Chapter

# MODELING MULTIFUNCTIONAL AGROFORESTRY SYSTEMS WITH ENVIRONMENTAL VALUES: DEHESA IN SPAIN AND WOODLAND RANCHES IN CALIFORNIA

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**Abstract**
The high environmental and amenity values of Mediterranean oak woodlands influence the response of the public and landowners to market forces and to public policies for the management of oak woodland areas. In California and in Spain, woodlands with a *Quercus* overstory open enough to allow the development of a significant grassy or shrubby understory harbor exceptional levels of biodiversity, provide watershed and habitat, sequester carbon, offer historically meaningful landscapes, and are pleasing to the eye. For historic reasons, and because of the social and environmental values of the woodlands for their owners, large private holdings based on sylvo-pastoral enterprises have and will have a crucial role in the future of the woodlands. Simple financial models for predicting landowner behavior based on response to market forces do not explain landowner retention of oaks without incorporation of landowner consumption of environmental and amenity values from the property, because landowner utility for oaks is not fully accounted for. By the same token, predicting the best afforestation approach considering carbon sequestration alone without consideration of the biodiversity and amenity values of native oaks risks an over-valuation of planting alien species that could have negative environmental and social consequences. Reforestation models for carbon sequestration that do not incorporate biodiversity and public amenity values might favor plantings of alien species such as eucalyptus, however, this does not take into account the high public and private consumption values of native oaks.

**Key Words**
Oak woodlands, optimization model, carbon sequestration, firewood, optimal control

## Introduction

Mediterranean oak woodlands have high environmental and amenity values. Woodlands with a *Quercus* overstory open enough to allow the development of a significant grassy or shrubby understory harbor exceptional levels of biodiversity, provide watershed and habitat, sequester carbon, offer historically meaningful landscapes, and are pleasing to the eye. Such woodlands are important throughout the Mediterranean area, and also in California, where the climate and vegetation formations are similar (Figure 1). Traditional sylvopastoral uses have proven to be in many cases essential and in others at least reasonably compatible with the continuance of these woodlands. For historic reasons, and because of the social and environmental values of the woodlands for their owners, large private holdings have had a crucial role. As a result, today the decisions of Spanish and Californian landowners will determine the fate of much of these woodlands. In this chapter we examine different economic models for predicting landowner response to various market forces, assessing the role of environmental and other social values for the landowner in explaining landowner decisions. We review the historical background of landownership and management; then examine the development of an optimal control model for explaining and predicting landowner stewardship of oaks; and finally examine a model assessing the potential future impact of carbon sequestration incentives for afforestation and reforestation, comparing approaches that do and do not internalize biodiversity and other values.

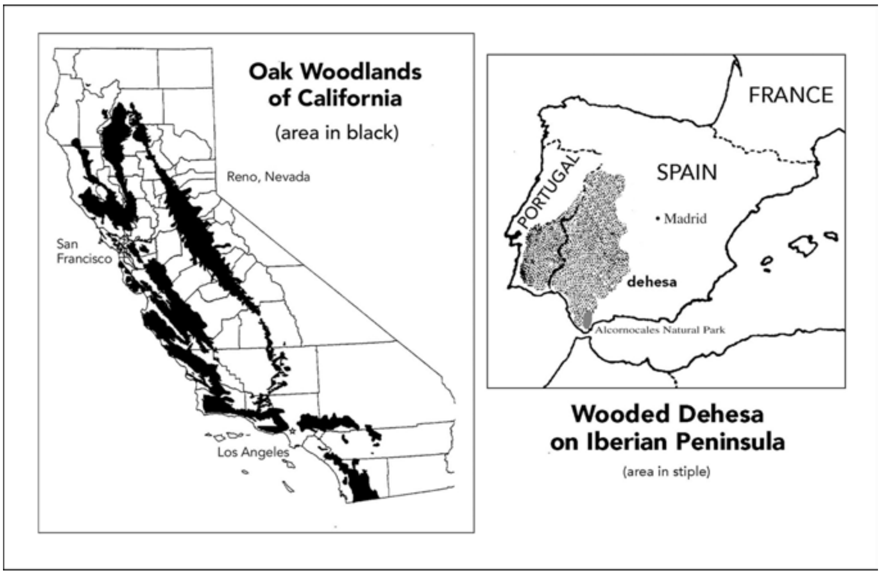


**Figure 1—**Californian oak woodlands and Spanish dehesa.

## Spanish woodland dehesa ownership and management history

Known as *dehesa* in Spain and *montado* in Portugal, sylvopastoral oak woodlands extend for more than 3.2 million hectares in the west and southwestern Iberian Peninsula. The typical dehesa is a private property larger than 500 ha (Campos, 1984) (Table 1). Dehesa is most common in areas where powerful knights and Castillian nobility were rewarded with the woodlands of re-conquered Muslim areas beginning in the 11th century and ending with the surrender of Granada to the Catholic kings in the 15th century (Hernández and Pulido, 2004).

Despite evidence that dehesa was extant in the first millennium AD when land was divided among retired Roman legionnaires near Merida, in Extremadura (Cerrillo, 1984), widespread development of dehesa is relatively recent. The historically sparse Castillian population could convert closed forest to dehesa only slowly, and manual clearing of the vigorous shrubby woodland understory to create dehesa pasture lasts only a few years without intensive grazing followed by renewed clearing or understory cereal cultivation (Díaz *et al.*, 1997). Dehesa extent peaked only in the early twentieth century (Linares and Zapata, 2003).

During the Middle Ages high quality merino wool was increasingly marketed to the textile industries of England, Flanders, and Genova. Livestock owners from the mountainous regions of Castilla and Leon found that the opened up Arab lands offered a solution for the problem of limited feed and winter forage. Transhumance, moving livestock to the south and southwest for the winter, began. The ease of collecting taxes and fees of all types as livestock threaded across bridges and through gates along the stockways meant that the transhumance was favored by the Crown. King Alfonso X the Wise sponsored a new association of transumant Castillian livestock producers under the powerful denomination of “Mesta” in 1273 (Klein, 1920).

The Mesta was for a long time one of the principal financial institutions of the Crown of Castille, and the sylvopastoralist enterprises originating from the allocation of woodlands among those considered responsible for the re-taking of the Muslim lands benefited. The leadership of the Reconquest of Castillian territory south of the Duero River and the creation of the vast majority of the dehesa is thus tightly linked. Such powerful interests were able to maintain large herds and extensive grazing lands for the wool production, limiting expansion of the subsistence crops of local rural populations. Only close to the towns are the lands mostly treeless and divided into small farms for intensive crop and pasture use.

In the nineteenth century, most church and municipal lands, and lands of the knighted orders, were expropriated by the state and sold at public auction. Some municipal common grazing lands (Dehesa Boyal), watersheds, and forests were excluded, and some nobility, able to weather the disentailment fever, retained their dehesa. The process of disentailing the dehesa increased the number of rich landowners, in a way that tended to benefit landowners from distant Madrid, Barcelona, or the Basque provinces more than local landowners.

This shared political history means that the Iberian countries developed the large private ownerships that are today’s dehesa and montado. They are the only countries on the Mediterranean that maintain livestock production integrated with oak and crop production. In continental Europe oaks were eliminated quickly from the ubiquitous medium and small properties to allow for crop production and grass pasture, just as oaks were eliminated by similar classes of Castillian landowners. The recent North African population explosion has led to an annual reduction in oak woodland cover of more than 1% per year (Campos, 2004). But in the Iberian peninsula the woodlands are stable in extent in large part because of owners of large properties who value the environmental and other benefits they get from owning the land.

Families with a history of dehesa and montado ownership consider them part of family identity and distinction. There is no doubt that newer landowners find dehesa and montado a means of achieving a higher social status. Dehesa and montado has persisted because owners have not responded to market signals that should have led them to clear oaks for cultivation. Instead they have kept their woodlands, profiting from woodland earnings, but profiting perhaps even more from family meaning, a second home, recreation, and the social status of rancher, a genteel status not enjoyed by other kinds of rich agriculturalists in Spain

**California oak woodland ranch ownership and management**

Oak woodlands with a developed understory cover more than 2 million hectares in California's Mediterranean climate zone, mostly in the rolling hills of the coast ranges or the Sierra Nevada foothills (CDF-FRAP 2003) (Table 1). Inhabited by more than 300 vertebrate species (Jensen et al. 1990), they are perhaps the most significant of the state's wildlife habitat when extent is considered. An "oak woodland ranch" is a livestock enterprise based on grazing the comparatively stable grass understory that, unlike in dehesa, most often persists without ongoing human intervention.

Prior to European settlement, California was home to an indigenous population of several hundred thousand with a long history of oak woodland management. What is known about the interaction of native management using fire and other methods, and the oak woodlands, is limited because of the widespread and rapid destruction of the indigenous way of life with the coming of Europeans (Keeley 2003), though California tribes today are making an effort to restore native management to some areas. The displacement and depopulation of native California opened up large areas to settlement, and as in Spain, the original land allocations were often made on the basis of service to the Kingdom.

It is California's Spanish and Mexican history that is largely responsible for the creation of large oak woodland ranch properties. California's coastal areas were settled starting in 1769 with missions, presidios, pueblos, and large land grants, called *ranchos*, used for livestock production. The foundation of the colonial economy, livestock hides and tallow were traded to Europe. About 30 *rancho* grants of thousands of hectares of expropriated lands were made, mostly to retired soldiers. When California became part of Mexico 1821, the new government broadened and accelerated the granting of lands in large parcels, with more than 770 grants to individuals, especially following the secularization and sale of mission lands in 1834 (Perez, 1982). In surrounding states, the U.S. government allocated lands to private holders in much smaller parcels, resulting in most forest, woodland, and desert remaining in public ownership.

With the end of the Mexican-American War in 1848, California became a territory of the United States. The ranchos were largely broken up because of legal disputes or owner impoverishment, but the ranch properties derived from these break ups, from the sale and granting of mission lands, and from various kinds of reclamation programs under the U.S. government, are relatively large, averaging 800-960 hectares in size (Table 1). The few original ranchos that remain are generally of thousands of hectares. Today, 82% of oak woodlands are privately owned (CDF-FRAP 2003).

**Table 1.** Summary of California oak woodland and Spanish dehesa characteristics.

| | **Californian oak woodland** | **Spanish wooded dehesa** |
|---|---|---|
| Extent | More than 2 m ha total oak woodlands and grasslands (CDF-FRAP 2003). | 2.2 oak wooded dehesa out of 7 m hectares of dehesa woodlands, shrublands, and grasslands (Díaz, Campos, and Pulido 1997; Campos 1984). |
| Most common oak | Blue oak (*Q. douglasii*) | Holm oak (*Q. ilex*) |
| Ownership | 82% + private (CDF-FRAP 2003). Public woodlands are part of large federal land holdings, utility corridors and watersheds, or regional, county, state and local parks. Sometimes they are leased for grazing. | 75-80 %+ private based on study of a representative area in Extremadura (Campos-Palacín 1984). "Public dehesas, dehesa boyal and other municipal woodlands, are those maintained for community use and are less then 20% of the woodlands. |
| Average ranch size | 800-960 ha (Huntsinger, Buttolph, and Hopkinson 1997; Sulak and Huntsinger, 2002). | 500 ha+ (Campos, 1984) |
| Amenity & investment ownership | Increasing owner self- consumption of environmental services | Increasing owner self-consumption of environmental services |
| Land use | Extensive sylvo-pastoral ranching over more than 60% of the woodland | Agro-sylvo-pastoral complex, "Dehesa" |
| Stocking rate of livestock (does not meet total animal demand) | 5-10 ha/A.U./year (Ewing and others 1988). | 4 ha/A.U./year in Extremadura (Campos 1997) |
| Large stock | 92% of animal demand is cattle (California Agricultural Statistics, 1990-2001) | 42 % of animal demand is cattle (Campos 1997). |
| Commodity products | Beef, lamb, wool, firewood, game, grazing resources. | Beef, Iberian pig, lamb, acorns, firewood, hay, cereal grain, grazing resources, wool, cabrito, goat milk, game, trufa, charcoal, cheese, fodder, honey, cork. |

The 1849 Gold Rush stimulated a huge short-term population increase, as gold seekers flooded in and then left. Already reduced by more than half under Spanish and Mexican governance, native populations continued a precipitous decline caused by disease, poverty, warfare, and genocide, all now extended to the mountainous mining regions. In the post-Gold Rush vacuum, transhumance into the mountains developed in the 1860's. Cattle and sheep were driven to montane summer range, helping to compensate for the loss of watered lowlands to crop production. Though management of California oak woodland ranches is far less intensive than management of the dehesa because of a general lack of aggressive shrub growth as well as rural labor, ranchers do have some history of oak thinning, brush clearing, and seeding of improved forage. A single clearing, followed by grazing, will often last indefinitely. The removal of oaks for more intensive livestock production was subsidized by the state government in the 1940's through 60's, as part of an effort to increase commodity production without consideration of environmental costs. Despite this, and an oak firewood market supported by the state's growing population, foothill and coastal ranches remain mostly oak woodland today, and studies have shown that ranchers do not often thin oaks or attempt to control them unless canopy cover becomes quite dense (Huntsinger, Buttolph, and Hopkinson 1997). More than 80% of ranchers live on the ranch with their families, and manage the enterprise themselves, with few, if any, employees. Some are multiple generation family owners, while others are wealthy individuals seeking a part- or full-time way of life that is widely admired by Americans. Numerous studies have shown that ranchers are highly motivated by lifestyle values, willing to accept considerable opportunity costs to remain in ranching, and often take off-ranch jobs or use off-ranch income to support the ranching operation (Liffmann et al. 2000).

**Goals of this analysis**

Dehesa and California's oak woodland ranches are the result of particular natural conditions, but also of a particular history of human intervention (Table 1). Thus, models aiming at the understanding of the evolution of these ecosystems need to pay particular attention to the implications of human intervention. The rest of the chapter presents two models developed to understand: (i) the current behavior of landowners in stewarding their oak woodlands in response to market forces and private environmental and social values (Model A), and (ii) the possible future development of these ecosystems under the development of carbon sequestration markets and policies (Model B).

Model A is an optimal control model developed for ranches in California. The basic model was found to severely overestimate the cutting of oaks, so a positive mathematical programming (PMP) approach (Howitt, 1995) was used to derive missing elements of the true costs and returns of oak harvest that were omitted from the original, normative model. As shown below, this permits estimate of the environmental values consumed by the owners themselves. There are environmental values, not internalized in markets as flows (although they are indeed internalized in the price of land), which explain the actual behavior observed in California of not clearing oaks. In Spanish dehesa this "owner auto-consumption" of environmental services is also important, as work by Campos and Mariscal (2003) has shown. In Spain, however this value was estimated using contingent valuation techniques, and not modeling tools.

Model B is a normative model proposed to evaluate the impact of carbon sequestration markets and policies in a model for two types of reforestations: cork oak (a native species with high environmental values, *Q. suber*) and eucalyptus (an alien species used in the past in Spain and California, *Eucalyptus globulus*). Although data for a full calibration are not available yet, the theoretical model is discussed to identify potential conflicts that may arise from internalizing only carbon sequestration, and not biodiversity and other values. Internalizing carbon sequestration only may imply an incentive to plant fast growing alien species and a lower incentive to maintain or increase oak woodlands, which may have negative impacts on biodiversity and/or on scenic and other public and landowner values. Results from a contingent valuation study estimating the impact of these two different kinds of reforestations in Spain are presented to illustrate this concern.

**Modeling landowner investment in environmental values (Model A)**

Models of likely silvopastoral management decisions must incorporate landowner values (utility), including landowner valuation of the environmental services from their lands. Poorly specified models based only on commodity production understate a manager's own consumption of amenity and environmental services, and lead to erroneous conclusions about likely management behavior and appropriate public polices.

Standiford and Howitt (1992) developed a normative dynamic oak woodland optimization model including cattle, firewood and hunting. The basic structure of the model is as follows:

$$\max NPV = \int_{t=0}^{t} e^{rt} \left\{ WR_t(WDSEL_t) + HR_t(WD_t, HRD_t, exog.) + LR_t\left[HRD_t, CS_t, FOR_t(WD_t, exog.)\right]\right\}$$

s.t.

$\dot{WD} = F(WD_t, exog.) - WDSEL_t$ (Equation of motion for oaks),

$H\dot{R}D = G(HRD_t, exog.) - CS_t$ (Equation of motion for livestock),

the initial conditions $WD_0 = INITWD$ and $HRD_0 = INITHRD$, and the non-negativity constraint $WDSEL \geq 0$.

Where: WD and HRD are the stock of wood and livestock (cows); WR, HR and LR are the net revenues of firewood, hunting and livestock respectively; WDSEL is the volume of firewood sold and CS a vector of the different classes of livestock sold; and FOR gives the number of forage quantity available.

This model was calibrated for the early nineties and concluded that markets at that time would lead landowners to clear their oaks to increase forage yield for livestock production (Standiford and Howitt, 1992). Although common in the 1940's to 1970's, this behavior was actually rare in the nineties, contradicting the prediction of the model (Standiford et al., 1996). The model's shortcomings were due to failure to accurately account for a landowner's desire to keep oaks for their amenity value. A positive mathematical programming (PMP) approach (Howitt, 1995) was used to derive missing elements of the true costs and returns of oak harvest that were not in the original normative model. The dynamic optimization model was constrained by actual landowner behavior to derive these missing values. The shadow prices from the behavior constraint represent the marginal benefit of retaining trees as it differs from what might otherwise be predicted.

The firewood net revenue developed from market information, and the hedonic pricing model calibrated from the actual behavior of oak woodland owners results in two curves (Figure 2). The difference between them is the environmental self-consumption value of retaining trees -- the value of oak trees to the landowner that cannot be explained by the price of wood or other commodity values. This specification incorporates actual landowner behavior, giving a more realistic assessment of landowner behavior than a model which omits the value of trees to the landowner (Figure 3) (Standiford and Howitt, 1992).

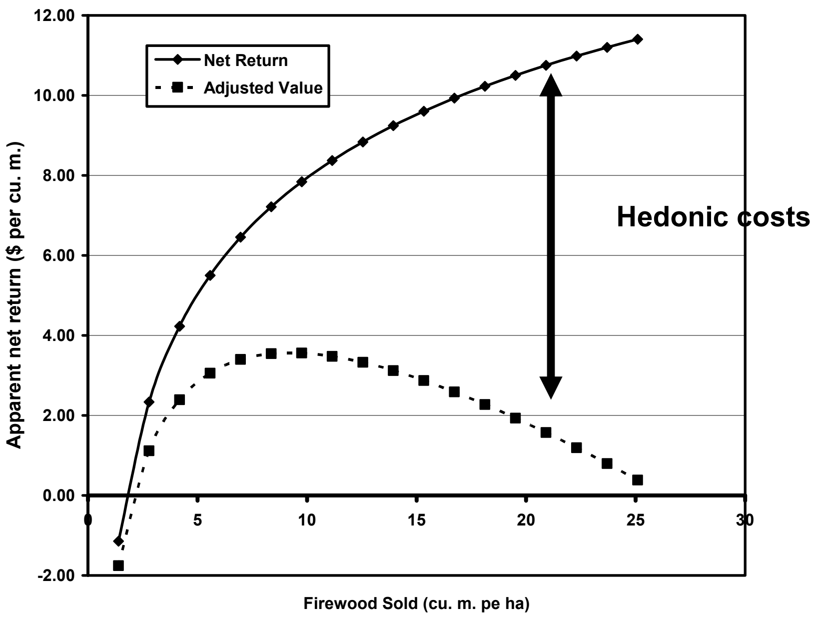


**Figure 2.** Net firewood return per cubic meter as function of amount of wood harvested Source: Standiford and Howitt (1992).

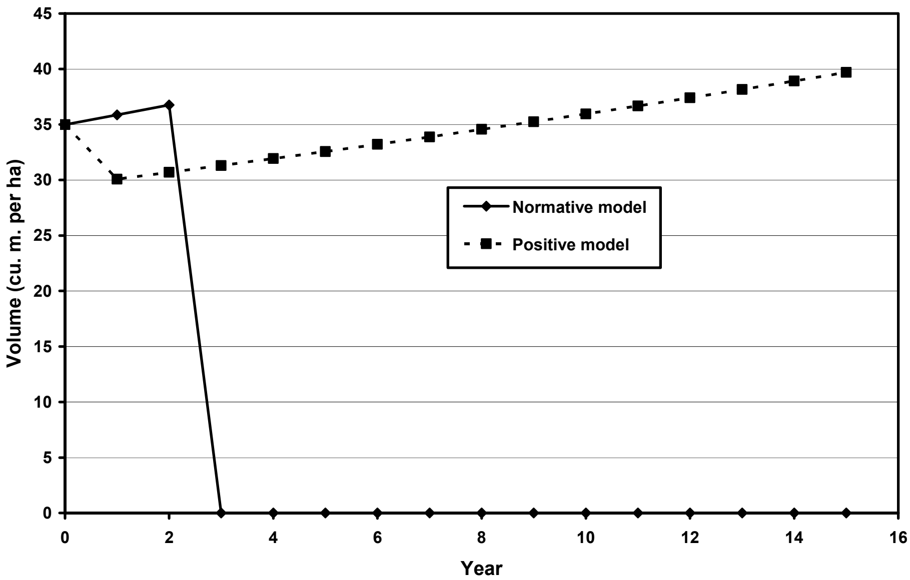


**Figure 3.** Oak volume levels in California oak woodlands under normative and positive modeling approaches (Standiford and Howitt 1992).

This optimization model, incorporating landowner utility, is used to evaluate oak cover, firewood harvest, and cattle grazing under different risk and land productivity conditions (Standiford and Howitt, 1992). Three major commercial enterprises typically contribute to total net present value of California oak woodlands (Figure 4). With an initial oak volume of 50 cubic meters per hectare

(Standiford and Howitt, 1993), cattle production on average has a positive economic value. Fee hunting can be an important enterprise, contributing from 40 percent (on good range sites) to 70 percent (on poor range sites) of the total silvopastoral value. The economic contribution of wood harvest is low.

The model showed that diversification of silvopastoral enterprises reduced tree harvesting and cattle grazing. The marginal value of retaining oaks for wildlife habitat for hunt clubs exceeded the marginal value of the extra forage or firewood harvest (Standiford and Howitt, 1992). Wood harvest is used in years with poor forage production or low livestock prices. The capital value of the trees is a hedge against years with low livestock profitability. Inclusion of a risk term shows that firewood harvest and livestock grazing intensity both increase. Policies reducing landowner risk, such as a subsidized loan program during poor forage production or low livestock price years, might reduce the need cut the trees for an infusion of capital.

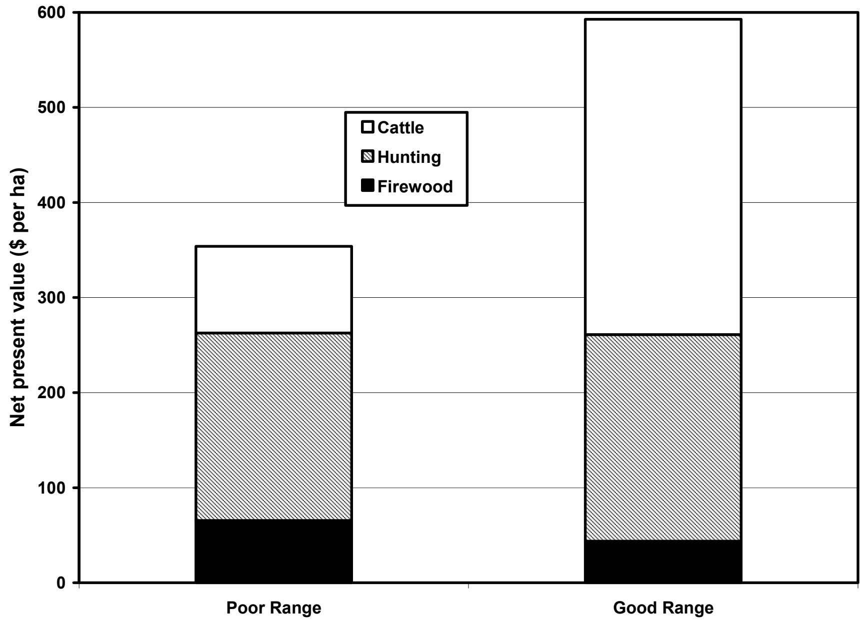


**Figure 4.** Net present value of California oak woodlands from various commercial enterprises. Initial oak volume is 50 cubic meters per ha (Standiford and Howitt 1993).

### Including carbon sequestration and biodiversity or scenic values in the analysis (Model B)

Countries ratifying the Kyoto Protocol, a development of the United Nations Framework Convention on Climate Change, will need to reduce their greenhouse gas emissions to an overall 5% below 1990 levels by 2012, though specific targets vary by country. One of the alternatives included in the Kyoto Protocol is to plant trees, since trees sequester carbon from the atmosphere by growing and thereby reduce carbon dioxide concentrations. This is 'afforestation and reforestation' in the terminology of the Kyoto Protocol and the Marrakech Accords, an agreement that completes the Protocol.

According to the Marrakech Accords, Parties can issue credits through ‘afforestation and reforestation’ by means of art. 3.3 of the Kyoto Protocol if the land is located in an Annex I country (OECD countries and former economies in transition) that ratifies the Protocol (or eventually via art. 6 and Joint Implementation), and by means of art. 12 (Clean Development Mechanism) if the land is located in any Non-Annex I Party. Thus, incentives will probably be created make carbon sequestration a forest management goal. The incentive scheme for afforestation and reforestation undertaken in an Annex-I country will probably associate payments with the actual carbon budget (since only the national budget is relevant) while for credits earned by CDM projects two methods have finally been accepted: the t-CERs and the l-CERs. The main difference between the two crediting procedures is the lifetime of the credit, 5 years (renewable) in the case of the t-CER and up to 30 years with the l-CERs. Therefore, three different crediting mechanisms are possible. However, for simplicity, the model presented below uses annualized values and a single framework, ensuring that the investment incentives are not changed, i.e., the model includes the constant annual income that would equalize the actual future stream of incomes generated for each value over the entire reforestation cycle. As a result the model presented is general enough to be applied to any of the three crediting mechanisms, and focuses on the additional income generated by reforestation with one or another species if carbon sequestration and/or biodiversity-scenic values are internalized.

It is usually accepted that biodiversity increases when degraded and agricultural lands are converted into forests (IPCC, 2000). However, this is only true of indigenous forests and not when the 'reforestation' is plantations of rapidly growing alien species like eucalyptus. It is also not true where existing land uses have high biodiversity values (IPCC, 2000). Matthews *et al.* (2002) have quantified bird biodiversity associated with reforestations in the United States and have found further evidence of the potential negative impacts of reforestation. As indicated in Jacquemont and Caparrós (2002), the 'afforestation and reforestation' alternative may conflict with the goal of the Convention on Biodiversity, since incentives to increase carbon sequestration may be negative for biodiversity under some conditions.

Van Kooten (2000) proposed an optimal control model to evaluate carbon sequestration via single species 'afforestation and reforestation', without taking into account biodiversity or scenic values. This model was extended in Caparrós and Jacquemont (2003) to include two species and biodiversity values. Nevertheless, since this paper focused on the legal and economic implications of the Protocol the model was not completely solved (only first order conditions were used) and not applied. In Caparrós *et al*. (2005) the model is discussed in depth from a theoretical point of view, and applications are currently being made in Spain and California. In what remains of this chapter we summarize the main theoretical findings of the model in Caparrós *et al.* (2005) and present some preliminary results of a contingent valuations study done to identify the willingness to pay (WTP) to favor a reforestation with oak trees and to avoid a reforestation with eucalyptus.

Moons *et al.* (2004) also deal, using a GIS-based model, with the establishment of new forests for carbon sequestration purposes, including recreation and other values in the analysis. Their model is solved numerically and highlights the empirical importance of taking into account recreational values. Thus, we will analyze not only impacts on biodiversity values but also potential impact on scenic values, since they are relatively similar from a modeling point of view (Caparrós *et al*., 2003).

Following Caparrós *et al*. (2005) we assume that the agent can choose between two types of forest, and that type 1 has greater biodiversity-scenic values while type 2 has greater carbon sequestration potential. A typical example of this situation is when reforestation with a natural indigenous species alternative (forest type 1) is compared with fast growing alien species (forest type 2). In Spanish *dehesa* or Californian oak woodland ranches we could see this model as comparing a reforestation program with oak trees (type 1) and with eucalyptus (type 2), a fast growing alien species used in the past in Spain as well as in California.

Define: $L$= total land available; $f_0(t)$ = pasture land at time $t$; $f_i(t)$= land of forest type $i$ ($i$=1,2);. To simplify, we can eliminate $f_0(t)$ from the model by setting $f_0(t) = L - f_1(t) - f_2(t)$ and leave $f_1(t)$ and $f_2(t)$ as state variables. Obviously, $f_i$ cannot have negative values. Nevertheless, for simplicity, Caparrós *et al*. (2005) analyze the problem without explicitly incorporating this restriction and check afterwards the results for non-negativity. Define further: $r$ = discount rate, $u_i(t)$= total area reforested at time $t$ of forest type $i$ ($i$=1,2) (control variables), and $K_i(u_i)$ = reforestation cost for forest type $i$ ($i$=1,2), a function of the amount of land reforested in a given year. The control variable $u_i(t)$ refers only to the amount of new land devoted to forest (or deforested) and not to the reforestation or natural regeneration needed to maintain the current forest surface. Assume $K_i'(u_i) > 0$ and $K_i''(u_i) > 0$ (e.g. as specialized labor becomes scarce, salaries increase). Finally, define $F_i(f_i)$ ($i$=0,1,2) as space-related functions showing the annual net capital income values for pasture land ($i$=0) or forest land of type $i$ ($i$=1,2), and assume $F_i'(f_i) > 0$ and $F_i''(f_i) < 0$. These functions are supposed to have three terms: $F_i(f_i) = W_i(f_i) + C_i(f_i) + B_i(f_i)$. Where: $W_i(f_i) > 0$, $C_i(f_i) > 0$ and $B_i(f_i) > 0$ represent annual net capital income associated with commercial uses (timber, cork, fire-wood, livestock breeding etc.), carbon sequestration and biodiversity-scenic values respectively. Note that forest-related data are sometimes strongly time-related but, for modeling reasons, it is interesting to annualize them, ensuring that investment incentives are not changed (Van Kooten, 2000). In the case of the Spanish *dehesa* this may be important, although in the case of Californian oak woodlands most of the data are already annualized.

The objective function is:

$$MaxV = \int_0^\infty \Pi(t) e^{-rt} dt$$

$$\Pi(t) = F_1(f_{1t}) - K_1(u_{1t}) + F_2(f_{2t}) - K_2(u_{2t}) + F_0(L - f_{1t} - f_{2t})$$

s.t.

$$\dot{f}_1 = u_1$$

$$\dot{f}_2 = u_2$$

And initial conditions: $f_1(0) = f_1^0$; $f_2(0) = f_2^0$.

$\Pi$ is a concave function, since it is the sum of concave functions and convex functions (with a negative sign). In addition, the equations of motion for the state variables are linear in the control variables. Thus, the Mangasarian sufficient conditions will hold. Using the current-value Hamiltonian and the Pontryagin maximum principle the following first order conditions can be obtained for the steady-state (Caparrós *et al*. 2005):

$$\frac{F_1(f_1^*)}{r} - K_1'(0) = \frac{F_0(L - f_1^* - f_2^*)}{r} \tag{1}$$

$$\frac{F_2'(f_1^*)}{r} - K_2'(0) = \frac{F_0(L - f_1^* - f_2^*)}{r} \tag{2}$$

Taking (1) and (2) together, and writing them out:

$$\frac{W_1(x) + C_1(x) + B_1(x)}{r} - K_1'(0) = \frac{W_2(x) + C_2(x) + B_2(x)}{r} - K_1'(0) = \frac{F_0(L - f_1^* - f_2^*)}{r} \tag{3}$$

The interpretation of equation (3) follows conventional lines. In the steady-state equilibrium the stream of net revenues associated with the reforestation of one additional hectare of forest type 1 has to be equal to the revenues associated to one additional hectare reforested with forest type 2, and to the revenues associated to the use of that hectare as pasture.

Caparrós *et al.* (2005) show, after setting $F_0' = \alpha \geq 0, \forall x$ (that is, the marginal value of pasture land is constant), that in the long-term equilibrium the amount of forest type $i$ is: $f_i^* = F_i'^{-1}(rK_i(0) + \alpha) > 0$. They also show that this equilibrium is a saddle point and that (i) if the initial amount of forest type $i$ is lower than the optimal amount $f_i^*$ the optimal approach is to reforest forest type $i$ (a positive $u_i$), and that (ii) if the initial amount of forest type $i$ is higher than $f_i^*$ the optimal approach is to reduce the amount of forest type $i$ (a negative $u_i$). The optimal approach never implies reforesting first and deforesting afterwards, so that the annualization of the revenues as described above does not change investment incentives.

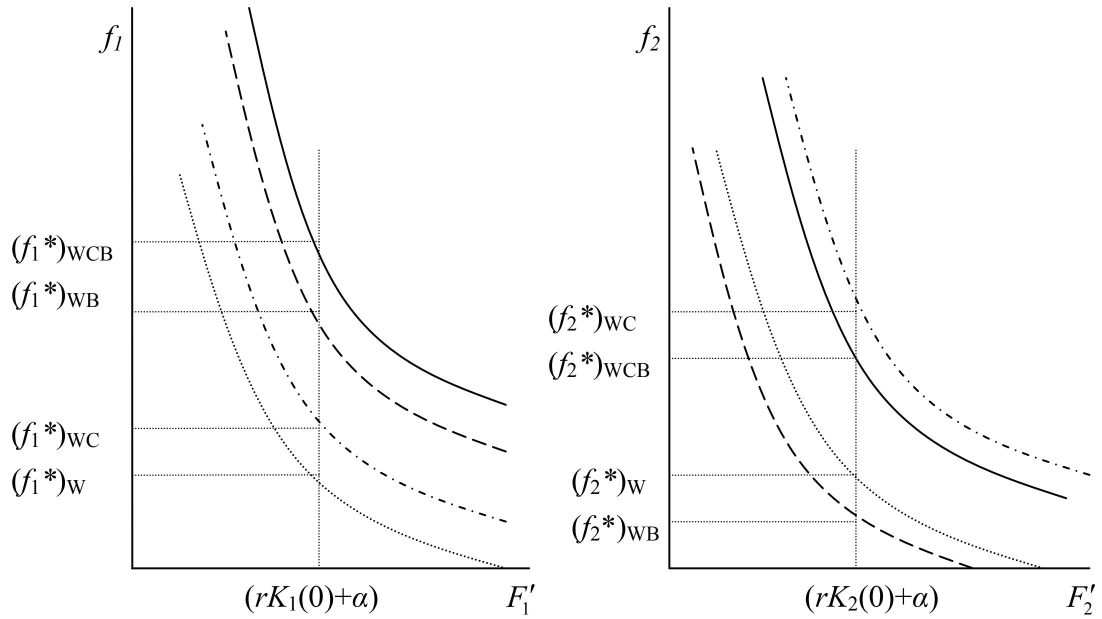


**Figure 5.** Equilibrium reforestation amounts for species 1 and 2. The functions shown are the marginal value functions for timber (dotted line), timber and carbon sequestration (dashed-dotted line), timber and biodiversity-scenic (dashed line) and timber, carbon and biodiversity-scenic (solid line) (Caparrós *et al* (2005).

So far we have discussed the system focusing on the overall valuation function ($F$). Now we will discuss the impact of different values for conventional commercial uses (timber, cork, firewood), carbon sequestration (a value that might become a market value in the future) and biodiversity values. To make things interesting, Caparrós *et al*. (2005) assume $B_1' > B_2'$ and $C_1' < C_2', \forall x$ (i.e. species 1 has higher marginal values for biodiversity and species 2 has higher marginal values for carbon sequestration). Recalling the additive form of the valuation function assumed, we can compare the optimal amount of space devoted to each species in the equilibrium considering different values. We will call $\left(f_i^*\right)_X$ the amount of species $i$ in equilibrium considering only the values indicated in the sub-index of the bracket (where $X$ can be any combination of the three values defined above: $W$, $C$ and $B$). In an arbitrary situation where commercial values are supposed to be equal for species 1 and 2, and carbon values are higher for species 2, biodiversity values for species 2, eucalyptus, are supposed to be negative (Figure 5). This is a reasonable assumption, as will be shown below.

For example, we might have a situation where future market forces (timber plus carbon) favor species 2, $\left(f_1^*\right)_{WC} < \left(f_2^*\right)_{WC}$, while present market forces equalize the amounts of both species, $\left(f_1^*\right)_W = \left(f_2^*\right)_W$, and social benefits (timber, carbon sequestration and biodiversity-scenic values) would favor species 1: $\left(f_1^*\right)_{WCB} > \left(f_2^*\right)_{WCB}$. If only timber and biodiversity-scenic values are taken into account (probably the social values currently considered) the relative amount of species 1 in equilibrium should even be bigger. In addition, these values (especially scenic values) are local by their nature while carbon sequestration benefits are global. Thus, implementing an incentive for carbon sequestration might, in this particular case, be counter to local benefits.

The discussion so far does not allow us to say if this situation is relevant to the real world. Our current research is focused on applications to multiple-use forests in Spain and California. Data for a complete calibration of this model are not available yet; however, preliminary data for scenic values in Spain suggest that cork oak reforestations are seen as highly positive by visitors while reforestations with eucalyptus are seen as negative (Tables 2 and 3). In acontingent valuation study with 900 interviews undertaken in the *Alcornocales* Natural Park (southwest Spain), half of the interviewees were asked about a reforestation with cork oak trees (showing them the evolution of this kind of reforestation in a booklet) and half were asked about reforestation with eucalyptus (giving them a similar booklet describing a reforestation with eucalyptus) (Table 2). The interviewees where then asked about their “Willingness to Pay” (WTP) to *ensure* a reforestation with cork oaks and about their WTP to *avoid* a reforestation with eucalyptus (Table 3).

**Table 2.** Subjective valuation of a reforestation with different species in the Alcornocales Natural Park (ANP) n=900 (Caparrós *et al.* 2005).

| | What is your opinion about a reforestation in the Natural Park with ….? | |
|---|---|---|
| | cork oaks (%) | Eucalyptus (%) |
| Very negative | 0.7 | 58.9 |
| Negative | 2.5 | 31.0 |
| Indifferent | 2.2 | 2.5 |
| Positive | 42.9 | 6.1 |
| Very positive | 51.8 | 1.6 |

**Table 3.** Willingness to pay to ensure reforestation with cork oaks and to avoid reforestation with eucalyptus (n=900)(Caparrós *et al.* 2005).

| | Reforestation to maintain current forest surface (compensate deforestation) | | Reforestation to increase 20% of the current forest surface | |
|---|---|---|---|---|
| | WTP to *ensure* this reforestation with *cork oaks (euros)* | WTP to *avoid* this reforestation with *eucalyptus* | WTP to *ensure* this reforestation with *cork oaks* | WTP to *avoid* this reforestation with *eucalyptus* |
| Total answers | 450 | 450 | 450 | 450 |
| Valid answers | 425 | 408 | 425 | 408 |
| Mean (€) | 26.96 | 24.21 | 30.49 | 29.68 |
| Median (€) | 12.00 | 10.00 | 12.00 | 10.00 |
| Std. deviation | 58.43 | 61.73 | 60.60 | 88.65 |

## Conclusion

Mediterranean forests in Spain and California have in common climate, Spanish historical influence, ownership structure and management. Thus, Spanish dehesa and California ranches are similar systems and present similar modeling challenges. Two optimal control models designed to incorporate environmental and social values into analysis of management options for Mediterranean forests were presented and discussed. The first model reveals that including the environmental goods valued as amenities by the landowner can better explain the fact that California landowners keep their oaks even if a simple financial model would suggest that the optimum action is to cut them down to maximize grazing resources. The second model includes carbon sequestration and biodiversity values in the analysis of reforestation alternatives for oak woodlands. The simple model suggests that fast-growing alien species are best for carbon sequestration. However, although data currently available are not enough for a full calibration of the model, the high biodiversity values of cork oak woodlands, and public preference for cork oaks compared to species such as eucalyptus, increases the benefit of cork oak reforestation. Care has to be taken not to promote aggressive incentives for carbon sequestration favoring alien species at the expense of oak woodlands. We find that at both the landowner and landscape scale, the values of landowners and the public render models that to do not incorporate what have been shown to be high amenity and other social and environmental values to be potentially misleading for policy development, and of limited explanatory value. However, further research in this area is needed.

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