Carbon sequestration or water yield? The effect of payments for ecosystem services on forest management decisions in Mediterranean forests

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Abstract

A decrease in net benefits from market-based forest products over the last three decades seems to be connected to the increasing abandonment of forestland in Mediterranean areas of Europe. In this paper, we estimate spatially distributed probabilities of continuing with forest management practices in mainly native pine and oak forests in Andalusia (southern Spain). The continuation of active forest management encompasses landowners' investment in operations that promote the recruitment of new individuals of the same species once a stand reaches its rotation age. On the contrary, abandoning forestry investment involves natural self-regeneration including tree densification, shrub encroachment and forest species replacement. We estimate the long-term effects of alternative scenarios of forest management on carbon sequestration and water provisioning services (blue water, i.e. surface discharge and ground water recharge). We examine the effect of payments for carbon sequestration, or alternatively for increasing blue water, on forest management decisions. Results are provided as spatially disaggregated values over different time frames up to 2100 and consider the sensitivity of forest investment decisions to a wide range of price levels and discounting rates.

Keywords: forestry investment abandonment; trade-offs and synergies; economic decision; Andalusia.

JEL: Q23, Q51.

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1 INTRODUCTION

Mediterranean environments are characterized by particular climatic conditions, whose varied topography has resulted in a range of microclimates that foster high biodiversity levels (Ruiz-Labourdette et al., 2013). However, the composition and structure of Mediterranean vegetation and landscapes has been profoundly influenced by human activities over the centuries (Scarascia-Mugnozza et al., 2000). The interaction between environmental and management factors has resulted in complex, but fragile, landscape patterns that provide a variety of market and non-market (i.e. amenities) goods and services (e.g., Croitoru, 2007; Gaspar, et al., 2007; Ovando et al., 2016), whose conservation demands constant and specific management (Plieninger et al., 2014). Since the mid-20th century, northern Mediterranean mountain areas have experienced an increased land abandonment, propelled mainly by changing socio-economic factors such as urbanization and rural depopulation processes, and a decline in the economic profitability of traditional agroforestry activities (Fragoso et al., 2011; Serra et al., 2008; Weissteiner et al., 2011). The abandonment of grazing and extensive farming uses has favored revegetation, with a consequent expansion in trees and shrub cover (Otero et al., 2011; Allard et al., 2013; Kuemmerle et al., 2016). Similarly, declining profitability rates of forestry activities – the result of high production costs and plunging prices for timber and non-timber products (Scarascia-Mugnozza et al. 2000) – has also prompted forestland abandoned in northern Mediterranean areas (Palahi et al., 2008).

The potential impacts of forestland abandonment on the provision of ecosystem services¹, such as water supply, biodiversity conservation, recreation, or carbon sequestration, are not fully understood. Some studies suggest that land abandonment might lead to landscape homogenization and biodiversity loss in Mediterranean areas (Amici et al., 2015). It has also been suggested that abandonment might lead to landscapes' re-naturalization, boosting biodiversity (Plieninger et al., 2014). While scientific evidence suggests a positive impact of forest cover on water quality (e.g., Abildtrup et al., 2013; Brogna et al., 2017), it also indicates that forest cover reduces water supply, in particular at the catchment level (Beguiría et al., 2003; García-Ruiz et al., 2011; Cáceres et al., 2015; Filoso et al., 2017). However, the role of forests in regulating water flows is still subject to debate (Bennett and Barton, 2018), especially as concerns the impact of forests on precipitation, water yield and the hydrological cycle at scales larger than local catchments (Ellison et al., 2012; Spracklen et al., 2012). While the potential beneficial relationship between forest and water yield is still debated, forests' carbon sequestration has been promoted as a way to mitigate climate change through afforestation/reforestation (Sonntag et al., 2016), and protecting carbon stocks by reducing emissions from forest degradation and deforestation (Canadell and Raupach, 2008). Forestland abandonment might increase forest carbon stocks (Ovando et al., 2017), but may also increase the risk of fire due to biomass accumulation (Allard et al., 2013; Fernandes et al., 2014), increasing the risk of soil erosion and landslides (García-Ruiz et al., 2011). The aggregate environmental consequences of changes in Mediterranean landscape structures due

¹Ecosystem services are here defined as “the contributions of ecosystems to benefits [goods and services] used in economic and other human activity” (United Nations et al., 2014, para. 2.23, p. 19).

to forestland abandonment seem to depend on uncertain and non-linear factors and relationships that are influenced by the scale of changes and complex local environmental factors (Lasanta et al., 2015).

Traditionally, land use management in Mediterranean areas was characterized by a combination of livestock rearing, agriculture, and forestry practices to satisfy the demand for agriculture and forestry products (Martin and Fernandez-Ales, 2006). More recently, rising living standards and urbanization processes have raised the demand for cultural (amenities) and environmental regulating services from Mediterranean forests (Campos et al., 2017; Caparrós et al. 2010; Palahi et al. , 2008). In spite of conservation scientists' and policymakers' increased attention on landscape management to enhance the provision of ecosystem services, there is still a lack of instruments to support planning, management, and decision-making (de Groot et al., 2010). This is exemplified by northern Mediterranean areas, where traditional agroforestry has progressively lost importance and marketability, while the demand for ecosystem services has not been translated into incentives to sustainable silvicultural practices (Scarascia-Mugnozza et al., 2000).

There needs to be a better understanding of the patterns, drivers, and impacts of land use changes over time and across space, as well as the synergies and trade-offs involved, to ensuring the sustainable management of cultural landscapes. This study aims to analyze the long-term effects of forest management decisions on two ecosystem services: climate change regulation through carbon sequestration and water provision. These ecosystem services are key to two of the Mediterranean's most relevant environmental challenges today: global warming and water scarcity. Mediterranean ecosystems are predicted to be exceptionally vulnerable to global warming, with climate change scenarios forecasting relevant increases in surface temperature and decreases in average precipitation (IPCC, 2014). Climate change is expected to magnify threats such as forest fires (Schröter et al., 2005) or seasonal droughts (Vicente-Serrano et al., 2014) in the coming years. Forests can play a role in mitigating climate change through capture and storage of carbon, but in arid and semi-arid areas they can also have adverse impacts on water supply (Chisholm, 2010; Whitehead, 2011), affecting diverse downstream economic activities such as agriculture (Nordblom et al., 2012). Making better land management decisions in the presence of trade-offs requires a systematic account of the relationships between forest management and the ecosystem services delivered, and their economic values. Indeed, a recurring recommendation of international forest and water experts has been to integrate the economic value of forest and water resources into decision-making in order to support more sustainable management strategies (FAO, 2016; MCPFE, 2009).

In this paper we develop and apply an environmental-economic forest management decision (EEFMD) model that integrates detailed forest, hydrological, and economic models and data. This modeling framework is applied to estimate the spatially distributed probabilities of forest management continuation or abandonment in Andalusia (southern Spain) for a range of forest product prices and incentives to deliver water and carbon services. The model is

further used to evaluate the consequences of those decisions on carbon sequestration and water provisioning services across space and over time (for the 2010-2100 period).

Andalusia constitutes an ideal case study to evaluate spatial and temporal trade-offs and synergies in the provision of climate change regulation and water provision services, given the spatial variability exhibited by the forests of this region, and in view of the water scarcity problems (Velázquez, 2007). Andalusia is indeed a very diverse region, with climatic conditions that range from one of the rainiest points on the Iberian Peninsula to desert areas, and altitudes ranging from sea level up to 3,400 m. The forested areas cover more than one third of the Andalusian territory (2.83 million hectares), consisting mainly of a mix of native slow-growing oak and conifer (mainly pine) species (Campos, 2015). We apply the EEFMD model to the area covered by pure and mixed forest of seven of the main native pine and oak species, accounting for explicit spatial forest attributes such as the existing forest structures and inventories and the productivity of the sites to grow different forest products and topography; this represents about 80% of the forested area of this region.

Payment for ecosystem services (PES) has gained popularity over the past two decades as a mechanism for creating new markets for ecosystem services, and translating societal benefits (at least in part) to financial incentives to the providers of ecosystem services (Engel et al., 2008). In this paper, we further simulate the potential effect of payments for carbon sequestration or water provision on forest management decisions and their environmental consequences. With the application of this model we intend to address three main research questions: (1) What are the expected dynamics and spatial distribution of managed and abandoned forest, and of the carbon sequestration and water services they provide, under different economic conditions? (2) What is the effect of payments for carbon sequestration and water provision on forest management decisions and the provision of these two services? (3) How do the trade-offs between carbon and water values can affect the design of incentives to enhance the effectiveness and efficiency of spatially targeted environmental and rural development policies for forest management? Based on the results of this assessment, we discuss the key findings and identify future research needs.

2 METHODOLOGY

2.1 Modeling forest growth, yields and management in Andalusia

Forest growth, yields, and structure (defined by the number of trees per unit of land, shrub cover and distribution of tree species and age classes) are predicted using the environmental-economic forest management decision model developed by Ovando et al. (2017). This model integrates detailed forest, hydrological, and economic data, as shown schematically in Fig. 1, that represent current forest state and management practices in Andalusia for the most important forest species in this region. The model is then applied to seven of its most relevant native forest species: *Quercus ilex*, *Q. suber*, *Pinus pinaster*, *P. halepensis*, *P. nigra*, *P. pinea*, and *P. sylvestris*. These species are found in an area of 2.52 million hectares (Table 1),

distributed over 48,895 polygons² of the Spanish forest map (SFM), and whose stock data corresponds to the stratified results of the third National Forest Inventory (IFN3) (Díaz-Balteiro et al., 2015).

[Figure 1 about here]

The EEFMD model integrates growth and yield functions for individual trees with forest management models for even-aged and uneven-aged stands, which are based on Montero et al. (2015). In total, 20 different forest management models (from two to five options by species) are applied, according to the productivity of the site for growing timber, cork, or pine nuts. The forest growth, mortality, and yield models are based on empirical data collected in different Mediterranean forests of Spain, while forest management standards consider the most common forest practices in Andalusia. The model is applied to each forest unit (SFM polygon) to predict the dynamics of forest structure and yields. Forest dynamics depend on the profitability that forest management has for the landowner at each SFM unit, which in turn would be affected by spatial-explicit attributes such as the initial existing inventories, the slope of the terrain (estimated using a digital elevation model), and the quality of the site for growing different species. In addition, the model accounts for species and province-specific wildfires risk rates, and species and management model-specific tree mortality functions, as detailed in the Supplementary online text.

The forest management model is coupled with an economic decision module to predict commercial harvesting and forest regeneration investment decisions. Here we simulate the rational decisions of forest land owners that maximize the monetary benefits accrued from forestry uses. Furthermore, the model provides information on how the forest structure dynamics impact on a spatially distributed hydrological model, applied to the 44 major water catchments (defined by a regulated water reservoir) in Andalusia. Two alternative situations are evaluated: (i) forest management is abandoned, or alternatively, (ii) the landowner invests in facilitating forest natural regeneration and its management to produce market-based forest products.

[Table 1 about here]

The continuation of active forest management encompasses landowners' investment in forestry operations that promote the recruitment of new individuals of the same species once each tree cohort (trees of same species and diametric class) reaches its rotation age. In this case we assume that the new forest stand will be affected by the same tree growth, yield, mortality, fire risks, and management functions and models as the older stand. We expect that abandonment of forest management will, in most cases, result in natural self-regeneration including tree and shrub encroachment and forest species replacement, with the consequent variation in the distribution of oak and conifer species at each forest unit. For the

² Each polygon constitutes a homogeneous forest unit with the same biophysical attributes (slope gradient, site quality, and forest structure), with areas varying from a couple of square meters to more than 500 ha.

abandonment scenarios, we simulate forest vegetation transitions from managed forests to more naturalized stands. Those scenarios define the maximum canopy cover for shrubs, oaks, and conifer species that would characterize abandoned forestland according to the dominant species and management model evaluated (Ovando et al., 2017). These were defined using IFN3 data for aged forest stands, and considered the advice of experts who suggested a transition period of up to 50 years after forestry activity is abandoned to reach the thresholds that define each forestland abandonment state. To simulate the evolution of trees and shrubs during the 50-year transition period, we assume linear changes between the state of each tree cohort at rotation age and the corresponding abandoned state (see Supplementary online text). After this transition period, we assume stable forest states.

2.2 Quantification of market and environmental benefit and costs

We quantify the stream of market revenues and environmental products³ (jointly referred to as “benefits” hereinafter) and management costs for each polygon of the SFM in the case of continued forest management or, alternatively, the flow of environmental benefits in the case of abandoning forest management. This study includes the benefits from traditional forest management products such as timber, cork, firewood, pine nuts, and pastures, and two ecosystem services: carbon sequestration in forest biomass (trees and shrubs), and water provision through both surface water discharge and ground water recharge. Although we quantify total surface water, we are particularly concerned with gauging the surface water portion that is of economic interest – that is, the share of surface water that is captured by forest and delivered to consumptive uses (Beguería et al., 2015). Ground water recharge is quantified only in physical terms, thus not included as an economic decision variable; the remaining market and environmental variables are accounted for to simulate future forest management decisions.

The products considered in this study are the main market-based products delivered by Andalusian forests (CMA, 2010). Other products such as mushrooms or wild edible plants are usually gathered without landowner’s permission, hence without any payment to landowners in most Spanish forests (Górriz-Mifsud, et al. 2015). The only forest product of market interest excluded from this analysis is the leasing of land for hunting purposes (e.g. Herruzo et al. 2016). This is because we ignore how forest management or abandonment would affect hunt leases; hence we would need to assume similar fees for the abandonment and management scenarios, with no effect on the decision problem.

Market benefits and costs

³ Strictly speaking, “environmental products” can include, labor cost, capital costs and manufactured inputs, for the purposes of this paper they are considered to be equivalent to “ecosystem services”, which represents the pure contribution of ecosystems to the production process. This paper considers carbon sequestration and water provisioning services as joint products of forest management, thus no labor or any manufactured input or capital is assigned to the provision of those services.

Timber, cork and pine nut yields are valued by species, product and quality class (for timber and cork) using the average stumpage prices (as of 2010) for those products in Andalusia over the 2008-2010 period (Diaz-Balteiro et al., 2015); while pastures are valued by forest species and province according to observed grazing environmental prices in Andalusia in 2010 (Campos et al. 2016). Forest costs depend upon the specific spatial attributes of each SFM polygon, particularly forest structure and slope gradient. Forest operations include scrub clearing, maintenance and structural pruning, thinning, as well as commercial harvesting of pine nuts, cork, firewood, or timber (see Supplementary online text for details). The evolution of forest stands, and their associated benefits and cost, are conditioned by the commercial harvesting and forest regeneration investment decisions, as discussed in section 2.3. In the forest management scenario, their costs include the opportunity cost of a 20-year livestock grazing exclusion period, which is needed to ensure tree recruitment in an aged forest stand. The benefits associated to forest management are estimated assuming that future prices and production costs would remain constant and equal the average prices observed in Andalusia in 2010 (defined as price level 1). We simulate variations of up to $\pm 50\%$ in the average prices of forest products, and real discount rates (dr) in the range of 2% to 5%. These discount rates are within the range usually applied to evaluating forest investment projects involving slow growing species (Brukas et al., 2001).

Carbon sequestration

Carbon sequestration accounts for the biomass growth of trees as well as shrub formation under the tree layer. Tree carbon sequestration is estimated by individual species using the Montero et al. (2006) equations, which allow us to estimate the carbon stored in both aerial and root biomass as a function of tree diameter at breast height. Net carbon sequestration depends on the annual variation of carbon stocks, which are in turn affected by tree diameter growth, mortality, forest fires, and scheduled tree thinning and harvesting operations. Carbon sequestration in shrub biomass is measured using Pasalodos-Tato et al. (2015) functions that describe shrub biomass growth according to the fraction of their canopy cover (FCC) and the average height of shrub formations (SH). Net carbon sequestration by shrub growth further accounts for potential carbon dioxide (CO_2) withdrawals due to forest fires (assuming the same risk as the specific tree layer) and shrub clearing (assuming that 2% of the understory shrub area is cleared every year)⁴. The spatial information on the variables (FCC and SH) used to estimate net carbon sequestration in shrub biomass is taken from Diaz-Balteiro et al. (2015). It is assumed that the shrub vegetation would maintain its current carbon stock and growth ability at each site in the future (steady state situation); but in the event of forestry activity abandonment, we expect shrub encroachment over a 50-year transition period before reaching a new steady state situation.

In this paper we analyze the effect of theoretical PESs for carbon sequestration (and water provision) on private forest management decisions. PESs are incentive-based (and voluntary)

⁴ That equates to a total clearing and regeneration period of 50 years as estimated by Ovando et al. (2015) using survey data from a sample of Andalusian forest landowners.

mechanisms that intend to correct negative environmental externalities (Engel et al., 2008), and consequently should account for the full social cost of those externalities. We consider that these PES schemes would only be carried out to encourage activities with positive externalities that would not have been implemented in the absence of these payments (Wunder, 2005). Therefore, while we estimate total carbon sequestration, we assume that PES for carbon would only recognize active forest management projects; there would be no carbon payments in case of forest abandonment. We further assume that forest owners who participate in this voluntary market would get paid for each CO₂ ton that is removed from the atmosphere (only by tree growth) and should pay a tax (equal to the unit carbon allowance price) for CO₂ withdrawals due to fires, thinning, and tree harvest (assuming instantaneous emissions).

Although we simulated the PES scheme here, there are some inherent difficulties in putting a monetary value to forest carbon sequestration, as forest carbon offsets do not have well-established markets. The European Union Emissions Trading Scheme (EU ETS) is the world's first, and remains the biggest, carbon market; however, forestry carbon abatement options are not eligible in this scheme. On the other hand, the UN Reduced Emissions from Deforestation and Forest Degradation (REDD+) initiative accounts for a relevant share of primary carbon offset markets, but it only considers forestry and land use projects in developing countries (Hamrick and Gallant, 2017). Carbon sequestration services can be alternatively priced at the social cost of carbon (SCC), which estimates the economic effects of an additional emitted ton of CO₂ in the atmosphere (Nordhaus, 2017). However, SCC estimates vary greatly, from a few dollars to hundreds of dollars per ton (Van Den Bergh and Botzen, 2015). In practice, connecting PES schemes with the social benefits and/or costs of any ecosystem service is challenging, since both the ecological functions and social values that affect the interactions between the beneficiaries and providers of ES are typically complex and uncertain (Muradian, 2013), and often characterized by market failures such as asymmetric information and incomplete property rights, but also by policy distortions (Engel, 2016).

As none of the carbon valuation option is free of controversy, in this paper we consider the EU ETS carbon price (€14 per CO₂ ton in 2010) to be the closest carbon market for European forestry projects. This analysis considers, nonetheless, variations of up to $\pm 50\%$ with respect to the prices observed in 2010 (€7 to €21 per t CO₂), as this market has shown a clear declining tendency since 2010 (SENDECO2, 2017). Voluntary forest and land use carbon offsets (out of Europe) attained values close to US\$5.1 per CO₂ ton in 2016 (Hamrick and Gallant, 2017), which is similar to the EU ETS carbon prices observed during this period. Meanwhile, a meta-analysis of SCC estimates suggests that governments using a 3% discount rate for climate change and other decisions should impose a carbon tax of between US\$25 and US\$50 per carbon ton (Tol, 2009). This tax would be in the range of

€5.1 to €10.2 per CO₂ ton⁵, with the CO₂ simulated prices covering the upper range of carbon taxes based on the SCC estimates.^{9o}

Surface and ground water

The water balance partition of each forest patch was estimated using the Soil and Water Assessment Tool (SWAT) model, calibrated for the Andalusian forests by Beguería et al. (2015). This model is based on numerical simulations of the forest water balance on hydrological response units (HRU) of 44 water reservoir catchments in Andalusia with minimum or no upstream regulation (natural flows), for which there was sufficient streamflow data for calibration and validation. The simulation uses daily climatic data as the main variables, and covers the period 2000-2009. Precipitation water (and superficial springs in some cases) constitutes the water input to each HRU that is transformed (partitioned) by forestland into the forest water outputs: evapotranspiration, surface flow, and groundwater recharge. SWAT is a spatially distributed hydrological model, and the Beguería et al. (2015) application allocates hydrological flows within each HRU as a function of the main forest stand attributes and their evolution over time at each SFM polygon, for both forest management and abandonment scenarios. The attributes considered are tree and shrub canopy cover and the share of conifers, oaks, and eucalyptus basal areas over the total basal area (in m²/ha). Tree canopy cover is estimated as a function of the basal area using specific equations that were estimated using IFN3 data (see Supplementary online material). Those variables are assessed for specific years in 10-year increments for the period 2010-2100 and integrated into the SWAT model to estimate the water balance partition at each SFM polygon.

Forest water can be either consumed within the HRU by the vegetation (evapotranspiration) or exported out (surface water run-off) of the HRU, as surface discharge and deep aquifer recharge. A fraction of the exported water flows can be regulated by the government water agency (collectible forest water surplus) and sold to users, which in the Andalusian case are mainly in the agricultural sector (85%), with a smaller proportion of industrial or household (15%) users (Beguería et al., 2015). Note that downstream farmers would not pay an environmental price for water; however, they do benefit from the environmental value of this water. This environmental capital value of water is embedded in the price that land buyers would pay when acquiring an irrigated plot of land in Andalusia.

Similarly, water prices should ideally reflect the value that water has for society. However, water resources exhibit high exclusion costs since they are common-pool resources (Booker et al., 2012), challenging the valuation on the basis of people's willingness to pay. In Andalusia, agriculture is the main consumptive use of water, and water fees are charged according to the number of irrigated hectares rather than the amount of water consumed. This arrangement, along with long-term (i.e., 50-year) water concessions granted to farmers,

⁵ Considering average an 2010 exchange rate of 1.3257 US\$ per euro (Eurostat, 2018), and that 1 kg of CO₂ contains 0.27 Kg of carbon.

introduces distorting elements in water resource management and valuation (Velázquez, 2007). Consequently, water resources are often underpriced, and at best only reflect the water provisioning cost, not the opportunity cost of water resource scarcity (Pulido-Velazquez et al., 2013). Berbel and Mesa (2007) estimate the capital value of water for its irrigation use in the Guadalquivir River Basin, using a simple hedonic price method that considers water as an external variable affecting the price of agricultural lands. The estimated water capital price ranges from €2.8 to €4.2 (2004 values). Berbel et al. (2011) estimate the average residual value of irrigation water in Andalusia, based on water productivity ratios and gross margins of irrigated crops in the same basin. The estimated average residual water value is €0.31 m⁻³ (2005 values), with 62% of this value representing a return to irrigation water, and the remaining 48% returning to rain-fed water (land), man-made capital and labor. Borrego-Marín et al. (2016) estimate a higher blue water value of €1.33 m⁻³ for the Guadalquivir basin for the period 2004-2012, when drought periods are accounted for.

For our calculations, we have chosen to use the value of forest blue water based on the average water capital value (P_w) estimated by Berbel and Mesa (2007), which was updated to €4.04 per cubic meter as of 2010. The price for each cubic meter of water (p_w) is calculated using the simulated real discount rates (dr) as the capitalization rates: $p_w = P_w * dr$. Likewise, we simulate water price variations of up to $\pm 50\%$ with respect to the estimated 2010 water capital value, and for discount rates ranging from 2% to 5%, which gives us a prices range of €0.04 to €0.30 m⁻³. These prices are further multiplied by the fraction of water that is regulated and implicitly sold to final users (termed “economic water”) at each SFM polygon, in order to obtain the value of the economic surface water flow.

2.3 The forest management economic decision model

Investment and abandonment decisions are simulated, spatially and temporally, using the EEFMD as indicated before. In this model, forest regeneration investment decisions are simulated at each SFM polygon as trees reach their theoretical rotation age, which varies spatially according to the initial age class distribution of the SFM polygon. The rotation age is estimated considering the average time needed by each species, under specific site conditions, to attain saw-wood diameter (in the case of pine species) or the diameter at which the forest management models of Montero et al. (2015) recommend the start-up of regeneration felling. These rotation ages are species-, site- and forest management model-specific (Fig. 2).

[Figure 2 about here]

The EEFMD model endogenously determines the probability (φ) of continuing forest management at each polygon, depending on the returns from natural regeneration investment, which change as a function of price levels (changes with respect to 2010 prices) and discounting rates in the simulated scenarios. We assume that the forest owner has perfect information about expected market and environmental forest benefits and costs. We further assume that the forest owner will invest in the regeneration treatments of an aged forest if

the net present value (NPV_M) of the expected market benefits (B_{M_s}^{ij}) of the new rotation equals or surpasses the NPV_M of its production costs and grazing opportunity costs (C_{M_s}^{ij}) along the entire rotation. The forest management probability is therefore a binary variable that takes a value of 1 or 0 depending on the NPV_M values. The regeneration investment probability (φ) equals 1 if the NPV_M is positive; otherwise it takes a value of 0:

$$\begin{aligned} \text{NPV}_M^{ij} &= \sum_{t=s}^{T^{ij}} \sum_{k=1}^K \delta^{-s} (\sum_{q=1}^Q p_e^{ij,q} q_{s,k}^{ij} - \text{cm}_s^{ij,q}) \geq 0, \quad \varphi=1 \\ B_{M_s}^{ij} &= p_e^{ij,kq} q_{s,k}^{ij,q}; \quad C_{M_s}^{ij} = \text{cm}_s^{ij,kq} \end{aligned} \quad (1)$$

where δ is the discount function of the type: 1/(1+dr)^{-s}, while p_e^{ij,q} and q_s^{ij,q} are the price and quantity, respectively, of a *k* market product, of a quality class *q*, of a species *i* and a management model *j*, at a price level *e* (0.5, 1, 1.5) in a period *s*. Prices are assumed to remain constant for each product and quality class, while quantities vary in each period, as do the management costs (cm).

The information provided by the management and growth models, as well as additional forest-related variables, is used to simulate the forest stand evolution. The stand evolution depends on the conditional survival probabilities (π_t^{ij}, where 0 ≤ π_t^{ij} ≤ 1) of each individual tree (of a species *i* and management model *j*) at every year of its rotation (T^{ij}). The conditional survival probabilities are affected by scheduled tree thinning and final harvesting (α_t^{ij}), natural tree mortality (θ_t^{ij}), and by forest fire risk (Q_t^{ij}). The variables α_t^{ij}, Q_t^{ij} and θ_t^{ij} represent the annual probabilities that trees will be felled, burnt, or will die, respectively:

$$\pi_t^{ij} = \begin{cases} 1 - \alpha_t^{ij} - Q_t^{ij} - \theta_t^{ij}, & \text{if } t=1, \\ \pi_{t-1}^{ij} - \alpha_t^{ij} - Q_t^{ij} - \theta_t^{ij}, & \text{if } t > 1, \end{cases} \quad \text{where } t = \{1, 2, \dots, T^{ij}\}. \quad (2)$$

The individual tree felling probability at each tree age (*t*, in years) is quantified as the ratio between the number of trees that the silvicultural models determine will be felled at that age (*h_t*), and the initial number of trees (*N_i*) according to the silviculture model: (α_t^{ij} = *h_t*/*N_i*). The risk of forest fire is estimated using the average historical forest fire ratios by species and province assessed for the period 1987-2006 (Diaz-Balteiro et al., 2015), while the mortality ratios are estimated as logarithmic functions of tree age, and vary by species and productivity of the site (see Supplementary online text).

A survival probability matrix (A^{ij}) has a dimension T^{ij} × T^{ij} in the forest management scenario and computes the conditional probability (π_{dt}^{ij}) that a tree of an age *d* is alive at each of the tree ages *t* that are to be reached (*d* < *t*): π_{dt}^{ij} = Pr(*d*/*t*) = π_t^{ij}/π_d^{ij}:

$$A^{ij} = \begin{bmatrix} \pi_1^{ij}/\pi_1^{ij} & 0 & \cdots & 0 & \cdots & 0 \\ \pi_2^{ij}/\pi_1^{ij} & \pi_2^{ij}/\pi_2^{ij} & \ddots & 0 & \ddots & 0 \\ \vdots & \vdots & \ddots & \vdots & \ddots & \vdots \\ \vdots & \vdots & \ddots & \vdots & \ddots & \vdots \\ \pi_{T^{ij}-1}^{ij}/\pi_1^{ij} & \pi_{T^{ij}-1}^{ij}/\pi_2^{ij} & \cdots & \pi_{T^{ij}-1}^{ij}/\pi_s^{ij} & \cdots & 0 \\ \pi_{T^{ij}}^{ij}/\pi_1^{ij} & \pi_{T^{ij}}^{ij}/\pi_2^{ij} & \cdots & \pi_{T^{ij}}^{ij}/\pi_s^{ij} & \cdots & \pi_{T^{ij}}^{ij}/\pi_{T^{ij}}^{ij} \end{bmatrix} \quad (3)$$

In the abandonment scenario, the survival probability matrix has a dimension $T^{ij} + \tau \times T^{ij} + \tau$, where τ accounts for a transition period of 50 years after the regeneration treatments were supposed to have started. Those survival matrices are used to simulate the evolution of forests and can 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. Future forest growth, yields, and benefits are affected by these probabilities in both the forest management and abandonment scenarios.

The net present value associated with ecosystem services (NPV_E) is given by: carbon sequestration and water provisioning services estimated in view of forest management abandonment/continuation probabilities (φ), the price of a k ecosystem service (p^k), and the annual production of this service in the forest management continuity scenario ($q_t^{k,m}$) until the rotation age is reached, the production of this service during the transition period ($q_t^{k,m+\tau}$) in the abandonment scenario, and the annual yield of the k service after the transition period ($c^{k,a}$):

$$NPV_E = \sum_{t=S}^{T^{ij}} \delta^t \cdot Y_t^{k,m} + \varphi \cdot \frac{\delta^{S-s}}{1-\delta} \sum_{t=1}^{T^{ij}} Y_t^{k,m} + (1-\varphi) \cdot \left(\delta^{S-s} \cdot \sum_{t=S-s}^{U-s} (\delta^t \cdot Y_t^{k,m+\tau}) + \delta^{S-s} \cdot \frac{\delta^{U-s}}{1-\delta} \cdot Y^{k,a} \right)$$

where: $S = T^{ij} + 1$; $U = S + 1 + \tau$;

$$Y_t^{k,m} = (p^k \cdot q_t^{k,m}); Y_t^{k,m+\tau} = (p^k \cdot q_t^{k,m+\tau}); Y^{k,a} = (p^k \cdot c^{k,a}). \quad (4)$$

As mentioned previously, we assume that after the transition period, the water and carbon yields remain constant. It is also assumed that if forest management continues, there will be no changes in forest species, and that current rotation will be followed by an infinite sequence of identical forest rotations of the same species, but accounting for the whole rotation in that case ($t = \{1, 2, \dots, T^{ij}\}$) and considering the expected initial tree densities defined by Montero et al. (2015).

3 RESULTS

3.1 Probability of forest management abandonment

At the initial period of our simulation, in 2010, around 12% of the evaluated forest area has trees that are 50 or more years older than the theoretical rotation ages defined in this study (by species and management model), and therefore are considered to be abandoned forest stands. We further estimate the probability of abandonment ($\varphi = 0$) or continuity ($\varphi = 1$) with forest management for each species, SFM polygon, and period, in view of the initial age of each tree cohort and the net present values derived from market forest products (solid black line in Fig. 3). The net present value of forest management benefits for the land owner (NPV_M^{ij}) fluctuates within the discount rates and the relative price levels of simulated market forest products. Our results show relevant variations in the probability of continuing with forest management, even though the trend towards abandonment seems clear even in scenarios with higher prices ($pt=1.5$) and lower discount rates ($dr=2\%$), with at least 40% of the forest area abandoned by 2100, when carbon sequestration payments are not integrated into the analysis ($pc=0$).

[Figure 3 about here]

Predictions of forest management continuity vary among the studied species and sites (note that our results show the share of forests that are managed). We forecast a higher abandonment trend in forests dominated by pine species, which present lower market profitability in comparison to oak species. In pine forests, such as those dominated by *Pinus halepensis* (Aleppo pine) and *P. nigra* (Black pine), this probability is less sensitive to changes in the market prices or discount rates. On the other hand, species such as *Quercus suber* (Cork oak) is highly sensitive to variations in cork prices and discount rates, and unless we simulate significant decreases (-50%) in the relative cork prices, our model forecasts a lower trend towards forest activity abandonment. Our results also show that about one third of current *Q. ilex* (Holm oak) forest area is expected to be abandoned in the future due to low expected market profitability across all the simulated scenarios, while the abandonment of forest management in the remaining two thirds of the *Q. ilex* area is sensitive to the simulated discount rates and price levels (Fig. 4).

[Figure 4 about here]

3.2 Consequences of abandonment of forest management

In most cases, continuing with forest management implies a reduction in the carbon sequestration potential, whereas abandonment is expected – at least in the medium term – to increase carbon stocks due to shrub encroachment and tree densification. It is estimated that

at the aggregated level, continuing with forest management would entail weighted⁶ average decreases of 16% in carbon sequestration levels in 2020, compared to forest abandonment in those tree cohorts and polygons that reach the rotation age in this period (see Fig 5). By 2050 it is expected that, in all cohorts that reach their rotation age by this period, continuing with forestry activity would decrease the carbon sequestration capacity by 40%. By 2100, on the other hand, we would expect forest sequestration capacity to increase by 21% relative to the abandonment scenario. This result is connected to the “rejuvenation” of forest stands due to regeneration treatments, which are expected to increase net carbon sequestration to a greater extent than the natural forests resulting from the forest management abandonment scenarios. Indeed, close to one half (49%) of the trees of the studied forest area in Andalusia would reach an age similar to, or higher than, the species- and management model-specific rotation ages by 2050 (Fig 2)⁷.

[Figure 5 about here]

Fig. 6 shows the spatial distribution of changes in annual carbon sequestration estimated for the periods 2020, 2050, and 2100 for the forest management scenario with respect to the abandonment scenario⁸. Carbon sequestration ability fluctuates, in most cases, between 1.5 and 10.0 CO₂ metric tons per hectare per year, being in many cases between 10% and 25% lower in the forest management scenario relative to abandonment. Our aggregated results confirm that there are, in general, higher carbon sequestration values in the abandonment scenarios relative to the management scenarios up to 2050; this result is inverted by 2100. There is however a wide spatial variability in the net carbon sequestration ability of abandoned forests relative to the managed ones. Likewise, our results show that in the very long term, continuing with forest management would enhance carbon sequestration services, although the opposite occurs in the medium to long term. As a general result, management leads to sustained or slightly increased carbon sequestration, with ups and downs due to the timing of rotation cycles, while abandonment leads to a progressive reduction in carbon sequestration (Fig. 5).

[Figure 6 about here]

Changes in economic water provision are also relevant, as shown in Fig. 5 and Fig. 6. With respect to the current situation (2010), it is estimated that the abandonment of forest management could lead, by 2050, to an aggregated reduction of 44.5% in the amount of economic surface water that flows out of the forest land and becomes available to the final

⁶ Weighted values according to the area occupied by each polygon with respect to the total studied area.

⁷ For example, note that about 70% of *Q. ilex* forest area would have trees of an age equal to or higher than the forest's rotation age by 2040 (Fig. 2), and that this species alone covers more than 55% of the studied forest area (Table 1). Hence, a large part of *Q. ilex* aged forests would need forestry treatments that induce natural regeneration in order to be renewed.

⁸ Those scenarios are based on forest management either being abandoned or continued through new investments on the total area that reaches the rotation age by 2050 and 2100.

users, with a similar value (43.7%) in 2100⁹. In the management scenario, on the other hand, this reduction drops to 9.3% in 2050 and 9.9% by 2100. The values depicted in Fig. 5 and Fig. 6 correspond to extreme scenarios in which all forest polygons undergo management or abandonment when the rotation age is reached, but not a mixed scenario in which both situations coexist, as in the simulations of Fig. 3 and Fig. 7. For a central scenario of a 3% discount rate and stable market prices (and no payments for carbon or water), the estimated reduction in economic surface water amounts to 24.5% and 26.0% in 2050 and 2100, respectively, corresponding to share of 51% and 36% of managed forests in Andalusia, respectively.

Our results show that forest management increases economic surface water in most of the polygons considered, especially in the Sierra Morena mountain range in north-western Andalusia, whereas forest management can increase economic water by more than 25% with respect to the abandonment scenario by 2050 and 2100. Forest abandonment is expected to increase biomass stock and consequently evapotranspiration, reducing the flow of water to regulated reservoirs (surface economic water), but also the total blue water made up by the sum of regulated and unregulated surface water and ground water recharge (Fig. 6). As shown in Fig. 5, evapotranspiration increases only moderately in the management scenario (1.5% in 2050 and 1.6% in 2100), but is significantly higher in the case of forest management abandonment (5.4% in 2050 and 2100). Ground water recharge is expected to decrease slightly in both scenarios, albeit at a lower rate in the management scenario (1.5% in 2050 and 2.9% in 2100) than in the abandonment scenario (2.7% and 3.0%). Given that evapotranspiration accounts for the largest share of the forest water budget in Mediterranean areas, representing almost 80% of the total water entering the system as precipitation, a small increase in relative terms can induce very important changes in the other components (surface discharge and ground water recharge).

3.3 Could payments for ecosystem services modify abandonment patterns?

PES are expected to be based on “additionality”, which refers to an increase in services compared with the absence of the project (Engel et al. 2008; Wunder, 2005). Measuring ex-ante additionality with respect to the baseline is challenging and surrounded by uncertainty, especially in the long term as in the case of forest interventions (Naeem et al., 2015). Our results indicate that preventing forestland abandonment is expected to have a significant and positive additional effect, in terms of both economic and non-economic water yield. Carbon sequestration is expected to be higher in the abandonment scenario over the next 20 to 30 years, but this situation is likely to be reversed in the long term (Fig. 5), generating in the management scenario additional carbon sequestration with respect to the abandonment scenario. There are also non-negligible uncertainties surrounding the additionality of carbon sequestration, as land abandonment is likely to increase wildfire risk (Fernandes et al., 2014).

⁹ This does not mean that the amount of water stored in the reservoir and therefore available to final users would decrease by the same margin, but it implies additional stress and difficulties for reaching the desired reservoir levels every season.

The integration of annual PES for carbon sequestration due to active forest management and payments for regulated (economic) surface water could generate incentives to invest in natural regeneration treatments. Additional carbon sequestration refers, in that case, only to carbon sequestration by tree growth of the new recruited individuals, assuming no carbon payments if forest management is abandoned. Our results suggest that the integration of payments for carbon sequestration in the new stands might be key for enhancing the multiple use of Mediterranean forests, (dashed lines in Fig. 3), in a way that encourages both market and environmental forest products, even with relatively low carbon price levels ($p_c=0.5$) and high discount rates ($dr=5\%$).

On the other hand, the effect of payments for economic surface water is marginal at the aggregated scale (Fig. 7). The reason is that only about 26% of the total forest land in Andalusia contributes to the economic surface water, while in the remaining areas the economic value of surface water is null (or very low) as these surface water flows occur (totally or partially) in non-regulated catchments (as shown in Fig. 6). This study does not account for payments for non-regulated surface and ground waters, which might bestow additional economic values over and above those connected to final water consumption by different agents and sectors in the economy. Also, those values might be related to the provision of other ES such as recreational values associated with water bodies, or the intermediate services that ecological water flows might provide for the maintenance of aquatic habitats, which in turn might also legitimize specific payments for maintaining certain levels of ecological water flows (Brauman et al. 2007)

[Fig. 7 about here]

When we look at the spatial effect of PES for carbon sequestration and for economic water, we find that carbon payments (for EU ETS 2010 prices and a 3% discount rate) decrease forest abandonment by 22% by 2050, with respect to the business as usual (BAU) scenario (no payments for carbon or water). Payments for economic water, on the other hand, would decrease abandonment only by 7% with respect to the BAU scenario. In general, PES for carbon sequestration generates more incentives to forest investment, except in very specific areas of north Andalusia (Cordoba province) where PES for water promotes more forest regeneration investment (Fig. 8).

[Fig. 8 about here]

We estimate the break-even carbon and water prices and their sensitivity to discounting rates and price levels of forestry products (Table 2). Water break-even prices are calculated considering different water yields, aside from the strictly economic. Those break-even prices are the minimum unit prices for carbon or water that would make land owners indifferent to investing in forest regeneration treatments or abandoning land management. Carbon and water break-even prices are quite sensitive to the discount and price scenarios simulated, with average fluctuations from €2 t CO⁻² to more than €12 t CO⁻² in the case of carbon, and

between $\text{€}0.06 \text{ m}^{-3}$ and $\text{€}0.88 \text{ m}^{-3}$ in the case of economic surface water. Carbon break-even prices are in the range of the EU ETS prices observed since 2010 (SENDECO2, 2017). In general, break-even prices for economic surface water are many times higher than the environmental water price estimated on the basis of the Berbel and Mesa (2007) hedonic water pricing study; however those prices fit within the irrigation water productivity ranges estimated by Borrego-Marín et al. (2016) for the Guadalquivir Basin ($\text{€}1.33 \text{ m}^{-3}$, for the period 2004-2012). The break-even prices for ecological water would fluctuate between $\text{€}0.01 \text{ m}^{-3}$ and $\text{€}0.14 \text{ m}^{-3}$, implying minimum annual payments between $\text{€}4.5 \text{ ha}^{-1}$ and $\text{€}45 \text{ ha}^{-1}$ either for carbon or water services. Those values are a simple reference, defining the minimum PES required to cover production costs, including the opportunity cost of using the forest for grazing purposes for 20 years, but excluding any transaction costs. Ideally, any payment for environmental water should account for the utility that watershed services have to individuals, which in turn is determined by their preferences based on their observed or stated willingness to pay for a given good or service (Mueller et al. 2013).

[Table 2 about here]

4 DISCUSSION

The integration of forest management and economic decision models with an eco-hydrological model to analyze the spatial and temporal effects of forest management decisions on water supply, carbon sequestration, and other forest provisioning services is a relevant contribution to the incipient literature addressing the interactions between forest and water resource decisions. Although significant progress has been made towards a better understanding of these interactions (Vose et al., 2011), to date there are few studies that integrate economic aspects into their analysis (e.g., Abildtrup and Strange 2000; Cademus et al. 2014; Nordblom, et al. 2012), especially in Mediterranean areas (e.g., Garcia-Prats et al. 2016; Kucuker and Baskent 2015). The regional spatial scale and detail of our work constitutes a reference point for studying the effects of economic forest management decisions on forest and water resources and the services that flow from them, to better inform management decisions and environmental policy design.

The results of this study suggest that the continuity of forest management in Andalusian forests can be significantly curbed by the low profitability of traditional market forest products and management models. The observed trend towards forest activity abandonment is slightly slowed or accelerated by changes in the relative prices of traditional forest products and inter-temporal consumption preferences that affect discount rates. Nonetheless, in general we expect that the abandonment of current forest management will be accentuated in the coming decades in the absence of additional incentives. Otherwise, future forestry activity might move towards more profitable forest species or production models, which are not analyzed in this study. Our results for the medium to long terms (i.e. up to 2050), similar to Cademus et al. (2014) and Kucuker and Baskent (2015), show trade-offs more than synergies in the production of climate change regulation services and water supply in Mediterranean forests, though managing the forest is likely to reverse this situation in the long term.

PES, especially for carbon sequestration resulting from active forest management, can generate relevant incentives for the continuity of multiple use forests in Andalusia. This result may seem paradoxical, as forest management abandonment is expected to increase the carbon stock in the medium to long terms (2020-2050) relative to continued management. In this case, it should be noted that the payments for carbon sequestration services as simulated in this study are designed to recognize carbon sequestration resulting from active forest management (those operations inducing forest regeneration), rather than natural processes such as revegetation that occur in the case of land use abandonment. Therefore, simulated carbon payments consider only carbon sequestration in new forest stands, and only if the regeneration investment is successful. Additional carbon payments might also favor the provision of other ES linked to silvopastoral landscape conservation such as public recreation, scenic values, and conservation of endangered biodiversity species. These aspects, not considered here, may legitimate government payments to encourage the sustainable management of forest resources.

PES for carbon sequestration may favor future water supply even more than specific payments for water yields. This result relates to the fact that, in a relevant share of the forest area of Andalusia, water is not regulated and therefore would not be entitled (in principle) to receive payments for the water supply service. More research is needed to determine people's demand and preferences regarding other recreational and biodiversity conservation values associated with water bodies, in order to justify PES beyond irrigation as the main water demanding sector in the region (Velázquez, 2007). In this paper we do not analyze the effect of simultaneous payments for water and carbon, given the trade-offs between these two ecosystem services. We also ignore the impact that non-monetary motivations may have on forest management decisions, such as amenities that a landowner may obtain from active forest management (Campos et al., 2017). The study also ignores the effect of landowner's risk preferences on long term forest investment decisions (Andersson and Gong, 2010).

The spatial analysis of the consequences of forest management and abandonment scenarios can help the design of incentives to enhance effective and spatially targeted environmental and rural development policies to boost either water or carbon sequestration services. Our spatially distributed results show, for example, that forest management can significantly increase the provision of economic water in some specific areas of north Andalusia, where PES for water are expected to favor more forest regeneration investment than payments for carbon (Fig. 8).

In this paper we also show that forest management clearly favors blue water yield, and that the payments to encourage forest management might be moderate (e.g. less than €50 per hectare per year). Water scarcity is a reality in Mediterranean areas, and it is likely to be magnified by climate change (IPCC, 2014). The definition of appropriate mechanisms to encourage new markets for water provision in Andalusia remains open, and current policy distortions affecting water resource management in this region cannot be overlooked (Velázquez, 2007). Yet, connecting upstream land use decisions with downstream water

demands within well-defined watershed boundaries seems desirable, and can help increasing the efficiency of water resource management in this region.

The simulated scheme of PES for carbon in this paper only recognizes active forest management, even though forest abandonment might increase climate regulation services (at least in the medium term). Nonetheless, the seeming lack of additionality of forest management on climate change regulating services is uncertain. Our carbon sequestration estimates for the forest abandonment scenarios consider a simplified model and assumptions that ignore complex ecological succession dynamics in forest ecosystems. Moreover, our estimates presume that the abandonment of forest management would not increase forest fires risk, even though increasing biomass stocks might raise potential wildfire risks (Fernandes et al., 2014). Accordingly, the results of the abandonment scenarios in terms of potential carbon sequestration might be overvalued.

Our forest growth, yield, and hydrological models neglect the potential effects of, and uncertainties involved in, changing climatic conditions, which are expected to be relevant in Mediterranean areas (García-Ruiz et al. 2011; IPCC, 2014). In order to partially overcome this limitation, this study provides a sensitivity analysis to different economic variables covering a wide range of potential variations affecting forest benefits, and hence management decisions. We examine the potential effect of increases or decreases in net market revenues and environmental benefits delivered by forests, which could in turn respond to increases or decreases in prices or cost levels, or in forest productivity due to market or technological innovations or climate change effects. The range and variation of the results obtained for different price and discounting scenarios highlight the relevant economic uncertainty concerning the future provision of forest ES.

5 CONCLUSIONS

The results of this study suggest a marked trend towards the abandonment of forest management in Andalusia in the absence of incentives other than those derived from forest market products. This is slightly affected by the assumptions on discount rates and price levels on the probability of abandoning traditional management in Andalusian forests.

Our results suggest trade-offs more than synergies in the production of climate change regulation services through carbon sequestration and water supply in Mediterranean forests in the short- to medium term (up to 2050), with synergies between carbon sequestration and water production in the longer term (from 2060 to 2100). The abandonment of forest management is expected to reduce surface water discharge and ground water recharge, as tree and shrub biomass stocks are expected to increase along with their evapotranspiration. Payments for additional carbon sequestration resulting from active forest management might be a key factor for enhancing the long-term conservation of multiple use of Mediterranean forests, favoring water production more than direct payments for regulated surface economic water. However, in contrast with carbon sequestration, forest investment clearly delivers additional water outcomes. More research is nonetheless needed to elicit individual

preferences and willingness to pay for enhancing the provision of environmental water as an alternative PES option.

This study represents a benchmark for developing an integrated hydro-economic model for simultaneous management of forest and water resources in Mediterranean forest; further research is required to make this model more comprehensive to better represent the complexity of interactions between natural and economic systems and inherent uncertainties. A key element in future research is the analysis of the effects of changing climatic conditions on forest production and hydrological balance, particularly with regards to the likely intensification of forest fires and seasonal water droughts. Also, more research is needed to examine production alternatives to the traditional forest management models analyzed in this study, responding to changes in the demand for market forest products and ecosystem services.

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TABLES

Table 1 Initial forest species distribution in Andalusia in 2010

Forest species		Pure and mixed forest area	
Common name	Scientific name	(ha)	(%)
Scots fir	<i>Pinus sylvestris</i>	30,182	1.3
Stone pine	<i>Pinus pinea</i>	195,750	8.3
Aleppo pine	<i>Pinus halepensis</i>	268,503	11.4
Black pine	<i>Pinus nigra</i>	90,397	3.8
Maritime pine	<i>Pinus pinaster</i>	145,972	6.2
Holm oak	<i>Quercus ilex</i>	1,394,281	59.0
Cork oak	<i>Quercus suber</i>	239,945	10.1
Total		2,365,030	100.0

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

Table 2 Break-even prices for carbon sequestration and water resources

Discount rate	Price level forestry products	Average unit annual payments					
		Carbon sequestration	Economic surface water	Total surface water	Ground water	Total Water ⁽¹⁾	Ecological water ⁽²⁾
		€/t CO ₂	€/m ³	€/m ³	€/m ³	€/m ³	€/m ³
2	50	2.79	0.12	0.03	0.10	0.02	0.04
	100	1.95	0.08	0.02	0.07	0.02	0.03
	150	1.49	0.06	0.02	0.05	0.01	0.02
3	50	5.89	0.28	0.07	0.22	0.05	0.09
	100	4.33	0.19	0.05	0.16	0.04	0.06
	150	3.51	0.15	0.04	0.13	0.03	0.05
4	50	9.79	0.47	0.11	0.37	0.08	0.14
	100	7.19	0.34	0.08	0.27	0.06	0.10
	150	5.94	0.26	0.06	0.22	0.05	0.08
5	50	12.51	0.88	0.16	0.52	0.12	0.19
	100	10.08	0.60	0.12	0.42	0.10	0.16
	150	8.36	0.49	0.10	0.34	0.08	0.13

Notes: ⁽¹⁾ Total water quantity accounts for the sum of ground and total surface water. ⁽²⁾ The quantity of ecological water is estimated as the difference between total surface water and the economic surface water.

FIGURES

Fig. 1 Model components, interrelations and outcomes

Rotation	<i>Q. ilex</i>	<i>Q. suber</i>	<i>P. sylvestris</i>	<i>P. pinea</i>	<i>P. halepensis</i>	<i>P. nigra</i>	<i>P. pinaster</i>
In years	42-190	103-141	105-131	103-126	80-151	84-105	74-86

Fig. 2 Share of forest area occupied by trees of an age equal to or higher than the rotation age by forest species and years*

*Note: that to construct this figure we assume that there will be no investment in facilitating forest natural regeneration.

Fig. 3 Prediction of the share of managed forests for different: discount rates (dr), price levels of market forest products (pt) and the level of carbon prices (pc) in Andalusia.

Note: The solid black line represents a business-as-usual situation, i.e. with no PES for carbon sequestration ($pc=0$), while the dashed lines represent situations with PES for carbon ($pc>0$).

Fig. 4 Prediction of the share of managed forest for different forest species, discount rates (dr) and price level of market forest products (pt) in Andalusia

Fig. 5 Expected evolution of evapotranspiration, carbon sequestration, economic surface water and ground water recharge for the forest management and abandonment scenarios ⁽¹⁾

Note: ⁽¹⁾ Weighted average values according to the surface area of each forest unit.

a) Changes carbon sequestration per hectare and year (in percentage)

b) Changes in economic water per hectare and year (in percentage)

c) Changes in total blue water per hectare and year (in percentage)

Fig. 6 Changes in carbon sequestration, economic water and total blue water of the forest management scenario with respect to the abandonment scenario by 2020, 2050 and 2100 in Andalusia

Fig. 7 Prediction of the share of managed forests for different discount rates (dr), price levels of market forest products (pt) and the level of water prices (pw) in Andalusia.

Note: The solid black line represents a business-as-usual situation, i.e. with no PES for water ($pw=0$), while the dashed lines represent situations with PES for water ($pw>0$).

b) Payments for carbon sequestration

b) Payments for economic water

Fig. 8 Distribution of managed and abandoned forest for the business as usual (BAU), payments for carbon sequestration, and payments for economic water in Andalusia by 2050.

Note: Data for the central scenario, average prices of 2010, and a discount rate of 3%. Potential avoided abandonment thanks to PES for carbon sequestration or for economic water indicate those areas which abandonment could be potentially avoided through those PES schemes.