest in Portugal is very scarce and fragmented, and most patches, at least the larger ones, occur in protected areas. While the protection of existent forests is in assured by conservation tools, such as the Habitat Directive, there is a lack of initiatives to promote their expansion, although this is already considered in the National Forest Strategy (DGRF 2007) and in the Permanent Forest Fund (Fundo Florestal Permanente, Decreto-Lei nº63/2004, de 22 de Março). However, the way to forest sustainability has necessarily to incorporate an increase in the area covered by native broadleaved species. Two lines of forest management could be followed towards this purpose: natural forest regeneration and reforestation using native broadleaved species (Chazdon 2008). The choice between both approaches will depend on the level of ecosystem degradation. Natural forest regeneration will be adequate when dispersion sources (natural forest patches) and faunal dispersal agents (e.g., jays, squirrels) are present. If dispersal sources or agents are absent then reforestation will be the only choice (Chazdon 2008). In both cases soil conditions will have to be adequate for forest developments (Chazdon 2008). For example, the extensive use of fire may cause the loss of superior soil horizons and inhibit the reestablishment of vegetation communities that require more profound soils (Torres et al. 2002) However, the pathways of forest reestablishment are still weakly understood and future research is needed to better identify sites' potential for natural regeneration and to support the choice of management strategies (Chazdon 2008, Aubin 2008, Cramer et al. 2008).

Natural regeneration should be preferred to reforestation whenever possible because it may restore more biodiversity and a larger range ecosystem services (Chazdon 2008, Benayas et al. 2008). However, while natural regeneration may have a lower cost compared to other alternatives, it has the major drawback of requiring a longer time interval (Chazdon 2008, Benayas et al. 2008). Intermediate solutions between natural regeneration and reforestation may contribute for a faster development of natural forest in abandoned land, while still considering the restoration of biodiversity and ecosystem services (Benayas et al. 2008).

Regardless of the relevance already granted to sustainable forest management and multifunctional forests (MCPFE 2007, DGRF 2007), the prevailing paradigm of forest management is still centred on fast timber production. The transition towards a paradigm of multifunctional forests is dependent on a shift in society valuation and demand for non-marketed forest services. Multifunctional forests and ecosystem services will only gain true relevance in forestry when society not only acknowledges the value of services like air purification and water retention but is also willing to subsidize efforts that promote the provision of those services (Patterson and Coelho 2009).

In summary, socio-economic changes in the last six decades have created the conditions for the natural regeneration of deciduous oak forests. While forest succession is taking place in abandoned fields there is an urgent need for actions that support the development of naturally regenerated forests. Management options that join efforts with natural succession may be the best and less expensive way to make the transition towards multifunctional forests (Benayas et al. 2008, Chazdon 2008). According to the results presented in this study, modified natural forests constitute chief habitats for forest species. Therefore, these forests may act, along with remaining patches of primary forests, as sources of species dispersion to semi-natural forests and to forest plantations, contributing for the restoration of biodiversity in these habitats and in the landscape.

References

Aubin I, Messier C, Bouchard A (2008) Can plantations develop understory biological and physical attributes of naturally regenerated forests? Biological Conservation 141:2461-2476.

Benayas JMR, Bullock JM, Newton AC (2008) Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. Frontiers in Ecology and the Environment 6:329-336.

Carvalho J (2007) Conservação, regeneração e exploração do carvalho-negral. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Chazdon RL (2008) Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands. Science 320:1458-1460.

Cramer VA, Hobbs RJ, Standish RJ (2008) What's new about old fields? Land abandonment and ecosystem assembly. Trends in Ecology & Evolution 23:104-112.

DGRF – Direcção Geral dos Recursos Florestais (2007) Resultados do IFN 2005/06. Direcção Geral dos Recursos Florestais. Available in http://www.dgrf.min-agricultura.pt.

Fernandes P (2007) Entender porque arde tanto a floresta em Portugal. In: Silva JS (ed.). Proteger a floresta: incêndios, pragas e doenças. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

MCPFE - Ministerial Conference on the Protection of Forests in Europe (2007) State of Europe’s Forests 2007: The MCPFE report on sustainable forest management in Europe. Ministerial Conference on the Protection of Forests in Europe Liaison Unit Warsaw.

Mendes A, Fernandes L (2007) Politicas e instituições florestais em Portugal – desde o final do Antigo Regime até à actualidade. In: Silva JS (ed.). Floresta e Sociedade. Uma história em comum. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Patterson TM, Coelho DL (2009) Ecosystem services: Foundations, opportunities, and challenges for the forest products sector. Forest Ecology and Management 257:1637-1646.

Radich MC, Alves AAM (2000) Dois Séculos da Floresta em Portugal, CELPA, Lisboa.

Silva JS, Deus E, Saldanha L (2008) Evolução dos incêndios florestais em Portugal, antes e depois de 2003. In: Silva JS, Deus E, Saldanha L (eds.) Incêndios Florestais, 5 anos após 2003. LPN, AFN.

Torres J, Honrado J, Caldas FB (2002) A vegetação da "Mata do Ramiscal" (Serra do Soajo, Parque Nacional da Peneda-Gerês). UGEV-ICETA, Universidade do Porto.

REFERENCES

References

Abelho M, Graça MAS (1996) Effects of eucalyptus afforestation on leaf litter dynamics and macroinvertebrates community structure of streams in central Portugal. Hydrobiologia (Historical Archive) 324:195-204.

Aguiar C, Maravalhas E (2003) A ecologia das borboletas. In: Maravalhas E (ed) As Borboletas de Portugal. Porto.

Aguiar C, Pinto B (2007) Paleo-história e história antiga das florestas de Portugal continental até à Idade Média. In: Silva JS (ed.). Floresta e Sociedade. Uma história em comum. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Aguiar C, Rodrigues O, Azevedo J, Domingos T (in press) Montanha. In: Pereira HM, Domingos T, Vicente L and V Proença (eds.) Ecossistemas e Bem-Estar Humano: Resultados da Avaliação do Milénio para Portugal. CELTA Editora.

Alves AL, Carvalho NS, Silveira SCB, Marques JP, Costa Z, Horta ALL (2003) O abandono da actividade agrícola. Ministério da Agricultura do Desenvolvimento Rural e das Pescas, Grupo de trabalho agro-ambiental, Lisboa.

Alves AM, Pereira JS, e Silva JMN (eds.) (2007) O Eucaliptal em Portugal. Impactes Ambientais e Investigação Científica. ISAPress, Lisboa.

Amman GD, Ryan KC (1991) Insect infestations of fire-injured trees in the greater Yellowstone area. USDA Forest Service, Intermountain Forest and Range Experiment Station. Research Note INT-398. Ogden.

Arrhenius O (1921) Species and Area. The Journal of Ecology 9:95-99.

Atauri JA, de Lucio JV (2001) The role of landscape structure in species richness distribution of birds, amphibians, reptiles and lepidopterans in Mediterranean landscapes. Landscape Ecology 16:147-159.

Aubin I, Messier C, Bouchard A (2008) Can plantations develop understory biological and physical attributes of naturally regenerated forests? Biological Conservation 141:2461-2476.

Baeza M, Valdecantos A, Alloza J, Vallejo V (2007) Human disturbance and environmental factors as drivers of long-term post-fire regeneration patterns in Mediterranean forests. Journal of Vegetation Science 18: 243–252.

Balmford A, Bruner A, Cooper P, et al. (2002) Economic reasons for conserving wild nature. Science 297 : 950-953

Barlow J, Gardner TA, Araujo IS, Ávila-Pires TC, Bonaldo AB, Costa JE, Esposito MC, Ferreira LV, Hawes J, Hernandez MIM, Hoogmoed MS, Leite RN, Lo-Man-Hung NF, Malcolm JR, Martins MB, Mestre LAM, Miranda-Santos R, Nunes-Gutjahr AL, Overal WL, Parry L, Peters SL, Ribeiro-Junior MA, da Silva MNF, da Silva Motta C, Peres CA (2007) Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proceedings of the National Academy of Sciences 104:18555-18560.

Bassi S, Kettunen M, Kampa E, Cavalieri S (2008) Forest Fires: Causes and Contributing Factors to Forest Fire Events in Europe Study for the European Parliament Committee on Environment, Public Health and Food Safety under contract IP/A/ENVI/FWC/2006-172/LOT1/C1/SC10.

Benayas JMR, Bullock JM, Newton AC (2008) Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. Frontiers in Ecology and the Environment 6:329-336.

Bengtsson J, Nilsson SG, Franc A, Menozzi P (2000) Biodiversity, disturbances, ecosystem function and management of European forests. Forest Ecology and Management 132:39-50.

Bennett AF, Radford JQ, Haslem A (2006) Properties of land mosaics: Implications for nature conservation in agricultural environments. Biological Conservation 133:250-264.

Bernaldez FG (1991) Ecological consequences of the abandonment of traditional land use systems in central Spain. Options Méditerranéenes - Série Séminaires 15: 23-29.

Bignal EM, McCracken DI (1996) Low-intensity farming systems in the conservation of the countryside. Journal of Applied Ecology 33: 413-424.

Blake JG, Schuette B (2000) Restoration of an oak forest in east-central Missouri: Early effects of prescribed burning on woody vegetation. Forest Ecology and Management 139: 109-126.

Blondel J, Aronson J (1999) Biology and wildlife in the Mediterranean region. Oxford University Press, Oxford.

Bond JW, Keeley JE (2005) Fire as a global ‘herbivore’: the ecology and evolution of flammable ecosystems. Trends in Ecology and Evolution 20: 387–394.

Bond WJ, Midgley JJ (2001) Ecology of sprouting in woody plants: the persistence niche. Trends in Ecology and Evolution 16: 45–51.

Bond WJ, van Wilgen BW (1996) Fire and Plants. Chapman & Hall, London.

Boteva D, Griffiths G, Dimopoulos P (2004) Evaluation and mapping of the conservation significance of habitats using GIS: an example from Crete, Greece. Journal for Nature Conservation 12:237-250.

Broncano MJ, Retana J, Rodrigo A (2005) Predicting the Recovery of Pinus halepensis and Quercus ilex Forests after a Large Wildfire in Northeastern Spain. Plant Ecology 180: 47-56.

Brooks TM, Mittermeier RA, da Fonseca GAB, et al. (2006) Global biodiversity conservation priorities. Science 313: 58-61

Brotons L, Herrando S (2001) Factors affecting bird communities in fragments of secondary pine forests in the north-western Mediterranean basin. Acta Oecologica. 22: 21-31.

Brown JK, Smith JK (2000) Wildland fire in ecosystems: effects of fire on flora. USDA Forest Service, Rocky Mountain Research Station. Gen.Tech. Rep. RMRS-GTR-42-vol.2. Ogden.

Brown RT, Agee JK, Franklin JF (2004) Forest Restoration and Fire: Principles in the Context of Place. Conservation Biology 18: 903–912.

Buhk C, Gotzenberger L, Wesche K, Sánchez Gómez P, Hensen I (2006) Post-fire regeneration in a Mediterranean pine forest with historically low fire frequency. Acta Oecologica 30: 288–298.

Caldas EC (1998) Agricultura na História de Portugal. EPN - Empresa de Publicações Nacionais, Lda., Lisboa.

Calvo L, Santalla S, Marcos E, Valbuena L, Tárrega R, Luis-Calabuig E (2003) Regeneration after wildfire in communities dominated by Pinus pinaster, an obligate seeder, and in others dominated by Quercus pyrenaica, a typical resprouter. Forest Ecology and Management 184: 209–223.

Calvo L, Santalla S, Valbuena L, Marcos E, Tárrega R, Luis-Calabuig E (2008) Post-fire natural regeneration of Pinus pinaster forest in NW Spain. Plant Ecology 197: 81–90.

Cantelaube P, San-Miguel-Ayanz J, Schmuck G, Biard F (2002) Towards forecasting forest fire risk:Comparison of Meteorological forecast data from MeteoFrance with measured meteorological data from the JRC-MARS database. JRC, IES, Land Management Unit. European Communities, Italy.

Carnus JM, Parrotta J, Brockerhoff E, Arbez M, Jactel H, Kremer A, Lamb D, O'Hara K, Walters B (2006) Planted forests and biodiversity. Journal of Forestry 104: 65-77.

Carvalho A (1997) Madeiras portuguesas: estrutura anatómica, propriedades e utilização. Instituto Florestal, Lisboa.

Carvalho J (2007) Conservação, regeneração e exploração do carvalho-negral. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Alves PC, Grosso-Silva JM, Santos TM (2007) Biologia e ecologia das florestas de carvalho-negral. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Alves PC, Grosso-Silva JM, Santos TM (2007) Biologia e ecologia do carvalho-roble. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Santos JA, Reimão D (2007) Utilizações do carvalho roble. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Carvalho J, Santos JA, Reimão D, Santos J (2007) A valorização dos carvalhais. In: Silva JS (ed.). Os carvalhais. Um património a conservar. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Castro EB, Casado MA, Costa M, Escribano R, García M, Génova M, Manzaneque AG, Moreno JC, Morla C, Regato P, Sáinz H (2001) Los Bosques Ibéricos: Una Interpretación Geobotánica. Editorial Planeta, Barcelona.

Chazdon RL (2008) Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands. Science 320:1458-1460.

Christian DP, Hoffman W, Hanowski JM, Niemi GJ, Beyea J (1998) Bird and mammal diversity on woody biomass plantations in North America. Biomass and Bioenergy 14: 395-402.

Ciancio O, Nocentini S (1997) The forest and man: the evolution of forestry thought from modern humanism to the culture of complexity. Systemic silviculture and management on natural bases. In: Ciancio O (Ed) The forest and man, Accademia Italiana di Scienze Forestali, Firenze, Italy, pp. 21-114.

Colwell RK (2005) EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.5. User's Guide and application published at: http://purl.oclc.org/estimates.

Costanza R, d'Arge R, de Groot R, et al. (1997) The value of the world's ecosystem services and natural capital. Nature 387: 253 – 260

Cramer VA, Hobbs RJ, Standish RJ (2008) What's new about old fields? Land abandonment and ecosystem assembly. Trends in Ecology & Evolution 23:104-112.

Daily GC, Ehrlich PR, Sanchez-Azofeifa GA (2001) Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11:1-13.

Daily GC, Söderqvist T, Aniyar S, et al. (2000) The value of nature and the nature of value. Science 289: 395–396

Davies KF, Margules CR, Lawrence JF (2000) Which traits of species predict population declines in experimental forest fragments? Ecology 81:1450-1461.

Davies KF, Melbourne BA, Margules CR (2001) Effects of within- and between-patch processes on community dynamics in a fragmentation experiment. Ecology 82:1830-1846.

Davies-Colley RJ, Payne GW, van Elswijk M (2000) Microclimate gradients across a forest edge New Zealand Journal of Ecology 24: 111-121.

DGF – Direcção Geral das Florestas (2001) Inventário Florestal Nacional. 3ª Revisão. Direcção Geral de Florestas, Lisboa.

DGRF – Direcção Geral dos Recursos Florestais (2007) Estratégia Nacional para as Florestas. Imprensa Nacional da Casa da Moeda, Lisboa.

DGRF – Direcção Geral dos Recursos Florestais (2007) Resultados do IFN 2005/06. Direcção Geral dos Recursos Florestais. Available in http://www.dgrf.min-agricultura.pt.

DGRF (2007) Resultados do IFN 2005/06. Direcção Geral dos Recursos Florestais. Available from http://www.dgrf.min-agricultura.pt.

Díaz S, Fargione J, Chapin FS, Timan D (2006) Biodiversity loss threatens human well-being. PLoS Biology 4: 1300–1305

Diaz-Delgado R, Lloret F, Pons X, Terradas J (2002) Satellite Evidence of Decreasing Resilience in Mediterranean Plant Communities after Recurrent Wildfires. Ecology 83: 2293–2303.

Dimitrakopoulos A, Papaioannou KK (2001) Flammability Assessment of Mediterranean Forest Fuels. Fire Technology 37: 143–152.

Doerr SH, Shakesby RA, Walsh RPD (2000) Soil water repellency: its causes, characteristics and hydro-geomorphological significance. Earth-Science Reviews 51:33-65.

Domínguez L, Calvo L, Luis E (2002) Impact of wildfire season on regeneration of Quercus pyrenaica forest and Pinus sp. stands. Journal of Mediterranean Ecology 3: 47–54.

EC – European Community (2004) Evaluating Financing of Forestry in Europe, Projecto EFFE, Consórcio European Forest Institute (Finlândia), University of Joensuu (Finlândia), University of Ghent (Bélgica), University of Freiburg (Alemanha), Polish Forest Research Institute (Polónia), Aristoteles University of Thessaloniki (Grécia) e Universidade Católica Portuguesa-CRP (Portugal).

EEA – European Environmental Agency (2003) Europe's environment: the third assessment. Office for Official Publications of the European Communities.

EEA – European Environmental Agency (2006) Progress towards halting the loss of biodiversity by 2010. EEA report 5/2006.

EEA – European Environmental Agency (2008) European forests - ecosystem conditions and sustainable use. EEA report 3/2008

Endels P, Adriaens D, Verheyen K, Hermy M (2004) Population structure and adult plant performance of forest herbs in three contrasting habitats. Ecography 27: 225- 241.

Fabbio G, Merlo M, Tosi V (2003) Silvicultural management in maintaining biodiversity and resistance of forests in Europe--the Mediterranean region. Journal of Environmental Management 67:67-76.

Fagan WF, Cantrell RS, Cosner C (1999) How Habitat Edges Change Species Interactions. The American Naturalist 153:165-182.

FAO (2006) Global Forests Resources Assessment 2005: Progress towards sustainable forest management. FAO, Rome.

Farrell EP, Führer E, Ryan D, Andersson F, Hüttl R, Piussi P (2000) European forest ecosystems: building the future on the legacy of the past. Forest Ecology and Management 132:5-20.

Fernandes P (2007) Entender porque arde tanto a floresta em Portugal. In: Silva JS (ed.). Proteger a floresta: incêndios, pragas e doenças. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Fernandes P (in press) Combining forest structure data and fuel modelling to classify fire hazard in Portugal. Annals of Forest Science.

Fernandes P, Botelho H, Rego F (2005) A Piroecologia do Pinheiro Bravo. Silva Lusitana 13: 233–248.

Fernandes PM, Rigolot E (2007) The fire ecology and management of maritime pine (Pinus pinaster Ait.). Forest Ecology and Management 241: 1–13.

Fernandes PM, Vega JA, Jiménez E, Rigolot E (2008) Fire resistance of European pines. Forest Ecology and Management 256: 246-255.

Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS (2004) Regime shifts, resilience and biodiversity in ecosystem management. Annual Review of Ecology and Systematics 35: 557–581.

Franco JA (1971) Nova Flora de Portugal. Volume I. Lisboa.

Fraterrigo JM, Turner MG, Pearson SM (2005) Previous land use alters plant allocation and growth in forest herbs. Journal of Ecology 94: 548-557.

Gleason HA (1922) On the relation between species and area. Ecology 3: 158-162.

Godefroid S, Koedam N (2003) How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? Global Ecology and Biogeography 12: 287-298.

Gomes I (2008) Valorização da Biodiversidade na definição de prioridades de ordenamento no Parque Nacional da Peneda-Gerês. Dissertação para obtenção do Grau de Mestre. Instituto Superior Técnico, Lisboa.

Gomes JFP (2006) Forest fires in Portugal: how they happen and why they happen. International Journal of Environmental Studies 63: 109-119.

González JR, Palahí M, Trasobares A, Pukkala T (2006) A fire probability model for forest stands in Catalonia (north-east Spain). Annals of Forest Science 63: 169-176.

Gracia M, Retana J, Roig P (2002) Mid-term successional patterns after fire of mixed pine–oak forests in NE Spain. Acta Oecologica 23: 405–411.

Graham CH, Blake JG (2001) Influence of patch- and landscape-level factors on bird assemblages in a fragmented tropical landscape. Ecological Applications 11: 1709-1721

Green RE, Cornell SJ, Scharlemann JPW, Balmford A (2005) Farming and the Fate of Wild Nature. Science 307: 550-555.

Grissom RJ (2000) Heterogeneity of Variance in Clinical Data. Journal of Consulting and Clinical Psychology. 68: 155-165.

Guo Q (2001) Early post-fire succession in California chaparral: Changes in diversity, density, cover and biomass. Ecological Research 16: 471–485.

Halpern CB (1988) Early Successional Pathways and the Resistance and Resilience of Forest Communities. Ecology 69: 1703–1715.

Hanes TL (1971) Succession after Fire in the Chaparral of Southern California. Ecological Monographs 41: 27–52.

Hartley MJ (2002) Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management 155: 81-95.

Hermy M, Verheyen K (2007) Legacies of the past in the present-day forest biodiversity: a review of past land-use effects on forest plant species composition and diversity. Ecological Research 22:361-371.

Herrando S, Brotons L (2002) Forest bird diversity in Mediterranean areas affected by wildfires: a multi-scale approach. Ecography 25: 161-172.

Honrado JJP (2003) Flora e Vegetação do Parque Nacional da Peneda-Gerês. Dissertação de candidatura ao grau de Doutor em Biologia, Faculdade de Ciências da Universidade do Porto, Porto.

Hughes JB, Daily GC, Ehrlich PR (2002) Conservation of tropical forest birds in countryside habitats. Ecology Letters 5:121-129.

ICN – Instituto para a Conservação da Natureza (1998) Primeiro Relatório de Portugal a submeter à Conferência das Partes da Convenção sobre a Diversidade Biológica. Instituto de Conservação da Natureza, Ministério do Ambiente, Lisboa.

ICN – Instituto para a Conservação da Natureza (2006) Plano Sectorial da Rede Natura 2000. Vol. II. Valores naturais. Fichas de caracterização ecológica e de gestão. ICN. Available from http://www.icn.pt/psrn2000/caracterizacao_valores_naturais/habitats/9230.pdf.

ICNB - Instituto da Conservação da Natureza e da Biodiversidade (2009) Parque Nacional da Peneda-Gerês. Available from http://portal.icnb.pt/ICNPortal/vPT2007-AP-Geres (accessed March 2009).

INM - Instituto de Meterorologia (2009) Clima de Portugal Continental Available from http://www.meteo.pt/pt/areaeducativa/otempo.eoclima/clima.pt/index.html (accessed March 2009).

INM - Instituto Nacional de Meterorologia (2009) Caracterização climática: Ano 2006. Instituto de Meterologia, I.P. Available from http://www.meteo.pt (accessed January 2009).

Jones EW (1959) Biological flora of the British Isles: Quercus L. The Journal of Ecology 47: 169-222.

Keeley JE, Fotheringham CJ (2003) Species area relationships in Mediterranean climate plant communities. Journal of Biogeography 30: 1629-1657.

Keeley JE, Fotheringham CJ, Baer-Keeley M (2005) Factors affecting plant diversity during post-fire recovery and succession of mediterranean-climate shrublands in California, USA. Diversity and Distributions 11: 525–537.

Kent M, Coker P (1994) Vegetation description and analysis: a pratical approach. John Wiley and Sons, Chichester

Koch NE, Skovsgaard JP (1999) Sustainable management of planted forests: some comparisons between Central Europe and the United States. New Forests 17: 11-22.

Kupfer JA, Malanson GP[, Franklin SB[(2006) Not seeing the ocean for the islands: the mediating influence of matrix-based processes on forest fragmentation effects. Global Ecology & Biogeography 15:8-20.

Kwok HK, Corlett RT (2000) The bird communities of a natural secondary forest and a Lophostemon confertus plantation in Hong Kong. Forest Ecology and Management 130: 227-234.

Ladd PG, Crosti R, Pignatti S (2005) Vegetative and seedling regeneration after fire in planted Sardinian pinewood compared with that in other areas of Mediterranean-type climate. Journal of Biogeography 32: 85-98.

Lavorel S (1999) Ecological diversity and resilience of Mediterranean vegetation to disturbance. Diversity and Distributions 5: 3–13.

Lawesson JE, de Blust G, Grashof-Bokdam C, Firbank L, Honnay O, Hermy M, Hobitz P, Jensen LM (1998) Species diversity and area-relationships in Danish beech forests. Forest Ecology and Management 106: 235-245.

Lentile LB, Smith FW, Shepperd WD (2005) Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forest Research 35: 2875–2885.

Lima ACPS (1996) Castro Laboreiro, povoamento e organização de um território serrano. Instituto da Conservação da Natureza, Parque Nacional da Peneda-Gerês e Câmara Municipal de Melgaço, Braga.

Lindenmayer DB, McIntyre S, Fischer J (2003) Birds in eucalypt and pine forests: landscape alteration and its implications for research models of faunal habitat use. Biological Conservation 110: 45-53.

Liu J, Diamond J (2005) China's environment in a globalizing world. Nature 435: 1179-1186.

Lloret F, Calvo E, Pons X, Díaz-Delgado R (2002) Wildfires and landscape patterns in the Eastern Iberian Peninsula. Landscape Ecology 17: 745–759.

Longino J, Colwell RK, Coddington JA (2002) The ant fauna of a tropical rainforest: estimating species richness three different ways. Ecology 83: 689-702.

Loreau MS, Naeem S, Inchausti P, et al. (2001) Biodiversity and ecosystem functioning: current knowledge and future challenges. Nature 294: 804-808

Lugo AE (1997) The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. Forest Ecology and Management 99: 9 -19.

MacDonald D, Crabtree JR, Wiesinger G., Dax T, Stamou N, Fleury P, Lazpita JG,. Gibon A (2000) Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. Journal of Environmental Management 59: 47-69.

Madeira MV, Fabião A, Pereira JS, Araújo MC, Ribeiro C (2002) Changes in carbon stocks in Eucalyptus globulus Labill. plantations induced by different water and nutrient availability. Forest Ecology and Management 171:75-85.

Magura T, Baldí A, Horvath R (2008) Breakdown of the species-area relationship in exotic but not in native forest patches. Acta Oecologica 33: 272 -279.

Malhi Y, Roberts TJ, Betts RA, Killeen TJ., Li W, Nobre CA (2008) Climate Change, Deforestation, and the Fate of the Amazon. Science 319: 169 -172.

McCann KS (2000) The diversity-stability debate. Nature 405: 228–233.

McNeely JA (1994) Lessons from the past: forests and biodiversity. Biodiversity and Conservation 3:3-20.

MCPFE - Ministerial Conference on the Protection of Forests in Europe (2007) State of Europe’s Forests 2007: The MCPFE report on sustainable forest management in Europe. Ministerial Conference on the Protection of Forests in Europe Liaison Unit Warsaw.

MENDES A, Dias R (2002) Financial instruments of forest policy in Portugal in the 80s and 90s. EFI Forest Policy Research Forum: Cross-Sectoral Policy Impacts on Forests, 4-6 April, Sanvonlinna.

Mendes A, Fernandes L (2007) Politicas e instituições florestais em Portugal – desde o final do Antigo Regime até à actualidade. In: Silva JS (ed.). Floresta e Sociedade. Uma história em comum. Árvores e Florestas de Portugal. FLAD, PUBLICO, LPN.

Michelsen A, Lisanework N, Friis I, Holst N (1996) Comparisons of Understorey Vegetation and Soil Fertility in Plantations and Adjacent Natural Forests in the Ethiopian Highlands. Journal of Applied Ecology 33: 627-642.

Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press,Washington, DC

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: current state and trends. Island Press, Washington, DC

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: health synthesis. World Health Organization, Geneva

Moreira F, Catry F, Duarte I, Acácio V, Silva J (in press) A conceptual model of sprouting responses in relation to fire damage: an example with cork oak (Quercus suber L.) trees in Southern Portugal. Plant Ecology.

Moreira F, Delgado A, Ferreira S, Borralho B, Oliveira N, Inácio M, Silva JS, Rego F, (2003) Effects of prescribed fire on vegetation structure and breeding birds in young Pinus pinaster stands of northern Portugal. Forest Ecology and Management 184: 225–237.

Moreira F, Ferreira PG, Rego FC, Bunting S (2001b) Landscape changes and breeding bird assemblages in northwestern Portugal: the role of fire. Landscape Ecology 16: 175-187.

Moreira F, Rego FC, Ferreira PG (2001) Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. Landscape Ecology 16:557-567.

Mouillot F, Rambal S, Joffre R (2002) Simulating climate change impacts on fire frequency and vegetation dynamics in a Mediterranean-type ecosystem. Global Change Biology 8: 423–437.

Murcia C (1995) Edge effects in fragmented forests: implications for conservation. Trends Ecological Evolution 10: 58–62.

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403: 853–858

Naveh Z (1975) The evolutionary significance of fire in the Mediterranean region. Vegetatio 29: 199–208.

Naveh Z (1994) The role of fire and its management in the conservation of Mediterranean ecosystems and landscapes. In: Moreno, J.M., Oechel, W.C. (eds) The Role of Fire in Mediterranean-Type Ecosystems, vol. 107 Ecological Studies. Springer-Verlag, pp. 163–186.

Neary DG, Klopatek CC, DeBano LF, Folliott PF (1999) Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management 122: 51–71.

Novotny V, Basset Y (2000) Rare species in communities of tropical insect herbivores: pondering the mystery of singletons. Oikos 89: 564 - 572.

Núñez MR, Bravo F, Calvo L (2003) Predicting the probability of seed germination in Pinus sylvestris L. and four competition shrub species after fire. Annals of Forest Science 60: 327–334.

Núñez MR, Calvo L, Pando V, Bravo F (2008) Floristic changes induced by fire on Pinus sylvestris plantations in northwest of Spain. Investigación Agraria: Sistemas y Recursos Forestales 17: 168–177.

Onofre N (1990) Impactes do eucaliptal sobre a fauna dos eucaliptais. In: M. Alves M, Pereira JS (eds) Impactes ambientais e sócio-económicos do eucaliptal em Portugal. DEF, UTL-ISA, Lisboa.

Ordóñez JL, Retana J, Espelta JM (2005) Effects of tree size, crown damage, and tree location on post-fire survival and cone production of Pinus nigra trees. Forest Ecology and Management 206: 109-117.

Ormeño E, Céspedes B, Sánchez IA, Velasco-García A, Moreno JM, Fernandez C, Baldy V (2009) The relationship between terpenes and flammability of leaf litter. Forest Ecology and Management 257: 471-482.

Patterson TM, Coelho DL (2009) Ecosystem services: Foundations, opportunities, and challenges for the forest products sector. Forest Ecology and Management 257:1637-1646.

Pausas J, Bladé C, Valdecantos A, Seva J, Fuentes D, Alloza J, Vilagrosa A, Bautista S, Cortina J, Vallejo R (2004) Pines and oaks in the restoration of Mediterranean landscapes of Spain: New perspectives for an old practice — a review. Plant Ecology 171: 209-220.

Pausas J, Vallejo VR (1999) The role of fire in European Mediterranean ecosystems. In: Chuvieco, E. (ed) Remote Sensing of Large Fires in the European Mediterranean Basin. Springer, New York, pp. 3-16.

Pausas JG, Llovet J, Rodrigo A, Vallejo R (2008) Are wildfires a disaster in the Mediterranean basin? – A review. Int. J. Wildland Fire 17: 713-723.

Pereira H, Daily GC, Roughgarden J (2004) A framework for assessing the relative vulnerability of species to land-use change. Ecological Applications 14: 730-742.

Pereira HM, Cooper DH (2006) Towards the global monitoring of biodiversity change. Trends in Ecology and Evolution 21: 123-129

Pereira HM, Daily GC (2006) Modelling biodiversity dynamics in countryside landscapes. Ecology 87:1877-1885.

Pereira JS, Correia AV, Correia AP, Branco M, Bugalho M, Caldeira MC, Souto-Cruz C, Freitas H, Oliveira AC, Pereira JCM, Reis RM, Vasconcelos MJ(2002) Forest and biodiversity. In: Santos FD, Forbes K, Moita R. (eds.). Climate Change in Portugal: Scenarios, Impacts and Adaptation Measures – SIAM Project, Gradiva. Lisboa.

Pimenta M, Santarém ML (1996) Atlas das Aves do Parque Nacional da Peneda-Gerês. ICN/NPPG, Porto.

PNPG – Parque Nacional da Peneda-Gerês (1995) Plano de Ordenamento do Parque Nacional da Peneda-Gerês. Relatório de síntese. ICN, Braga.

Prieler S, Lesk'o A, Andberg S (1998) Three Scenarios for Land-Use Change: A Case of Study in Central Europe, International Institute for Applied Systems Analysis.

Quinn GP, Keough MJ (2002) Experimental Design and Data Analysis for Biologists. Cambridge University Press, Cambridge.

Radich MC, Alves AAM (2000) Dois Séculos da Floresta em Portugal, CELPA, Lisboa.

Ramil-Rego P, Muñoz-Sobrino C, Rodríguez-Guitián M, Gómez-Orellana L (1998) Differences in the vegetation of the North Iberian Peninsula during the last 16,000 years. Plant Ecology 138:41-62.

Raunkaier C (1937) Plant life forms. Clarendon Press, Oxford.

Reyes O, Casal M (1995) Germination behaviour of 3 species of the genus Pinus in relation to high temperatures suffered during forest fires. Annals of Forest Science 52: 385–392.

Ribeiro O, Daveau S, Lautensach H (1988) Geografia de Portugal: II – O ritmo climático e a paisagem. Ed. J. Sá da Costa, Lisboa.

Ricketts TH, Daily GC, Ehrlich PR, Fay JP (2001) Countryside Biogeography of Moths in a Fragmented Landscape: Biodiversity in Native and Agricultural Habitats. Conservation Biology 15:378-388a

Ricklefs RE, Miller G (1999) Ecology. W. H. Freeman

Rigolot E (2004) Predicting postfire mortality of Pinus halepensis Mill. and Pinus pinea L. Plant Ecology 171: 139-151.

Rosenzweig ML (1995) Species diversity in space and time. Cambridge University Press, Cambridge, UK.

Rothermel RC (1983) How to predict the spread and intensity of forest and range fires. U.S. Forest Service General Technical Report INT-143.

Santos T, Tellería JL, Carbonell R (2002) Bird conservation in fragmented Mediterranean forests of Spain: effects of geographical location, habitat and landscape degradation. Biological Conservation 105: 113-125.

Saunders DA, Hobbs RJ, Margules CR (1991) Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18-32.

Scheiner SM (2003) Six types of species-area curves. Global Ecology and Biogeography 12: 441-447.

Schoennagel T, Veblen TT, Romme WH (2004) The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54: 661–676.

Sequeira N (1995) Plano de gestão florestal da mancha florestal do Mezio. Parque Nacional da Peneda-Gerês, Arcos de Valdevez.

Shvidenko A, Barber CV, Persson R (2005) Forest and woodland systems. In: Hassan R., Scholes R., Ash N. (eds) Ecosystems and human well-being: Current state and trends. Island Press, Washington D. C., pp. 585-621.

Silva JS, Deus E, Saldanha L (2008) Evolução dos incêndios florestais em Portugal, antes e depois de 2003. In: Silva JS, Deus E, Saldanha L (eds.) Incêndios Florestais, 5 anos após 2003. LPN, AFN.

Silva JS, Moreira F, Vaz P, Catry F, Godinho-Ferreira P (in press) Assessing the relative fire proneness of different forest types in Portugal. Plant Biosystems.

Sobrino C, Ramil-Rego P, Rodriguez Guitián MA, (2001) Vegetation in the mountains of northwest Iberia during the last glacial-interglacial transition. Vegetation History and Archaeobotany 10: 7-21.

Tabarelli M, Mantovani W, Peres CA (1999) Effects of habitat fragmentation on plant guild structure in the montane Atlantic forest of southeastern Brazil. Biological Conservation 91:119-127.

Tapias R, Climent J, Pardos J, Gil L (2004) Life histories of Mediterranean pines. Plant Ecology 171: 53–68.

Tietje W (2001) Live oak saplings survive prescribed fire and sprout. California agriculture 55: 18-22.

Tjorve E (2002) Habitat size and number in multi-habitat landscapes: a model approach based on species–area curves. Ecography 25:17–24.

Tjorve E (2003) Shapes and functions of species–area curves: a review of possible models. Journal of Biogeography 30: 827-835.

Torres J, Honrado J, Caldas FB (2001) A vegetação da "Mata do Ramiscal" (Serra do Soajo, Parque Nacional da Peneda-Gerês). UGEV-ICETA, Universidade do Porto.

Trabaud L (1982) Effects of past and present fire on the vegetation of the French Mediterranean region. In: Symp. dynamics and management of Mediterranean-type ecosystems, USDA.

Triantis KA, Mylonas M, Lika K, Vardinoyannis K (2003) A model for the species–area-habitat relationship. Journal of Biogeography 30: 19–27

Turner MG, Romme WH (1994) Landscape dynamics in crown fire ecosystems. Landscape Ecology 9: 59–77.

Turner MG, Romme WH, Gardner RH (1999) Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire 9: 21–36.

Ugland KI, Gray JS, Ellingsen KE (2003) The species accumulation curve and estimation of species richness. Journal of Animal Ecology 72: 888-897.

USDI, National Park Service (2001) Fire Monitoring Handbook. Nation Interagency Fire Center, Boise.

Valbuena L, Nuñez R, Calvo L (2001) The seed bank in Pinus stand regeneration in NW Spain after fire. Web Ecology 2: 22–31.

van Vuuren DP, Sala OE, Pereira HM (2006) The future of vascular plant diversity under four global scenarios. Ecol. Soc. 11: 25. Available from http://www.ecologyandsociety.org/vol11/iss2/art25/.

Verdú M, Pausas J (2007) Fire drives phylogenetic clustering in Mediterranean Basin woody plant communities. Journal of Ecology 95: 1316-1323.

Wade TG, Riitters KH, Wickham JD, Jones KB (2003) Distribution and causes of global forest fragmentation. Conservation Ecology 7: 7. [online] http://www.consecol.org/vol7/iss2/art7.

Whelan R (1995) The Ecology of fire. Cambridge University Press. Cambridge, England.

Zapata BR, Correia AI, García MJG (1995) Evolución del paisaje en la Sierra de Peneda (NO Portugal) durante los últimos 3500 BP, Geogaceta 18: 105-108.

Zurita GA, Rey N, Varela DM, Villagra M, Bellocq MI (2006) Conversion of the Atlantic Forest into native and exotic tree plantations: Effects on bird communities from the local and regional perspectives. Forest Ecology and Management 235: 164-173.