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Paola Ovando
Alejandro Caparrós
Luis Díaz-Balteiro
María Pasalodos
Santiago Beguería
José L. Oviedo
Gregorio Montero
Pablo Campos

INSTITUTO DE POLÍTICAS Y BIENES PÚBICOS – CSIC

Instituto de Políticas y Bienes Públicos
Consejo Superior de Investigaciones Científicas
C/ Albasanz, 26-28
28037 Madrid (España)

Tel: +34 91 6022300
Fax: +34 91 3045710

http://www.ipp.csic.es

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

Paola Ovando[1*], Alejandro Caparrós[2], Luis Diaz-Balteiro[3], María Pasalodos[4], Santiago Beguería[5], José L.Oviedo[2], Gregorio Montero[4], Pablo Campos[2]

∗ E-mail:P.Ovando-Pol@lse.ac.uk

[1] Grantham Research Institute, London School of Economics and Political Sciences (LSE), [2]Institute of Public Goods and Policies, Consejo Superior de Investigaciones Científicas (CSIC), [3] School of Forestry, Technical University of Madrid (UPM),[4] Forest Research Centre, The National Institute for Agricultural and Food Research and Technology (INIA), [5] Estación Experimental de Aula Dei, Consejo Superior de Investigaciones Científicas (CSIC)

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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 the environmental asset figures, at the same time we identify potential forestry abandonment problems when continuing with forestry activity is unprofitable for the landowner. Our results show a relevant spatial variability that depends on heterogeneous biophysical factors, such as forest species distribution and structure. We examine the likely trade-offs between forestry provisioning services, water and carbon sequestration services. The results also point towards the significant effect of economic assumptions about discount rates and prices on environmental asset values and the prediction of the forestry activity abandonment.

Keywords: ecosystem services, natural capital, forest conservation, silvicultural models, Mediterranean forest.

(JEL Q23, Q51, Q57)

1 Introduction

Recent initiatives for moving towards a green economy triggered the interest in developing environmental accounting to analyze and track the state of ecosystems and the services they provide (MA, 2005; UN et al., 2014a,b). In recent years, there has been a noticeable effort to consider explicitly the spatial configuration of the provision of various ecosystem services (ES) (see Wolff et al., 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 ES at relevant spatial scales (Bateman et al., 2013; Lawler et al., 2014). Nonetheless, and despite recent attempts at ES quantification and mapping, these have been rarely translated into the valuation of environmental assets (EA)[1], in a way that has sense for decision makers (Fenichel and Abbott, 2014).

Forest ecosystems are spatially heterogeneous areas in which the provision of ES 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 specially pertinent in forest ecosystems, as tree growth, forest depletion and forestry operations might affect the dynamics of ES supply (Biber et al., 2015; Ovando et al., 2015). This study focuses on Andalusia, a region in the south of Spain whose forests are mainly of the Mediterranean type. This type of forests form a unique mosaic of terrestrial ecosystems shaped by diverging climatic (often extreme), geomorphological and anthropogenic factors, and that are frequently characterized by their multi-functionality (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 inter-temporal provision of ES and the potential trade-offs involved.

The benefits associated with market-priced and non-market forest products, such as private amenities, biodiversity-scenic values, public recreation

[1] Here we use the term Environmental Asset as Natural Capital is considered to be a broader measure that would include the stock of all environmental assets, including ecosystems assets, mineral and energy resources (UN et al., 2014a).

and carbon sequestration have been estimated for different Mediterranean forests, mainly at the forest products case study level (Bernués et al., 2014; Campos and Caparrós, 2006; Caparrós et al., 2003, 2010; Ovando et al., 2010). Those benefits have been also 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 showed the various ways forests contribute to human and economic activities, but do not delve deep into the spatial and temporal distributions of benefits and asset values associated with the provision of forest ES.

In this study we develop an environmental asset 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 to develop EA accounts for individual natural resources such as timber or water. Our approach, in contrast to SEEA-CF, considers the forest as a functional unit that supplies multiple products, entailing trade-offs amongst the provision functions of 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 two regulating services, comprising water flow regulation and a climate regulating service through carbon dioxide (CO_2) sequestration (carbon hereinafter). The model estimates the EA values derived from the provision of the aforesaid ES in a group of 567 private silvopastoral farms that are distributed across Andalusia (see Fig.A.1 in the Appendix). The application integrates spatially-explicit biophysical and economic data at farm level for the main forest species in this region: *Quercus ilex, Quercus suber, Pinus pinea, Pinus halepensis, Pinus pinaster* and *Eucalyptus globulus* and *E. camadulensis* (jointly referred to as *Eucalyptus sp.*), as well as for treeless shrub-land 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 assumption on price levels and discounting rates, as well as, in accordance with the expected forest management 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 precipitations received.

Our results reveal a noticeable spatial variability in EA values, and point towards potential trade-offs associated to silvopastoral provisioning 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 as current forest rotations come to and end. Likewise, the model allows for the exploration of the effect of annual payments for carbon sequestration 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.

2 Materials and methods

2.1 Silvopastoral farms case studies and area of study

Andalusia is a very diverse region with altitudes ranging from sea level up to 3,400 meters and from the rainiest point in the Iberian Peninsula to the desert of Almeria. This region covers 84,023 km^2, which is similar size to Austria. About 53% of this territory is covered by Mediterranean forests and grasslands. These are complex ecosystems in which tree, shrub and herbaceous vegetations have been traditionally managed jointly to obtain raw materials such as cork, timber, firewood, pinenuts, hunting and grazing resources, which ascribes them as silvopastoral systems[2]. Private ownership

[2]Those systems comprise a deliberate growing of woody perennials on the same unit of land as livestock in interacting combinations for multiple products from the same management unit (Nair, 1993).

dominates (73%) the area covered by silvopastoral systems in Andalusia (Diaz-Balteiro et al., 2015).

The 567 silvopastoral farms included in this study are distributed across Andalusia, and were taken from a survey to 765 forest owners, whose properties where randomly selected in this region (see Oviedo et al. (2015) and Appendix). They jointly occupy an area of 2,975 km^2 (9.3% of total private farms in Andalusia), and have an average size of 525 ha (SD$\pm$ 849 ha). The seven forest species included in this study represent 67% of the farms area, while shrubs and grasslands make up a share of 19%. Other forest species (4%) and crops (10%) occupy the remaining area.

2.2 Pricing ecosystem services and environmental assets

From an environmental accounting standpoint, EA is defined as the naturally occurring biotic (whether natural, semi-natural or modified) and non-biotic components of the Earth that provides a flow of ES; which in combination with labor and manufactured assets contribute to generate products used in human and economic activities (UN et al., 2014a,b; Obst and Vardon, 2014). Markets for environmental assets and the services they provide are often incomplete or missing, specially for stocks and goods with weak exclusion (Fenichel and Abbott, 2014), such as public products. The EA and ES values are 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 ES 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 associated ES. Hedonic pricing models might be useful to estimate the land asset value associated with different commercial forest benefits (Zhang et al., 2013) when statistical information on forest properties' sales and their attributes is available. This is not the case of land prices statistics for forest properties in Andalusia (Campos et al., 2009), which demand for alternative asset valuation methods, as we detail later on. Notwithstanding, that the hedonic price approach would not be able to capture public non-market values as the market does not assign the property rights on these products to landowners.

Land leasing and forestry products prices embed the value of provisioning services such as grazing resources or timber. Likewise, 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. The difference between market prices and the unit labor and manufactured input and full capital costs would render the unit natural resource rent (UN et al., 2014a,b), and this unit price is used to value the provisioning services considered in this study.

For those ES 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 for forestry carbon in Andalusia, and their 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 environmental asset price of water estimated by (Berbel and Mesa, 2007, p. 141) using an hedonic price model for irrigated agricultural lands in Andalusia. This model uses land price statistics that in Andalusia are only available for agricultural lands (CAP, 2011), and not for forest lands. The environmental asset price of water (P^w), updated to year 2010 attains a value of 4.04 euro/m^3, and the water ES prices (p_w) is estimated using real discounting rates (r) ranging from 2% to 6%: $p^w = P^w \cdot r$.

Output prices and forestry operation costs included in this study do not account for subsidies and taxes on production. We assume that prices for output and forestry operations remain constant, as well as that the returns to scale are constant. This study consider the most common forestry practices in Andalusia (Montero et al., 2015), assuming the continuation of the business-as-usual (BAU) scenario. The baseline prices correspond to those observed in Andalusia and in the EU ETS markets for forest provisioning services and carbon, respectively in year 2010 (see Appendix for details).

EA values are quantified as the discounted net present value (NPV) of the stream of ES (estimated as a resource rent) that a forest ecosystem is expected to yield in the future, in an infinite time horizon. The NPV approach is the standard rule for pricing assets in a deterministic case (Dixit and Pindyck, 1994) and follows the SEEA-CF recommendations (UN et al., 2014a). The SEEA-CF recommends to estimate EA values by capitalizing the flow of resource rents over the life of assets. This resource rent represent the economic rent accrued in relation to environmental assets, and should ideally account for the remuneration to those assets as production factor and their depletion (UN et al., 2014a).

The ES monetary value we estimate represents the returns to EA 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 benefits; while capital costs embrace a normal return to manufactured assets used in the production process[3] (Ovando et al., 2015). Our model implicitly computes for both the potential EA depletion and improvement by anticipating existing tree inventory withdrawals due to forest fires, natural mortality or management and entries due to tree growth and recruitment (see section 2.3).

Provisioning and regulating services depending on tree growth

For multi-periodical products such as timber, cork and firewood, the expected pattern of ES is quantified considering a simplified approach[4], based

[3]In our application this normal return to manufactured assets equals to 3% for the main scenario, and varies within the discount rate applied

[4]A more comprehensive approach for estimating ES, as an environmental income (Ovando et al., 2015), would consider natural timber/cork/firewood growth as an output

on the value of expected extractions minus forestry operating and full capital costs (as it was detailed earlier).

We extend the asset valuation approach applied by Caparrós et al. (2003) to price timber stock, to multi-product outputs such as cork, pinenuts or firewood and carbon sequestration due to tree growth[5]. EA accounts for both the present forest rotation (EA_{T1}) and for the expected ES after this rotation. Our asset valuation approach could be applied to both even and uneven-aged forest, regardless of the initial forest structure and species distribution, as is detailed in Section 2.3.

The EA_{T1} is estimated as:

$$\begin{gathered} EA_{T1} = p' \cdot Q \\ p'_p = \left(p_p^1, p_p^2, ...p_p^d, ...p_p^n, \right) \\ \text{Being: } p_p^d = \textstyle\sum_{j=s}^{T} (p_f^d - p_m^d) \cdot \gamma_{dt} \cdot \beta_{dt} \cdot \delta^{(t-d)} \text{ for each } d=\{1,2,...,T\}. \\ \gamma_{dt} = \tfrac{q_t}{q_d}. \end{gathered} \tag{1}$$

Where p'_p is a vector of unit resource rent (euro per cubic meter or per metric ton). This price vector includes for its T rows the standing price (p_f^d) of the product and the cost of forestry treatments (p_m^d)[6], including the opportunity cost of manufactured capital. β_{dt} represents the conditional probability that a tree of an age d is logged at any of the t age classes to be reached ($d \leq t$) (see Section 2.3 for details). Q is a vector that records the existing stock of forestry products or carbon for each age class at the initial period (2010). γ_{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 (qt). Finally, δ represents the discount function:

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 and Caparrós, 2006; Ovando et al., 2015).

[5]Carbon sequestration due to tree growth is estimated as a function of tree diameter (Montero et al., 2006).

[6]The forestry treatments refer to those operations scheduled for the years that are left before reaching the rotation of a particular forest species.

$\delta = (1+r)^{-1}$.

The provision of ES after the present rotation depends on the probability of forest regeneration investment (φ), which equals 1 if current forestry activity continues in the future (assuming the same species and silvicultural model at each forest unit[7], and equals 0 if this activity is abandoned. We assume that forestry activity will continue if the NPV of the expected benefits of the new rotation surpasses the NPV of its costs.

The EA in an infinite time horizon is then estimated as:

$$EA = EA_{T1} + \left\{ \begin{array}{l} \varphi \cdot (\delta^{T+1-s} \cdot (1-\delta^{T})^{-1} \cdot EA_{T2}) + \\ (1-\varphi) \cdot (\delta^{T+1-s} \cdot (1-\delta)^{-1} \cdot y_t^l). \end{array} \right. \tag{2}$$

Where s is the starting valuation period and T the rotation age. EA_{T2} represents the EA 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 EA_{T1} using Eq.(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. y_t^l represents the annual ES of the alternative land use l in the event of forestry abandonment.

The model considers that forestry abandonment would lead to shrub encroachment and would change the present distribution of forest species[8] Grazing resources, carbon sequestration and water will be the only ES delivered by this land use.

The EA associated to the provision of silvopastoral products would take a zero value, in the event the NPV of net benefits associated to 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, whit no return to the environmental asset. The ES related to carbon are estimated each period as the difference between gross CO_2 sequestration and

[7]A forest unit is is defined by homogeneous forest species and ages classes distribution, slope gradient, silvicultural model and municipality.

[8]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).

release, and as we assume that carbon sequestration does not involve any manufactured assets or labor, a negative EA would indicate loss in carbon environmental stock value.

Grazing resources

Grazing resources include acorns (only for *Quercus ilex*) and grass (swards, browses and fruits) produced in forest, shrub 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 number of forage units obtained by dominant vegetation (e) and province (j) in Andalusia. Those prices and quantities are taken from a survey of 765 agroforestry farm owners in this region (Oviedo et al., 2015) that includes the sample of 567 farms considered in this study.

We estimate the EA associated with grazing resources (EA_g^i) at farm level using the average land leasing price (p_g^{ej}) per forage unit of a dominant vegetation[9] 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 farm level over the forest rotation and would only change in the event of forestry activity abandonment in a land unit i:

$$\begin{gathered} EA_g^i = \sum_{t=s}^{T} \delta^t \cdot Y^{ek} + \varphi \cdot \frac{\delta^{S-s}}{1-\delta} \cdot Y^{ek} + \\ (1-\varphi) \cdot \left(\delta^{S-s} \cdot \textstyle\sum_{t=S-s}^{U-s} \left((1-\alpha_t \cdot) \, Y^{ek} - \alpha_t \cdot Y^{lk} \right) + \frac{\delta^{U-s}}{1-\delta} \cdot Y^{lk} \right). \\ \text{where:} \\ \mathrm{S = T+1; U = S+1+\tau; \alpha_t = t/U.} \\ \mathrm{Y^{ek} = \omega^k \cdot \left(p_g^{ej} \cdot q_g^k - cm_g^j \right); Y^{lk} = \omega^k \left(p_g^{lj} \cdot q_g^{lj} - cm_g^j \right) ..} \end{gathered} \tag{3}$$

The first term of Eq.(3) refers to the asset value of grazing resources for present inventory until the trees reach their rotation age. The second term of

[9] The classification of farms for estimating grazing EA considers the vegetation that occupies the largest part of the farm.

Eq.(3) represents the grazing resources value for an infinite sequence of forest rotations of the same species and silvicultural model in a land 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 equal to those observed in farms dominated by shrub at each one of the Andalusian provinces. The third term Eq.(3) represents the transition from q_g^k to q_g^{lj} considering a period (τ) of 50 years after forestry abandonment, and that this transition is linear. Finally, we consider that after the period τ, q_g^{lj} and p_g^{lj} would remain constant over time.

EA_g^i estimation additionally considers, as subtrahend, the operating and capital manufactured costs involved in the supply of grazing resources (cm_g^j), as well as, an additional correction factor defined by ω^k. This factor indicates the probability of the farm k being used for livestock grazing in the future (hence, $\leq 0\ \omega^k \leq 1$)[10].

Carbon sequestration in shrub biomass

Carbon sequestration in shrub biomass is estimated using Pasalodos et al. (2015) functions that relate shrub biomass growth to the fraction of their canopy cover and the average height of shrub formations. Net carbon sequestration by shrub growth further considers potential CO_2 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 Diaz-Balteiro et al. (2015), for both shrub formations under the tree layer and treeless shrub-lands. 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

[10] In the case that grazing resources are currently consumed by livestock in a farm, ω^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 (more details in the Appendix).

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 τ period after forestry abandonment (see Appendix). We assume a linear transition for the shrub carbon growth from the present situation to the one expected 50 years after forestry abandonment.

The estimation of the EA associated follows Eq.(3), whereas we replace the price variables of this equation by a single carbon price, (p_c). Likewise, we substitute the Eq.(3) quantity variables, by (q_c^k) and (q_c^l), which represent the annual net carbon sequestration in shrub for the forest regeneration scenarios and forestry abandonment scenario, respectively. After the transition period τ 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 (HRU) 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 the water output *forest water*. 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 the *green water*. A fraction of these flows can be regulated by the water agency (collectible surplus of forest water) and later be sold to the users. The forest water with economic value is thus made up of the superficial water run-off that reaches a reservoir in Andalusia and is sold to the final users (Beguería et al., 2015).

Estimations of the forest water balance depend, amongst other factors, on 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 (stationary conditions), and would only change 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 using an equation similar to (3), but replacing the price and quantity variables by a single and constant water environmental price (p_w) and constant quantities of economic water flows for the regeneration (q_w^k) and forestry abandonment scenarios (q_w^l), respectively.

2.3 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 (see Supplementary on-line text for details). The information provided by the silvicultural models allows the estimation of individual tree survival functions. These functions specify the survival probability (π_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 the scheduled tree thinning and final logging (α^{ij})[11], the natural tree mortality (θ^{ij}), and also by the forest fire risk (ρ^{ij}). The variables: α_t^{ij}, ρ_t^{ij} and θ_t^{ij} represent the annual probabilities that trees will be felled, burnt or die, respectively:

$$\pi_t^{ij} = \begin{cases} 1 - \alpha_t^{ij} - \rho_t^{ij} - \theta_t^{ij}, & \text{if t=1} \\ \pi_{t-1}^{ij} - \alpha_t^{ij} - \rho_t^{ij} - \theta_t^{ij}, & \text{if t >1, where t=}\{1,2,...T^{ij}\} \end{cases} \tag{4}$$

The individual tree felling probability at each period t is quantified as

[11] Note that the conditional probability of tree logging β_{dt} of Eq.(1) is estimated as: $\beta_{dt} = \alpha_{dt} \cdot \pi_{dt}$.

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 tree density according to the silviculture model: ($\alpha_t^{ij} = h_t/N_1$). The mortality ratios are estimated as logarithmic functions of tree age, while the future risk of forest fire depends upon the average historical forest fire ratios by species and province estimated for the period 1987-2006 (Diaz-Balteiro et al., 2015). The rotation age by species and silvicultures is exogenously defined by Montero et al. (2015)'s models (see Appendix).

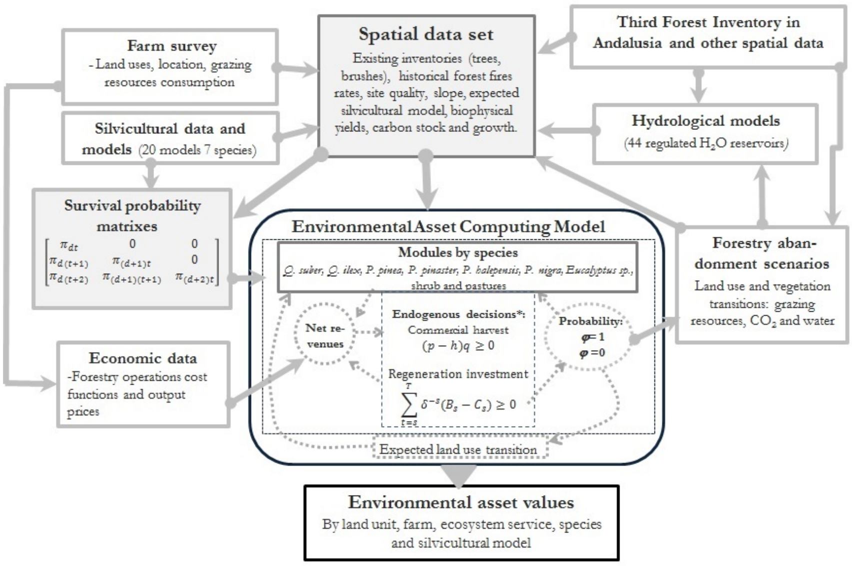
A survival probability matrix has a dimension T^{ij}xT^{ij} and computes the conditional probability (π_{dt}^{ij}) that a tree of an age d is alive at each one of the tree ages t that are to be reached ($d < t$): $\pi_{dt}^{ij} = \Pr(d/t) = \pi_t^{ij}/\pi_d^{ij}$.

The EA model includes 152 different probability matrices, one for each of the 19 silviculture model and each one of the 8 provinces of Andalusia. Those 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 classes 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 (SFM) using the latest National Forest Inventory in Andalusia (MARM, 2013) and digital elevation maps (Diaz-Balteiro et al., 2015). 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 SFM 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, shrubs, grasslands and other land uses is specific to the farm. A homogeneous forest unit is defined in terms of species composition, density, age classes 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. Fig.1 shows a scheme of the interrelated components of this computing model and the sources of biophysical and economic data.

Figure 1: Components of the environmental assets valuation model



3 Results

3.1 Environmental asset for provisioning and regulating services

Average values at farm and vegetation levels

Table 1 shows the estimated EA (in euros per hectare) by forest species, ecosystem service and silvicultural model for the baseline scenario: 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 on output prices from ± 25% and ± 50%, while production cost remain constant.

Table 1: Average environmental asset value by ecosystem service, vegetations and silvicultural model(1)

Species	Environmental asset value by species and silvicultural model (euro/ha, year 2010)(2)					Standard deviation
	S1	S2	S3	S4	Total Mean	(SD)
Quercus ilex	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
Quercus suber	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
Pinus pinea	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
Pinus halepensis	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
Pinus nigra	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
Pinus pinaster	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
Eucalyptus 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(3)					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
Carbon trees					635	593
Carbon shrub					452	480
Water					654	779

Notes:(1) EA results are provided for the main scenario (discount rate 3% and average prices of 2010), and corresponds to weighted average values according to the area of each forest unit.
(2) $S1_j$ to $S4_j$ are referred to the four potential silvicultural models applied to each one of the j species.
(3) Other vegetation includes grazing resources in treeless shrubs, grassland, crops and other forests species.

The aggregated EA value for provisioning services, water and carbon amounts to 2,813 euros per hectare (SD ± 2,383 euro/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 shrubs net growth) and water 23% of this value. Timber, pinenuts and firewood slightly account for the remaining 1%.

The EA values for timber and pinenuts display a higher variability amongst the farms than any other ecosystem service. Variability in cork values across farms is also relevant, as it is small in terms of grazing resources. The relative homogeneity in EA_g values is due to the fact that available data on grazing leasing prices only differ by dominant vegetation and province, without connection to other spatial factors. The variability of the EA value associated with the provision of water is also small for the group of silvopastoral farms, but higher for specific vegetations in particular for the category other vegetations, which mainly includes treeless shrubs and grass lands. Silvopastoral provisioning services are relevant land price factors, while land price would in principle not be affected by forest water (as landowners do not get any payment for this ES). We estimate that the aggregated value of the provisioning services derived from silvopastoral activity would account for 25% of the average land price for non-irrigated pastures (4.294 euro/ha) in 2010 in Andalusia (CAP, 2011)[12].

The 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 non-market private amenities (Campos et al., 2009) that affect forestland prices, but due to data limitations those are not considered in this paper. *Quercus suber* is the species that offers the highest aggregated EA value, with cork adding almost the half of this figure. The contribution of timber EA is negligible amongst *Eucalyptus* and pine species, after covering labor and manufactured costs associated with timber

[12] There is no statistical data on forest and shrub-land prices in Andalusia, and the price of non-irrigated pastures is the only proxy land price statistic available for silvopastoral farms.

production. Grazing resources and, particularly, carbon explain the largest part of the aggregated EA values for pine species and *Q. ilex*. The EA value of water is significantly smaller in forest units covered by pine species (except for *Pinus nigra*, which is primarily located in mountainous areas) than in those covered by oak trees.

The estimated carbon EA 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 vegetations include those resources obtained in shrub and grass lands, other forests and crops. Carbon sequestration in those other vegetations considers CO_2 fixation due to shrub biomass growth, and CO_2 release due to shrubs clearing and forest fires affecting this vegetation. As expected, our results confirm that carbon sequestration potential is bigger in forested areas than in treeless shrub-lands. They also show that the value of potential carbon storage in the tree stratum is higher than the storage in the shrub stratum (Table 1).

Spatial distribution of EA values

Fig.2 shows the spatial distribution of EA values for aggregated silvopastoral provisioning services (EAPr), CO_2 regulating service (EAC), 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 the forest or the quality of the sites for growing cork and timber, as is detailed later.

Our results indicate higher EAPr values in Western Andalusia and in the areas with a relevant extent of *Quercus suber* woodlands. The lowest EAC values are observed in Eastern Andalusia, where shrub-land is the predominant vegetation. EAC values are lower for those areas where *Quercus suber* is the dominant species and higher for the areas dominated by pine species. Those areas dominated by pines depict, however, lower EAPr values. Those results suggest a trade-off between carbon sequestration and forestry provisioning services for cork oaks and pine species in Andalusia.

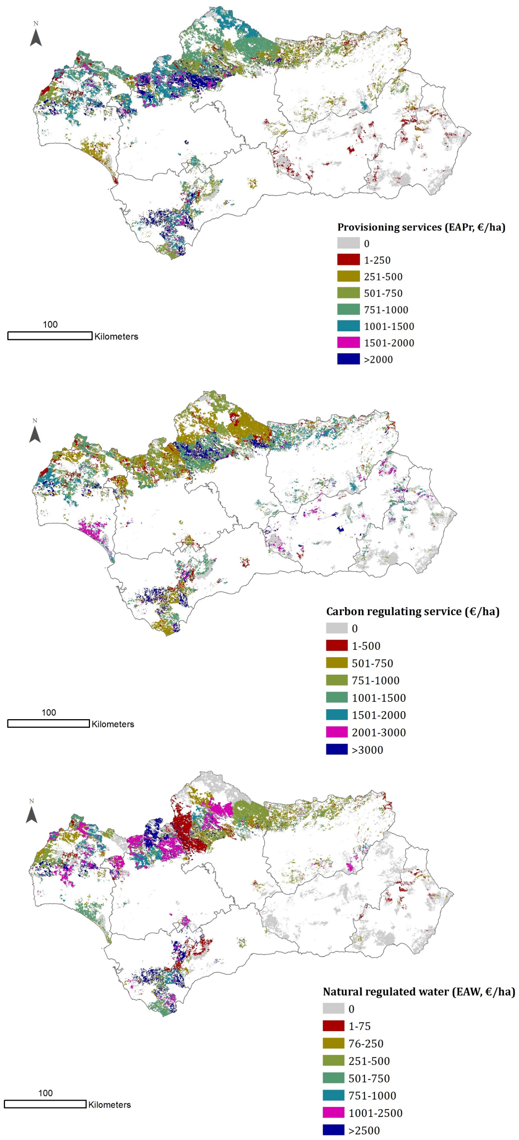
EAW values also depict a relevant spatial variability, with lower values for provinces of Eastern Andalusia. The mountainous areas of Seville and Cádiz provinces show higher EAW values. We also observe that areas with a higher EAC value depict, at the same time, moderate to low EAW values, which seems to be related to a higher evapotranspiration rate in forests with higher CO_2 sequestration potential. On the other hand, medium-to-high EAW values tend to coincide with medium-to-high EAPr ones, which is likely associated with *Quercus suber* distribution[13].

The variables that operate in the EA valuation model are diverse, and depend on multiple interactions between spatial and non-spatial biophysical and economic factors. To examine the magnitude of the effect of spatial 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 and forest unit levels. Table 2 shows the results of these functions for the sample of farms[14], which examines the effect of the slope gradient, the density of the forest and the share of oak woodlands, pine species and treeless shrubs and grass lands on total EA values.

The EAPr values at the farm level are increased within the share of oak (*Quercus suber* and *Quercus ilex*) and pine species, while the share of treeless shrubs and grass lands reduces the EA value of provisioning services. The effect of the slope is not significant at the farm level, while the basal area (as a proxy indicator of tree density) negatively affects the EAPr values. On the other hand, EAC values decrease within the share of oak species and increase within the share of pines at farm level. This latter result confirms the trade-off between EAPr and EAC regarding oak species.

[13] *Quercus suber* is a species better suited for more humid areas.
[14] See Appendix for details

Figure 2: Distribution of provisioning services, carbon and water environmental asset values in the municipalities where the studied farms are located



Note: Data for main scenario (r=3%, p=1.0).

Table 2: Environmental asset functions for the farm's sample for the main scenario

Variables(1)	Main scenario (r=3% , p=1.0)(2)								
	EAPr			EAC			EAW		
	Coefficient		SE(3)	Coefficient		SE	Coefficient		SE
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
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^2	0.68			0.78			0.49		

***p<0.01, **p<0.05, *p<0.10. Number of observations 567.
Notes: (1) The functions estimate the EA value in euros/ha. Slope is estimated as a percentage value, BA, refers to the initial basal area (in m^2/ha). The share (S) variables indicate the proportion of the farm area occupied by different land use classes (in %): SQI: Share of *Quercus ilex*; SQS: Share of *Quercus suber*; SPP: Share of pine species; SSP: Share of shrub and grass lands. (2) EAPr: environmental asset value of silvopastoral provisioning services, EAC: environmental asset value for carbon sequestration, EAW: environmental asset value for water provisioning service.(3) Robust standard error.

In the case of EAW values, we observe that the share of oak species negatively affects the estimated EAW values, being the effect of pine species not significant. The main reason for such results is that forested areas (in contrast to treeless shrubs and grasslands) have higher evapotranspiration rates, which reduces the forest water flows that can be regulated at each HRU. The size of the farm has no significant effect on the EA values considered, and this is related to the assumption of constant returns to scale, and therefore it is not consider as a variable in the regression model.

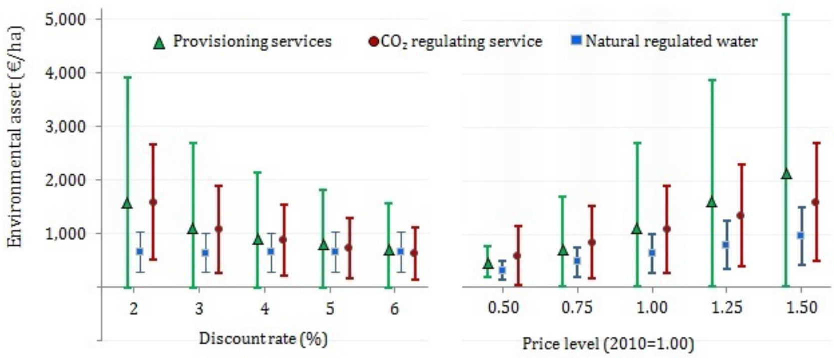
Sensitivity to discount rates and output prices

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 central discount rate (3%) (Fig.3). Similarly, the average EA value for carbon ranges from -41% to 46% with respect to the main scenario. Higher discount rates have a greater effect on the EAPr figures than they do on the estimated EAC ones; while they do not affect EAW values, as we apply a single environmental asset price for water that does not depend on the discount rate.

The sensitivity analysis includes variations in output prices with respect to the 2010 prices for silvopastoral products, carbon and water, while forestry costs remain constant. 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 raises 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 central price 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 with a zero water economic value (Fig.2), which makes the EAW less sensitive to changes in output prices.

Figure 3: Sensitivity of the environmental asset value for provisioning services and carbon to discount rates and changes on output prices



Note: Lines show the standard deviation (SD) of the mean. The central price level for different discount rates is 1.0, whilst the discount rate for different price levels is 3%.

3.2 The forest conservation question

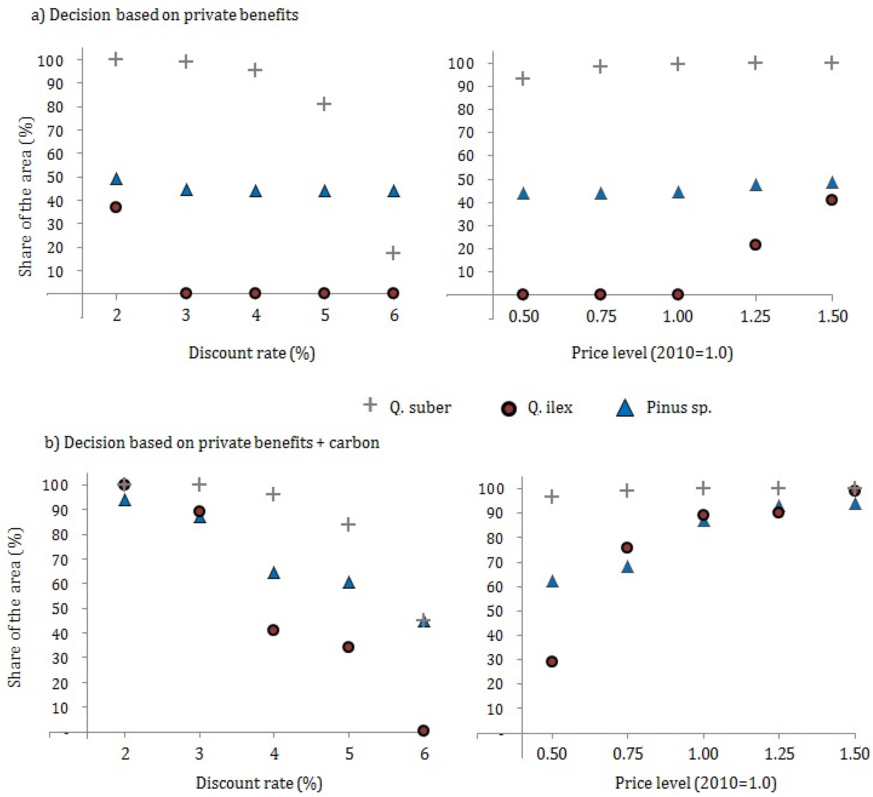
The forest regeneration investment decision depends on the NPV of expected private benefits from the new rotation exceeding its costs. Those private benefits are related exclusively to silvopastoral provisioning services. We acknowledge that contrasting the NPVs of future benefits and costs to forecast 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 getting any products of commercial interest, and the opportunity cost of land and other non-monetary variables that may affect landowners' preferences (Campos et al., 2009; Diaz-Balteiro et al., 2009). Nonetheless, and 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 and price scenario. The share of the land currently occupied by *Quercus ilex, Quercus suber* and different pine species that is expected to remain as managed forests of the same species after current trees reach their rotation age is shown in Fig.4.

We found that *Quercus 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 *Quercus 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 (Fig.4.a).

The integration of an annual payment for additional carbon sequestration (due to tree growth) into the natural regeneration investment decision modeling, changes the overall forestry abandonment picture, especially for pine species (Fig.4.b). 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 *Quercus ilex* ones in the main scenario. Those results are, however, quite sensitive to variation in carbon prices. Indeed, under the low carbon price scenario (p=0.50) natural regeneration

Figure 4: Share of the land that is expected to remain as forest the next rotation by species, discount and price scenarios when only private benefits and private benefits and carbon payments are considered



investment would only be profitable in 39% of *Quercus 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 at each forest unit. This minimum compensation is only estimated for those forest units with a zero regeneration investment probability (φ =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 decrease in grazing resources EA value due to a 20-year grazing set-aside period (Ovando et al., 2010). For *Quercus ilex* this minimum compensation averages 996 euros per hectare (SD± 501 euro/ha) for the main scenario. The minimum compensation to pine species is slightly higher (1,386 euros per hectare, SD± 713 euro/ha). The estimated minimum compensations diverge spatially (see Fig.A.2 in the Appendix).

4 Discussion and conclusions

The EA valuation approach developed in this study is in line with the SEEA-CF recommendations (UN et al., 2014a, 192-193), although it goes further than this system in terms of the spatial and silviculture modeling details and the variety of ES included (forestry 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 inter-temporal 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 environmental asset values for provisioning services, such as timber, cork, firewood, pinenuts, grazing resources, as well as for water and CO_2 regulating service. Our results also point towards potential trade-offs in the provision of those ES, which will depend on the complex interaction of different biophysical variables (forest species distribution, soil type and slope gradients or tree 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 inter-temporal preferences and expected price levels have a large effect on the estimated EA values. Likewise, changes on economic assumptions lead to quite different representation of potential forestry abandonment, which also denotes a high economic uncertainty concerning future provision of ES. 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 BAU 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 BAU scenario also assumes that forest growth and yields, mortality and fire 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 in Southern European forests (García-Ruiz et al., 2011; Reyer et al., 2014). More research is needed to analyze the potential fluctuations in forest growth, yields, mortality, forest fire patterns and adaptive forest management in response to changing climate condition, and their impacts on ES dynamic.

The information produced in this research 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 areas 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 ES. Further research and new approaches will be also needed to integrate a larger set of ES, such as those related with biodiversity conservation and cultural services connected with to forest attributes, as new scientific information becomes available.

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Appendix

1 Supplementary material

1.1 Spatial distribution of silvopastoral farms

The spatial distribution of land uses and vegetations within Andalusia is shown in Fig.A.1. Forested areas account for close to 2.9 million ha, of which 56% is dominated by broad-leaves and the remaining 44% by conifers and mixed forest. *Quercus ilex* and *Quercus suber* are the main broad-leaf species, which jointly account for 43% of the area that is covered by trees. *Pinus pinea, Pinus halepensis, Pinus pinaster* and *Pinus nigra* together embrace 43% of this total area, while *Eucalyptus* sp. cover 4% (MARM, 2013). Treeless scrub-lands occupy some 1.6 million ha and grasslands 0.3 million ha (CMA, 2010).

Fig.A. 1: Silvopastoral farms distribution in Andalusia

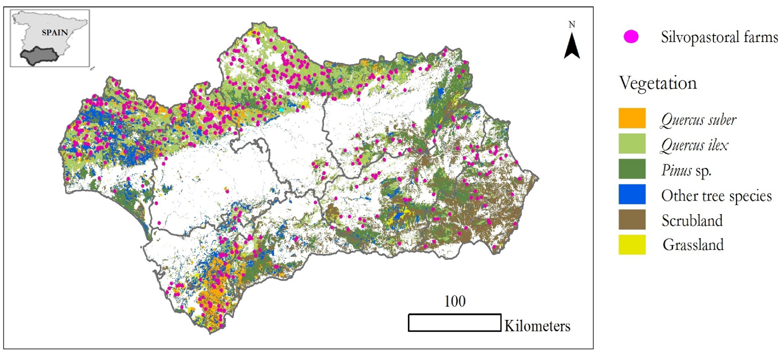


Fig.A.1 also shows the distribution of the 567 silvopastoral farm case studies. Those case studies are taken from a larger forest farms sample (765 farms) that were randomly selected in Andalusia (Oviedo et al., 2015), and embrace those farms that have at least one of the seven forest species considered in this study. There are five main diverging spatial conditions across farms: land use, vegetation distribution, slope, tree density and age class distribution. Land use distribution varies amongst the Andalusian provinces (Table A.1). The share of conifer forests is higher in the majority of Eastern Andalusia's provinces (Almería, Jaén and Granada), whilst the oak (*Quercus*) woodlands share is higher in Western Andalusia (Cádiz, Córdoba, Huelva and Seville) and Málaga provinces. Note that the share of conifers in private farms is lower than the regional distribution of these species, since a relevant part of conifer forests are located in publicly owned lands.

Table A. 1: Main attributes and land use distribution of the silvopastoral farms sample by province

Class	N(1)	Average spatial variables(2)				Land uses distribution (%)							
		Area (ha)	Slope (%)(3)	BA (m^3 /ha)		Q.i-lex	Q.su-ber	Pin-us sp.	Euc. sp.	Oth. for	Sh.& GR(4)	Cro-ps	Oth. uses
				FOR	TOT								
Almería	33	357	30	27	12	12.1	-	21.1	-	19.1	39.5	7.6	0.5
Cádiz	55	469	18	31	14	2.5	42.6	0.7	-	6.9	32.1	14.4	0.8
Córdoba	152	440	23	20	17	67.3	6.7	8.3	-	0.5	10.1	7.1	0.1
Granada	39	577	32	45	25	36.9	-	14.5	-	6.6	16.3	24.8	0.8
Jaén	113	445	25	19	16	53.3	9.6	0.9	12.4	0.0	21.0	2.6	0.2
Huelva	54	487	35	32	21	52.7	-	9.6	-	6.7	22.1	8.0	0.7
Málaga	19	365	39	31	17	24.6	17.2	1.5	-	0.0	46.2	6.6	3.9
Seville	102	852	26	14	11	57.2	9.0	0.1	0.5	9.2	11.4	12.6	0.0
Andalusia	567	525	26	24	16	49.6	9.9	4.9	2.2	4.3	18.6	10.1	0.4

Notes:(1) N is the number of farms per province. (2) FOR refers to the basal area (BA) of the forest, whilst TOT to the total basal area for the total farm area. (3) The slope and basal area have been estimated as a weighted average value according to the relative area that each species and silvicultural model occupies in the farm. (4) Sh & Gr: treeless shrubs and grass lands.

Basal area is considered an indicator of the tree density. Most of the farms (53%) have a moderate tree density, that is to say a basal area between 15 to 30 m^2 per hectare, 15% of the farms have a basal area less than 15 m^2/ha and 22% higher than 30 m^2/ha. Similarly, most of the farms (48%) have a moderate slope (10% to 30%), 37% of them are located in areas with sharp slopes (> 30%), and only 15% of the farms are located in relatively flat areas (< 10%) (Table A.1). Farm size classes are more variable across the farms. Close to one third (34%) of farms have an area smaller than 100 ha. About 25% of the farms have an area between 100 and 250 ha, 20% an area between 250 and 500 ha, and 21% areas higher than 500 ha.

1.2 Silvicultural models, growth and yield functions

The EA computing model considers 19 different silvicultural schemes applied to seven different forest species. These silvicultural models were developed by Montero et al. (2015) and vary depending on the forest structure (even or uneven-aged), the quality of the site, the soil type and the forest density. Table A.2 shows the main features of the silvicultural models applied, indicating the rotation length defined by Montero et al. (2015) and the diametric class at which different forestry treatments are to be applied. These rotation ages might be overestimated for species and provinces with a higher historical incidence of forest fires. In such cases the rotation ages are reappraised taking into account the individual trees' survival probabilities,and these adjusted rotation ages are further reapplied in order to estimate the survival functions and other EA-related estimates

Table A. 2: Silvicultural model main forestry operation by diametric class

Species[(1)]	Model	Rotation age (yr)	Definition[(2)]	Diametric class for the treatment		
				Thinning	Pruning	Shrub clearing
Eucalyptus	1	12	FL-AS	-	-	every 6 yr.
sp.	2	16	MA-AS	-	-	every 6 yr.
P.halepensis	1	80	EA-HMS	5,10,15,20	-	10,20
	2	151	EA-MLS	5,10,20,25	-	10,20
P.nigra	1	84	EA-MLS	5,10,15,20	-	10,20
	2	105	EA-HMS	10,15,20,25	-	10,20
P.pinaster	1	78	EA-MLS-HD	15,25	-	10,20
	2	86	EA-MLS-MD	15, 25	-	10,20
	3	74	UEA-AS	15, 20, 25	-	10,20
P.pinea	1	126	EA-MLS-FL	10,15,20,25		10,20
	2	107	EA-HMS-FL	10,15,20,25		10,20
	3	117	EA-HMS-MA	10,15,20,25		10,20
	4	103	EA-MLS-MA	10,15,20,25		10,20
Q.ilex	1	138	EA-MLS	5,10,15,20,30	10-65	5,10
	2	190	EA-HMS	10,15,20,30	10-75	5,10
	3	42	CP-AS	20,25,35	15-75	5,10
	4	164	UEA-AS	5,10,15,20,30	10-60	5,10
Q.suber	1	141	EA-AS	5-45	10,15	5,10
	2	103	UEA-AS	5-45	10,15	5,10

Source: *Own elaboration* based on Montero et al. (2015).
Notes: [(1)] P (*Pinus*), Q. (*Quercus*). [(2)] AS: all sites; CP: coppice forest; EA: even-aged forest; FL: flat lands; HD: High density, HMS: high-medium site quality; MA: Mountain areas; MD: Medium density; MLS: medium-low site quality; UEA: uneven-aged forest.

Table A.3 shows the parameters for growth function in diameter and in volume associated with each species and silvicultural model, which has been estimated using Montero et al. (2015) primary data and models. Carbon stock per individual tree depends on the volume of trees, according to a ratio that relates tree volume to carbon content in above and below-ground tree biomass, which are estimated using Montero et al. (2006) functions.

Table A. 3: Diameter and volume growth function and carbon stock by species and silvicultural model

Species$^{(1)}$	Model	Diameter growth $(y)^{(1)}$ $y = \beta_1 x + \beta_2 x^2$		Volume growth $(v)^{(1}$ $v = \beta_0 + \beta_1 x + \beta_2 x^2 + \beta_3 x^3$			Carbon stock
		β_1	β_2	β_1	β_2	β_3	t CO_2/m^3
Eucaly-	1	1.08	$-6.4*10^{-3}$	$4.38*10^{-4}$	$4.99*10^{-4}$	$5.00*10^{-6}$	2.49
ptus sp.	2	$9.61*10^{-1}$	$-4.5*10^{-3}$	$4.38*10^{-4}$	$4.99*10^{-4}$	$5.00*10^{-6}$	2.49
P.hale-	1	$4.69*10^{-1}$	$-1.2*10^{-3}$	$2.24*10^{-4}$	$-1.14*10^{-5}$	$-2.04*10^{-5}$	3.02
pensis	2	$3.52*10^{-1}$	$-8.9*10^{-4}$	$2.24*10^{-4}$	$-1.14*10^{-5}$	$-2.04*10^{-5}$	3.02
P.nigra	1	$5.53*10^{-1}$	$-1.5*10^{-3}$	$9.47*10^{-5}$	$-2.12*10^{-6}$	$2.02*10^{-5}$	2.26
	2	$5.62*10^{-1}$	$-1.8*10^{-3}$	$9.47*10^{-5}$	$-2.12*10^{-6}$	$2.02*10^{-5}$	2.26
P.pina-	1	$7.84*10^{-1}$	$-2.5*10^{-3}$	$-2.57*10^{-2}$	$1.97*10^{-3}$	$-1.82*10^{-5}$	1.57
ster	2	$8.91*10^{-1}$	$-3.6*10^{-3}$	$-2.57*10^{-2}$	$1.97*10^{-3}$	$-1.82*10^{-5}$	1.57
	3	1.0416	$-5.0*10^{-3}$	$-2.57*10^{-2}$	$1.97*10^{-3}$	$-1.82*10^{-5}$	1.57
	4	$5.28*10^{-1}$	$-7.0*10^{-4}$	$-2.57*10^{-2}$	$1.97*10^{-3}$	$-1.82*10^{-5}$	1.57
P.pinea	1	$3.58*10^{-1}$	$-5.5*10^{-4}$	$-4.28*10^{-3}$	$4.47*10^{-4}$	$7.95*10^{-6}$	3.02
	2	$6.51*10^{-1}$	$-2.0*10^{-3}$	$-4.28*10^{-3}$	$4.47*10^{-4}$	$7.95*10^{-6}$	3.02
	3	$6.51*10^{-1}$	$-2.0*10^{-3}$	$-4.28*10^{-3}$	$4.47*10^{-4}$	$7.95*10^{-6}$	3.02
Q.ilex	1	$4.15*10^{-1}$	$-6.5*10^{-4}$	$-1.92*10^{-2}$	$1.20*10^{-3}$		1.40
	2	$5.10*10^{-1}$	$-8.5*10^{-4}$	$-1.92*10^{-2}$	$1.20*10^{-3}$		1.40
	3	$8.11*10^{-1}$	$-1.4*10^{-3}$	$-1.92*10^{-2}$	$1.20*10^{-3}$		1.40
	4	$4.31*10^{-1}$	$-4.9*10^{-4}$	$-1.92*10^{-2}$	$1.20*10^{-3}$		1.40
Q.suber	1,2	$5.83*10^{-1}$	$-10.0*10^{-4}$	$-6.42*10^{-3}$	$3.21*10^{-4}$	$-4.72*10^{-7}$	4.43

Source: *Own elaboration* based on Montero et al. (2006, 2015).
Notes: $^{(1)}$ The variable x refers to the tree age (in years), while the variable y to the tree diameter (in cm). The volume (v) is measured in m^3. $^{(2)}$ The tree volume in case of *Q. ilex* includes trunk and coppice, thus the carbon stock ratio is lower.

Table A.4 presents the functions used to estimate cork, pinenut and acorn yields per individual tree. Cork production is estimated as the product of cork density (kg/m^2) and harvesting area (in m^2), which in turn is estimated as the product of tree diameter (y, in m) and harvest height (in m). The density of cork is taken from Montero et al. (1996) for different regions of Andalusia, whilst the cork harvest height is estimated using Montero et al. (2015) functions that relate the harvesting height (up to a threshold of 3 m) to the diameter of cork trees. pinenuts (cones) for *P. pinea* and acorns for *Q. ilex,* are also taken from Montero et al. (2015) In both cases the individual tree yield of those fruits depends on the diameter (y, in cm) of pine or oak trees.

Table A. 4: Coefficients to estimate cork, pinenuts and acorns yields

Class[(1)]	Cork yield (in kg per tree)			pinenuts		Acorns	
	Cork density	Harvest height(H)(m) $H = \beta_0 + \beta_1 y$		kg of cones per tree (C) $C = \beta_0 y^{\beta_1}$		kg of acorns per tree (A) $A = \beta_1 + \beta_2 y^2$	
	kg/m^2	β_0	β_1	β_0	β_1	β_1	β_2
Function 1	10.18	0.111	0.079	7.0*10^{-8}	4.7112	1.873*10^{-1}	1.1*10^{-3}
Function 2	11.70	-1.222	0.108	5.0*10^{-5}	3.9552		
Function 3	10.53	0.606	0.069	5.5*10^{-3}	1.4475		
Function 4	8.32	0.988	0.058				

Source: *Own elaboration* based on Montero et al. (1996, 2015).

Notes: [(1)] The variable y indicates the diameter of the tree. This diameter is estimated in meters (m) for the cork yield functions, and in cm for the pinenuts and acorns functions. In case of cork yield the function are applied to the following provinces: 1 is applied to *Q. suber* forest in Almería and Granada; 2 in Cádiz and Málaga; 3 in Córdoba, Jaén and Seville; and 4 in Huelva. In case of pinenuts, the functions 1, 2, and 3 are applied to the three silvicultural models of *P. pinea* presented (in that order) in Table A.2.

1.3 Mortality and forest fire risk ratios

Individual tree survivals functions depend on tree felling, natural mortality and forest fire risk. Natural mortality is estimated as a logarithmic function of tree age, for each species and silvicultural model (see Table A.5). These functions were estimated using Montero et al. (2015) mortality estimates by species, silviculture type and diametric class. In the case of Eucalyptus we assume a natural annual mortality rate of 0.05%. Forest fire risk ratios correspond to average ratios between the burned area and the total area by forest species and province between 1987 and 2006 (Diaz-Balteiro et al., 2015). In many cases there are no significant differences (p$\leq$0.05) between the average forest fire ratios at provincial and regional (Andalusia) levels. We use the province ratios only if they differ significantly from the regional one(Table A.6).

Table A. 5: Natural mortality function by species and forestry model

Species	Model	Annual mortality rate ($\theta_t = \beta_0 + \beta_1 * ln(t)$)[(1)]		
		β_0	β_1	R^2
P. halepensis	1,2	$5.80*10^{-4}$ ($1.53*10^{-5}$)	$-3.75*10^{-5}$ ($3.44*10^{-6}$)	0.41
P. nigra	1,2	$3.73*10^{-3}$ ($1.43*10^{-5}$)	$4.26*10^{-4}$ ($1.84*10^{-6}$)	0.99
P. pinaster	1	$9.59*10^{-3}$ ($2.62*10^{-4}$)	$-1.95*10^{-3}$ ($6.11*10^{-5}$)	0.86
	2	$9.06*10^{-3}$ ($2.44*10^{-4}$)	$-1.81*10^{-3}$ ($5.69*10^{-5}$)	0.86
	3	$8.44*10^{-3}$ ($2.52*10^{-4}$)	$-1.71*10^{-3}$ ($5.87*10^{-5}$)	0.70
	4	$2.55*10^{-3}$ ($2.75*10^{-4}$)	$-4.75*10^{-4}$ ($6.43*10^{-5}$)	0.75
P. pinea	1,3	$4.29*10^{-3}$ ($1.82*10^{-4}$)	$-8.35*10^{-4}$ ($4.12*10^{-5}$)	0.67
	2	$3.54*10^{-3}$ ($1.50*10^{-4}$)	$-7.34*10^{-4}$ ($3.39*10^{-5}$)	0.70
Q. ilex	1	$9.17*10^{-4}$ ($5.45*10^{-5}$)	$-1.60*10^{-4}$ ($1.11*10^{-5}$)	0.42
	2	$1.52*10^{-3}$ ($3.91*10^{-5}$)	$-2.64*10^{-4}$ ($8.00*10^{-6}$)	0.79
	3	$6.61*10^{-3}$ ($4.11*10^{-5}$)	$-1.24*10^{-3}$ ($8.41*10^{-5}$)	0.43
	4	$1.40*10^{-2}$ ($1.98*10^{-4}$)	$-2.55*10^{-3}$ ($4.04*10^{-5}$)	0.93
Q. suber	1	$2.76*10^{-3}$ ($1.05*10^{-4}$)	$-4.19*10^{-4}$ ($2.24*10^{-5}$)	0.58
	2	$9.26*10^{-3}$ ($2.46*10^{-4}$)	$-1.82*10^{-4}$ ($5.26*10^{-5}$)	0.83

Source: *Own elaboration* based on Montero et al. (2015).
Notes: [(1)] Where t is the age of the tree. Standard errors in parenthesis.

Table A. 6: Average annual forest fire ratios by forest species and province in Andalusia (Period 1987-2006)

Species	Annual forest fire ratio (ρ_j) in ha/100 ha [(1)]								
	AL	CA	CO	GR	JA	HU	MA	SE	AND
Eucalyptus sp.	0.68	0.68	0.68	0.68	0.68	0.68	0.68	0.68	0.68
P. halepensis	0.17	0.17	0.17	0.17	0.00*	0.17	0.17	0.17	0.17
P. nigra	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
P. pinaster	0.42	0.42	0.42	0.42	0.42	0.42	0.42	0.42	0.42
P. pinea	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.82*	0.25
Q. ilex	0.27*	0.16	0.05*	0.16	0.16	0.16	0.16	0.16	0.16
Q. suber	0.00*	0.37	0.04*	0.92*	0.37	0.37	0.37	0.37	0.37

Source: *Own elaboration* based on Diaz-Balteiro et al. (2015).
Notes: [(1)] The ratio ρ_j indicates the number of hectares out of every 100 ha that are expected to burn every year. AL (Almería), CA (Cádiz), CO (Córdoba), GR (Granada), HU (Huelva), JA (Jaén), MA (Málaga), SE (Seville), AND (Andalusia). *Ratios that exhibit a significant differences ($p < 0.05$) from the regional average value.

1.4 Forestry operation cost and output prices

The standing market prices for timber, firewood, cork and pinenuts (p_w) were obtained from a large data set on sales of forestry products recorded by the government of Andalusia (CMA, 2011; Diaz-Balteiro et al., 2015). Timber prices depend on tree species and diameter, cork prices on the cork quality index and grazing resources on both the dominant vegetation and province. We use a single regional price for firewood and pinenuts, since we have found no relevant differences in prices for those products across the Andalusian provinces. Table A.7 depicts the average prices for different forestry products according to tree diameter or quality of products. Outputs are integrated as exchange values (without including consumers surplus) in such a way that they are consistent with national accounting outcomes (Obst and Vardon, 2014).

Table A. 7: Forestry products standing prices (euro, year 2010)

Class[1]	Unit	Price 1		Price 2		Price 3		Price 4	
		cm	e/u	cm	e/u	cm	e/u	cm	e/u
Eucalyptus sp.-timber[2]	m^3	< 10	5.0	> 15	13.6	> 25	26.7		
P. halepensis-timber[2]	m^3	< 15	3.5	15-25	12.6	> 25	26.1		
P. nigra-timber[2]	m^3	< 15	2.7	15-25	9.1	> 25	31.0		
P. pinaster-timber[2]	m^3	< 15	3.3	15-25	12.0	> 25	30.3		
P. pinea-timber[2]	m^3	< 15	4.2	15-25	10.8	> 25	30.4		
P. pinea-nuts[3]	kg	All	0.05						
Q. ilex-firewood	m^3	All	58.3						
Q. suber-first cork[2,4]	Mkg	Q_1	56.9	Q_2	112.2	Q_3	153.4	Q_4	172.6
Q. suber-reproductive cork[2,4]	Mkg	Q_1	474.1	Q_2	934.7	Q_3	1,278	Q_4	1,726
Carbon credits (CO_2)[5]	Mg CO_2		13.7						

Source: *Own elaboration* based on Diaz-Balteiro et al. (2015).
Notes: [1] cm refers to the diameter of trees while e/u to the unit price in euro. [2] Average prices for 2008-2010 (updated to 2010, using the consumer price index) (Diaz-Balteiro et al., 2015).
[3] pinenuts price refers to the unit price of pine cones (between 4 and 5 kg of pine cones will yield 1 kg of pinenuts).
[4] Cork prices depend on the quality of cork from the worst quality (Q1) to the best (Q4). There are different prices for the first or virgin cork and the cork obtained from second and successive cork harvests.
[5] The average 2010 carbon price of the European Union Emissions Trading System (EU ETS) was 14.3/Mkg CO_2 (SENDECO2, 2015) which attains to 13.7/ Mkg CO_2 (considering a correction factor of 0.96). This correction takes into account Sohngen and Mendelsohn (2003) results; which suggest a decrease of 4% in the carbon price by the period 2010-2020 in a scenario in which forestry carbon credits are allowed with respect to a scenario without these credits.

Forestry operation costs depend on the slope gradient of the site and tree density. The unit costs of forestry operations function are shown in Table A.8. All forestry operation costs, except scrub clearing, are attributed to the production of forestry products (timber, cork, firewood and pinenuts). Shrub clearing is attributed to grazing resources. Half of *Q. ilex* pruning costs are attributed to grazing resources since this operation is intended to enhance acorn production. Our cost estimations assume constant returns to scale, that is to say, the size of the forest unit does not affect the cost functions, as we further consider that the fixed costs at the farm level are very low since landowners will normally either rent specific equipment or machinery or else hire a specialized firm to perform various forestry tasks on their lands.

Table A. 8: Forestry operation costs (euro, year 2010)

Class	Tree diameter (cm)	Unit (y)	Type of function([1]	β_0	β_1)
Oak woodlands					
Thinning	$\leq$ 12.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	33.48	27.53
Thinning	12.5-22.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	9.45	18.87
Thinning	22.5-32.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	9.45	29.27
Thinning	$\geq$ 32.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	14.41	19.68
Structural pruning	7.5-17.5	euro/ha	$y = \beta_0 x + \beta_1 x$	326.4	1,185.2
Maintenance pruning					
Oaks afforestation		euro/ha	$y = \beta_0 x + \beta_1 x$	917.9	726.2
Eucalyptus sp.					
Afforestation		euro/ha		850	
Re-sprouting	$\geq$ 17.5	euro/ha		219	
Weed control	Every 6 yrs.	euro/ha		432	
Coniferous forest					
Pines thinning	$\leq$ 7.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	22.88	40.30
Pines thinning	7.5-27.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	10.13	29.12
Pines thinning	$\geq$ 27.5	euro/m^3	$y = \beta_0 x + \beta_1 x$	8.80	30.12
Pines pruning	7.5-17.5	euro/tree	$y = \beta_0 x + \beta_1 x$	0.624	0.197
Pines Afforestation		euro/ha	$y = \beta_0 x + \beta_1 x$	505.3	210.34
Shrub clearing	All	euro/ha	$y = \beta_0 x + \beta_1 x$	497.8	726.2

Source: *Own elaboration* based on Diaz-Balteiro et al. (2015).

Notes: [1] The variable x refers to the slope gradient (in %).

Finally, Table A.9 presents the grazing resources prices by province and dominant species and the share of farms that are currently used for livestock grazing. The manufactured costs involved in the provision of grazing resources include the fixed capital consumption (amortization) of manufactured assets used for livestock grazing (i.e. fences, troughs) and their associated opportunity cost (Ovando et al., 2015). This manufactured cost amounts to 2.46 euro/ha in the provinces of Cádiz, Córdoba, Huelva, Málaga and Seville, and to 0.44 euro/ha in the remaining provinces.

Table A. 9: Average grazing resources price and the share of land currently used for livestock grazing

Provinces	Price (euro/FU[2])						Share of land used for livestock grazing (ω, %)					
	Q_i	Q_s	P	Eu	Sh	Gl	Q_i	Q_s	P	Eu	Sh	Gl
Almería	0.08		0.07		0.01	0.07	29				6	
Cádiz	0.06	0.13	0.07		0.03	0.07	100	72	67		79	89
Córdoba	0.05	0.07	0.07		0.07	0.07	87	40	57		44	100
Granada	0.07	-	0.07		0.01	0.07	48		55		43	67
Huelva	0.08	0.11	0.07	0.01	0.10	0.07	96	47	33	25	55	100
Jaén	0.02	-	0.07		0.11	0.07	63		45		42	43
Málaga	0.14	0.09	0.07		0.05	0.07	89	80	-		83	67
Seville	0.08	0.07	0.07		0.05	0.07	95	89	-		33	100
Total farms	0.07	0.10	0.07	0.01	0.05	0.07	85	69	36	25	33	81

Source: *Own elaboration* based on Oviedo et al. (2015).

Notes: [1] Eu: *Eucalyptus sp.*, Qi: *Q. ilex*, Qs: *Q. suber*, P: *Pinus sp.*, Sh: Shrubland; Gl: grassland. [2] A forage unit (FU) is equivalent to the metabolic energy content of a kg of barley (Oviedo et al., 2015, for details).

1.5 Land use transition scenarios

The effect of forestry activities abandonment is expected to diverge amongst forest species. It is difficult to predict the evolution on tree species and brushes distribution, but an analysis of a number of attributes of forest inventories (Diaz-Balteiro et al., 2015), such as the fraction of canopy covered by trees and scrubs, the average age class and the tree species distribution, allows us to outline a simplified set of scenarios that define what the the maximum fraction of scrubs canopy cover (FCC_S) by species and silvicultural models will be in a period of 50 years after the natural regeneration investment should have taken place. Those scenarios also define what share of specific *Pinus* sp. (FCC_P)will be after the aforesaid period and the maximum fraction of trees canopy cover (FCC_T) (Table A.10). The share of the land covered by scrubs and trees will not correspond to the maximum value for all the forest land units, rather it will depend linearly on the initial values of FCC_S, FCC_P and FCC_T.

Table A. 10: Expected fraction of Shrubs and forest species canopy cover for forestry abandonment scenarios by species and silviculture model

Species	Model	Fraction of canopy cover (FCC) %			
		Maximum Shrubs		Pines	All tree species
		FCC_S (range)		Min FCC_P	Max FCC_Ts
P. halepensis	1	55	60	50	90
	2	60	70	40	90
P. nigra	1	55	60	30	95
	2	60	70	15	45
P. pinaster	1	55	60	-	95
	2	60	70	15	45
	3	45	40	35	95
	4	55	60	35	45
P. pinea	1	58	65	45	90
	2	60	70	35	95
	3	55	60	50	80
	4	60	70	40	90
Q. ilex	1	55	60	-	95
	2	53	55	-	95
	3	58	65	-	95
	4	60	70	20	95
Q. suber	1	55	60	-	95
	2	60	70	20	95

1.6 Forest water balance

The Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) was used for computing the water balance of the Andalusian forest parcels .

SWAT is a basin scale hydrological model designed to assess the impact of different management options on water, sediment, and contaminant flows. It was originally developed by the US Agricultural Research Service (ARS), and it has gained wide international acceptance as a robust tool for watershed modeling. A comprehensive review of the SWAT model components and their interdisciplinary applications can be found in (Gassman et al., 2007).

In SWAT, a basin is divided into hydrologic response units (HRUs) with homogeneous land use (vegetation), soil and topographic characteristics. In each HRU the complete water balance is calculated, involving the partition of the input precipitation (plus natural spring outflows and anthropic irrigation inputs, if these are present) into different water flows, including those internal to the HRU (such as storage in the canopy, soil profile and shallow aquifer) and external HRU flows such as evapotranspiration, runoff and deep aquifer recharge.

We applied SWAT to 44 basins across Andalusia, covering the most important reservoirs of the region and a wide variety of climatic, topographic, soil and vegetation conditions. The input data included the gtopo30 digital elevation model, with a spatial resolution of 90 x 90 m ((Data available from the U.S. Geological Survey)); the land use / land cover map of Andalusia (JA, 2003); the soil class and soil properties map of Andalusia (CSIC-IARA, 1989); and daily maximum and minimum temperatures and precipitation data from more than 1,000 climatic stations (Agencia Española de Meterología, AEMET; and Confederación Hidrográfica del Guadalquivir, CHG), over the period 2000-2009.

The simulations were validated against monthly stream-flow data and mean annual aquifer recharge data (CHG). Data from more than 2000 forest HRUs spanning Andalusia were collected, and the hydrological balance was used for determining the main outgoing water flows from the forest farms analyzed in this study.

2 Supplementary results

2.1 The role of spatial variables in environmental asset estimations

Table (A.11) shows the results of the simple linear regression 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.*). The independent variables include the slope gradient, basal area (density), the quality of the site for growing cork or timber, forest structure (even or uneven aged forest) and specific Pine or oak species. The linear regressions for specific forest species are estimated only for the main scenario, whilst the 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 the forestry cost rise within the slope gradient, it is expected that the EAPr values are negatively affected by this variable in the cases of *Pinus* and *Quercus ilex* species, although the slope is not a significant variable. In the case of *Quercus suber* we observe higher EAPr values for an area with a higher slope gradient, which seem to be explained by the distribution of the quality of the sites for growing cork. The tree density (BA) positively affects EAC and EAPr values, and we observe that EAW values are higher in areas where *P. nigra* and *Q. suber* are located . Water values depend, however, on other factors besides the slope or the density, such as climatic factors and the characteristics of the catchment areas, that are not considered in the estimated regressions.

Table A. 11: Environmental asset functions for *Pinus sp.*, *Q. ilex* and *Q. suber*[(1)]

Variables[(2)]	*Pinus sp.*						*Quercus ilex*					
	EAPr			EAC			EAPr			EAC		
	Coef.		SE[(3)]	Coef.		SE	Coef.		SE	Coef.		SE
Const.	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
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^2	0.35			0.59			0.52			0.80		
N obs.	171			171			1,317			1,317		
	Quercus suber						*Quercus sp.*			*Pinus sp*		
	EAPr			EAC			EAW			EAW		
	Coef.		SE	Coef.		SE	Coef.		SE	Coef.		SE
Const.	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
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^2	0.38			0.50			0.06			0.25		
N obs.	720			720			2,037			171		

***$p<0.01$, **$p<0.05$, *$p<0.10$. EA values for the main scenario (r=3% , p=1.0)

. Notes: [(1)] EAPr: environmental asset value of silvopastoral provisioning services, EAC: environmental asset value for carbon sequestration, EAW: environmental asset value for water provisioning service. [(2)] The variables referred to are: Slope gradient is estimated as a percentage, BA is the basal area in m^2/ha, and the remainder are dummy variables: LMQ (Low-medium quality); HMQ (High-medium quality); EAF (even-aged forest), Suber, Halep, Nigra and Pinst are referred to as *Q. suber, P. halepensis, P. nigra* and *P. pinaster* forest, respectively, and Q_1, Q_2 and Q_4 are the cork quality index (being Q_1 the lowest and Q_4 the highest). [(3)] Robust standard error.

2.2 Forestry abandonment and minimum compensations

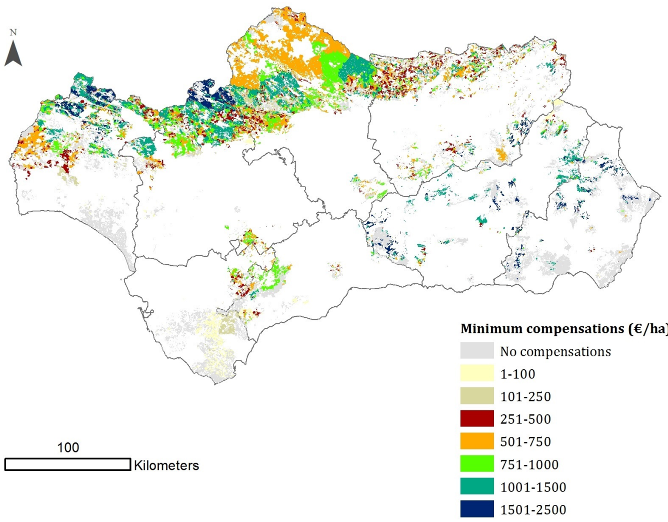
Table A.12) shows the minimum compensation to *Q. ilex* and pine species for the main scenario and for the low and high discount and price scenarios. The estimated minimum compensations diverge spatially (Fig.A.2).

Table A. 12: Minimum compensation to natural regeneration investment in *Q. ilex* and pine species forest units (euro/ha, year 2010)

Scenario	*Pinus sp.*			*Quercus ilex*		
	Mean	SD	CV (%)	Mean	SD	CV (%)
Central	1,386	713	51	996	501	50
Low discount	2,009	1,121	56	543	509	94
High discount	806	375	46	1,078	380	35
Low price	1,616	766	47	1,929	763	40
High prices	1,279	676	53	348	341	98

Note: SD: Standard deviation, CV: Coefficient of variation (CV=SD/mean).

Fig.A. 2: Distribution of the minimum compensations to natural regeneration investment



Note: Data for main scenario (r=3%, p=1.0).

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