# Chapter 7
# The Functioning, Management and Persistence of Dehesas

**G. Moreno[1]* and F.J. Pulido[2]**

**Abstract** Dehesas are the most widespread agroforestry systems in Europe, where they cover 3.1 million hectares. They are multipurpose open woodlands, mostly created by clearing the natural forests, where livestock rearing, cereal cropping, cork and firewood harvesting, and hunting are combined. In dehesas, trees can be seen as "ecosystem engineers", as they allow the maintenance of grass production in poor soils under a semiarid climate. We summarize the most outstanding results on both the effect of trees on the production and quality of the understorey (crop and native grasses) and also on the consequences of reduced tree density for the physiological condition and production of trees. The ecological basis of tree-understorey interactions is explained based on spatial distribution and use of above and belowground resources. Dehesas have been considered habitats to be preserved because they maintain a high biological diversity including several globally endangered animal species. They are considered an example of sustainable land use, although their conservation has been threatened in the last few decades. Excessive tree cutting, including complete elimination in some cases, has taken place as a consequence of increased mechanisation and stocking rates. This has caused a lack of natural regeneration and tree death in over-aged stands. We make a critical analysis of the ecological stability and sustainability of the system following four different approaches related to current problems: (i) historical evolution of the dehesa range, (ii) soil degradation and erosion, (iii) plot and farm-level factors precluding tree regeneration, and (iv) economic profitability of the dehesas. From these analyses, we derive a number of recommendations for dehesa management aimed at ensuring both its multifunctional role and its sustainability. The critical role of the shrub understorey for the ecological function, nutritional contribution and biodiversity is emphasized.

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[1] Dpto de Biología Vegetal, Ecología y Ciencias de la Tierra, I.T. Forestal (Forestry School), Universidad de Extremadura

[2] Ingeniería del Medio Agrícola y Forestal, I.T. Forestal (Forestry School), Universidad de Extremadura

* Corresponding author: Centro Universitario, Avd. Virgen del Puerto, Plasencia 10600 (Cáceres), Spain, e-mail: gmoreno@unex.es

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## Introduction

Dehesa is an agrosilvopastoral system originating from clearing of evergreen woodlands where trees, native grasses, crops and livestock interact positively under specific management practices (Campos 1992; Montero et al. 1998; Joffre et al. 1999). They are among the most prominent and widespread agroforestry land-use systems in Europe (Grove and Rackham 2001; Papanastasis 2004). At present, dehesas occupy 2.3 million hectares in Spain and 0.7 million hectares in Portugal, where they are called "montados" (MAPYA 2004; Pereira et al. 2004).

Dehesas result from a simplification, in structure and species richness, of Mediterranean forests and shrublands, and are attained by reducing tree density, eliminating *matorral* cover, and favouring the grass layer by means of grazing and crop culture (Montero et al. 1998). Dehesas are characterized by the rearing of traditional livestock breeds at low stocking densities and careful exploitation of evergreen oaks (Plieninger and Wilbrand 2001). The environmental setting of the Iberian dehesas is influenced by the Mediterranean climate, the low fertility of soils, and the usually undulating topography, that make arable farming unprofitable. Under these circumstances, dehesas have arisen as the only feasible way of productive land use (Montero et al. 1998). Dehesas are among the best preserved low-intensity farming systems in Europe, and in them the integration of traditional land-use and biodiversity conservation is considered exemplary land use management (Gómez-Gutiérrez and Pérez-Fernández 1996; Blondel and Aronson 1999; Plieninger and Wilbrand 2001).

The importance of dehesas rests on both environmental and socio-economic values. First, dehesas play a prominent role in the economy of rural areas in southwestern Spain (Escribano and Pulido 1998; Campos 2004; Pereira et al. 2004), because they occupy about 50% of grazing lands (Campos and Martín-Bellido 1997). In addition, dehesas are a fundamental component of regional identity, and are the source of high-quality food products derived from livestock production. On the other hand, dehesas have been valued at an international policy-making level for their biodiversity, aesthetic qualities and potential for tourism and recreation (Shakesby et al. 2001; Schnabel and Ferreira 2004). Dehesas support a large number of species and a diversity of habitats (Díaz et al. 1997), qualifying them to be listed in the EU habitat directive as being of community-wide interest.

Nevertheless, over the last few decades, dehesas and other agrosilvopastoral systems in Europe have faced several threats due to intensive land use imposed by a concomitant change in the technological and socio-economic conditions and common agricultural policies (Escribano and Pulido 1998; Papanastasis 2004; Pereira et al. 2004). These changes have resulted in a shift from traditional farming systems with very low external inputs to a simplified system involving intensive management techniques and decreased diversity of land uses (Schnabel and Ferreira 2004). Indeed, during the second half of the 20th century, around six million trees were removed (Elena-Roselló et al. 1987). A significant decrease in the area of distribution of dehesas and in the tree density has been occurring as a result of increased mechanisation, stocking rate and death of trees in over-aged stands (Fernández-Alés et al. 1992; Plieninger et al. 2003; Papanastasis 2004; Pereira et al. 2004). Additionally, the loss of traditional agrosilvopastoral practices has increased at least three sources of environmental degradation: (i) soil erosion rates due to changes in vegetation, soil properties and hydrological processes (Coelho et al. 2004; Schnabel and Ferreira 2004); (ii) over-aged oak stands due to a prolonged lack of regeneration (Montoya 1998; Pulido and Díaz 2005) and (iii) loss of diversity at various spatial scales (Díaz et al. 1997; Plieninger and Wilbrand 2001).

In this context, the sustainability of the dehesa system has been seriously questioned (Montoya 1993; Hernández 1996; Montero et al. 1998), and a considerable debate concerning the long-term persistence of dehesas has emerged, because most stands have over-aged trees and saplings are extremely scarce. Some authors have indicated that lack of regeneration is an inherent feature of grazed dehesas (e.g. Pulido et al. 2001; Plieninger et al. 2003). Others argue that the present lack of tree regeneration is mostly linked to the intensification of dehesa management and loss of multiple uses and management practices (e.g. Llorente-Pino 2003). Regardless of past dehesa regeneration patterns, at present there are no tested management practices for ensuring tree regeneration (based either on traditional or scientific knowledge). Hence, the following questions arise: (i) are dehesas declining? (ii) are dehesas a well-designed agroforestry system or a phase of forestland degradation? (iii) how much does dehesa persistence depend on management practices and/or dehesa structure? and (iv) to what extent are trees important for dehesa functioning?

Here we suggest that a mosaic-type structure of dehesa with a combination of grazed, shrubby and cultivated open woodland and dense forest (called *manchas*) plots is the only way to maintain the function and persistence of dehesas. In this paper, we address three central issues of dehesa literature, namely ecological function, productivity and persistence, and analyze the consequences of the different management practices on them. As far as possible, we have relied on quantitative information from recent literature and from our own studies. Thus, this chapter is conceived as an insight to new challenges for dehesa management in the face of new socio-economic status of the *local society* and environmental needs of the *global society*.

## Structure and Management of Dehesas

The history of human management of dehesas has resulted in a complex form of current exploitation. Dehesa structure at three spatial scales, referred to as in-plot, in-farm and off-farm are described. Attention is focussed on recent changes in management practices that could affect dehesa structure and function.

## *In-Plot Structure: Components of an Integrated Land-Use System*

Most dehesas are characterised by a two-layered vegetation structure, with the presence of a savanna-like open tree layer and an understorey pasture or crop in the same land unit. The tree layer is dominated by the evergreen holm oak (*Quercus ilex* subsp. *ballota* (Desf.) Samp.) and cork oak (*Q. suber* L.) and, to a much lesser extent, by the deciduous *Q. pyrenaica* Willd., *Q. faginea* Lam., and *Fraxinus angustifolia* Vahl. These tree species have a density ranging from 5 to 80 trees per hectare (usually 15–45 trees per hectare) and 21–40% canopy cover, this variation depending on its main use: lower densities occur in intercropped areas and higher densities in areas devoted to big game hunting (Montero et al. 1998; San Miguel 1994). Holm oak stands are regularly thinned and pruned for multiple purposes, such as enhancing herbage growth, ensuring maximum yield of acorns and obtaining browse, firewood and charcoal (San Miguel 1994). Most of the pasture species are annual herbs, with two non-vegetative periods in summer and winter (Montero et al. 1998). Although there are many species varying enormously among dehesas and also within each dehesa (because of the topography and the presence of trees), some of the more ubiquitous species are: *Aira caryophyllea* L.*,* *Airopsis tenella* (Cav.) Asch. & Graebn.*,* *Psilurus incurvus* (Gouan) Schinz & Tell and *Bromus* sp. among grasses, *Ornithopus compressus* L., *Biserrula pelecinus* L.*,* *Lathyrus angulatus* L. and several species of *Trifolium* among legumes, and *Xolantha guttata* (L.) Raf.*,* *Geranium molle* L.*,* *Spergularia rubra* (L.) J. Presl & C. Presl*,* *Silene inaperta* L.*,* *S. portensis* L.*,* *Cerastium glomeratum* Thuill.*,* *Tolpis barbata* (L.) Gaertn and *Bellis annua* L. of other families (Devesa 1995). In late successional and more fertile pastures, especially beneath tree canopies, perennials gradually replace annuals (Puerto 1992). Here, pasture is dominated by *Poa bulbosa* L. and *Trifolium subterraneum* L., frequently accompanied by *Trifolium bocconei* Savi*,* *Bellis perennis* L., *Erodium botrys* (Cav.) Bertol., *Parentucellia latifolia* (L.) Caruel. and different species of *Ranunculus* L. and *Plantago*.

Livestock are the main tool for maintaining stable understorey vegetation. According to Montero et al. (1998), the main functions of livestock are: (i) preventing colonization of pastures by invading shrubs; (ii) improving grassland quality; (iii) ameliorating soil fertility; and (iv) quickening the nutrient cycle. Different types of livestock (cattle, sheep, goats, pigs, horses) are common in dehesas, with some seasonal differences, to obtain an optimum yield from its varied structure (San Miguel 1994; Escribano and Pulido 1998). Briefly, sheep are the most suited species for exploitation of most dehesas. Cattle are found in the most humid dehesas, while goats are often used as a complement to make better use of woody fodder. Finally, pigs are introduced in the dehesa during October-January to take advantage of the abundance of acorns (San Miguel 1994). In recent decades, a noticeable increase of stocking rates in dehesas has taken place (Fig. 7.1).

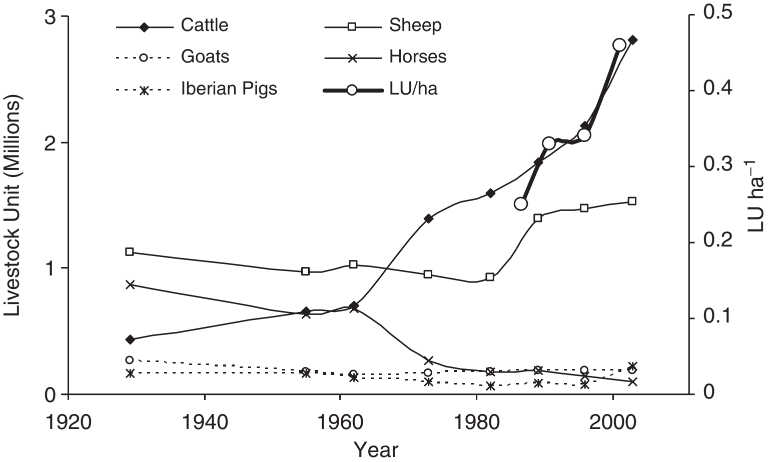


**Fig. 7.1** Temporal evolution of livestock numbers in south-western Spain. Data are expressed as livestock units (LU), according to the following equivalence: 1 cow = 1 LU; 1 sheep = 0.15 LU; 1 goats = 0.15 LU; 1 pig = 0.45 LU; 1 horse = 1 LU; 1 mule = 0.75 LU and 1 donkey = 0.5 LU. Author's estimates from data available in the Spanish National Annuals of Agricultural Statistics (MAPYA 1929 to 2004). Stocking rate (LU ha$^{-1}$) from Pulido (2002). Note: Due to the lack of statistics on stocking rate in dehesas, livestock data are aggregated for 13 provinces (Badajoz, Cáceres, Salamanca, Huelva, Sevilla, Cordoba, Ciudad Real, Toledo, Zamora, Avila, Madrid, Cádiz, Jaen)

## *In-Farm Structure: Rotational Cycles in Mosaic-Like Estates*

Dehesas are an unstable vegetation type that requires continuous human intervention to prevent shrub encroachment. Undergrazing encourages the invasion of various species of shrubs (e.g. *Cistus salviifolius* L.*, C. ladanifer* L.*, C. monspeliensis* L.*, Genista hirsuta* Vahl*, Ulex eriocladus* C. Vicioso*, Retama sphaerocarpa* (L.) Boiss.*, Citysus* L.), which will eventually replace the understorey grasslands (Montero et al. 1998). The importance of the shrub layer is reflected in the diversity of feed it provides for domestic animals during the periods of grass shortage in summer (Patón et al. 1999; Hajer et al. 2004).

Periodic or rotational ploughing is also a common practice in dehesas to control shrub encroachment, avoid soil compactation, and obtain a fodder complement through sowing. The system is therefore referred to as being agro-silvopastoral, because it combines crops, pasture and trees, shifting irregularly over successive years (every 3–12 years). After 3 years the number of species found in the ploughed pastures is usually similar to that found in the neighbouring unploughed pastures (Casado et al. 1985). Gradually, the improvement of pastures (*posío*) leads to reduced cropping, and even to elimination of tillage when livestock can stop the encroachment of woody vegetation (Montero et al. 1998). In some areas, where edaphic amelioration takes place due to nutrients gathered and excreted by livestock, very dense grassland of annuals and perennials of high nutritional value (called *majadal*) results. Pasture yield and quality is also increased through mineral fertilization and sowing of native and alien species (e.g. INIA/SEAIADG 1984; Olea et al. 2005). Therefore, there is a seasonal replacement of food sources for livestock in the dehesas: pasture in spring and autumn, acorn, tree and shrub browse in winter, and fodder crops in summer and winter (San Miguel 1994).

This trend has resulted in a sharp decrease in the arable area under cereal cultivation, which became increasingly unprofitable. At present, the most representative image of dehesa landscape is that of a vast savanna lacking any bushy understorey or croplands, and nearly half of the dehesa estates have only a grassland understorey (Campos et al. 2002).

## *Off-Farm Structure: An Adaptive Management to Cope with Seasonality*

In spite of the amelioration of pasture yield and quality and the in-farm resource integration mentioned above, dehesas are not currently self-sufficient because the feeding of livestock depends on neighbouring systems in periods of food shortage, mainly summer (Montero et al. 1998). The strong seasonality and variability of pasture herbage and its generally poor quality increase this problem (Escribano et al. 1996; Olea et al. 2005). On three representative farms of south-western Spain Escribano et al. (1996) have shown that as a whole, dehesas provide 57–73% of feed needs for ruminants, but only 43–47% for Iberian swine. Fodder scarcity in summer was traditionally overcome by *transhumance* livestock migration over some 300–500 km to mountain pastures. However, the abandonment of transhumance practice due to the use of external fodder and concentrates has resulted in an increasing presence of livestock in summer.

## *Recent Changes*

Dramatic changes in dehesa management schemes have occurred in the last decades. Gómez-Gutiérrez (1992), Plieninger and Wilbrand (2001), Campos et al. (2003a), Linares and Zapata (2003), and San Miguel (2005) have summarized these changes: (i) massive emigration of the rural population, with a labour shortage on many dehesa estates, a five-fold increase in the salary of workers, a reduction of specialised hand labour (herdsmen, shearers, pruners, and charcoal burners, among others) and increased mechanization; (ii) loss of land use diversity, with a dramatic decrease in the use of tree products (charcoal, firewood, browse and wood), strong decrease in crop cultivation, and loss of self-sufficiency due to dependence on external food, fertilizer and agro-chemical inputs; and (iii) partial substitution of extensive, low-intensity grazing for a semi-intensive management regime, with partial substitution of traditional breeds by artificial crossing, abandonment of shepherding (replaced by large-scale free-range grazing), partial substitution of sheep with cattle (as a result of the lack of shepherds and Common Agricultural Policy (CAP) subsidies; Fig. 7.1) and abandonment of periodic transhumance to summer mountain pastures. Since Spain and Portugal joined the EU in 1986, subsidies for ewes and suckler cows were granted as headage payments, having stocking rates equal to those needed for more productive northern (Atlantic biogeographic) regions in spite of lower productive Mediterranean environments, thus encouraging further increase of livestock numbers and dehesa overgrazing (Campos 2004).

## Ecological Function: Interactions Between Dehesa Components

### *Spatial Distribution of Resources: The Role of Trees*

A key issue for sustainable management of dehesas is to understand the function of isolated trees in the ecosystem. Their effects can be understood in terms of stabilisation and productivity (Gómez-Gutiérrez and Pérez-Fernández 1996; Montero et al. 1998). The influence of trees is reflected in the spatial distribution of above- and below-ground resources (light, soil water and nutrients and forage biomass), which vary with the distance to the tree.

### *Microclimate and Light Availability in Scattered Oak-Trees*

The low tree density in dehesas allows most of the light to reach the understorey, with values of 78% of full sunlight for a stand with 24 mature trees per hectare and 13% of canopy cover (Montero and Moreno 2005). These values are considered enough for optimum understorey production, according to the common values reported for herbaceous plants in temperate regions (around 70% of full sunlight; Montard et al. 1999). Nevertheless, the presence of scattered trees implies a strong heterogeneity in the spatial distribution of the light (Fig. 7.2). Considering 25% light reduction as a threshold and 25 trees per hectare, almost 20% of the surface could be significantly affected by shading (Montero and Moreno 2005).

The decrease of light availability in the vicinity of the tree canopy can be seen as a beneficial or detrimental effect on understorey yield (McPherson 1997). In dehesa, the decreased solar radiation beneath the canopy has a positive effect on both air and soil temperature (Nunes et al. 2005; Fig. 7.2). Temperature was significantly lower beneath than beyond the tree canopy on warm days, whereas on cold days the reverse was true (Moreno et al. 2007).

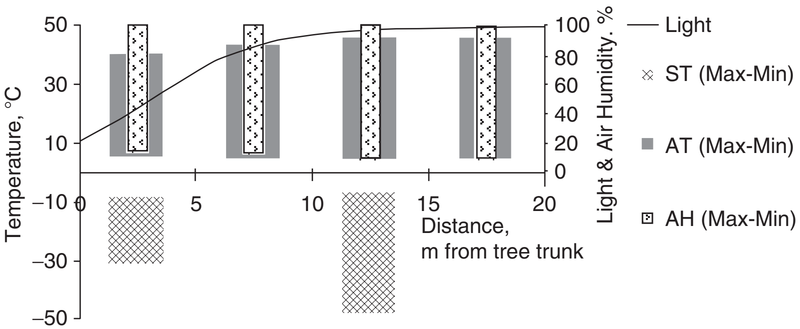


**Fig. 7.2** Light distribution and microclimate parameters beneath and beyond holm-oak canopies in a mature dehesa stand in central-western Spain. ST (soil temperature at 0–10 cm depth); AT and AH: Air temperature and humidity, respectively (Elaborated from Montero and Moreno (2005) for light and Moreno et al. (2007) for microclimate)

## Water Dynamics

The geographic range of dehesas is characterized by low rainfall, high PET (potential evapotranspiration), and high rainfall variability within and between years. Thus, one of the major ecological factors affecting dehesas is water availability (Infante et al. 2003). Puerto (1992) and Joffre and Rambal (1993) found that soil water content was always higher beneath than beyond the tree canopy in southern and northern subhumid dehesas, respectively. Joffre and Rambal (1988) estimated that maximum soil water storage was between 40 to 110 mm higher beneath than beyond trees in three southern subhumid dehesas. This increased soil moisture occurred in spite of the soil beneath the canopy receiving significantly less water than the area between trees as consequence of rainfall interception by trees (between 58% and 71.1% of annual rainfall Luis-Calabuig (1992) and Mateos and Schnabel (2002), respectively).

These results indicate an improved microclimate and soil physical properties beneath tree cover. A positive effect of trees on soil organic matter, dry bulk density (1.51 vs 1.58 g cm$^{-3}$, beneath and beyond canopy, respectively), infiltration rate, available soil water (243 vs 155 mm, respectively), and texture (increasing the abundance of fine particles) has been found in dehesas (Joffre and Rambal 1988; see also Fig. 7.3). Other authors (e.g. Escudero 1985; Cubera and Moreno 2007b) did not find that the canopy had any significant effect on soil texture. Anyway, as a result of the physical changes, the onset of drought is usually delayed by 1 month (Joffre and Rambal 1988) or by 1.5 month (Puerto 1992).

A recent study in semi-arid dehesas (annual rainfall around 500 mm) has shown that soil beneath than beyond the tree cover dried at nearly the same rate (Cubera and Moreno 2007b; Fig. 7.4). A similar pattern was reported by Nunes et al. (2005) in an area with an annual rainfall of 666 mm year$^{-1}$. Thus, the widely accepted idea that trees improved soil water status in dehesas is not certain in all dehesas, especially in the driest ones. In these latter cases, the volume of water extracted by tree roots must have had a major effect on the spatial and temporal changes in soil moisture. Indeed, trees can reach water located beyond the canopy cover (Joffre and Rambal 1993), even that located up to 20 m away from the tree (Cubera and Moreno 2007b). Soil moisture can be also be affected by the understorey vegetation and hence shrub encroachment in dehesas can significantly reduce soil moisture to values below those of an adjacent dense forest, at least in the first metre of soil depth (Fig. 7.4).

The presence of trees also affects the water balance, as Joffre and Rambal (1993) have shown. Trees significantly increase water consumption by transpiration, whereas water is easily lost by deep drainage and/or surface runoff beyond the tree canopies. In their study, water yield (excess of soil water) occurred with 570 and 200 mm of annual rainfall beneath and beyond the canopy, respectively.

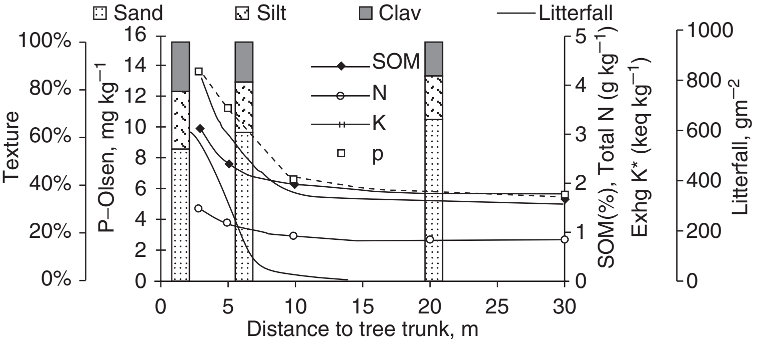


**Fig. 7.3** Summary of the main consequences of the presence of trees on the heterogeneity of soil properties. SOM (soil organic matter) (Elaborated from Puerto 1992; Escudero et al. 1985; Moreno and Obrador-Olán 2007)

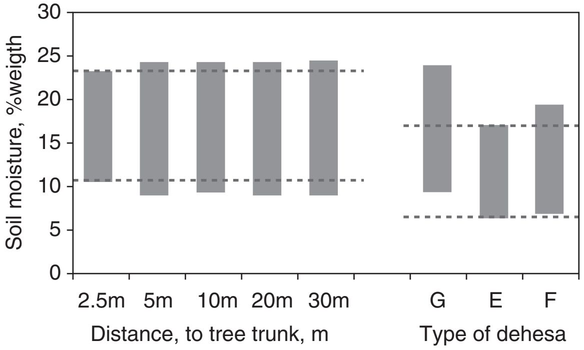


**Fig. 7.4** Maximum and minimum values of soil moisture measured (monthly) during a 3-year period. Soil moisture was measured at different distance of the trees and in different habitat types (G: grazed dehesa; E: Shrub encroached dehesa; F: Forest) of CW Spain. Soil moisture data were averaged from the first metre of soil depth (Elaborated from Cubera and Moreno 2007a, b)

## Nutrient Cycling and Soil Fertility

In semi-arid ecosystems isolated trees have an important effect on the spatial and temporal heterogeneity of soils, which can determine the structure and function of the herbaceous and animal communities in the soil (Gallardo et al. 2000). Trees immobilize large amounts of nutrients in their living and dead tissues (Table 7.1), which can be a detrimental short-term effect for understorey but a favourable long-term effect for nutrient storage (Escudero 1992). Tree roots bring up nutrients from deep in the soil profile that are inaccessible to herbaceous vegetation, and extract nutrients laterally from areas beyond the canopy (McPherson 1997; Scholes and Archer 1997). As a result, more than 50% of the nutrients are annually recycled beneath the canopy despite a canopy cover of only 20% of the dehesa surface (Table 7.1).

The role of trees in nutrient dynamics is critical because dehesas have a mostly internal nutrient cycle (Escudero 1992). Both nutrient inputs *via* atmospheric deposition and output *via* animal harvesting are very low when compared with internal fluxes (Table 7.1). Litterfall in dehesa is unusually high, even higher than in dense holm oak forest (1,900 and 1,600 kg ha$^{-1}$, respectively; Escudero 1992). Thus litterfall comprises an annual input to soil of 0.30–1.43% of the soil pool of N beneath the canopy, 21–59% of available P, 1.8–9.5% for exchangeable K, and 1.1–9.9% for exchangeable Ca (Escudero et al. 1985).

Additionally, the turnover rate of nutrients on the soil surface of dehesa ecosystems is also unusually high (Escudero et al. 1985). Dehesa litterfall decomposes up to 24 times faster that in dense forest (Escudero 1992). The amount of litterfall accumulated on the soil surface was estimated at 400 kg ha$^{-1}$ and 8,000 kg ha$^{-1}$ in

**Table 7.1** Main nutrient pools and fluxes of dehesas

| POOL and FLUX | | N | P | K | Ca | Mg |
|---|---|---|---|---|---|---|
| TREES | Pools kg ha$^{-1}$ | 44.3 | 5.54 | 49.6 | 301.2 | 22.7 |
| PASTURE | | 25.0 | 2.98 | 16.9 | 4.7 | 2.5 |
| Atmospheric input[a] | External cycle kg ha$^{-1}$ year$^{-1}$ | 7.7 | 0.76 | 2.2 | 7.7 | 1.6 |
| Output[b] | | 2 | 0.8 | – | – | – |
| Litterfall[c] | Internal cycle (Turnover) kg ha$^{-1}$ year$^{-1}$ | 15.2 | 1.21 | 4.6 | 13.7 | 2.3 |
| Pasture beneath canopy[c] | | 6.4 | 0.8 | 6.3 | 1.7 | 0.8 |
| Pasture beyond canopy[d] | | 19.0 | 2.0 | 15.3 | 6.7 | 2.7 |
| Canopy leaching[a] | | <0 | 0.40 | 7.95 | 1.2 | 2.9 |
| Turnover beneath canopy | % per year[e] | 53% | 55% | 55% | 71% | 69% |

Adapted from

[a] Moreno and Gallardo (2002)

[b] Escudero (1992)

[c] Escudero et al. (1985)

[d] Escudero et al. (1983)

[e] Turnover beneath canopy = (Litterfall + canopy leaching + pasture beneath)/Total turnover

**Table 7.2** Main N pool and fluxes in dehesas (Adapted from Gallardo et al. 2000)

| Nitrogen variable | Under trees | Between trees |
|---|---|---|
| Net N mineralization rate, μg $g^{-1}$ $d^{-1}$ | 4.77 | 2.09 |
| Net ammonification rate, μg $g^{-1}$ $d^{-1}$ | –0.46 | –0.20 |
| Net nitrification rate, μg $g^{-1}$ $d^{-1}$ | 5.32 | 2.28 |
| Available ammonium, μg $g^{-1}$ soil | 19.3 | 11.4 |
| Available nitrate, μg $g^{-1}$ soil | 20.2 | 13.2 |
| Microbial biomass-N, μg $g^{-1}$ soil | 122.2 | 73.1 |

dehesa and dense forest, respectively (Escudero et al. 1985). This rapid decomposition may be explained by the action of herbivores, which can recycle up to 85% of the phytomass (Escudero et al. 1985). Trees play a prominent role in the process, because net mineralization is higher beneath than beyond the canopy cover, as Gallardo et al. (2000) reported for N dynamics (Table 7.2).

As a result of the nutrient dynamics in dehesas, soils beneath the tree canopy are richer in soil organic matter (SOM) and nutrients than soil beyond the canopy (Fig. 7.3), (González-Bernáldez et al. 1969; Escudero 1985; Puerto 1992; Gallardo 2003; Nunes et al. 2005; Moreno and Obrador-Olán 2007). Although the effect of the trees is usually observed in the whole soil profile (Joffre and Rambal 1988), significant differences in soil properties beneath and beyond canopy are usually only found for the uppermost soil layer (from 0 to 20–30 cm) (Escudero 1985; Moreno and Obrador-Olán 2007).

Beside trees, shrubby vegetation may significantly modify soil fertility, although the information available on this is very scarce. Moro et al. (1997) has shown a positive effect of Mediterranean shrubs on soil fertility. In encroached dehesas Moreno and Obrador-Olán (2007) reported an increase in SOM, total N and exchangeable $Ca^{2+}$ and $K^+$ but a decrease of available P and mineral N.

## *Oak Tree Competitive Effects*

Trees exert a series of positive effects on dehesa resources, but trees can also compete for resources (light, nutrients, and water) with understorey vegetation. Like all agroecosystems, dehesa is a non-equilibrium system and only the persistence of grazing or ploughing disturbances allows its maintenance (Díaz et al. 2003). Nevertheless, in dehesas some mechanisms of plant-to-plant interaction used to explain the coexistence of trees and grasses can be invoked, that is, niche separation by the different rooting systems and phenological differences (Scholes and Archer 1997).

Annual and perennial grasses take water mostly from the upper 40–60 cm of soil (Joffre et al. 1987), whereas holm-oak has been reported to extract water from depths of 3 m (Cubera and Moreno 2007b), to 13 m (David et al. 2004). Holm-oak seedlings exhibit a high stomatal conductance and photosynthetic rate (Mediavilla and Escudero 2004) during the rapid development of the root system (more than 1 m depth during the first 3–4 months after germination; Cubera 2006). It seems that trees can easily avoid competition with grasses for water. But, can grasses easily escape this competition? Moreno et al. (2005) have shown a certain degree of spatial separation between grass and tree root systems in dehesas, as root length density (RLD) of grasses were five times higher than RLD of trees in the first 40 cm of soil depth. Phenological segregation could also be acting to some extent in dehesas. Maximum amounts of nitrate and ammonium occur in October–November and December–January, respectively (Gallardo et al. 2000), when roots of grasses are developing (Joffre et al. 1987) and trees are inactive (trees sprout in April–May; Oliveira et al. 1994). As a result, it appears that trees and grass are not highly competitive for nutrients and water, although more specific studies are still needed, especially for some key periods (e.g. early spring).

Moreno et al. (2007) have shown that (i) tree growth and acorn production did not increase significantly with soil fertilisation and irrigation, (ii) nutritional status of trees was not enhanced by fertilisation, and (iii) tree foliar nutrient contents did not correlated significantly with the nutrient content of the uppermost soil layer, while herbaceous plants did. In fact, herbaceous understorey responds positively to both irrigation and fertilisation (Nunes et al. 2005). These results indicate a low dependence of holm oak and a high dependence of herbaceous plants on the resources in the uppermost soil layer. By contrast, the dense foliage of evergreen oak becomes a limiting factor for forage production given that the photosynthetically available radiation (PAR) was reduced by around 25% (Nunes et al. 2005), but only in the vicinity of the trunk (Fig. 7.2). Indeed, maximum pasture yield is found with a canopy cover of 30%, (Qarro et al. 1995). Overall, results abovementioned indicate that the combination of holm oak with herbaceous plants could be an example of competition avoidance. The rooting system and the low growth rate of holm oaks could determine a low competitive potential of holm oak with grasses.

## *Tree Physiological Status: Benefiting from Dehesa Structure*

Dehesa trees have to cope with the high variability of the Mediterranean climate (Joffre et al. 1999). These authors demonstrated that dehesa structure could be interpreted as the result of an ecological adjustment, the distribution of tree density being to a great extent controlled by water availability because as rainfall increased, mean tree density increased. According to Rambal (1993), evergreen oaks are ‘regulator’ species, with three mechanisms for drought resistance: stomatal control, deep rooting and reduced leaf area. This set of functional strategies allows evergreen oak species to survive dry environments, but at a cost of very low rates of water transpiration and photosynthesis (Mediavilla and Escudero 2003).

Despite this, *Quercus ilex* L. growing in a dense forest reached predawn water potential below −4 MPa (Sala 1999), and suffered dieback in severe drought episodes (e.g. Peñuelas et al. 2001). By contrast, the low tree density of dehesas allows high per capita water availability for isolated individual trees. Infante et al. (1999), David et al. (2004) and Montero et al. (2004) recorded predawn leaf water potential remaining relatively high throughout summer: above −1.9, 2.16, −0.75 and −1.1 MPa, respectively, in dehesas with tree canopy cover between 20% and 40%. These results indicate that holm oak experienced moderate stress in dehesas as compared to that suffered by holm oaks in dense forest (Damesin and Rambal 1995; Sala 1999; Savé et al. 1999). Daily and seasonal transpiration patterns analysed at the leaf and whole-tree levels have also shown that prolonged drought hardly affected the water relations of holm oaks in dehesas (Infante et al. 2003; David et al. 2004). The advantage of the reduced canopy cover has been demonstrated by Cubera (2006), analysing leaf water potential in trees with distinct availability of soil, e.g. different tree densities (Fig. 7.5).

Tree clearance can also affect the nutritional status of trees. Úbeda et al. (2004) reported an obvious benefit of forest clearance on the leaf nutrient concentration of *Quercus suber* L. (Table 7.3). Finally, overstorey structure can also have a significant effect on leaf nutrient concentration of trees. Shrub encroachment was associated with a significant increase in K and P in *Quercus ilex* leaves of dehesas in west-central Spain, while the concentration of N, Ca and Mg was negatively affected (Table 7.3).

As a result of the improved water and nutritional status of trees in dehesas, holm-oak produced 13 times more acorns in open than in dense stands Pulido and Díaz (2005). Nevertheless, the determination of the most suitable density for optimizing the productivity of the dehesa is a controversial and an under-researched topic in dehesas. The complex combination of products and the influence of tree cover on understorey yield and quality make determination of an optimal density a very difficult task (Montero et al. 1998).

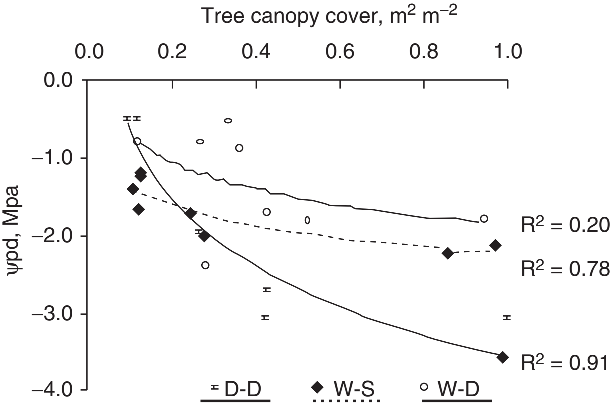


**Fig. 7.5** Leaf water potential (predawn) as a function of both tree density (expressed as relatively available soil for each individual tree) and edaphoclimatic conditions. Values for early September. D-D: Dry climate and deep soil; W-S: Wet climate and shallow soil; W-D: Wet climate and deep soil (Elaborated from Cubera 2006)

**Table 7.3** Leaf nutrient concentration of Mediterranean oaks growing with different stand densities and understorey structures

| | | | N | P | K | Ca | Mg |
|---|---|---|---|---|---|---|---|
| Source | Species | Plots | mg $g^{-1}$ | | | | |
| Úbeda et al. (2004) | Quercus suber | Dense | 7.8 | 0.60 | 3.1 | 2.4 | 0.30 |
| | | Open | 7.6 | 0.90 | 3.6 | 2.6 | 0.39 |
| Obrador-Olán et al. (2004) | Quercus ilex | Encroached | 10.0 | 0.65 | 5.0 | 6.5 | 1.25 |
| Moreno et al. (2007) | | Grass | 11.5 | 0.48 | 4.0 | 9.0 | 1.5 |

## Dehesa Production

Dehesa is a multipurpose system providing direct and indirect products and benefits. Among the former, pasture, browse, acorns, firewood, cork, and game are most important. Indirect benefits involve the high recreational and landscape value and very high levels of biodiversity in dehesas. Additional benefits include prevention of fire hazards, protection of soil and vegetation and an enormous historical and cultural value (Montero et al. 1998).

### *Pasture Yield*

Pasture yield in dehesa is usually low and shows a huge spatial variation (both at regional and local scales) and temporal variation (both annual and seasonal). The range reported by Puerto (1992) for northern dehesas is 400–9,000 kg DM $ha^{-1}$ $year^{-1}$ in the driest and wettest areas, respectively, with mean values around 2,400–3,500 kg DM $ha^{-1}$ $year^{-1}$. Figures for southern dehesas given by San Miguel (1994) are 300–3,000 kg DM $ha^{-1}$ $year^{-1}$. Pasture yield increases from 1,440 kg DM $ha^{-1}$ $year^{-1}$ in natural pasture to 2,238 and to 2,670 kg DM $ha^{-1}$ $year^{-1}$ with P fertilisation and P fertilisation plus seeding, respectively, in dehesas of Extremadura. Most of the primary production of this annual grassland is concentrated in spring, with a minor peak in autumn, depending of the amount of precipitation (Fig. 7.6). Nevertheless, the high spatial variability within each dehesa determines also certain temporal replacement of grassland types (Fig. 7.6). By contrast, the strong year to year variability (more than 250% in a 5 year period), imposes a serious drawback for livestock management (Olea et al. 2005).

The effect of trees on pasture is a controversial issue. Many authors have reported a positive effect on pasture yield (e.g. González-Bernáldez et al. 1969; Puerto and Rico 1988), nutritional quality (Puerto 1992; Pérez-Corona et al. 1995; Vázquez de Aldana et al. 2000; Moreno et al. 2007; Fig. 7.7), composition (greater

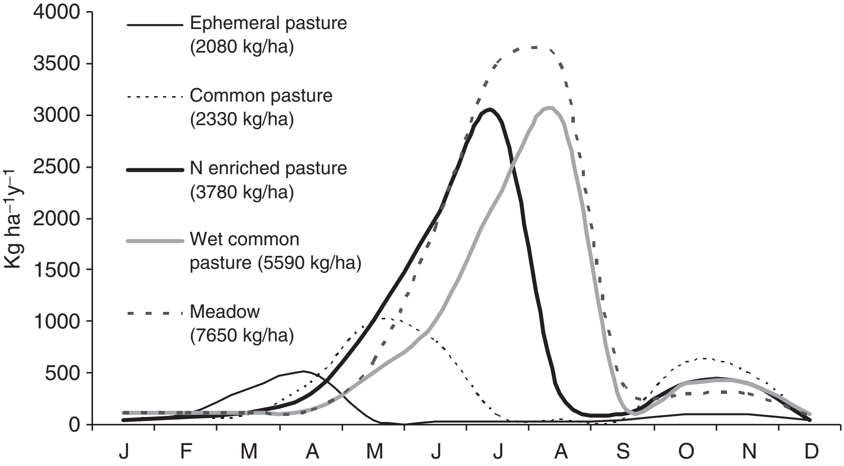


**Fig. 7.6** Seasonal variation of pasture yield at different locations of common northern dehesas (Salamanca province, Spain). (Elaborated from Gómez-Gutiérrez and Luis-Calabuig 1992.) Parenthetical Values in parentheses express mean annual yield of different types of pasture: ephemeral pasture (annuals in shallow soils), common pasture (mostly annuals in medium-depth soil), N-enriched pasture (annual and perennial grasses and legumes in soil enriched in manure), wet common pasture and meadow (mostly perennial grasses alive in early summer or along all the summer, respectively). Note that the onset of drought is earlier in central and southern dehesas. The onset of drought can occur between early May and the end of June

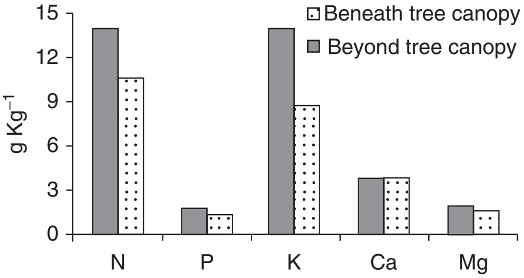


**Fig. 7.7** Effect of the trees on the nutrient concentration (quality) of pasture in dehesas of Salamanca province (Elaborated from Escudero et al. 1983)

abundance of perennials beneath canopy; Marañón 1986; Puerto 1992), length of growing season (Joffre et al. 1987; Puerto and Rico 1989), and stability against climatic variability (Puerto 1992).

The nature of the tree-crop interaction can vary among years and sites according to soil water availability. For instance, Puerto (1992) reported several cases where trees reduced the pasture yield (Fig. 7.8), but he also found that in the poorest soils or driest years, yield was homogeneous across distances or even it was higher

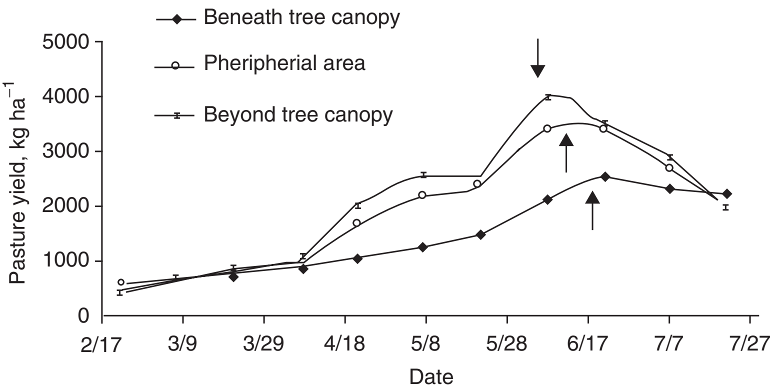


**Fig. 7.8** Temporal evolution of pasture yield at three distances from the tree. Note the decrease of pasture yield beneath the canopy, but also the temporal difference on the maximum yield (arrows indicate the period of maximum pasture production at each area) (Elaborated from Puerto 1992)

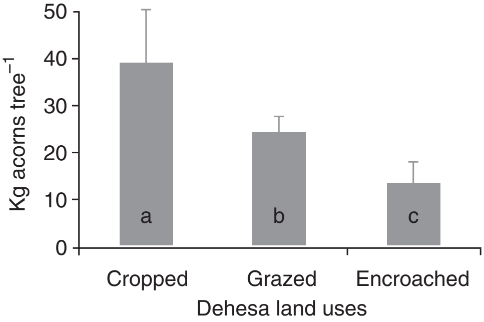
beneath than beyond the tree cover. Similarly, we have found a strong influence of fertility on the spatial pattern of crop yield in intercropped dehesas. Without fertilisation crop yield was higher beneath the canopy than in the open, while it was higher beyond the canopy in fertilised crops (Moreno et al. 2007).

## *Tree Production*

### Acorn Production

Acorns are the main winter food for several wild birds and mammals and domestic animals (pigs and others) inhabiting dehesas. In addition, both the initial number of acorns produced and the rate of removal by vertebrates strongly influence oak recruitment and potential for future production of acorns (Pulido and Díaz 2005). In holm oak dehesas, mean number of fully grown acorns (either viable or not) produced by trees and year was 3,773 and 5,851 in 2 years in grazed plots (García-López 2005). Mean weight of total acorns produced by individual trees in grazed dehesas was 18.12 kg (range: 6.0–28.0, SD = 7.2, n = 15 sites within the whole dehesa range). Mean production per hectare was 420 kg (range: 234–674, SD = 142, n = 9 grazed sites). In the only study available testing the effects of understorey management, cropping was shown to significantly increase production, while shrub encroaching caused a slight decrease as compared to grazed sites (Moreno et al. 2007; Fig. 7.9). Management affected total production through its effect on tree size and tree density, but also by increasing or reducing the probability of successful transition from flower to sound acorns (García-López 2005; Pulido and Díaz 2005). Finally, in the few studies available for cork oak grazed dehesas mean production per tree was 7.66 kg (range: 6.7–8.4, SD = 0.87, n = 3 sites), while

**Fig. 7.9** Comparison of acorn production per tree in three distinct dehesa land uses. Elaborated from Moreno et al. (2007). Bars indicate standard deviations, and different letters indicate significant differences ($p < 0.05$)



mean production per hectare was 315 kg (range: 240.5–448.5, SD = 115.6, n = 3 sites). Besides understorey characteristics, tree pruning has been shown to increase holm oak acorn production 2–6 years after branch elimination (Porras-Tejeiro 2002), though the presumed effects of pruning, tree density and site quality have not been tested with appropriate experimental designs.

The main environmental traits determining oak fecundity and acorn availability for animals are weather factors related to fertilization of pistilate flowers, the existence of leaking fruits (those showing abnormal sap exudates causing early abortion) fruits, and the infestation by borer insects. In a dehesa stand only 28% of flowers produced fruits, and the incidence of abiotic (weather-related) factors was much higher (90%) than that of borer infestation (10%) as a source of pre-dispersal losses (García-López 2005; Pulido and Díaz 2005). All these losses, and also those caused by episodic caterpillar outbreaks, are subjected to the effects of management on tree condition. Hence acorn production could be improved by appropriate management of the understorey and the tree canopy (García-López 2005).

## *Cork Production*

Cork is exploited periodically throughout the life of cork-oak trees, and the average production of cork per adult tree in each 9-year cycle varied from 5 kg (young trees) to 71 kg (mature trees) (Montero et al. 2003), i.e. 480 kg ha$^{-1}$ year$^{-1}$ (Pereira et al. 2004). The production of cork has been decreasing in the last few decades in the Spanish dehesas. However, the economic potential of cork has increased markedly in the past 2 decades (Fig. 7.10), and presently, thousands of hectares of arable and pastureland are being afforested with cork oaks. In Portugal, where production has been rather stable in the last 50 years, the value of annual cork production is comparable to that of the national wood production, 258 vs €222 million respectively in 1998, (Pereira et al. 2004).

As long as the international markets continue to consider cork as the most efficient bottle-stopper, the future of cork-oak woodlands should be assured.

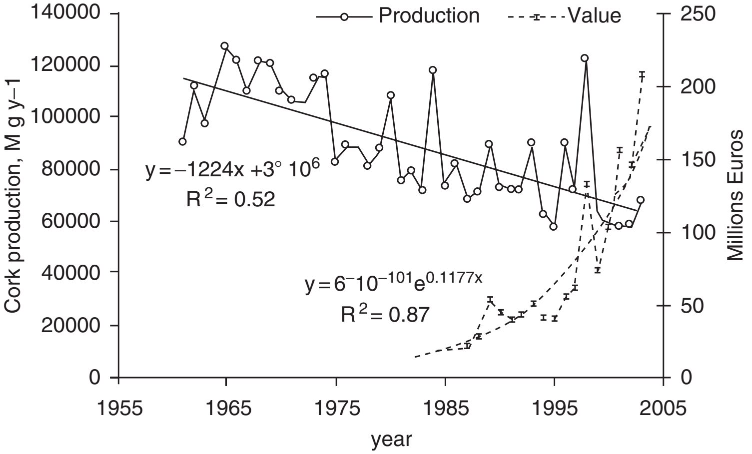


**Fig. 7.10** Historical data of cork production in Spain and the value of total national production. Note a linear decrease in cork production ($R^2 = 0.52$), but an exponential increase in the commercial value of total cork production ($R^2 = 0.87$) (Elaborated from Montero et al. 1994; MAPYA 2004)

However, future drops in the price of cork, its substitution by synthetic materials, and the degradation of cork oak stands, may threaten the maintenance of vast cork-oak woodlands (Pereira et al. 2004).

## *Other Direct Tree Products*

Dehesa trees are periodically pruned, and lopped branches are used for firewood or charcoal production and as fodder in winter. Several prunings are carried out during the life of the oaks, traditionally performed in the year preceding arable cultivation to increase light availability for the crops. Due to the sclerophyllous evergreen nature of dehesa trees, they represent substantial fodder reserves for wildlife and livestock (San Miguel 2005). A rational pruning can yield 300–500 kg $ha^{-1}$ $year^{-1}$ of dry browse material (Cañellas et al. 1991).

However, the economic costs of traditional light or moderate pruning are very high, and there are attempts to compensate these costs by obtaining income from firewood, charcoal or virgin cork. This generally implies an increase in the intensity of pruning, which can be too intense and cause damage to the tree (Cañellas et al. 2007). There is a traditional belief that pruning increases acorn production (San Miguel 1994; Gómez-Gutiérrez and Pérez-Fernández 1996) but a recent study has shown that, overall, pruning decreases acorn production (Cañellas et al. 2007). They found that pruning significantly decreased acorn production when production was above the average, whereas production was not affected by pruning the years that acorn yield was below the average. Hence, the effect of pruning in Mediterranean oak woodland is still controversial and more information based on research is needed to form an objective and rational opinion upon the response of trees to this important silvicultural practice (Cañellas et al. 2007).

### Other Direct Products

Big game has become one of the most important direct benefits of the dehesa, and has also great potential, because it is a high quality product, compatible with dehesa conservation (Carranza et al. 1991; Vargas et al. 1995). Red deer ingest a high proportion of browse in summer during dry years (0.83% to 0.89% of total diet) and also in wet years (0.47%; Bugalho and Milne 2003; see also Fig. 7.11). However, few attempts to quantify the effects of game on dehesa vegetation and sustainability have been carried out (Patón and Pulido 1999). Special attention should be given to the transformation of the vegetative cover, food supplementation, population structure and disease and genetic effects caused by the uncontrolled transference of animals between hunting estates (Carranza 1999; San Miguel 2005). Another direct product, agrotourism, represents an important growing source of income in dehesas, especially those located close to nature reserves, where recreation can account for a considerable proportion of total income (Campos 1998). The number of estates offering entertainment services is growing rapidly as a result of increasing demand by visitors, especially in naturally protected areas. In this way, environmental values of dehesas will be increasingly internalized by landowners as a source of income values.

Finally, other direct products commonly exploited in dehesas are honey (especially in encroached areas), and a variety of medicinal and edible herbs and fungi.

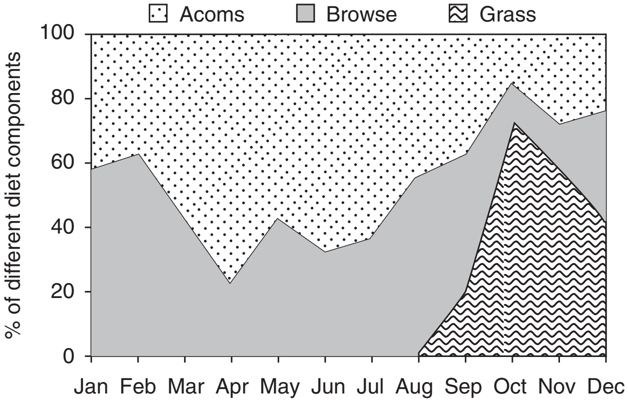


**Fig. 7.11** Seasonal variation in the proportion of acorns, browse and grass in the diet of deer grazing dehesa systems (Elaborated from San Miguel and Pérez-Carral 1996)

## *Wildlife and Biodiversity*

Dehesas serve as the main habitat for several endangered species and for very high diversity of animals and plants. The Spanish imperial eagle (*Aquila adalberti* C. L. Brehm), the black vulture (*Aegypius monachus* L.), the black stork (*Ciconia nigra* L.) and the Iberian lynx (*Lynx pardina* Temminck) use dehesas as feeding habitats and adjacent forests and scrublands for breeding, and a noticeable fraction of their world populations depends on dehesas (Díaz et al. 1997). Many bird species, notably common cranes (*Grus grus* L.), use dehesas as their preferred winter habitat. As a consequence, a large proportion of the dehesa range has been included in the Natura 2000 European web for nature conservation, and dehesa grasslands are also a habitat to be protected by the EU Habitats Directive (Díaz et al. 2003). In addition, dehesas sustain a high species richness of several contrasting taxonomic groups. For example with vascular plants, research has described 135 species in 0.1 ha in holm oak dehesas or 60–100 species per 0.1 ha in cork oak stands (Marañón 1986). Values of species richness of this and other taxa are much higher than those of other European man-made habitats. Also, diversity values of plants, birds and butterflies have been shown to be similar, or even higher, to those found in natural or semi-natural habitats located nearby (Díaz et al. 2003). As the only example available of a comprehensive biodiversity survey is from a 220 ha montado farm where, 264 fungi, 75 bryophytes, 304 vascular plants and 121 vertebrate species were recorded (Santos-Reis and Correia 1999).

The main explanation for the diversity values found in dehesas is the intimate mixture of habitats at various scales. First, at the very fine scale the presence of trees increases habitat heterogeneity and plant richness as compared to treeless grasslands. Second, within the same management type (pasture, crop or shrubland), differences in tree density or age and topography determine local variations in animal and plant diversity, respectively. Third, the habitat mosaic generated by the combination of land-use units enhances farm-level diversity by favoring a combination of habitat specialists and generalists via the “hybrid habitat” hypothesis (Díaz et al. 2003). According to this hypothesis, bird diversity values have been shown to follow a nested pattern in dehesas, that is, the number of forest species increases with tree density while the number of grassland specialists remains unchanged. From these results, it follows than the anthropogenic maintenance of multi-scale habitat heterogeneity is crucial for biological diversity in dehesas (Tellería 2001; Díaz et al. 2003). Also, globally threatened species, which have large home ranges, are clearly favored by landscape diversity, as they simultaneously exploit different habitat types (Donázar et al. 1997).

Nevertheless, the effect of dehesa land use on diversity remains a controversial issue that deserves further investigation. Thus, for example, the abundance of lizards increased when understorey bushy vegetation increased, while grasslands or cereal fields were scarcely colonised even if holm oak tree were present (Martín and López 2002). This and other less studied taxonomic groups may experience a reduction in species diversity as a result of forest clearance and grazing. Also, even if species diversity is enhanced by management, human practices may affect species which have a critical role in ecosystem function, as has been described for acorn dispersal on which oak regeneration relies (see Díaz et al. 2003; Pulido and Díaz 2005). Hence, the net effect of dehesas on diversity is not fully understood, and the assumed value of dehesa for the Mediterranean biota is more based on its importance for threatened species than on diversity values.

## Are Dehesas Sustainable?

In the last 2 decades, an intense debate about the sustainability of the dehesa system has gathered momentum in view of the lack of oak regeneration in dehesas. Plieninger et al. (2003) showed that the mean age of trees is closely related to the age of the dehesa formation, indicating that the maintenance of dehesa structure or management lead to a lack of tree recruitment, to the ageing of the tree population and, eventually, to its disappearance. In fact, it has been estimated that in the absence of artificial regeneration dehesas would have disappeared in 80 years at the rate of decrease estimated for the middle 20th century (Elena Roselló et al. 1987). By contrast, other authors argue that the present lack of tree regeneration is mostly linked to recent changes of dehesa management, regarding both soil and trees (e.g. Llorente-Pino 2003). He has documented cases of very old dehesas that currently maintain tree cover; therefore they must have experienced episodes of regeneration in the last 5 centuries.

### *Temporal Evolution of Dehesa Range*

Silvopastoral practices intended to transform dense evergreen forests and shrublands into wooded pastures have been used for centuries in lowland areas of the Mediterranean (Stevenson and Harrison 1992; Blondel and Aronson 1999). In southwestern Spain, recent historical analyses show that the increase in the area covered by dehesas parallels the growth of human populations from the 18th century onwards as a consequence of need for arable and grazing lands (Linares and Zapata 2003). The process of dehesa establishment was accelerated by advances in mechanization in the 20th century. This process was considered complete by the middle of the 20th century, when almost all natural habitats in flat areas had been converted into open dehesas (Fig. 7.12). During the period 1940–1970 an intensification of agricultural practices and socio-economic changes led to a crisis in the traditional dehesa system (Díaz et al. 1997). Consequently, the dehesa range suffered a sharp decrease due to tree cutting and lack of tree regeneration, a process that ceased during the 1980s as a result of new regulations. For instance, a specific law for dehesa management was created in 1986 in Extremadura (Law 1/1986), which forbids the cutting-down of oaks and the transformation of the dehesas into

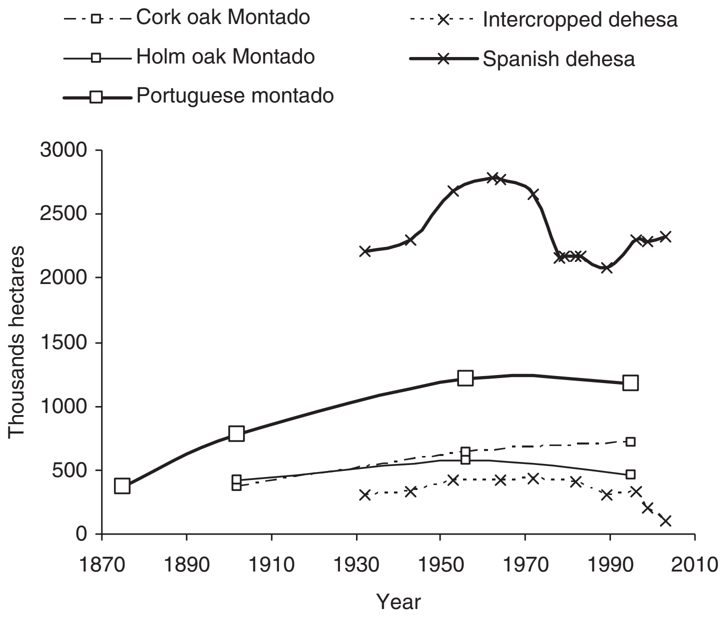


**Fig. 7.12** Historical variation in the area occupied by Spanish dehesas (total and intercropped) and Portuguese montados (both dominated by holm-oak and cork-oak). Elaborated from Annual of Agricultural Statistics for Spanish dehesas (MAPYA 1929 to 2004) and Radich and Monterio Alves (2000) for Portuguese montados. NOTE: Estimated by summing the area (thousands of hectares) covered by open woodlands in 13 provinces located in sothwestern Spain: Cáceres (502), Badajoz (403), Cordoba (275), Salamanca (273), Huelva (194), Sevilla (167), Ciudad Real (116), Cádiz (88), Jaén (74), Toledo (71), Ávila (61), Madrid (50) and Zamora (32). Open woodlands are defined by MAPYA (2004) as oaklands with a fractional canopy cover between 0.05 and 0.20 and whose main use is grazing

other land uses. Before that, 23% of oak trees were lost in Extremadura, and 9.6% of the dehesa area disappeared due to intensification, with a loss of around 5.7 millions oak trees over 1960–1985 period (Elena-Roselló et al. 1987). Nevertheless, dehesa landscape was relatively stable during the second half of the 20th century (García del Barrio et al. 2004) with less than 10% of dehesas experiencing any significant change in the last 50 years. After this phase of decrease in dehesa area and tree density, encroached areas and tree density have experienced moderate to high increases from the middle 1980s onwards (García del Barrio et al. 2004; Lavado et al. 2004; Roig et al. 2005; Plieninger 2006; Fig. 7.12). However, an apparently high oak tree mortality is presently occurring (Peñuelas et al. 2001; Sánchez et al. 2002), through a poorly defined process called *seca,* where both biotic (e.g. *Phytophthora cinnamomi* Ronds) and abiotic (e.g. sharp alternation of wet and dry periods) causes seem to be involved (Brasier et al. 1993; Tuset and Sánchez 2004).

At present, from the above information, it seems that the trend toward tree loss due to lack of regeneration and dieback in ageing oak stands is more than compensated for by recovery in areas that have been abandoned, protected or devoted to big game hunting. Nevertheless, due to a lack of spatially explicit historical information, exact timing of a specific dehesa creation is unknown (Plieninger et al. 2003) and it is difficult to confirm if old dehesas have regenerated (e.g. been abandoned) periodically or have been replaced by new ones elsewhere.

## *Soil Degradation and Soil Loss*

Land degradation is recognized as a significant problem in many of the dehesas and montados, implying impoverishment of the pasture cover, accelerated soil erosion and physical soil degradation (Murillo et al. 2004). Although the mean soil erosion rate on hill slopes is usually not too high (540 kg ha$^{-1}$ year$^{-1}$), it is considered excessive because of the degraded state of the soils. Nonetheless, it is clearly higher than that usually considered typical for non-managed or disturbed systems (4–200 kg ha$^{-1}$ year$^{-1}$), and around 30 times higher than values reported for a dense holm-oak forest in Catalonia (Schnabel 1997).

**Table 7.4** Effect of land use, management practices and tree cover on dehesa soil conservation

| | Land use[a] | | | Management practices[b] | | | | Tree cover[c] | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Soil parameter | Forest | Dehesa | Crop | NP | TS | DS | FP | HTD | MTD | Tree-less | BC |
| Soil cover, % | 57 | 21 | 12 | 50 | 58 | 58 | 58 | 85 | 55 | 35 | 80 |
| Soil compactation, kg cm$^{-2}$ | 3.08 | 3.95 | – | – | – | – | – | – | – | – | – |
| Overland flow, % rainfall | 6.5 | 36.3 | 16.0 | 12.3 | 12.7 | 2.3 | 18.6 | 12.0 | 30.5 | 36.3 | 9.6–13.2 |
| Erosion rate[a,b,c] | 5.0 | 87.5 | 200 | 98.4 | 14.5 | 12.7 | 27.8 | 3.1 | 5.2 | 5.8 | 0.9 |

Adapted from

[a] Coelho et al. 2004; Erosion rate determinates by Rainfall simulator (g h$^{-1}$ m$^{2}$)

[b] Murillo et al. 2004; Management practices: NP: Natural pasture, TS: Traditional seeding, DS: Direct seeding, FP: Fertilised pasture; Erosion rate determinates by erosion plots of 0.5 ha (g m$^{-2}$ year$^{-1}$)

[c] Schnabel 2001; Tree cover: HTD: High tree density, MDT: Medium tree density, BC: Beneath canopy; Erosion rate determinates by Gerlach traps (g m$^{-2}$ year$^{-1}$):

Moreover, the increased stocking rate of dehesa could have led to an increased risk of soil and pasture degradation (Coelho et al. 2004; Schnabel and Ferreira 2004; Shakesby et al. 2001). Heavily grazed evergreen oak sites both in Portugal and Morocco had a significant increase in soil compactation, overland flow and erosion rates (Table 7.4). Furthermore, ploughing for control of understorey vegetation and/or to improve pasture (seed sown) increased overland flow and erosion (Coelho et al. 2004; Table 7.4).

As previously stated, erosion rate is lower beneath the canopy than in open spaces (Table 7.4). Thus, to minimize the risk of erosion, apart from the maintenance of the tree cover, it is advisable to reduce the stocking rate, especially in summer to avoid excessive degradation of pasture cover before the onset of heavy autumn rainfall. Hence, the maintenance of transhumance would be of great benefit for soil conservation because dehesas would be destocked during the summer. Another recommendation is to improve pasture yield through fertilization and/or sowing selected native species (mostly legumes), but avoid soil ploughing on medium and steep slopes (Schnabel 2001).

## *The Lack of Tree Regeneration*

Several authors (Montoya 1998; Pulido et al. 2001; Plieninger et al. 2003) have pointed out that the forest cycle has been disrupted in most dehesas, where the lack of regeneration is an inherent problem to their exploitation. Disruption begins as each dehesa farm is developed from forest, and it has been exacerbated by the recent intensification of the agroforestry use of dehesa. Undisturbed oak forests, where oak recruitment occurs regularly, have size or age structures consistent with a negative exponential (inverse J-shaped) function (Pulido et al. 2001; Fig. 7.13). Recent studies conducted in oak stands, from local to regional scales, have revealed strong departures from the natural pattern of tree regeneration due to the lack of saplings and juveniles. In the case of cork oak dehesas, 72% of the stands showed regeneration failure (n = 159 stands; Institute of Cork, Wood and Charcoal of Extremadura, 2001 unpublished data), while the corresponding figure for holm oak dehesas is 87% (n = 60 stands; Naveiro et al. 1999; see also Pulido et al. 2001; Plieninger et al. 2003). In a comparative analysis of holm oak recruitment capacity in natural forest and dehesas (Pulido and Díaz 2005), the probability of establishment of new saplings was 75 times lower in dehesas (0.00150 and 0.00002, respectively; see also Fig. 7.13). This disparity was the result of differences in the success of seed dispersal to suitable sites and the lack of shrubs that have been found to exert a nurse effect in natural forests. This finding, confirmed by other studies (Plieninger et al. 2004b, García-López 2005), explains the general lack of natural regeneration of dehesas as compared to the holm oak forest. Both dispersal and safe site limitations are related to the lack of shrub cover associated with intensive dehesa management, thus, shrub encroachment predictably results in higher recruitment rates as compared with dehesa grazing or cropping (García-López 2005).

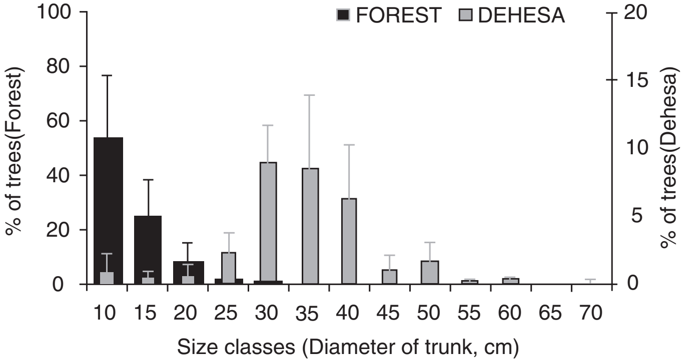


**Fig. 7.13** Size structure of the populations of mature trees in *Quercus ilex* dehesas and adjacent forests. Bars indicate standard deviations (Elaborated from Pulido et al. 2001)

Plieninger et al. (2004b) found that juveniles and saplings are highly associated with mature trees, shrubs, and rock outcrops, a result of directional dispersal, facilitation of seedling establishment and sheltering from browsing by livestock.

For decades dehesa researchers and technicians have agreed that regeneration failure is the main problem for the long-term persistence of dehesas, and that ensuring tree turnover should be a requisite for sustainable management (Montero et al. 1998). Nevertheless, the short timeframe in which land owners make management decisions implies that tree regeneration was not customarily considered as an important constraint. In fact, measures devoted to tree conservation have only been adopted under public intervention through subsidies for afforestation under EU regulations (Campos et al. 2003b). Subsidized oak plantations within large livestock exclosures have been a substantial source of income over the last 15 years, despite the cost of reduced grazing area. Over 186,000 ha were planted in the period 1996–2002 (MAPYA 2004). Plieninger et al. (2003) have showed that dehesa degradation is easily reversible if abandonment is periodically practiced, but specific measures promoting owners' interest in natural regeneration after partial exclusion of livestock have been anecdotal. They can be expected to become more widespread as the perception of subsidies became more dependent on the fulfillment of agri-environmental objectives.

## *Dehesa Profitability*

Dehesa is an extensive but labour-intensive land-use system. Thus, the increased labour costs in Europe threaten dehesa profitability and hence persistence (Gómez-Gutiérrez 1992). Commercial profitability of direct dehesa products is usually low

(Campos 2004). Applying the conventional net value added (NVA); dehesa profitability is very low, even in many cases negative (Escribano and Pulido 1998), with a range of −14.7 to 9.7 of NVA. Only in some cases, e.g. for cork oaks with low livestock grazing and red-deer hunting, is commercial profitability clearly positive (Campos et al. 2001). However, according to the total economic value theory (Campos et al. 2001), economic analysis based only on NVA produces an incomplete annual forestland income assessment. Whenever there are multiple uses of renewable resources, a new operative approach called Agroforestry Accounting System (AAS) can be used to incorporate environmental goods and services (Campos et al. 2001).

In the last 25 years, dehesas and montado have attained large capital gains; e.g. the price of dehesa land in Extremadura has increased at a real cumulative annual rate adjusted for inflation of 5%, (Campos 2004; see also Escribano and Pulido 1998). The constant rise of land prices of dehesas, at a time when commercial profitability of dehesa farming is declining, is thought to be largely due to the revaluation of self-consumed private environmental services (indirect products); in other words, 'leisure has become a product of the dehesa' (Pardal 2002; Campos 2004). Indeed, several studies carried in Spain showed that private environmental services used by landowners themselves account for 33–43% of the market price of land reported by landowners (Campos 2004). There is a consumptive value associated with ownership of rural land, reflecting innate desires to own land, live in a rural environment, obtain or maintain the lifestyle of a farmer, engage in outdoor recreation, get back to nature, and partake of any other real or perceived benefits of rural land ownership (Campos and Caparrós 2005).

As a result, dehesas have an unexpectedly high discount rate of 4.5% on average, which is higher than that of many European forests (around 2–3%; Campos et al. 2003b). Considering both capital gains and direct product-included subsides, which account for between 43% and 80% of the commercial income in a common dehesa (Calvo et al. 1999), the total private real profitability of dehesas is in the range of at least 3–5%, not including hypothetical incomes from public environmental services (Campos 2004). These public direct goods and services, and environmental functions are insufficiently known and are not fully incorporated to the present accounting systems (Escribano and Pulido 1998; Campos et al. 2003b).

The fact that current profitability depends less on income from direct productivity than on capital appreciation, and the low capital flux and long waiting period needed for financial returns from most forest operations, has very negative effects on dehesa conservation. Land owners are usually more interested in obtaining economic profit than in the rational long-term exploitation of dehesa resources. This attitude leads to a lack of capital to finance the management and improvements needed to exploit direct products in a sustainable way (Montero et al. 1998). As previously mentioned, only the implementation of direct policies for sustainable management of dehesas (subsides) seem to be contributing to solve this problem. According to Campos et al. (2003b) these subsidies can be justified in terms of economic efficiency and social fairness.

## Future Prospects for Sustainable Dehesas

From the experimental results presented here, there is a true opportunity for dehesa sustainability and conservation becomes apparent, because oaks are able to regenerate after several years of set-aside, e.g., systematic abandonment of agricultural and grazing activities according to a rotational scheme (Plieninger et al. 2003). The time needed to ensure natural regeneration excluding grazing activity has been roughly estimated for different livestock species but we still lack quantitative regeneration models accounting for the whole variability in the dehesa scenarios (Montero et al. 2003). Even assuming a set-aside period of 20 years and a mean holm oak lifetime of 200 years, 10% of any estate would be required. The establishment of long-term experiences allowing tree regeneration in fenced portions involving pilot farms is largely needed for to develop more accurate models.

The present total private economic profitability of dehesas and montados appear to be moderate to high. Nevertheless, this is mainly due to the income through livestock subsidies – as direct productivity – the self-consuming environmental services – as indirect productivity – and to the capital gains, with a low commercial operating profitability (Campos 2004). Landowners are aware of regeneration failure on their farms but they are reluctant to give up part of the moderate cash-flow to ensure the future profitability of the system by adopting a less intensive management (Campos 2004; Plieninger et al. 2004a). Hence sustainable management of dehesas should be encouraged by the national agencies through subsidies, justified by the social goods and services delivered and conditioned to the maintenance of the environmental functions of dehesas.

Under the appropriate EU regulations, biological diversity can be considered as environmental value contributing to economic sustainability of dehesas. There is an urgent need to correct deficient environmental regulations to guarantee the sustainability of dehesas and montados. For example, livestock income, the primary driver of overgrazing in dehesas, could be replaced by *set-aside regeneration reserve* subsidies. These should involve the creation of a mosaic-type farming, where shrubland patches (called *manchas*) should be also included. This type of landscape mosaic is assumed to have positive impact on biodiversity and sustainability (natural regeneration) of the dehesas (Pineda and Montalvo 1995; Carranza 1999; Plieninger and Wilbrand 2001; Pulido and Díaz 2005); however, the implications of this approach on economic returns should be explored in pilot farms before its widespread application. Similar criteria for forest management in support of the conservation and productive role of the landscape have been proposed by Fullbright (1996).

Dehesa management and structure should go beyond that of a simple concept of a two layered agroforestry system *sensu* Nair (1993), even beyond a combination of spatially and economically interacting plots with different vegetation structure *sensu* Etienne (1996). Dehesa should be managed as a *temporal* sequence of a set of plots, with distinct vegetation structure, integrated in a rotational cycle, as a modification of the traditional, crop-pasture cycle, to an expanded crop-pasture-shrub cycle. This expanded, long-term rotational cycle would allow both soil restoration and tree regeneration. The coexistence of two-layered plots, with multilayered plots (encroached open woods) and mono-layered plots (either dense forest or mono-pasture/monocrops) would give a mosaic at both estate and landscape scales. This would ensure the maintenance of a high structural diversity in dehesas. In this way, the environmental value of the dehesa could be maintained, and contribute to profitability in the context of the total economic value *sensu* Campos (2004).

## A Research Agenda

Explicit long-term strategies must be designed to promote management practices that ensure dehesa conservation; however, to convince landowners, administration and policy-makers, more knowledge is needed. For instance, spatially explicit studies on tree population dynamics and temporal regeneration trends as influenced by ecological and management variables are needed. A suitable demographic model for oak replacement has not yet been developed. The optimal tree density of dehesas under different uses and ecological constraints is unknown. Insufficient information exists on the stocking rates that can sustain dehesa regeneration. Stocking rate at each dehesa should be based on the overall forage availability and its seasonal pattern, but also on the need to have regular or periodical tree recruitment through avoiding grazing in summer or for periods of several years. Shrub encroachment is certainly favourable for tree regeneration, but it is doubtful whether this would maintain stand function (e.g. hydric and nutritional tree status, biodiversity) and profitability (e.g. livestock carrying capacity) of dehesas. Biodiversity, soil conservation, $CO_2$ fixation, landscape amenity, etc. are objectives of interest to the EU and affect society as a whole. The cumulative influence of these environmental functions of dehesas is also a crucial issue to be studied given that future dehesa profitability depends mostly on these indirect incomes. Finally, the stability of the dehesa system in the face of long-term climatic change will need to be studied. A projected increase in the probability of extreme climatic events could have dramatic consequences in driest dehesas (Joffre et al. 1999). Finally, more studies are needed on the origin and consequence of *la seca* and its relationship to global change (increase of climate aridity, soil compaction, and shrub encroachment).

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