

# Spatial Valuation of Forests' Environmental Assets: An Application to Andalusian Silvopastoral Farms

*Paola Ovando, Alejandro Caparrós, Luis Diaz-Balteiro,  
María Pasalodos, Santiago Beguería, Jose L. Oviedo,  
Gregorio Montero, and Pablo Campos*

**ABSTRACT.** *We develop a model that estimates spatially allocated environmental asset values for the simultaneous provision of seven ecosystem services. We examine the effect of heterogeneous spatial and economic factors on asset figures, and identify potential forestry abandonment problems when continuing with forestry activity becomes unprofitable for the landowner. Our results show a relevant spatial variability according to forest species distribution and structure. We examine potential trade-offs among silvopastoral provisioning services, water, and carbon sequestration services. Results forecast the abandonment of forestry activity and quantify the significant impact of discount rates and prices on asset values. (JEL Q23, Q51)*

## I. INTRODUCTION

Recent initiatives for moving toward a green economy triggered the interest in developing environmental accounting to analyze and track the state of ecosystems and the services they provide (Millennium Ecosystem Assessment Board 2005; UN et al. 2014a, 2014b). In recent years, there has been a noticeable effort to consider explicitly the spatial configuration of the provision of various ecosystem services (ESs) (see Wolff, Schulp, and Verburg 2015 for a review) and natural stocks. Likewise, there has been an appreciable progress in the integration of biophysical and economic land use models to simulate the spatial and temporal patterns of provision of different ESs at relevant spatial scales (Bateman et al. 2013; Lawler et al. 2014). Nonetheless, and despite recent attempts at ES quantification and mapping, these have rarely been

translated into the valuation of environmental assets (EAs)<sup>1</sup> in a way that is meaningful for decision makers (Fenichel and Abbott 2014).

Forest ecosystems are spatially heterogeneous areas in which the provision of ESs is not distributed uniformly, either in space or over time (Häyhä et al. 2015; Lawler et al. 2014; Schaafsma et al. 2014; Yuan et al. 2012). Thus, moving from ES to EA values is especially pertinent in these ecosystems, as tree growth, forest depletion, and forestry operations might affect the dynamics of ES

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<sup>1</sup> Here we use the term “environmental asset,” as “natural capital” is considered to be a broader measure that would include the stock of all EAs, embracing ecosystem assets and mineral and energy resources (UN et al. 2014b).

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The authors are, respectively, postdoctoral researcher, Department of Environmental Social Sciences, Swiss Federal Institute of Aquatic Science and Technology, Dübendorf, Switzerland; director and senior researcher, Institute of Public Goods and Policies, Consejo Superior de Investigaciones Científicas (CSIC), Madrid, Spain; professor, School of Forestry Engineering and Natural Resources, Technical University of Madrid (UPM), Madrid, Spain; researcher, Forest Research Centre, National Institute for Agricultural and Food Research and Technology (INIA), Madrid, Spain; researcher, Estación Experimental de Aula Dei, Consejo Superior de Investigaciones Científicas (CSIC), Madrid, Spain; researcher, Institute of Public Goods and Policies, Consejo Superior de Investigaciones Científicas (CSIC), Madrid, Spain; researcher, Forest Research Centre, National Institute for Agricultural and Food Research and Technology (INIA), Madrid, Spain; and professor, Institute of Public Goods and Policies, Consejo Superior de Investigaciones Científicas (CSIC), Madrid, Spain.

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supply (Biber et al. 2015; Ovando, Oviedo, and Campos 2016). This study focuses on Andalusia, a region in the south of Spain whose forests are mainly of the Mediterranean type. This type of forest forms a unique mosaic of terrestrial ecosystems shaped by diverging climatic (often extreme), geomorphological, and anthropogenic factors, and that is frequently characterized by its multifunctionality (Scarascia-Mugnozza et al. 2000) and high levels of biodiversity (Myers et al. 2000). The Andalusian case is a good example for illustrating the spatial variation in the intertemporal provision of ESs and the potential trade-offs involved.

The benefits associated with market-priced and nonmarket forest products, such as private amenities, biodiversity-scenic values, public recreation, and carbon sequestration, have been estimated for different Mediterranean forests, mainly at the forest case study level (Bernues et al. 2014; Campos and Caparrós 2006; Caparrós, Campos, and Montero 2003; Caparrós et al. 2010; Ovando et al. 2010; Ovando, Oviedo, and Campos 2016). Those benefits have also been estimated for larger spatial scales such as regions and countries, although, in a very aggregated manner (Merlo and Croitoru 2005). Both case study and regional approaches show the various ways forests contribute to human and economic activities, but do not delve deeply into the spatial and temporal distributions of benefits and asset values associated with the provision of forest ESs.

In this study we develop an EA valuation model that extends the System of Environmental and Economic Accounts Central Framework (SEEA-CF) criteria, in terms of its production function boundaries. The SEEA-CF offers an internationally accepted statistical standard for environmental accounting, and provides the guidelines necessary to develop EA accounts for individual natural resources such as timber or water. Our approach, in contrast to the SEEA-CF, considers the forest as a functional unit that supplies multiple products, entailing trade-offs among the provision functions of a single ES. In this manner, we aim to contribute to the scientific debate on ecosystem assets accounting (UN et

al. 2014b) and to provide a practical model for its spatial valuation.

Our model simultaneously computes for five silvopastoral provisioning services, including timber, cork, firewood, pinenuts, grazing resources, and the provision of water<sup>2</sup> and one climate regulating service through carbon dioxide (CO<sub>2</sub>) sequestration (carbon hereinafter). The model estimates the EA values derived from the provision of the aforesaid ESs in a group of 567 private silvopastoral farms that are distributed across Andalusia.<sup>3</sup> The application integrates spatially explicit biophysical and economic data at the farm level for the main forest species in this region (*Quercus ilex*, *Q. suber*, *Pinus pinea*, *P. halepensis*, *P. pinaster*, and *Eucalyptus globulus* and *E. camadulensis*, jointly referred to as *Eucalyptus* sp.), as well as for treeless shrubland and grassland.

The EA model estimates the expected temporal pattern of benefits and costs linked to silvopastoral, carbon, and water production functions by forest species and farm. Those benefits and costs are time-varying figures that fluctuate with the assumptions on price levels and discounting rates, as well as in accordance with the expected forest management practices and tree growth and with explicit spatial attributes such as the slope gradient, existing tree and shrub inventories, the quality of the sites for growing timber or cork, soil structure, and precipitation levels. Our benefit and cost estimations consider the spatial-explicit age class distribution of present forests and two alternative management options once current forest rotations come to an end: carrying out forest regeneration investment or, alternatively, forestry activity abandonment, the choice of which depends on the profitability of those options for the landowner.

Our results reveal a noticeable spatial variability in EA values and indicate the potential trade-offs associated with silvopastoral pro-

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<sup>2</sup> Ecosystems regulate the flow and purification of water, while forests influence the quantity of water available locally; in this sense water is considered as a provisioning service (Haines-Young and Potschin 2013; TEEB 2010).

<sup>3</sup> See Figure A1 in the online supplement, available at <http://le.uwpress.org>.

visioning services, carbon, and water. EA values are highly dependent on future forest evolution and management. Therefore, an additional outcome of our model is that it identifies potential forestry abandonment at the site level, as a result of an expected unprofitable forest regeneration investment. Likewise, the model allows for the exploration of the effect of payments for ESs on forest investment decisions. Finally, our results also highlight the significant effect of economic assumptions regarding discount rates and prices, on both the EA values and the extent of anticipated forestry abandonment.

## II. MATERIALS AND METHODS

### Silvopastoral Farms Case Studies and Area of Study

Andalusia is a very diverse region, with altitudes ranging from sea level up to 3,400 m and with climatic conditions that vary from the rainiest point in the Iberian Peninsula to the desert of Almería. This region covers 84,023 km<sup>2</sup>, which is similar in size to Austria. More than 50% of this territory is covered by Mediterranean forests, consisting mainly of a mix of native slow-growing oaks, pine species, shrubs, and grasses (CMA 2010). These are complex ecosystems in which tree, shrub, and herbaceous vegetation have been traditionally managed jointly to obtain raw materials such as cork, timber, firewood, and pinenuts and to provide hunting and grazing resources, which ascribes them as silvopastoral systems.<sup>4</sup> Private ownership dominates (73%) the area covered by silvopastoral systems in Andalusia (Campos 2015).

The 567 silvopastoral farms included in this study are distributed across Andalusia and were taken from a survey of 765 forest owners whose properties were randomly selected in this region (see Oviedo et al. [2015] and the online supplement<sup>5</sup>). They jointly occupy an area of 2,975 km<sup>2</sup> (9.3% of total private farms

in Andalusia) that is distributed across 193 municipalities. The farms have an average size of 525 ha (standard deviation 849 ha). The seven forest species included in this study represent 67% of the farms' area, while shrubs and grasslands make up 19%. Other forest species (4%) and crops (10%) occupy the remaining area.

### Pricing ESs and EAs

From an environmental accounting standpoint, an EA is defined as the naturally occurring biotic (whether natural, seminatural, or modified) and nonbiotic components of the Earth that provide a flow of ESs, which, in combination with labor and manufactured assets, contribute to the generation of products used in human and economic activities (UN et al. 2014a, 2014b; Obst and Vardon 2014). Markets for EAs and the services they provide are often incomplete or missing, especially for stocks and goods with weak exclusion, such as public products (Fenichel and Abbott 2014).

The EA and ES values are, however, not directly observable even for stocks and goods with strong exclusion, since those are embedded in the market price for assets and products, respectively. Market asset prices would internalize the value of ESs associated with forest products, as landowners hold the property rights on them. The challenge for economic valuation is to split up the asset value into the single contribution of each forest benefit and its associated ES. Hedonic pricing models might be useful to estimate the land asset value associated with different commercial forest benefits (Zhang, Meng, and Polyakov 2013) when statistical information on forest properties' sales and their attributes is available. This is not the case with land price statistics for forest properties in Andalusia (Campos et al. 2009), which require alternative asset valuation methods, as we detail later on. In any case, the hedonic price approach would not be able to capture public nonmarket values, as the market does not assign the property rights over these products to landowners.

Land leasing and forest products' prices embed the value of provisioning services such as grazing resources, cork, or timber. Like-

<sup>4</sup> These systems comprise a deliberate growing of woody perennials on the same unit of land as livestock in interacting combinations to obtain multiple products from the same management unit (Nair 1993).

<sup>5</sup> Available at <http://le.uwpress.org>.

wise, there is usually a quantifiable human input in terms of both labor and manufactured assets, which is combined with the relevant ES to produce benefits to humans (UN et al. 2014b). The difference between market prices and the unit labor, manufactured input, and full capital costs would render the unit natural resource rent (UN et al. 2014a, 2014b), and this unit price is used to value the provisioning services considered in this study. For those ESs whose property rights are not attributed to the landowner, such as water and carbon in the forests of the study area, we use surrogate market prices. We further assume that carbon and water are joint benefits of forest management, thus no labor and manufactured costs are attributed to their production functions.

Forest carbon is not included in the European Union Emissions Trading Scheme (EU-ETS). Nonetheless, the EU-ETS is the closest market available for forestry carbon in Andalusia, and its emission allowance (EUA) prices can be used and are preferable to prices obtained from completely simulated markets. We use a single regional environmental price to estimate the economic value of water flows. This price corresponds to the unit EA price of water estimated by (Berbel and Mesa 2007, 141) using a hedonic price model for irrigated agricultural lands in Andalusia. This model uses land price statistics that, in Andalusia, are available only for agricultural lands (CAP 2011) and not for forestlands. The EA price of water ( $P^w$ ), updated to year 2010, attains a value of 4.04 €/m<sup>3</sup>, and the water ES price ( $p^w$ ) is estimated using real discounting rates ( $r$ ) ranging from 2% to 6%:  $p^w = P^w r$ .

Output prices and forestry operation costs included in this study do not account for subsidies and taxes on production. Our benefit and cost projections assume constant unit prices for output and forestry operations, as well as that the returns to scale are constant. The baseline prices correspond to those observed in Andalusia and in the EU-ETS markets for silvopastoral provisioning services and carbon, respectively, in year 2010. Timber, cork, and pinenut yields are valued using average stumpage prices observed in Andalusia in the period 2008–2010 (updated to 2010) by species, product, and quality classes.

On the other hand, this study considers the most common forestry practices in Andalusia, assuming the continuation of the business-as-usual forest management. Forestry operations include shrub clearing, pruning, thinning, and commercial harvesting of pinenuts, cork, or timber, with probabilities of occurrence defined by silvicultural models (Montero et al. 2015). Forestry costs depend on specific spatial attributes of the forests, in particular their structure (species, density, and age class distribution) and the slope gradient, and they account for unit wage and input prices observed in Andalusia in 2010.<sup>6</sup>

Our cost and benefit projections consider that the relative prices (output/costs) will remain constant in the future. We acknowledge that this might be a strong assumption in view of the price tendencies observed over the last decade,<sup>7</sup> but, on the other hand, there is no robust evidence to back the idea that those price tendencies will persist over time, especially so since our model accounts for slow-growing and long-rotation forest species. Therefore, we opt for a more conservative scenario in which the unit prices for silvopastoral products, carbon, and water and the unit production costs are constant over time. However, we further check the sensitivity of EA values to increases and decreases up to 50% in the net benefit<sup>8</sup> obtained from silvopastoral products, carbon sequestration, and water as the result of changes in the price level of those outputs with respect to the baseline 2010 prices, while production costs are assumed to remain constant.

EA values are quantified as the discounted net present value (NPV) of the stream of ESs (estimated as resource rent) that a forest eco-

<sup>6</sup> See the online supplement for details, available at <http://le.uwpress.org>.

<sup>7</sup> The prices of forestry products have experienced a marked decrease, with a compound annual growth (CAG) rate of -3.3%, over the last decade (2005–2014). By contrast, agricultural basic input prices and wages have increased at a CAG rate of 3.3% and 2.1%, respectively, over the same time period (SGAPC 2014; MAGRAMA 2014; MARM 2009).

<sup>8</sup> Estimated as the difference between the benefits accrued from sales of forestry products and leasing the land out for grazing, the imputed value of net carbon sequestration and economic water, minus forestry production costs.

system is expected to yield in the future considering an infinite time horizon. The NPV approach is the standard rule for pricing assets in a deterministic case (Dixit and Pindyck 1994; Fenichel and Abbot 2014) and follows the SEEA-CF recommendations (UN et al. 2014a). The SEEA-CF recommends estimating EA values by capitalizing the flow of resource rents over the life of assets. This resource rent represents the economic rent accrued in relation to EAs and should ideally account both for the remuneration to those assets as production factors and for their depletion (UN et al. 2014a).

The ES monetary value we estimate represents the returns to EAs after covering all the operating and full manufactured capital costs. The operating costs include labor, intermediate manufactured inputs (raw materials and services), and the depletion of manufactured assets involved in the production process of different forest products; while capital costs embrace a normal return to manufactured assets used in this production process<sup>9</sup> (Ovando, Oviedo, and Campos 2016). Our model implicitly computes for (1) the potential EA depletion, by anticipating existing tree inventory withdrawals due to forest fires, natural mortality, or management, and (2) improvements (entries) due to tree growth and recruitment (described later in this section). The expected ES values of timber, cork, and firewood are quantified under a simplified approach,<sup>10</sup> based on the value of expected extractions minus operating and full capital costs.

*Provisioning and Regulating Services Depending on Tree Growth*

We extend the asset valuation approach applied by Caparrós, Campos, and Montero

(2003) to price timber stock, to multiperiodical outputs such as cork, pinenuts, or firewood and carbon sequestration due to tree growth.<sup>11</sup> EAs account for both the present forest rotation ( $EA_{T1}$ ) and for the expected stream of ESs after this rotation. Our asset valuation approach could be applied to both even- and uneven-aged forests, regardless of the initial forest structure, species distribution, and rotation age, as detailed later in this section.

The  $EA_{T1}$  is estimated as follows:

$$EA_{T1} = \mathbf{p}'_p \cdot \mathbf{Q},$$

$$\mathbf{p}'_p = (p^1_p, p^2_p, \dots, p^d_p, \dots, p^n_p),$$

being

$$\mathbf{p}^d_p = \sum_{j=s}^T [(p^d_f - \mathbf{p}^d_m) \cdot \boldsymbol{\gamma}_{dt} \cdot \beta_{dt} \cdot \delta^{(t-d)}]$$

for each  $d = \{1, 2, \dots, T\}$ ,

$$\boldsymbol{\gamma}_{dt} = \frac{q_t}{q_d}, \tag{1}$$

where  $\mathbf{p}'_p$  is a vector of unit resource rent (euros per cubic meter or per metric ton). This price vector includes for its  $T$  rows the standing price ( $\mathbf{p}^d_f$ ) of the product and the cost of forestry treatments ( $\mathbf{p}^d_m$ ),<sup>12</sup> comprising the opportunity cost of manufactured capital.  $\beta_{dt}$  represents the conditional probability that a tree of an age  $d$  is logged at any  $t$  age class to be reached ( $d \leq t$ ).  $\mathbf{Q}$  is a vector that records the existing stock of forest products or carbon for each age class at the initial period (2010).  $\boldsymbol{\gamma}_{dt}$  is a vector of expansion/contraction factors that relate the unit stock of a tree at age class  $d$  ( $q_d$ ) and the unit stock of that same tree at the age class  $t$  ( $q_t$ ). Finally,  $\delta$  represents the discount function  $\delta = (1+r)^{-1}$ .

The provision of ESs after the present rotation depends on the probability of forest regeneration investment ( $\varphi$ ), which equals 1 if current forestry activity continues in the future and equals 0 if this activity is abandoned. The decision about forest regeneration invest-

<sup>9</sup> In our application this normal return to manufactured assets equals 3% for the main scenario and varies within the discount rate applied.

<sup>10</sup> A more comprehensive approach for estimating ESs as an environmental income (Ovando et al. 2015) would consider natural timber/cork/firewood growth as an output of each period, the standing value of the woody products that are harvested as an intermediate cost (input) in the form of work-in-progress used, and the revaluation of those (holding gains) woody products along the accounting period (Campos 2015; Ovando, Oviedo, and Campos 2016).

<sup>11</sup> Carbon sequestration due to tree growth is estimated as a function of tree diameter (Montero, Ruiz-Peinado, and Muñoz 2006).

<sup>12</sup> Forestry treatments refer to those operations scheduled for the years that are left before reaching the rotation of a particular forest species.

ment is simulated at each forest unit as trees reach their theoretical rotation age (Montero et al. 2015), which varies spatially according to the initial age class distribution of the forest unit.<sup>13</sup> We assume that the forestry activity will continue if the NPV of the expected benefits of the new rotation surpasses the NPV of its costs. This probability changes within the price level and discounting rate simulated scenarios.

We assume that forest regeneration investment will result in a new forest rotation of the same species at each forest unit. Forest regeneration investment includes operations such as shelterwood cutting to promote seedling and recruitment of new individuals, weeding out, a grazing set-aside period of up to 20 years, and the clear-cutting of mature trees after this set-aside period (Ovando et al. 2010). On the other hand, we consider that forestry abandonment would lead to shrub encroachment and would change the present distribution of forest species.<sup>14</sup> Grazing resources, carbon sequestration, and water will be the only ESs delivered by this land use.

With an infinite time horizon the EA is then estimated as

$$EA = EA_{T1} + \begin{cases} \varphi \cdot (\delta^{T+1-s} \cdot (1 - \delta^T)^{-1} \cdot EA_{T2}) + \\ (1 - \varphi) \cdot (\delta^{T+1-s} \cdot (1 - \delta)^{-1} \cdot y_1^1), \end{cases} \quad [2]$$

where  $s$  is the age of the trees at the starting valuation period and  $T$  their rotation age, which varies among forest species and silvicultural models.  $EA_{T2}$  represents the EA value associated with the rotation that follows the present one if there are no economic restrictions to tree regeneration. The measurement of  $EA_{T2}$  is similar to that of  $EA_{T1}$  using equation [1], although in that case, the model accounts for the complete forestry rotation (from year 1 to  $T$ ), assuming that the second rotation is followed by an infinite sequence of identical rotations. The variable  $y_1^1$  represents the annual ES of the alternative land use 1 in case of forestry abandonment.

The EA associated with the provision of silvopastoral products would take a zero value in the event that the NPV of net benefits associated with the production of a silvopastoral product is negative (UN et al. 2014a, 158). The negative net benefits are then redistributed as returns to manufactured investment, with no return to the EA. The ESs related to carbon are estimated each period as the difference between gross CO<sub>2</sub> sequestration and release, and as we assume that carbon sequestration does not involve any manufactured assets or labor, a negative EA value would indicate loss in carbon environmental stock value.

For the main (business-as-usual) scenario we assume there will be no relevant technical innovations in forest management that increase the net benefits associated with forestry activity and tree carbon sequestration. This assumption implies that the growth and yields of the new forest rotation (if  $\varphi = 1$ ) will be similar to those of the former one. This assumption may not be unrealistic in Mediterranean forests, which are characterized by low commercial profitability and productivity rates (Campos and Caparrós 2006; Campos et al. 2008), and a low adoption of technological innovations such as the use of genetically modified trees (Montero et al. 2015). Nevertheless, through the sensitivity analysis we examine the effect of relative increases in the value of forestry benefits with respect to the cost, which could be due to an increase either in prices or in productivity (as results of technological innovations). Alternatively, we examine the effect of relative decreases in forestry benefits, which in turn may represent a decline in forest productivity due to adverse climatic conditions or a decline in prices of outputs with respect to the production costs.

#### Grazing Resources

Grazing resources include acorns (only for *Q. ilex*) and grass (swards, browses, and fruits) produced in forest, shrublands, and grasslands and that are consumed by livestock, game, and other wild species. The economic value of grazing resources depends upon the market opportunity cost of leasing the land out for livestock grazing and the

<sup>13</sup> See definition later in this section.

<sup>14</sup> The abandonment of forests and rural areas is a common trend in northern Mediterranean countries and can increase the risk of wildfires (Allard et al. 2013).

number of forage units obtained by dominant vegetation ( $e$ ) and province ( $j$ ) in Andalusia. The prices and quantities used are taken from a survey of 765 agroforestry farm owners in this region that includes the sample of 567 farms considered in this study (Oviedo, Campos, and Caparrós 2015).

We estimate the EA value associated with grazing resources ( $EA_g^i$ ) at the farm level using the average land leasing price ( $p_g^{ej}$ ) per forage unit of a dominant vegetation<sup>15</sup> and the total forage units ( $q_g^k$ ) produced by each  $k$  farm (Ovando et al. 2015). We assume that  $p_g^{ej}$  and  $q_g^k$  would remain constant at the farm level over the forest rotation and would change only in the event of forestry activity abandonment in a forest unit  $i$ :

$$EA_g^i = \sum_{t=s}^T \delta^t \cdot Y^{ek} + \varphi \cdot \frac{\delta^{S-s}}{1-\delta} \cdot Y^{ek} + (1-\varphi) \cdot \left( \delta^{S-s} \cdot \sum_{t=S-s}^{U-s} ((1-\alpha_t) \cdot Y^{ek} - \alpha_t \cdot Y^{lk}) + \frac{\delta^{U-s}}{1-\delta} \cdot Y^{lk} \right), \quad [3]$$

where

$$S = T + 1; U = S + 1 + \tau; \alpha_t = t/U; Y^{ek} = \omega^k \cdot (p_g^{ej} \cdot q_g^k - cm_g^i); Y^{lk} = \omega^k \cdot (p_g^{lj} \cdot q_g^{lj} - cm_g^i).$$

The first term of equation [3] refers to the asset value of grazing resources for the present inventory until the trees reach their rotation age. The second term of equation [3] represents the grazing resources value for an infinite sequence of forest rotations of the same species and silvicultural model in a forest unit  $i$ , if the regeneration investment takes place ( $\varphi = 1$ ). We expect that the forestry abandonment scenario ( $\varphi = 0$ ) would lead to changes in the provision of grazing resources units. The variables  $q_g^{lj}$  and  $p_g^{lj}$  define, respectively, the quantity and price of grazing resources in that scenario, which we assume to be equal to those observed in farms dominated by shrubland in each one of the Andalusian provinces.

The third term of equation [3] represents the transition from  $q_g^k$  to  $q_g^{lj}$  considering a period ( $\tau$ ) of 50 years after forestry abandonment, and that this transition is linear. Finally, we consider that after the period,  $q_g^{lj}$  and  $p_g^{lj}$  would remain constant over time.

The  $EA_g^i$  estimation additionally considers, as subtrahend, the operating and capital manufactured costs involved in the supply of grazing resources ( $cm_g^i$ ), as well as an additional correction factor defined by  $\omega^k$ . This factor indicates the probability of the farm  $k$  being used for livestock grazing in the future (hence,  $0 \leq \omega^k \leq 1$ ).<sup>16</sup>

### Carbon Sequestration in Shrub Biomass

Carbon sequestration in shrub biomass is estimated using Pasalodos-Tato et al.'s (2015) functions that relate shrub biomass growth to the fraction of shrub canopy cover and the average height of shrub formations. Net carbon sequestration by shrub growth further considers potential CO<sub>2</sub> withdrawals due to forest fires and shrub clearing. The spatial information on the variables used to estimate net carbon sequestration in shrub biomass is taken from Díaz-Balteiro et al. (2015), for both shrub formations under the tree layer and treeless shrublands. It is assumed that the shrub vegetation would maintain its current carbon stock and growth ability at each site in the future, except in the event of forestry activity abandonment.

Forestry abandonment would imply, in most cases, changing the present fraction of shrub canopy cover. Our estimations consider a set of scenarios concerning forest species and silvicultures that define the maximum fraction of land covered by shrub in a transition period  $\tau$  after forestry abandonment.<sup>17</sup> We assume a linear transition for shrub carbon

<sup>15</sup> The classification of farms for estimating grazing EA value considers the vegetation that occupies the largest part of the farm.

<sup>16</sup> In the case that grazing resources are currently consumed by livestock on a farm,  $\omega^k$  would take a value of 1. Alternatively, this probability would represent the average share of farms that are currently being used for livestock grazing according to their dominant vegetation and province (see the online supplement for more details, available at <http://le.uwpress.org>).

<sup>17</sup> See the online supplement, available at <http://le.uwpress.org>.

growth from the present situation to the one expected 50 years after forestry abandonment.

The estimation of the associated EA value follows equation [3], but we replace the price variables of this equation with a single carbon price,  $p_c$ . Likewise, we substitute the equation [3] quantity variables with  $q_c^k$  and  $q_c^l$ , which represent the annual net carbon sequestration in shrub for the forest regeneration and forestry abandonment scenarios, respectively. After the transition period it is assumed that  $q_c^l$  remains constant over time.

*Forest Water*

Water flow figures come from Beguería et al. (2015) and are based on numerical simulations of the forest water balance on hydrological response units (HRUs) in 44 reservoir catchments in Andalusia. The simulation uses daily hydrological and climatic data and covers the period 2000–2009. Precipitation water (and superficial springs in some cases) constitutes the input of water to each HRU that is transformed by forestland into different water output flows. Forest water can be either consumed within the HRU by the vegetation (evapotranspiration flow) or exported out of the HRU (surface discharge and deep aquifer recharge flows).

In the water economics literature, blue water usually defines the fresh surface and ground water (i.e., water in rivers, lakes, and aquifers), while the water that is temporarily stored in the soils to be eventually evaporated or transpired by the plants is termed green water. A fraction of blue water flows can be regulated by the water agency (collectible surplus of forest water) and later be sold to the users. The forest water with an economic value is thus made up of the superficial water runoff that reaches a reservoir in Andalusia and is allocated between the final users (Beguería et al. 2015).

Estimations of the forest water balance depend on, among other factors, soil and climatic conditions, the distribution of oaks, conifers, and other forest species, and the fraction of tree canopy cover within the HRU. We assume that the average estimates of economic forest water for the period 2000–2009 would remain constant in the future (station-

ary conditions) and would change only in the event of forestry abandonment.

The abandonment scenario would imply variations in the estimated forest water due to changes in the forest species distribution and the fraction of tree canopy cover. We estimate water EA values using an equation similar to equation [3] but replacing the price and quantity variables with a single and constant water environmental price ( $p_w$ ) and constant quantities of economic water flows for the forest regeneration ( $q_w^k$ ) and forestry abandonment scenarios ( $q_w^l$ ), respectively.

**Silvicultural Models and Tree Survival Probability Functions**

The EA model considers a set of 19 simplified silvicultural models applied to seven different species that reproduce the most common forestry practices in Andalusia.<sup>18</sup> The information provided by the silvicultural models allows the estimation of individual tree survival functions. These functions specify the survival probability ( $\pi_t^{ij}$ , where  $0 \leq \pi_t^{ij} \leq 1$ ) of a tree that belongs to a species  $i$  and a silvicultural model  $j$  at each one of the  $t$  years of the forest rotation ( $T^{ij}$ ). This probability is affected by scheduled tree thinning and final logging ( $\alpha_t^{ij}$ ),<sup>19</sup> natural tree mortality ( $\theta_t^{ij}$ ), and also by forest fire risk ( $\rho_t^{ij}$ ). The variables  $\alpha_t^{ij}$ ,  $\rho_t^{ij}$ , and  $\theta_t^{ij}$  represent the annual probabilities that trees will be felled, burned, or die, respectively:

$$\pi_t^{ij} = \begin{cases} 1 - \alpha_t^{ij} - \rho_t^{ij} - \theta_t^{ij}, & \text{if } t = 1 \\ \pi_{t-1}^{ij} - \alpha_t^{ij} - \rho_t^{ij} - \theta_t^{ij}, & \text{if } t > 1, \end{cases} \quad [4]$$

where  $t = \{1, 2, \dots, T^{ij}\}$ .

The individual tree felling probability at each period  $t$  is quantified as the ratio between  $h_t$ , the number of trees that the silvicultural models determine will be felled in that period, and  $N_1$ , the initial stand density according to the silviculture model: ( $\alpha_t^{ij} = h_t/N_1$ ). The mortality ratios are estimated as logarithmic

<sup>18</sup> See the online supplement for details, available at <http://le.uwpress.org>.

<sup>19</sup> Note that the conditional probability of tree logging  $\beta_{dt}$  of equation [1] is estimated as:  $\beta_{dt} = \alpha_{dt} \cdot \pi_{dt}$ .



functions of tree age, while the future risk of forest fire depends upon the average historical forest fire ratios by species and province assessed for the period 1987–2006 (Díaz-Balteiro et al. 2015). The rotation age by species and silvicultures is exogenously defined by Montero et al.'s (2015) models.<sup>20</sup>

A survival probability matrix has a dimension  $T^{ij} \times T^{ij}$  and computes the conditional probability ( $\pi_{dt}^{ij}$ ) that a tree of an age  $d$  is alive at each one of the tree ages  $t$  that are to be reached ( $d < t$ ):  $\pi_{d\tau}^{ij} = \Pr(d/t) = \pi_t^{ij}/\pi_d^{ij}$ .

The EA model includes 152 different probability matrices, one for each of the 19 silviculture models and each one of the eight provinces of Andalusia. These matrices are used to simulate the evolution of forests and could be applied to any initial condition, which is defined by the distribution of the existing trees by species and age class in a forest unit. Initial forest inventories and other spatial variables at the farm level were estimated for the polygons of the Spanish Forest Map using the latest National Forest Inventory (IFN3) in Andalusia (MARM 2013) and digital elevation maps.

The IFN3 data were gathered between 2006 and 2008 in Andalusia and were updated to the beginning of 2010, considering species and site-specific growth function (see Díaz-Balteiro et al. 2015 for details). Those variables were assigned to the 567 farms according to the weighted average values by forest species and silvicultural model at the municipality level.

The reason for assigning values at the municipality level is that we ignore which Spanish Forest Map polygons correspond to each farm; rather we observe the municipality in which the farm is located and its land use distribution (as stated by farm owners). The farm area is shared out into a set of homogeneous forest units that represent the distribution of the forest inventories and silvicultural models of private lands in the municipality, while the area covered by each forest species, shrubland, grassland, and other land uses is specific to the farm. A homogeneous forest unit is de-

finied in terms of species composition, density, age class distribution, slope gradient, the silvicultural model assigned, and the quality of the site for growing timber or cork.

The EA valuation model is developed in Matlab R2014a.<sup>21</sup> Figure 1 shows a scheme of the interrelated components of this computing model and the sources of biophysical and economic data.

### III. RESULTS

#### EA Estimations for Provisioning and Regulating Services

##### *Average Values at Farm and Vegetation Levels*

Table 1 shows the estimated EA value (in euros per hectare) by forest species, ESs, silvicultural model for the main scenario, that is, with a discount rate of 3% and the average prices of 2010. This is the main scenario, although we analyze the sensitivity of results to discount rates in the range of 2% to 6% and variations in net benefits, due to changes in prices from  $\pm 25\%$  and  $\pm 50\%$ .

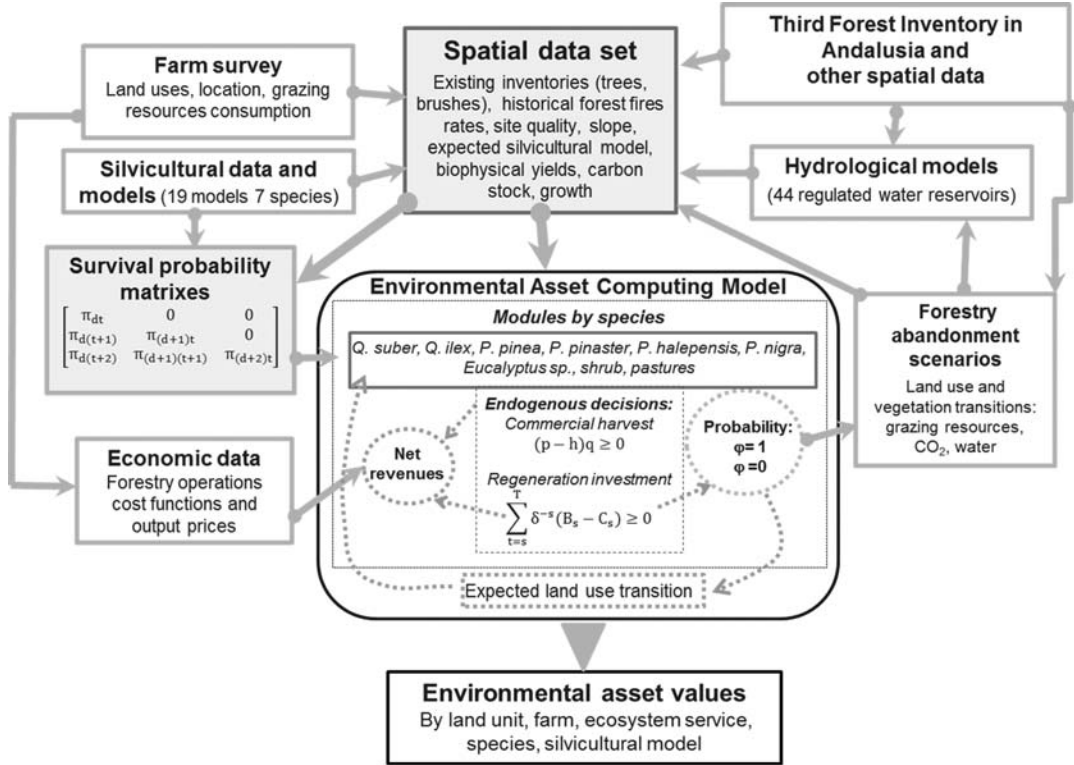
The aggregated EA value of silvopastoral provisioning services, water, and carbon amounts to 2,813 €/ha (standard deviation = 2,383 €/ha), on average for the 567 farms included in the analysis. Cork and grazing resources represent 9% and 28% of this average EA value, respectively; carbon contributes 39% (58% of which is due to tree net growth, and 42% due to shrub net growth) and water 23% of this value. Timber, pinenuts, and firewood account for the remaining 1%.

The EA values of timber and pinenuts display a higher variability among the farms than any other ES. Variability in cork values across farms is also large, while the variability of results in terms of grazing resources is small. The relative homogeneity in EA values of grazing resources is due to the fact that available data on grazing leasing prices differ only by dominant vegetation and province, without connection to other spatial factors. The variability of the EA value associated with the

<sup>20</sup> See the online supplement, available at <http://le.uwpress.org>.

<sup>21</sup> Matlab is available from MathWorks ([www.mathworks.com](http://www.mathworks.com)), and the specific code used was developed by the authors.

FIGURE 1  
Components of the Environmental Asset Valuation Model



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provision of water is also small for the group of silvopastoral farms, but higher for specific types of vegetation, in particular for the category “other vegetation,” which includes mainly treeless shrubs and grasslands. The EA value of carbon sequestration due to tree and shrub net growth varies among species and management models and is, in general, higher for pine than for oak species.

*Q. suber* is the species that offers the highest aggregated EA value, with cork making up almost half of this figure. The contribution of timber EAs is negligible among *Eucalyptus* and *Pinus* species, after covering labor and manufactured costs associated with timber production. Grazing resources and, particularly, carbon explain the largest part of the aggregated EA values of *Pinus* s. and *Q. ilex*. The EA value of water is significantly smaller in forest units covered by *Pinus* sp. than in those covered by oak trees.

The estimated carbon EA value for *Eucalyptus* trees is small in comparison to other

species, which may be surprising for a fast-growing species. Nonetheless, this result is consistent with managed forests close to a steady-state situation, where net carbon sequestration tends to zero as biomass gross natural growth equals extractions.

Grazing for the category “other vegetation” includes those resources obtained in shrub- and grasslands, other forests, and crops. Carbon sequestration in these other types of vegetation considers CO<sub>2</sub> fixation due to shrub biomass growth, and CO<sub>2</sub> release due to shrub clearing and forest fires affecting this vegetation. As expected, our results confirm that carbon sequestration potential is bigger in forested areas than in treeless shrublands. They also show that the value of potential carbon storage in the tree stratum is higher than the value of storage in the shrub stratum (Table 1).

Silvopastoral provisioning services are relevant land price factors, while land price would in principle not be affected by forest

TABLE 1  
Average Environmental Asset Value by Ecosystem Service, Vegetation, and  
Silvicultural Model

Species	Environmental Asset Value by Species and Silvicultural Model (euros/ha, year 2010)					Total Mean	Std. Dev.
	S1	S2	S3	S4			
<i>Quercus ilex</i>	2,580	3,543	2,744	3,238	2,951	1,467	
Firewood	0	442	0	131	107	181	
Grazing resources	888	766	521	728	712	296	
Carbon trees	403	898	976	804	702	508	
Carbon shrub	234	257	242	232	240	140	
Water	1,055	1,181	919	1,046	1,031	1,228	
<i>Q. suber</i>	6,236	6,383			6,307	4,150	
Cork	3,060	2,846			2,957	3,806	
Grazing resources	1,002	997			999	275	
Carbon trees	523	909			710	416	
Carbon shrub	494	514			504	430	
Water	1,156	1,118			1,137	1,092	
<i>Pinus pinea</i>	2,077	1,704	1,794		1,858	594	
Timber	8	9	1		6	7	
Pinenuts	109	57	152		106	90	
Grazing resources	492	593	486		524	312	
Carbon trees	660	261	236		386	374	
Carbon shrub	426	483	447		452	130	
Water	384	301	472		385	273	
<i>P. halepensis</i>	2,125	969			1,516	1,149	
Timber	46	9			26	28	
Grazing resources	323	312			317	335	
Carbon trees	1,456	357			877	897	
Carbon shrub	203	202			202	57	
Water	97	90			93	287	
<i>P. nigra</i>	2,478	5,847			3,826	1,978	
Timber	24	30			27	17	
Grazing resources	640	640			640	0	
Carbon trees	558	3,976			1,925	1,964	
Carbon shrub	125	125			125	12	
Water	1,131	1,075			1,109	804	
<i>P. pinaster</i>	112	4,081	2,615	2,053	3,150	1,902	
Timber	1	171	101	8	121	105	
Grazing resources	54	820	753	640	711	389	
Carbon trees	11	1,670	637	566	1,110	1,026	
Carbon shrub	33	789	823	411	742	404	
Water	13	631	301	398	466	616	
<i>Eucalyptus</i> sp.	2,235	2,251			2,389	710	
Timber	0	0			0	0	
Grazing resources	957	800			859	414	
Carbon trees	290	249			386	271	
Carbon shrub	839	1,100			1,029	602	
Water	149	103			115	256	
Other vegetation <sup>a</sup>					1,694	1,243	
Grazing resources					676	392	
Carbon shrub					964	1,025	
Water					54	210	
All species					2,813	2,383	
Timber					2	53	
Cork					262	818	
Firewood					23	53	
Pinenuts					4	27	
Grazing resources					781	946	

(table continued on following page)

TABLE 1  
Average Environmental Asset Value by Ecosystem Service, Vegetation, and  
Silvicultural Model (*continued*)

Species	Environmental Asset Value by Species and Silvicultural Model (euros/ha, year 2010)				Total Mean	Std. Dev.
	S1	S2	S3	S4		
Carbon trees					635	593
Carbon shrub					452	480
Water					654	779

*Note:* Environmental asset results are provided for the main scenario (discount rate 3% and average prices of 2010), considering the average results of 567 silvopastoral farms. S1<sub>j</sub> to S4<sub>j</sub> refer to the four potential silvicultural models applied to each one of the *j* species.

<sup>a</sup> Other vegetation includes grazing resources in treeless shrubs, grassland, crops, and other forest species.

water and carbon sequestration (as landowners do not get any specific payment for these ESs). The presence of water bodies (lakes or streams) may well affect the price of the property (Campos et al. 2009, 246), though our forest water estimations account for the quantity of water that reaches reservoirs that are regulated by the water authority, and that currently do not involve any payment to landowners for the water ESs. For the main scenario, we estimate that the aggregated value of the provisioning services derived from silvopastoral activity would account for 25% of the average land price for nonirrigated pastures (4,294 €/ha) in 2010 in Andalusia (CAP 2011).<sup>22</sup> This later share would fluctuate from 21% to 35% when EA values are estimated using discount rates of 4% and 2%, respectively. The main reason our EA values scarcely represent even a relatively small share of land market prices is that there are other final products such as hunting (Hussain et al. 2013) or nonmarket private amenities (Campos et al. 2009) that affect forestland prices, but due to data limitations those are not considered in this paper.

Campos et al. (2009) estimate an average cork oak woodland market price of €8,451/ha in 2002 (10,467 €/ha when updated to 2010<sup>23</sup>), based on information provided by landowners in Cádiz (Andalusia), and that cork and the aggregated livestock and graz-

ing rents would explain, respectively, 28% and 18% of this price. Nonmarket private amenities explain 36% of this price and other market factors (hunting rent, crops, and others) the remaining 18%. A recent survey of silvopastoral farm owners in Andalusia further suggests that 55% of the land price would be explained by nonmarket private amenities, while close to 30% would be explained by market-based silvopastoral products (Oviedo, Campos, and Caparrós 2015). We estimate, for the main scenario, that cork and grazing resources would account, respectively, for 28% and 10% of the updated *Q. suber* woodland price in 2010, which is in line with the land price distribution estimated by the aforesaid studies.

#### *Spatial Distribution of EA Values and Trade-offs*

Figure 2<sup>24</sup> shows the spatial distribution of EA values of aggregated silvopastoral provisioning services (EAPr), CO<sub>2</sub> regulating service (EAC), and forest water (EAW) per hectare in the municipalities where the studied farms are located. The spatial variability of EA values is connected to heterogeneous biophysical factors such as the slope gradient, the distribution of forest species, the density of forest stands, or the quality of the sites for growing cork and timber, as detailed later.

Our results indicate higher EAPr values in western Andalusia and in the areas with a relevant extent of *Q. suber* woodlands (Figure

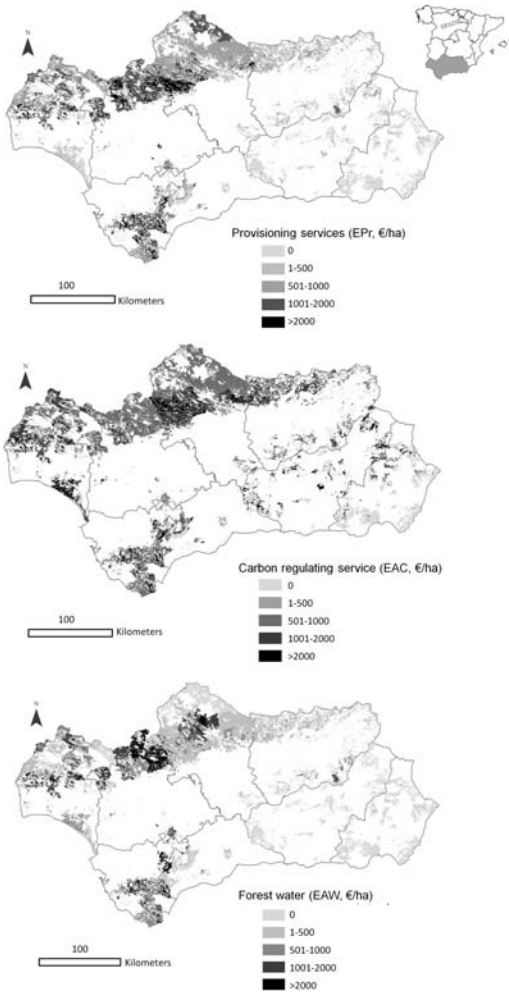
<sup>22</sup> There are no statistical data on forest and shrubland prices in Andalusia, and the price of nonirrigated pastures is the only proxy land price statistic available for silvopastoral farms.

<sup>23</sup> Value updated using the consumer price index.

<sup>24</sup> Data for main scenario: discount rate = 3%, price level of 2010 ( $p = 1.0$ ).

FIGURE 2

Distribution of Silvopastoral Provisioning Services, Carbon, and Water Environmental Asset Values in the Municipalities Where the Studied Farms Are Located



2a). The lowest EAC values are observed in Eastern Andalusia, where shrubland is the predominant vegetation (Figure 2b). EAC values are lower for those areas where *Q. suber* is the dominant species and higher for the areas dominated by *Pinus* sp., which in contrast depict lower EAPr values. Those results suggest a trade-off between carbon sequestration and silvopastoral provisioning services between cork oaks and pine species in Andalusia.

EAW values also depict a relevant spatial variability, with lower values for provinces in eastern Andalusia (Figure 2c). The mountainous areas of Seville and Cádiz provinces show higher EAW values. We also observe that areas with a higher EAC value show, at the same time, moderate to low EAW values, which seems to be related to a higher evapotranspiration rate in forests with higher CO<sub>2</sub> sequestration potential. On the other hand, medium-to-high EAW values tend to coincide with medium-to-high EAPr values, which is likely associated with *Q. suber* distribution.<sup>25</sup>

The variables that operate in the EA valuation model are diverse and depend on multiple interactions between spatial and nonspatial biophysical and economic factors. To examine the magnitude of the effect of spatial biophysical variables, we adjusted simple linear regression models that relate the average EA values (in euros per hectare) for provisioning services, carbon, and forest water to a number of spatial attributes at the farm level (Table 2) and the EA values estimated for forest units dominated by pine and oak species (Table 3).

At the farm level, we examine the effect of the slope gradient, the density of the forest (using the basal area as a proxy indicator of the stand density), and the share of oak woodlands, pine species, and treeless shrubs and grasslands on total EA values. The EAPr values at this level are increased within the share of oak (*Q. suber* and *Q. ilex*) and pine species, while the share of treeless shrubs and grasslands reduces the EA value of provisioning services. The effect of the slope is not significant at the farm level, while the stand density negatively affects the EAPr values. On the other hand, EAC values decrease within the share of oaks and increase within the share of pines at the farm level. This latter result confirms the trade-off between EAPr and EAC regarding oak species, although this trade-off would depend on other variables, such as the

<sup>25</sup> *Q. suber* is a species better suited to more humid areas.

TABLE 2  
Environmental Asset Functions for the Farms Sample for the Main Scenario  
(Number of Observations = 567)

Variable <sup>a</sup>	Main Scenario (Discount Rate = 3%, Price Level: 2010 = 1.0)					
	EAPr <sup>b</sup>		EAC <sup>c</sup>		EAW <sup>d</sup>	
	Coef.	Std. Err. <sup>e</sup>	Coef.	Std. Err. <sup>e</sup>	Coef.	Std. Err. <sup>e</sup>
Constant	67.52	47.37	-208.34***	70.76	178.09	122.24
Slope	-34.11	70.10	416.90***	102.10	780.70***	204.14
Density (BA)	-14.47***	1.24	49.15***	2.13	50.87***	2.32
SQI	966.87***	60.76	-110.35	78.32	-380.32***	111.80
SQS	1,248.82***	82.57	-393.02***	101.97	-270.82**	150.83
SPP	705.27***	97.37	726.54**	279.21	179.35	329.18
SSP	-43.88	58.50	50.17	74.76	676.65***	210.16
R <sup>2</sup>	0.68		0.78		0.49	

<sup>a</sup> The functions estimate the environmental asset value (in euros per hectare). Slope is estimated as a percentage value, density (BA) refers to the initial basal area (in square meters per hectare). The share (S) variables indicate the proportion of the farm area occupied by different land use classes (in percentage): SQI, share of *Quercus ilex*; SQS, share of *Q. suber*; SPP, share of *Pinus* sp.; SSP, share of shrublands and grasslands.

<sup>b</sup>EAPr: environmental asset value of silvopastoral provisioning services.

<sup>c</sup>EAC: environmental asset value of carbon sequestration.

<sup>d</sup>EAW: environmental asset value of water provisioning service.

<sup>e</sup> Robust standard error.

\*  $p < 0.10$ ; \*\*  $p < 0.05$ ; \*\*\*  $p < 0.01$ .

stand density (Figure 3<sup>26</sup>). In the case of EAW values, we observe that this value decreases within the share of oak species and increases within the share of treeless shrubs and grasslands, while the effect of pine species is not significant. The main reason for such results is that forested areas (in contrast to treeless shrubs and grasslands) have higher evapotranspiration rates, which reduce the forest water flows that can be regulated at each HRU. Finally, it should be noted that the size of the farm has no significant effect on the EA values considered, and that this is related to the assumption of constant returns to scale, and therefore farm size is not considered as a variable in the regression models.

Table 3 presents the results of simple linear regressions that relate EA values with the main biophysical variables that characterize the forest units of three specific forest species (*Q. ilex*, *Q. suber*, and *Pinus* sp.) for the main scenario. The independent variables include

the slope gradient, density of the stand, quality of the site for growing cork or timber, and forest structure (even- or uneven-aged forest). EAW regressions are analyzed for the aggregated oak and pine species, and specific oak and pine species enter as variables of the regression models.

As forestry costs rise with the slope gradient, it is expected that the EAPr values of *Pinus* sp. and *Q. ilex* are negatively affected by this variable, although this variable is significant only for *Q. ilex*. On the other hand, we observe higher EAPr values of forest units dominated by *Q. suber* with a higher slope gradient, which seems to be explained by the distribution of site qualities for growing cork (Díaz-Balteiro et al. 2015). The stand density has a positive effect on EAC and EAPr values in forest units dominated by *Q. suber* and *Pinus* sp., as higher carbon sequestration and timber and cork productivity are expected. By contrast, higher *Q. ilex* densities would reduce the EAPr values associated with this species. A relevant reason for these results is that *Q. ilex* in western Andalusia is normally managed in open woodlands (known as dehesa systems), with scattered trees and a well-de-

<sup>26</sup> EAPr: environmental asset value of silvopastoral provisioning services; EAC: environmental asset value of carbon sequestration; EAW: environmental asset value of water provisioning service. Data for main scenario: discount rate = 3%, price level of 2010 ( $p = 1.0$ ).

TABLE 3  
Environmental Asset Functions for *Pinus* sp., *Quercus ilex*, and *Q. suber* for the Main Scenario

Variable <sup>a</sup>	<i>Pinus</i> sp.				<i>Quercus ilex</i>			
	EAPr		EAC		EAPr		EAC	
	Coef.	Std. Err. <sup>(3)</sup>	Coef.	Std. Err.	Coef.	Std. Err.	Coef.	Std. Err.
Constant	227.75***	51.22	303.10***	86.41	1,253.78***	18.28	91.43***	30.89
Slope	-407.48	438.46	-87.94	1,603.64	-217.67*	122.45	658.69***	115.47
Density (BA)	12.64***	2.48	79.32***	10.73	-15.29***	0.74	48.22***	1.29
HMQ			-424.64**	165.71	429.93***	24.91	436.15***	22.67
EAF					-200.39***	15.12	-236.72***	19.39
Pinea	454.69***	62.82						
R <sup>2</sup>	0.35		0.59		0.52		0.80	
Number of observations	171		171		1,317		1,317	

	<i>Quercus suber</i>				<i>Quercus</i> sp.		<i>Pinus</i> sp.	
	EAPr		EAC		EAW		EAW	
	Coef.	Std. Err.	Coef.	Std. Err.	Coef.	Std. Err.	Coef.	Std. Err.
Constant	1,713.79***	483.80	303.43***	68.35	678.26***	73.36	317.27***	52.25
Slope	5,433.83*	3,184.98	99.91	313.61	3,411.47***	495.71	-355.48	431.01
Density (BA)	53.79***	15.99	33.97***	2.24	-8.21***	2.23	2.12	3.63
Q1	-1,981.28***	354.82						
Q2	-1,382.77***	270.37						
Q4	4,726.77***	464.97						
Suber					473.78***	59.73		
Halep							-218.42***	48.26
Nigra							779.04**	340.53
Pinst							121.93	139.07
R <sup>2</sup>	0.38		0.50		0.06		0.25	
Number of observations	720		720		2,037		171	

Note: EAPr, environmental asset value of silvopastoral provisioning services; EAC, environmental asset value of carbon sequestration; EAW, environmental asset value of water provisioning service. Data for the main scenario: discount rate = 3%; price level 2010 = 1.0. Robust standard error.

<sup>a</sup> Slope gradient is estimated as a percentage; density (BA) is the basal area in m<sup>2</sup>/ha. The remainder are dummies: HMQ is high-medium quality; EAF is even-aged forest; Suber, Halep, Nigra, and Pinst are *Quercus suber*, *Pinus halepensis*, *P. nigra*, and *P. pinaster* forests, respectively; and Q1, Q2, and Q4 are the cork quality indexes (Q1 being the lowest and Q4 the highest).

\*  $p < 0.10$ ; \*\*  $p < 0.05$ ; \*\*\*  $p < 0.01$ .

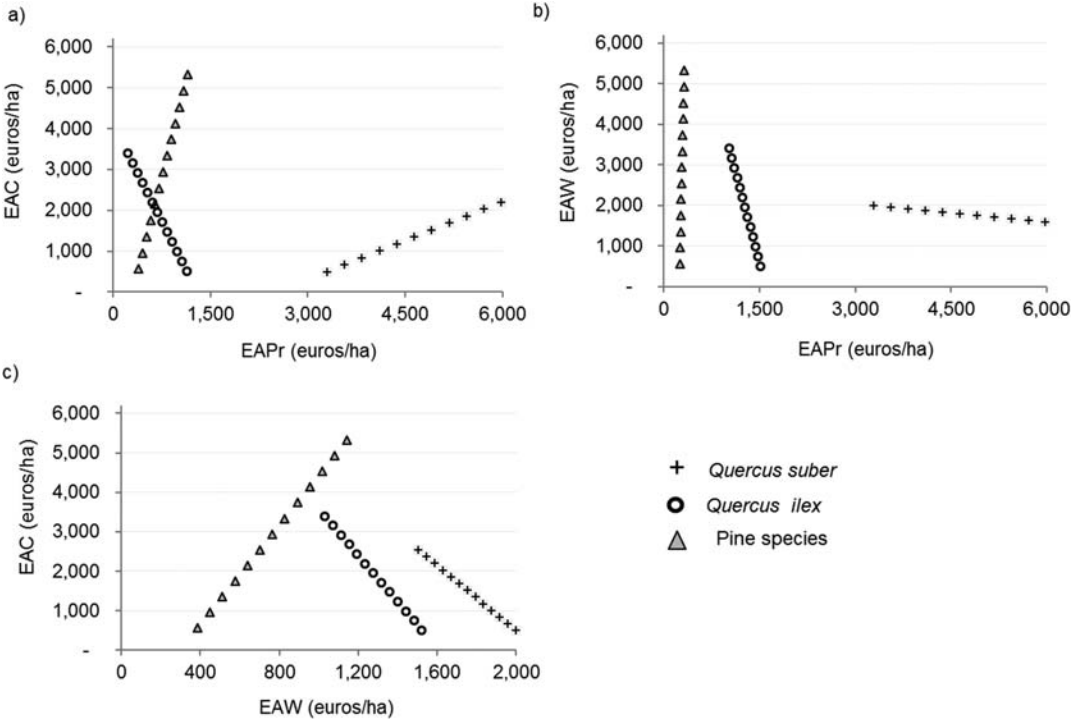
veloped herbaceous stratum that is mainly used for ranching, whereas grazing values are higher in open woodlands than in denser ones. We also observe that EAW values are higher in areas where *P. nigra* and *Q. suber* are located, concurring in many cases with mountainous areas. Water values depend, however, on other variables besides the slope or the density, such as climatic factors and the characteristics of the catchment areas that are not considered in Table 3's regressions, which explains the limited prediction abilities of the estimated EAW models (Table 3).

Figure 3 shows potential trade-offs between EAPr, EAC, and EAW values of *Q.*

*suber*, *Q. ilex*, and *Pinus* sp. These asset values are estimated using Table 3's regressions, considering basal areas ranging from 5 to 65 m<sup>2</sup>/ha and assuming forest units with average slope gradient (26%), and the weights of pine species and site qualities for growing timber, firewood, and cork observed in the sample of 567 silvopastoral farms. EAPr and EAC values increase with stand density for both *Q. suber* and *Pinus* sp. (Figure 3a). Hence, there is not really an intraspecific trade-off between these two services for either of these two species; it is, rather, an interspecific trade-off, as *Q. suber* would deliver higher EAPr and lower EAC values than *Pinus* sp. In contrast,

FIGURE 3

Trade-offs between the Environmental Asset Values of Silvopastoral Provisioning Services, Carbon, and Water in *Quercus suber*, *Q. ilex*, and *Pinus* sp. Forest Units



we observe a clear trade-off between EAPr and EAC services in *Q. ilex* forests when the density of the stand is increased.<sup>27</sup>

For pine species there are, as well, no real trade-offs within the other two pairs of ESs considered (EAW and EAPr, and EAW and EAC, Figure 3b and 3c). This implies that the three services can be maximized jointly by increasing the tree density of pine species; note, however, that the EAW results for this species have to be interpreted carefully, as the effect of the stand density is not significant (see Table 3). On the other hand, for *Q. suber* and *Q. ilex* we observe a clear trade-off between EAW and EAPr and between EAW and EAC (Figure 3b and 3c), as EAPr and EAC values increase with the stand density while the EAW value decreases.

*Sensitivity to Discount Rates and Output Prices*

The average EA value of silvopastoral provisioning services for all the studied farms fluctuates from -36% to 44% for the higher (6%) to the lower (2%) discount rate scenarios, with respect to the main scenario's discount rate (3%) (Figure 4<sup>28</sup>). Similarly, the average EAC ranges from -41% to 46% with respect to the main scenario. Higher discount rates have a slightly stronger effect on the EAPr figures than they do on the estimated EAC ones in terms of the variability of results, while they do not affect EAW values, as we apply a single EA price for water that does not depend on the discount rate.

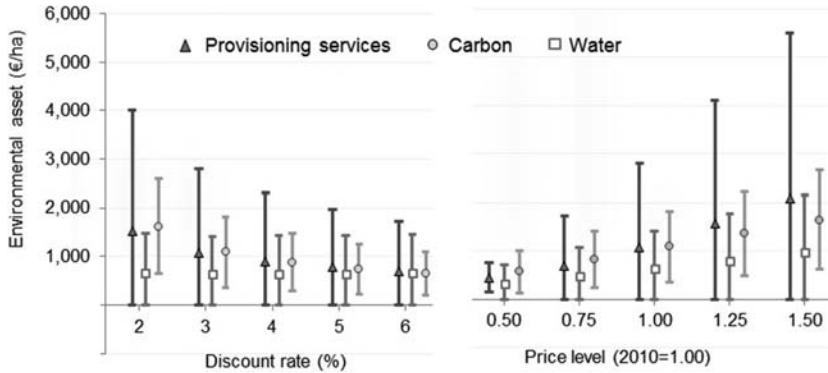
<sup>27</sup> See the online supplement for details, available at <http://le.uwpress.org>.

<sup>28</sup> Lines show the standard deviation of the mean. The central price level for different discount rates is 1.0, while the central discount rate for different price levels is 3%.



FIGURE 4

Sensitivity of the Environmental Asset Value for Silvopastoral Provisioning Services, Carbon, and Water to Discount Rates and Changes in Output Prices



The sensitivity analysis includes variations in output prices with respect to the 2010 prices for silvopastoral products, carbon, and water. A variation in output prices would have a larger effect on EAPr. Average EAPr values are expected to increase from 46% to 93% for relative rises in output prices of 25% and 50%, respectively. Correspondingly, EAPr is expected to decrease by up to 57% in the case of an extreme drop of 50% in silvopastoral product prices with respect to the baseline scenario.

We estimate that a drop in carbon prices of up to 50%, which is close to the fall observed in EU-ETS prices between 2010 and 2014 (SENDECO2 2015), reduces the EAC by 46% with respect to the main scenario. Average EAW values are largely affected by the number of forest units in which the economic value of water is zero (Figure 2c), which makes the EAW less sensitive to changes in output prices for water.

### The Forest Conservation Question

The forest regeneration investment decision depends on the NPV of expected private benefits from the new rotation exceeding its costs. Those private benefits are related exclusively to silvopastoral provisioning services. We acknowledge that contrasting the NPVs of future benefits and costs to forecast potential forestry abandonment is a very simplistic methodology. This approach neglects, for example, liquidity constraints due to the fact that

the landowner will have to wait many years before her investment yields any products of commercial interest, and the opportunity cost of land and other nonmonetary variables that may affect landowners' preferences (Campos et al. 2009; Díaz-Balteiro, González-Pachón, and Romero 2009). Despite its drawbacks, this approach indicates in which cases and under what conditions investment in facilitating natural regeneration would be profitable for landowners, and in which cases landowners would need additional incentives to invest in regeneration treatments.

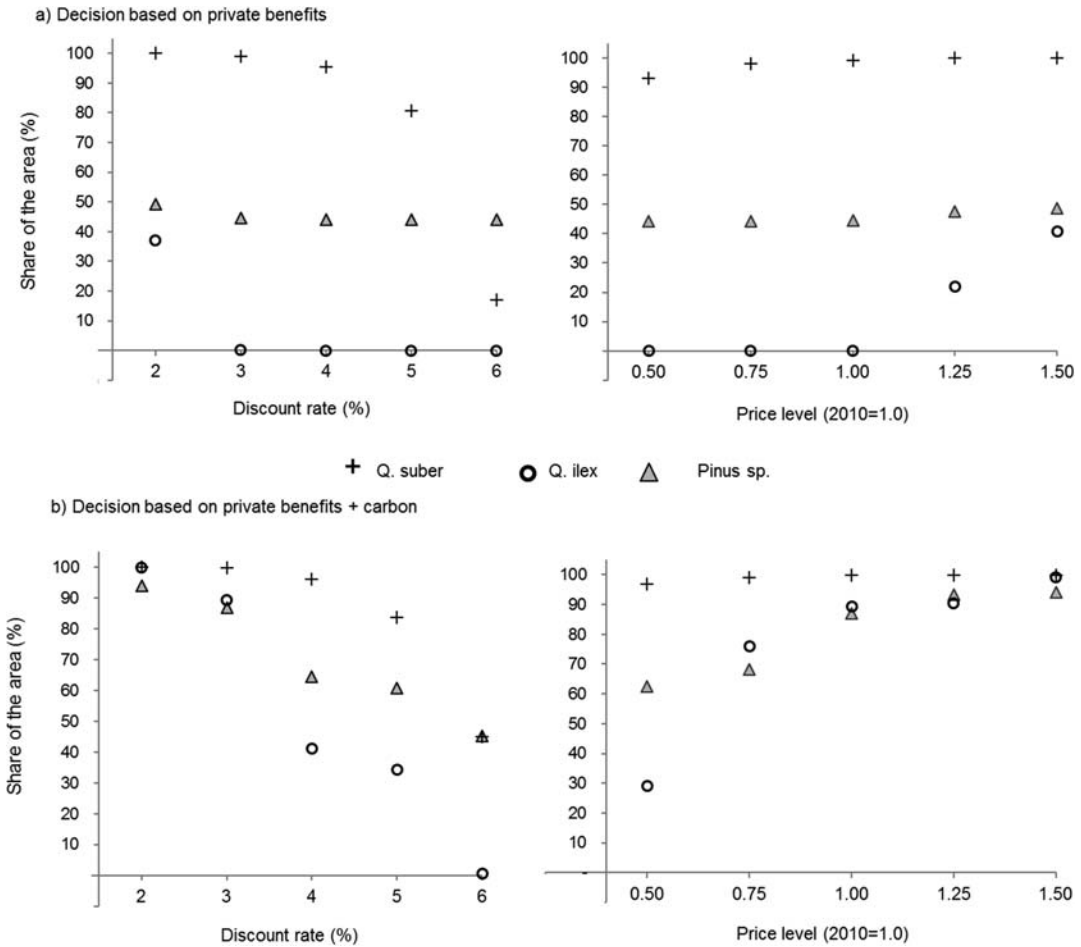
The probability of landowners investing in forest regeneration was estimated for each forest unit, discounting rate, and price scenario. The share of the land currently occupied by *Q. ilex*, *Q. suber*, and different *Pinus* sp. that is expected to remain as managed forests of the same species after current trees reach their rotation age is shown in Figure 5.

We found that *Q. suber* forests would not, in principle, face relevant forestry abandonment problems across the analyzed scenarios, except in the case of high discount rates ( $r = 6\%$ ). Meanwhile, investment in facilitating *Q. ilex* natural regeneration becomes efficient only for relatively high increases in the prices of firewood and grazing resources. The share of current pine forests that will face economic limitations to natural regeneration investment in the future is rather high for all the prices and discounting scenarios (Figure 5a).

The integration of an annual payment for additional carbon sequestration (due to tree

FIGURE 5

Share of Land Expected to Remain as Forest in Next Rotation by Species, Discount, and Price Scenarios When Only Private Benefits (*top panels*) and Private Benefits and Carbon Payments (*bottom panels*) Are Considered



growth) into the natural regeneration investment decision modeling changes the overall forestry abandonment picture, especially for pine species (Figure 5b). The internalization of carbon payments would make the regeneration investment decision an efficient option in 87% of pine forest units and in 89% of the *Q. ilex* forest units in the main scenario. Those results are, however, quite sensitive to variation in carbon prices. Indeed, under the low carbon price scenario ( $pr = 0.50$ ), natural regeneration investment would be profitable in only 39% of *Q. ilex* forest units, while still

being an efficient option for 62% of pine forest units.

One practical application of the EA accounting model is that it allows the estimation of the minimum compensation that would render natural regeneration investment an efficient option for each forest unit. This minimum compensation is estimated for only those forest units with a zero regeneration investment probability ( $\varphi = 0$ ) and is set equal to the difference between the NPV of costs and the NPV of private benefits of a new forest rotation starting today. Costs include a de-

crease in grazing resources EA value due to the 20-year grazing set-aside period (Ovando et al. 2010).

The estimated minimum compensations diverge spatially<sup>29</sup> and are relatively lower for *Q. ilex* than for *Pinus* sp., as carbon payments are not included in these estimations. For *Q. ilex* this minimum compensation averages 996 €/ha (standard deviation = 501 €/ha) for the main scenario. This figure goes up to 1,386 €/ha (standard deviation 713 €/ha) for *Pinus* sp. in the same scenario. *Q. ilex* would generate more private benefits to landowners than *Pinus* sp.; therefore, the minimum compensations are lower for this species. Nonetheless, as the carbon sequestration ability is higher for pine species than for the oak species, we find that the integration of carbon payments would increase the probability of regeneration investment in the pine forest units (Figure 5b).

#### IV. DISCUSSION AND CONCLUSIONS

The EA valuation approach developed in this study is in line with the SEEA-CF recommendations (UN et al. 2014a, 192–93), although it goes further than this system in terms of the spatial and silviculture modeling details and the variety of ESs included (forest products, grazing resources, carbon, and water). The EA valuation model can be scaled up and adapted to compile accounting structures such as the one proposed by the SEEA-CF. The estimated EA values represent the opening stock of timber, forest water, other provisioning services, and carbon.

This study extends our understanding about the role of heterogeneous spatial forest attributes and expectations on output prices and intertemporal preferences in the long-term supply of forest provisioning and regulating services. Geographic and biophysical conditions such as the slope gradient, land use distribution, forest structure, and productivity of the sites play an important role in portraying the spatial variation of EA values of provisioning services such as timber, cork, firewood, pinenuts, and grazing resources, as well

as of forest water and CO<sub>2</sub> regulating services. Our results also show potential trade-offs in the provision of those ESs, which will depend on the complex interaction of different biophysical variables (forest species distribution, soil type, and slope gradients or stand density).

Evolution of prices is a source of uncertainty in decision making regarding forest resources (Yousefpour et al. 2012). To generate plausible scenarios of potential future economic conditions, we estimate EA values for different discounting and price scenarios. The assumptions made regarding the intertemporal preferences and expected benefit levels (i.e., change in output prices while the production costs remain constant) have a large effect on the estimated EA values. Likewise, changes in economic assumptions lead to quite different representations of potential forestry abandonment, which also denote a high economic uncertainty concerning future provision of ESs.

Environmental accounting may provide useful information for examining sustainability questions, but it needs a prior understanding of underlying assumptions beyond accounting figures (Obst and Vardon 2014). In this study, we analyze these questions from the perspective of forestry abandonment and the consequent future reduction in the supply of provisioning services related to forestry activity. As indicated before, the EA estimations offered in this study outline a business-as-usual scenario. This scenario presupposes that silvopastoral farms will be managed in the future as they have been run in the past. The business-as-usual scenario embraces, on the other hand, no significant technological or commercial innovations that will alter the production frontiers of silvicultural products, water, and carbon.

This business-as-usual scenario also assumes that forest growth and yields, mortality, and wildfire risk rates are not significantly altered by changing climatic conditions. These are likely strong assumptions and include high levels of uncertainty (Keenan 2015), since changing climatic conditions may have a large effect on southern European forests (García-Ruiz et al. 2011; Reyer et al. 2014). More research is needed to analyze the potential fluctuation

<sup>29</sup> See Figure A4 in the online supplement, available at <http://le.uwpress.org>.

tuations in forest growth, yields, mortality, forest fire patterns, forest water, and adaptive forest management in response to changing climate conditions, and their impacts on ES dynamics.

Our results do not consider land use changes, since a relevant limitation to drawing reliable and policy-sound land use change options in Andalusia is that switching the use of land from forest to agriculture or urban uses involves legal restrictions under the Spanish forest laws. Our modeling framework does not consider the possibility of changing forest species as present forest rotations come to an end, either. Although this perspective is interesting, before reliable forest species substitution scenarios can be constructed more research is needed, such as establishing the distribution of the site qualities for growing different species than the ones that are currently grown on Andalusian silvopastoral farms.

The information produced in this paper can support private and social decision making and the design of payments for ecosystem services (PES) schemes. Our results show that PES for carbon might be key to encouraging the long-term conservation of multiple-use forestry. Those payments might, however, benefit those areas and species with a higher carbon sequestration potential, thereby reducing water availability, which is a limiting factor in Mediterranean areas (Allard et al. 2013).

The EA model developed in this work might be seen as a benchmark that could be extended to compile a wider range of ESs. Further research and new approaches will also be needed to integrate a larger set of ESs, such as those related to biodiversity conservation, and cultural services with connections to forest attributes, as new scientific information becomes available.

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