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ORIGINAL RESEARCH

Valuation of Ecosystem Services: A Source of Financing Mediterranean Loss-Making Forests

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Abstract

Forests provide market and non-market priced ecosystem services (ES). Mediterranean forests, with low timber productivity, have frequently negative economic balances, despite their significant ES production. Forest planning tools accurately account for the investments required to maintain forests, but not for benefits, because they only include market ES but not non-market ones. The aim of this study is to analyse the economic balance for five Spanish forests, incorporating actual operating and maintenance costs, and benefits from both market ES (which are currently being accounted for) and non-market ES (currently not considered). Non-market priced ES included are carbon sequestration, erosion control, watershed protection, biodiversity conservation, landscape protection and recreation. At present, all forests studied are loss-making, with losses of 60–370 €/ha·yr. The valuation and inclusion into the economic balances of all ES would result in a positive balance of 130–938 €/ha·year, which would imply an opportunity cost of using the land in forestry of 3%, higher in public forests than in private ones due to the recreational use of the former. Market-priced ES only represent around 1% of the total, due to the lack of timber production. Valuation of ES is a useful tool to highlight the benefits of forest ecosystems, and the need to maintain them. A major challenge is to convert this economic valuation into actual income.

Keywords Forest management · Forest economy · Environmental goods · Ecosystem services · Spain

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Introduction

Forests have multiple uses, providing environmental goods and services. Costanza et al. (1997) introduced the concept "ecosystem services" (ES) defining 17 main ones, which included both ecosystem goods and services. MEA (2005) proposed a widely used classification, which divides ES into provisioning (products and materials), regulating (regulation of ecosystem processes and the environment), cultural (non-material benefits) and supporting (underpinning services that enable other services to function). The Common International Classification of Ecosystem Services proposed a five-level hierarchy for ES, with three main sections, provisioning, regulating and maintenance, and cultural (Haines-Young and Potschin 2018).

Assessment of ES can be done quantitatively or qualitatively, each approach with benefits and drawbacks; the former requires accurate data, allowing ES to be monetised and considered in decision-making (although some ES may be undervalued), while the latter allows for a more comprehensive analysis as a whole, although less practical (Busch et al. 2021).

ES have economic value (Pearce 1998), but it is usually not easy to determine, especially when they have not material benefits or market value (Small et al. 2017); often the importance of some ES is appreciated only upon their loss (Daily et al. 2000). ES valuation attempts to measure changes in welfare through willingness to pay (WTP) or willingness to accept (WTA) compensation (Pearce 1998; Bockstael et al. 2000; DEFRA 2007 it is an instrumentalist, utilitarian and homocentric economic approach (Randal 1987), which aims to indicate the effect of a marginal change in the provision of ecosystem services in terms of a rate of compensation relative to other things people value (Turner et al. 2003). Although ES valuation is difficult, it is essential for decision-makers (Kumar and Kumar 2008). ES economic value is related with its contribution to human welfare, which depend on each individual's own assessment (Bockstael et al. 2000). Market imperfections may cause that ES market prices fail to reflect consumers' WTP for them. ES valuation—and the very concept of ES—has been criticised for being anthropocentric and for promoting the exploitation or commodification of nature (Schröter et al. 2014).

Total economic value (TEV) of an environmental asset is defined by the net sum of WTP and WTA, including use and non-use or passive values (Pearce et al. 2006; Riera et al. 2012). WTP and WTA are used to measure marginal changes in welfare. However, there may be large differences between individuals, and even antagonistic opinions, as detected by Aguilar et al. (2018) on WTP for watershed conservation in a survey in US households. The valuation can be done using market-based methods—market prices, production function, avoided damages, replacement costs—or non-market-based methods—revealed or stated preferences—(Mitchell and Carson 1989; Kumar and Kumar 2008; Pröbstl-Haider 2015). Values calculate marginal changes in welfare for beneficiaries of ES in specific locations, and methods capture different components of TEV, so there is a wide disparity of ES values in the literature.

Market prices, which seem more reliable, often do not reflect all the social costs (Daily et al. 2000). In turn, the economic valuation of non-market-priced

ES is an abstraction, precisely because there is no market for them. There are also other methodological difficulties, such as the risk of double counting (Fu et al. 2011; Turner et al. 2003). Consequently, each valuation method has benefits and drawbacks (Pascual et al. 2010), and may be subject to uncertainties and biases, but they are useful for decision-making (National Research Council 2005).

There are two approaches for valuing ES (Pagiola et al. 2004). The first is the valuation of the total flow of benefits, which clarifies contributions that ecosystems make to economic activity (a form of accounting). The second one is the valuation of changes in flows, used to examine the consequences of ecosystem degradation or assess the benefits of a conservation intervention (necessary for policy making). The first approach has difficulties such as drawing spatial and temporal links boundaries, and limitations in assessing the benefits of an intervention. However, it is useful for identifying the benefits that an ecosystem (such as a forest) provides to the society (Pagiola et al. 2004); this is the approach followed in this study, analysed from the perspective of society, hence it is not a private analysis. As Le et al. (2012) point out financial viability is restricted to private cash returns only, while economic viability is determined from the perspective of the community or society.

Forests need investments for conservation and improvement and produce goods and services. Well-managed forest has planning tools that define investments and benefits; however, accounting for investments is often accurate, while accounting for benefits is poor, because only market-priced ES are taken into consideration (and with the limitations noted above on their valuation).

In the Mediterranean region, many forests have low timber productivity, but nevertheless require expensive silvicultural treatments and preventive measures against wildfires (FAO-Plan Bleu 2018). Non-timber products, such as cork, mushrooms or pine nuts, or activities such as grazing and hunting, may produce some income, but forests in this region are frequently loss-making. In addition, there has been a decline in the benefits associated with forest products over the last decades (Ovando et al. 2019). Negative results in Mediterranean forests discourage active forestry (Górriz-Mifsud et al. 2016); compared to the agricultural sector, forest owners have very few economic incentives (Bösch et al. 2018). For example, many cereal crops are loss-making, but they are maintained thanks to subsidies from the European Union's common agricultural policy, which is not the case of forests. Forests provide ES with economic value to society, but their owners/managers are not financially (or privately) compensated for their provision (Valls et al. 2012; Bösch et al. 2018); despite their environmental value, forests can be a burden to their owners.

This is a major problem for forest management. In private forests the lack of profitability leads to abandonment (Valls et al. 2012), unless subsidies are received. In public forests, governments may assume losses, but when the balance is negative investments are frequently reduced. Adequate valuation of the ES is necessary, so that managers, politicians and decision makers know the real contribution of ecosystems to economic activity. A forest without investment is being "abused" because it is producing much more benefits than those associated with market-priced ES. An objective valuation of ES would allow more realistic forest planning.

Several studies have analysed the ES of forests in the Mediterranean region (Croitoru and Merlo 2005; Croitoru 2007; Górriz-Mifsud et al. 2016), in Spain (Campos et al. 2005; Esteban 2010), or in certain ecosystems, such as agroforests (Campos and Mariscal 2003; Mesa et al. 2016; Campos et al. 2020) or pine forests (Caparrós et al. 2001; Campos et al. 2021), and there are also partial studies focusing on aspects such as soil and erosion (Colombo and Calatrava 2003; Hein 2007) or recreation (Caparrós and Campos 2002; Voces et al. 2010). Other studies compare management alternatives associated with payments for ecosystem services (PES), such as carbon sequestration versus water harvesting (Ovando et al. 2019) or timber production (Enriquez-de-Salamanca 2021).

Although there is growing knowledge about ES, there is a significant gap between the macro and the local level, and between theoretical conception and practice; valuation approaches are useful to explain ES at a macro level, but local valuations relevant for decision making were hindered by data-scarcity (Pandeya et al. 2016). Despite a growing presence of ES assessment in the literature, its actual usefulness in policy change is little known (TEEB 2009; Laurans and Mermet 2014). Edens and Hein (2013) propose starting with pilot studies focusing on specific ecosystem services with ample data available.

The aim of this study is to partly fill this gap between the macro and local levels, and between a global conception of ES valuation and a more practical one, applied to forest management. For this purpose, five forests located in Central Spain have been selected, carrying out an assessment and valuation of the main ES provided. The reasons for selecting these forests are several: (i) all have forest management plans, with real information on stocks, growth, income and expenditure; (ii) they are representative of Mediterranean Spain, as dominant species are among the most frequent in the region; (iii) they are located nearby, at a maximum distance of 35 km; (iv) they include public and private forests; (v) they differ in terms of public use, allowing for a more diverse sample; (vi) all are loss-making forests, with negative economic results if the appropriate management work is undertaken, a common occurrence in the region.

A novelty of this paper is to incorporate the valued ES into the real forests economic balances, connecting a topic usually addressed from a conceptual point of view, with the actual financing need of these ecosystems. Several reviews about ES valuation (Czúcz et al. 2018; Acharya et al. 2019) conclude that studies on ES tend to focus on regulating services, especially carbon sequestration, and cultural services, especially recreation. We have integrated provisioning services with the main regulating and cultural services, proposing calculation methods and valuation based on the extensive literature on this topic. The ultimate aim is to demonstrate the positive contribution of these forest ecosystems to the local economy, which could eventually motivate more detailed analyses to justify payments to their managers for proper management and conservation.

Methods

Research Questions and Approach

The questions that this research intends to answer are: What are the ecosystems services of the studied forests? What is its contribution to the regional economy and to the welfare of people? According to Pagiola et al. (2004), these kinds of questions correspond to a valuation of the total flow of benefits of the forests. The forests have management plans approved by the Regional Government and based on scientific and technical criteria. As a consequence, it is not intended to evaluate management alternatives, valuing changes in flows between them.

The conservation of all these forests depends on public investment; although two of them are private, they have consortia with the Regional Government, who is in charge of their conservation. Although the forests are loss-making, they are maintained because the Regional Government assumes the losses; it is a political decision, which responds to a social demand of conservation. The Regional Government really knows that these forests provide ES that society demands, and invests for their conservation, but it neither knows what these ES really are nor their value. Consequently, the aim of this paper is to identify and value the total flow of net benefits from the ES of these forests, in order to clarify their contributions to economic activity from the perspective of the society.

Study Area

The study area included five forests located in the Community of Madrid (Central Spain), three publicly owned and two privately, although with consortia with the regional government. The consortia of the private forests were signed in the 1950s by the State (there were no regional governments until 1984), as a result of a State`s decree for the forced reforestation of deforested areas. These consortia implied that the State was in charge of the reforestation, and owned the trees, while owners retained ownership of the land; the management of the stand was carried out by the State, and the profits, minus reforestation and maintenance costs, were shared equally. In 1984 the consortia passed to the Community of Madrid, the regional government. In practice, these forests have never made a profit (nor are they expected to), but the maintenance of the forest and the prevention of forest fires is the responsibility of the regional government.

Average altitude is 700–1340 m, annual rainfall 459–884 mm and mean average annual temperature 13.1–14.3 °C. All the forests are on siliceous terrain, and were reforested, at least in part (Fig. 1, Table 1).

Boadilla-Encinas (Fig. 2A) is a public forest surrounded by built-up areas, with intense local recreational use. It is a forest of *Quercus rotundifolia* Lam., with some areas reforested with *Pinus pinea* L. in the mid-twentieth century. In 1991 a wildfire affected 70 ha. Despite the intense recreational use there are nests of threatened birds. It only has one management plan, from 2019.

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Fig. 1 Study area

Cuerda Herrera (Fig. 2B) is a private forest, reforested with *Pinus pinea* in 1956 through a consortium with the government. It lacks recreational use, and the one its owners do is moderate. There is only one management plan, from 2009.

Jurisdicción (Fig. 2C) is one of the most emblematic public forests in Madrid. Located in a tourist municipality and with a scenic landscape, it was reforested twice, in the early and mid-twentieth century. It has an intense recreational use. In 1999 a wildfire affected 100 ha. The first management plan was from 1956, and has had four revisions, the last one in 2014.

Monte Agudillo (Fig. 2D) is a public forest covered mainly by *Pinus pinea* and *P. pinaster* Aiton. It was harvested for pine nuts and resin until 1966, when the forest suffered a terrible wildfire, losing most of the trees. Reforested in 1967, *P. pinaster* was not successful, but *Quercus rotundifolia* colonized large parts of the planted areas. Recreational use is very scarce. The first management plan is from 1902, and has had seven revisions, the last one in 2016.

Ventilla-Vinatea (Fig. 2E) is a group of private forests, reforested with *Pinus pinea* between 1954 and 1961 through a consortium with the government. *Quercus rotundifolia* have regrown in many areas after clear cuttings were

Table 1 Main characteristics of the studied forests

Forest name	Area (ha)	Property	FMP	E (m)	S (%)	R (mm)	T (°C)	L	Main tree species
Boadilla—Encinas	910.4	Public	2019	704	7	520	14.1	Arkose	*Quercus ilex*, *Pinus pinea*, *Fraxinus angustifolia*, *Quercus faginea*
Cuerda Herrera	207.1	Private with consortium	2009	790	20	619	13.8	Gneis	*Pinus pinea*, *Juniperus oxycedrus*, *Quercus ilex*, *Fraxinus angustifolia*
Jurisdicción	847.5	Public	2014	1340	40	1109	9.4	Gneis	*Pinus sylvestris*, *P. pinaster*, *P. nigra*, *Fraxinus angustifolia*
Monte Agudillo	1,212.2	Public	2016	800	35	630	13.8	Granite	*Pinus pinea*, *P. pinaster*, *Quercus ilex*, *Fraxinus angustifolia*
Ventilla—Vinatea	586.2	Private with consortium	2009	700	20	515	14.1	Granite, arkose	*Pinus pinea*, *Juniperus oxycedrus*, *Quercus ilex*, *Fraxinus angustifolia*

FMP Forest Management Plan; *E* Average elevation; *S* Average slope; *R* Average annual rainfall; *T* Average annual temperature; *L* Lithology

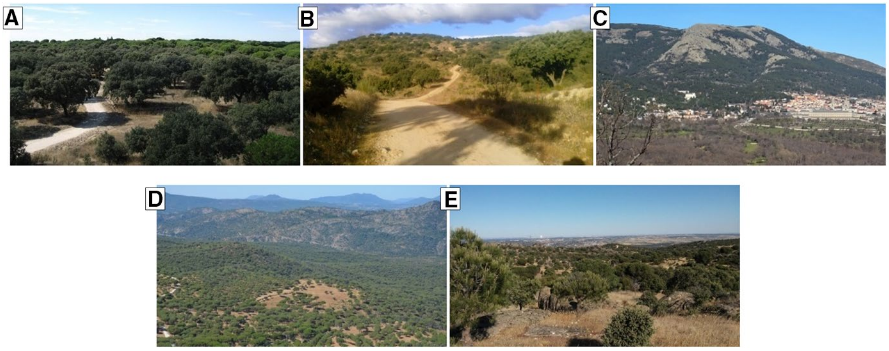

Fig. 2 Studied forests. **A** Boadilla-Encinas; **B** Cuerda Herrera; **C** Jurisdicción; **D** Monte Agudillo; **E** Ventilla-Vinatea

abandoned decades ago. There is neither recreation, nor by their owners. It has only one management plan, from 2009.

There are many beneficiaries of the ES of these forests, as they are located in Madrid, the most populous region in Spain, with more than 6.5 million inhabitants. The forests contribute to the conservation of the landscape and nature of the region, which benefits the population, and can generate a WTP for that conservation. In addition, the public forests have free access, which makes it possible to determine a WTP for their use, especially important in Jurisdicción (one of the best known and most publicly used forest in Madrid), and in Boadilla (with a population of 60,000 inhabitants bordering with the forest).

Current Economic Balance

Prices, including those obtained from management plans or from the literature, refer to € of 2020. Results are expressed in €/ha·yr, to make easier comparison. Current economic balances were obtained from management plans, except staff costs, specifically calculated. The concepts included were:

- *Expenses* (i) Silviculture (thinning, pruning). (ii) Reforestation and vegetation improvement (plantation, sowing). (iii) Forest fire prevention (firebreaks). (iv) Pest control and wildlife protection. (v) Infrastructure maintenance (tracks, water points). (vi) Cleaning (waste removal) and recreation regulation (signalling, barriers). (vii) Staff: management (administration, studies, projects) and protection (rangers).
- *Income* (i) Timber from thinning (poor quality: irregular in dimensions, partly from dead trees, expensive to extract, and without regular production, so the prize is lower than regular timber in the region). (ii) Non-timber products: honey (minimal value in these forests) and pine nuts (limited by pine regeneration problems); others, such as mushrooms or berries, do not produce income, because there are no permit fees for their collection. (iii) Grazing (annual concessions); the existing livestock were cows in Jurisdicción, sheep in Boadilla-Encinas

and goats in Monte Agudillo, while the private forests have no livestock use. (iv) Hunting (decennial concessions).

The management plans establish all the necessary investments for forests' maintenance and improvement and determine permitted uses and exploitation (e.g., annual timber extraction or permissible livestock density), so it is not possible obtaining additional income from forest goods. We have excluded from this study non-ES, such as easements and occupation permits, because they are not inherent to the ecosystem functioning; if economic sustainability depends on income from services unrelated to forest ecosystem functions, this opens the way to potentially destructive processes (urbanisation, agriculture, occupation, etc.).

Valuation of Non-market Priced Ecosystem Services

The most representative non-market priced ES have been selected: carbon sequestration, erosion control, watershed protection, biodiversity conservation, landscape protection and recreation. This selection includes the most important ES, but at the same time is concise enough to avoid double counting, a common problem in this type of valuation (Riera et al. 2012). For each ES we included a calculation method, an analysis of the values proposed in the literature, and a valuation proposal. The information is included in the results section because it is more coherent than dividing it in the methods, results and discussion sections. Results have been incorporated into the economic balances.

Results

Current Balance

Expenses in the studied forests ranged from 84 to 358 €/ha·yr (mean of 228 €/ha·yr); the highest values were in private forests, without management plans until 2009, where no investment had been made for decades; when pending investments were made, the annual costs will be lower. Maintenance costs of nearby public forest (25–60 km distance from the studies forests) based on management plans (Velazquez 2008; Cabrera 2010) and estimated staff costs, range from 94 to 251 €/ha·yr.

Current income from market ES ranged from 1 to 22 €/ha·yr. Values are really low because the forests do not produce timber. Although they are mostly pine plantations, their management focus in protection, and timber production is limited to conservation thinning. In nearby forests income ranges from 16 €/ha·yr (main income from grazing with limited timber production) to 192 €/ha·yr (important timber production, rare in the region).

Current balances showed losses in all the forests, ranging from 62 to 356 €/ha·yr, with the highest figures in private ones (Table 2). In the nearby forests mentioned above losses range from 58 to 71 €/ha·yr; even the forest producing quality timber is loss-making. Górriz-Mifsud et al. (2016) report losses in the forests of NE

Table 2 Current forests' budget (€/ha·yr)

Concept	Boadilla—Encinas	Cuerda Herrera	Jurisdicción	Monte Agudillo	Ventilla—Vinatea
Expenses	−153.06	−358.48	−211.39	−84.44	−332.23
Income	2.96	2.60	17.49	22.11	1.33
Balance	−150.10	−355.78	−193.90	−62.33	−330.90

Detailed information in Table 13

Table 3 Valuation of carbon sequestration

Forest name	Sequestration	
	t CO_2/ha·yr	€/ha·yr
Boadilla-Encinas	1.11	27.47
Cuerda Herrera	2.60	64.35
Jurisdicción	2.62	64.85
Monte Agudillo	1.21	29.95
Ventilla-Vinatea	2.07	51.23

Spain of 7–125 €/ha·yr depending on scenarios, with the lowest value in passive management.

Carbon Sequestration

Carbon sequestration is the amount of CO2 removed from the atmosphere by vegetation and fixed in plant tissues. The calculation was done using forest inventory data and growth equations (IFN3 2000) and valuation using the average price of Emission Rights in the EU in 2020 (24.75 €/t CO2); results were 30–65 €/ha·yr (Table 3). There is a huge disparity of values in the literature (e.g., 9 €/ha·yr in Spain, Caparrós et al. 2001; 14–21 €/ha·yr in northern boreal forests, Turner et al. 2003; 895 €/ha·yr in Southern Europe, Sukhdev 2008).

Erosion Control

Vegetation cover reduces soil losses, erosive damage, and silting in deposition areas. Water erosion can be determined through the Universal Soil Loss Equation, USLE (Wischmeyer and Smith 1978), revised (RUSLE) by Renard et al. (1991); it is considered the best available method to estimate soil losses for erosion inventories and mapping (MMA 2002). Several publications and mapping servers include erosion values throughout Spain (ICONA 1987; MMA 2002; MAPA 2021; MITECO 2021a). Using this information, we calculated current soil losses and those resulting if the forest disappeared; the difference is the erosion avoided by the forest. Guerra et al. (2014) follow a similar scheme in Portugal.

The economic valuation of the avoided erosion is more complex. For example, Colombo and Calatrava (2003) valued the reduction of erosion in S Spain; Darmendrail et al. (2004) the economic impacts of soil degradation; Hein (2007) the local costs of land degradation in a catchment in southeast Spain; Esteban (2010) through avoided costs of reservoir cleaning as a result of loss of capacity through sediment deposition; and Kuhlman et al. (2010) the cost of agricultural practices for erosion control (Table 4).

On-site or private costs focus on the site experiencing erosion, and include loss of soil fertility, changes in crop yields, uprooting of plants or formation of rills and gullies. Off-site or social costs occur outside the site affected by erosion, including damage to property and infrastructure, water pollution, reduction of the soil's water retention capacity, alteration of runoff or disruption of natural ecosystems (Darmendrail et al. 2004); social costs exceed private costs. The average value of private costs obtained was 43 €/ha and the average value of private and social costs 156 €/ha; for this study, the latter has been considered, as it takes into account all impacts associated with erosion. The average annual soil loss due to erosion in Spain in 2019 was 12.2 t/ha·yr (MITECO 2020), which would imply a private and social cost of erosion of 12.80 €/t·yr. Considering the erosion levels of the forests, the results obtained were 16–166 €/ha·yr for erosion control (Table 5); the lowest value was obtained in a forest with gentle topography and low erosive problems, while the highest occurs in an steep forest, where the loss of the tree cover would have catastrophic results.

Watershed Protection

A widely recognized ES is hydrological protection, considered in different ways in the literature (Table 6). Prieto et al. (1999) indicated that forests increase infiltration by 90 m^3/ha; to value this ES we use an average water price in Spain of 0.40 €/m^3, obtained weighting water for agricultural use (80%) at an average price of 0.02 €/m^3, and for urban and industrial use (20%) with an average price of 1.91 €/m^3 (INE 2020; CEDEX 2021). Croitoru and Merlo (2005) and Croitoru (2007) included values for watershed protection. Esteban (2010) valued the provision of water for

Table 4 Examples of economic valuation of erosion

Area	€/ha·yr	€/t·yr	References
EU 25	120.00–296.00[a]	29.60–60.00	Kuhlman et al. (2010)
France	54.43–57.46[a]		Darmendrail et al. (2004)
France	105.84–128.52[b]		Darmendrail et al. (2004)
Spain	75.35–210.20[b]		Colombo and Calatrava (2003)
Spain	1.40–60.44[a]	0.20–0.50	Hein (2007)
Spain	19.05–25.78[c]		Esteban (2010)

[a]On-site or private cost

[b]On-site and off-site, or private and social cost

[c]Partial off-site cost

Table 5 Valuation of erosion control

Forest name	Erosion (t/ha·yr)			Value €/ha·yr
	Current	Without forest	Forest reduction	
Boadilla-Encinas	0.35	1.61	1.26	16.13
Cuerda Herrera	1.45	8.94	7.49	95.87
Jurisdicción	6.23	19.20	12.97	166.02
Monte Agudillo	2.11	8.76	6.65	85.12
Ventilla-Vinatea	9.65	19.20	9.55	122.24

Table 6 Examples of economic valuation of watersheds

Area	€/ha·yr	Criteria	References
France	8.00	Watershed protection	Croitoru and Merlo (2005)
Italy	104.00	Watershed protection	Croitoru and Merlo (2005)
Italy	189.00	Forest management compensation	Muys et al. (2014)
Mediterranean	17.40	Watershed protection	Croitoru and Merlo (2005)
Portugal	24.00	Watershed protection	Croitoru and Merlo (2005)
South Mediterranean	35.32–42.39	Watershed protection	Croitoru (2007)
Spain	36.00	Increased infiltration	Prieto et al. (1999)
Spain	236.50	Water for agriculture from forests	Esteban (2010)

agriculture from forests. In the Romagna area (Italy) part of the water tariff (1–3%) is used to compensate forest owners for changing management practices, reducing erosion and the amount of nitrogen in the water, so that the owners increase their income, and the water company reduces the cost of purification (Potenella et al. 2012; Muys et al. 2014).

Water balance (annual rainfall minus evapotranspiration/runoffs) is widely used to assess watersheds ES (e.g., Xue and Tisdell 2001; Llerena 2003; Biao et al. 2010; Mashayekhi et al. 2010; Ninan and Inoue 2013; Ninan and Kontoleon 2016). The water surplus proposed by Thornthwaite and Mather (1957), and used in this study, provides a more accurate calculation as it determines the water drained in an area by considering not only the precipitation and the actual evapotranspiration but also the soil water reserve. For valuation we used the average cost of water collection in Spain (0.027 €/m^3; Maestu and Villar 2007). Results ranged from 26 to 178 €/ha·yr (Table 7), the highest in the rainiest and most steep forest.

Biodiversity Protection

Forests are wildlife refuges, providing an ES for biodiversity conservation; the most diverse and unique the wildlife, the greater the value of the ES provided. The value of animal and plant biodiversity protection must be valued separately,

Table 7 Valuation of watershed protection

Forest name	Drainage m^3/ha·yr	Value €/ha·yr
Boadilla-Encinas	1,005.79	27.16
Cuerda Herrera	2,040.20	55.09
Jurisdicción	6,606.20	178.37
Monte Agudillo	2,157.60	58.26
Ventilla-Vinatea	956.60	25.83

since it is not necessarily associated: areas with a low botanical value may have unique animal species, and vice versa. Two valuation criteria were used: diversity or species richness, and uniqueness or presence of rare or threatened species. We calculated indicators combining both criteria, ranging from 0 (no biodiversity) to 1 (maximum possible biodiversity).

To assess animal richness, the number of vertebrate species has been used. MITECO (2021b) has databases and GIS of vertebrates in Spain, using 10 km UTM grids (there are no exhaustive data on invertebrates), establishing five richness categories: very low (< 50), low (51–80), medium (81–110), high (111–140) and very high (> 140). Following this scale, a value of 0–1 was established for each forest. To assess uniqueness, we used the presence of species included in the Spanish Catalogue of Endangered Species (BOE 2011), assigning additional values of 10 points to endangered species, and 5 points to vulnerable species.

Plant species total 7069 in Spain (Pando et al. 2021) and 6280 in the Iberian Peninsula (Domínguez and Schwartz 2005). Moreno (2011) analysed plant richness using 10 km UTM grids on a sample of 1670 species, detecting that the richest grid, in the Pyrenees, had 206; in Aragon (including Pyrenees) grids with more than 500 species were considered very rich (IPE 2005); in Burgos, a mean of 202 species/grid was registered, with a maximum of 813 (Alejandre et al. 2009); in Alicante, normal values were 200–500 species/grid (Serra 2007). We considered that above 500 species the diversity is very high, establishing a proportional scale: very low (< 200); low (201–300); medium (301–400); high (401–500); very high (> 500). Following this scale, a richness indicator was established for each forest. The uniqueness would be valued like in the fauna, but there were no threatened plant species in these forests.

These indicators assessed the relative importance of the biodiversity. To obtain an economic value, it was necessary to multiply them with a basic biodiversity value. There were numerous examples of valuation in the literature, using different criteria, but only a few in the Mediterranean region (Table 8).

The mean ($\overline{X}$) value of biodiversity was 263 €/ha, with a standard error ($\sigma_{\overline{X}}$) of ± 183 €/ha. The average value including other areas of Europe (Hanley et al. 1998, Scotland, 51.49–170.64 €/ha using WTP; Willis et al. 2003, Great Britain, 620.11 €/ha as annual biodiversity value; Lindhjem 2007, Finland, Norway and Sweden, 149.00 €/ha using WTP; Juutinen 2008, Finland, 281.62 €/ha using a preferences survey; Ding et al. 2010, Scandinavian Europe, 122.40–253.78 €/ha

Table 8 Examples of economic valuation of biodiversity

Area	€/ha·yr	Criteria	References
Mediterranean	1.41–84.78	Different methodologies	Croitoru (2007)
Mediterranean Europe	354.29–612.05	Passive use. Meta-analysis	Ding et al. (2010)

using passive use) was 245 €/ha; it is a consistent result, as the Mediterranean is the most biodiverse region on the continent, so its value is somewhat higher.

We have considered that an ecosystem with high biodiversity (a value of 1 on the assessment scale established) would have a value of 446 €/ha ($\overline{X}+\sigma_{\overline{X}}$). This value has been divided 50% for fauna and 50% for flora and was multiplied by the forests' quality indicator. The results were 203–386 €/ha·yr (Table 9); Jurisdicción had the higher result, due to its rich wildlife and the presence of unique species.

Landscape Protection

Forests contribute to landscape quality, conservation and naturalness. The valuation of the landscape is often mixed with other ES (e.g., Hermann et al. 2011), or associated with recreational use (e.g., Croitoru and Merlo 2005); that is not possible in this case because some forests have landscape value but not intrinsic recreational use (only for external observers).

BLM (1986) established a landscape assessment method, widely applied, based on visual quality, calculated by landform, vegetation, water, colour, adjacent scenery, singularity and cultural modifications. This method establishes three classes, A, B and C, with results ranging from 0 to 33 points; we have converted these figures into a 0–1 scale.

Table 9 Valuation of biodiversity

Forest name	Group	Div[1]	Sing[2]	Ind[3]	Value (€/ha·yr)	
Boadilla-Encinas	Fauna	91	2E,1 V	0.64	142.72	202.93
	Flora	237	–	0.27	60.21	
Cuerda Herrera	Fauna	109	1 V	0.63	140.49	202.93
	Flora	240	–	0.28	62.44	
Jurisdicción	Fauna	142	1E,3 V	0.90	200.70	385.79
	Flora	587	–	0.83	185.09	
Monte Agudillo	Fauna	132	2E,3 V	0.90	200.70	278.75
	Flora	274	–	0.35	78.05	
Ventilla-Vinatea	Fauna	107	1 V	0.62	138.26	207.39
	Flora	253	–	0.31	69.13	

[1]Diversity (number of species)

[2]Singularity (E: Endangered species, V: Vulnerable species)

[3]Biodiversity indicator

Martínez (1996) proposed an equation to assess the loss of landscape values in forest fires in Spain, $P=0.65\ S_{ru}\ ((1+r)^{n-1}-1)/(1+r)^{n})$, where S_{ru} is the value of rural land, r the interest rate and n the time in years to mitigate the impact. The result is the total value of the landscape impact, so to determine the annual value we divide P by n.

Public forests are unsaleable, so there are no market prices; we have applied for calculations the same price as for private forests. The sale value of forest land in 2020 in Central Spain ranged from 3,000 to 69,000 €/ha, with a mean of 18,308 €/ha; in the nearest forests prices ranged between16,100–19,500 €/ha. The interest rate (r) can be established for this calculation as the expected return of a forest estate. Ramírez and García (2003) reported returns of 4–4.5% in irrigated crops, 3.5–4% in rainfed crops, 3–3.5% in pastures and 2% in forestlands, but in 2003 the interest rate was 2.8%, and currently it is negative; current market data show a return in cereal crops of 0–2.5%, depending on rainfall and EU subsidies. We proposed a moderate interest rate for calculations, 1%. Martínez (1996) proposes a time to mitigate the impact (n) of 20 years for permanent alteration of the landscape. The value obtained for landscape degradation—used as a valuation of landscape ES—is 107 €/ha·yr. The product of this figure by the landscape quality indicators provided results of 50–87 €/ha·yr (Table 10), the highest in forests located in mountainous areas, with steep and spectacular landscapes.

Recreation

There are numerous works dedicated to the valuation of recreation as an ES, based on revealed preferences or on market or simulated valuations, with disparate results, from a few euros to several thousand per hectare. Recreational use is highly variable, as shown in the studied forests: in two of them is intense, in one very scarce and in the other two is absent (there is no recreational use on private forests because access is forbidden).

An objective criterion to value recreation is the number of visitors. According to the regional government of Madrid the three closest protected areas (four of the studied forests are included within them) had between 1.6 and 2.4 million visitors in 2019, with 38–84 visits/ha on average. The results were higher than those collected by Caparrós et al. (2001) for pine forests in Central Spain (15 visits/ha) and

Table 10 Valuation of landscape

Forest name	BLM[1]	Ind[2]	Value €/ha·yr
Boadilla-Encinas	15	0.47	50.46
Cuerda Herrera	16	0.50	53.69
Jurisdicción	25	0.78	83.75
Monte Agudillo	26	0.81	86.97
Ventilla-Vinatea	16	0.50	53.69

[1]BLM (1986) method results

[2]Landscape indicator

by Campos et al. (2005) for natural areas in Spain (12–37 visits/ha), which can be justified by the high population of Madrid and the popularity of these parks. These figures distribute the visits homogeneously within all the parks' area, which is not true; most visitors are concentrated in a few places (such as Jurisdicción). In addition, these data focus on external visitors, but in two of the studied forests the daily influx of local visitors is very important. External visitors are those who do not live in the vicinity of the forests; their visits involve a trip by public or private transport, and generally stay at least a few hours in the area, mainly at weekends. Local visitors reside in the vicinity of the forests, accessing on foot, bicycle or short trips in transports, and frequently visit the area for short duration activities, although usually frequent (walking, jogging, dog-walking…). Estimates of use are 150 visits/ha·yr in Boadilla (95% local, 5% external); 100 visits/ha·yr in Jurisdicción (70% local, 30% external); and 2 visits/ha·yr in Monte Agudillo (50% local, 50% external).

The value of a visit can be estimated through a WTP approach, for example using the travel cost method (Table 11); the average value obtained from the literature is 6.24 €/visit. These valuations are based on visitors who specifically travel to visit an area, but not on local visitors who frequently visit an area close to their home for a short walk; in the latter case the WTP is lower due to the shorter duration of the visit and the repetition over time. Visitors' surveys in protected areas in E Spain show an average visit duration of 5 h, while recurrent local visits usually last 0.5 to 1 h; considering a figure of 0.75 h (45 min), the value of a short visit would be 15% of a conventional one (0.94 €/visit).

In private forests it can be considered an auto-consumption of ES by the owners; some examples of values are 92 €/ha·yr in agroforests of Western Spain (Campos and Mariscal 2003), 116 €/ha·yr in agroforests of Southern Spain (Campos et al. 2020), 32 €/ha·yr in agroforests of Central Spain (Mesa et al. 2016) or 85 €/ ha·yr in conifer farms of Southern Spain (Campos et al. 2021). In this study an auto-consumption value has been applied for recreation on private forests. In Cuerda Herrera, with several owners, auto-consumption was estimated to be equivalent to 10 visits/ha·yr; in Ventilla-Vinatea, owned by several companies and not visited by the owners, it was equivalent to 0.1 visit/ha·yr.

Table 11 Examples of economic valuation of visits and recreation

Area	€/visit	€/ha	References
Central Spain	1.16	276.21	Caparrós et al. (2001)
Central Spain	5.83		Caparrós and Campos (2002)
Central Spain	7.64	0.06–195.39	Voces et al. (2010)
Developed world	5.39		Markandya et al. (2008)
France	2.72–3.73		Scherrer (2002)
Meta-analysis	10.23–12.10		Grilli et al. (2014)
N Mediterranean (mean)		39.81	Croitoru and Merlo (2005)
N Mediterranean (range)		1.24–143.06	Croitoru and Merlo (2005)
Spain	7.36		Campos et al. (2005)
Spain		195.12	MAGRAMA (2014)

Results in forests with recreational use ranged from 7 to 253 €/ha·yr (Table 12), depending on visitors' intensity, consistent with values for Central Spain, 0–276 €/ha·yr (Table 11). In private forests the owners' auto-consumption (Table 12) ranged from 1–62 €/ha·yr, being the values for agroforests and conifer farms in Spain 32–116 €/ha·yr; Ventilla-Vinatea has a lower value due to the scarce private use.

Balance Including Non-market Priced Ecosystem Services

Incorporating to the current balance the valued non-market ES, a positive result is obtained in all the forests: expenses were covered, and a return was obtained (Table 13). The balance ranged from 162–992 €/ha. Jurisdicción stood out: an emblematic mountain forest, with scenic landscape, in an area of high recreational interest, and with rich and unique wildlife. At the opposite extreme was Ventilla-Vinatea, a private forest practically unused, with average biodiversity and landscape interest, and a historical deficit of investments.

The total ES value was 462–1,149 €/ha·yr, with a weighted average of 666 €/ha·yr. Attending to values proposed in the literature (Table 14), the mean is 502 ± 74 €/ha, for Mediterranean forests, and 811 ± 215 €/ha including also temperate forests.

Market-priced ES account for 1.4%, while non-market-priced ES represent 98.6%. These percentages differ significantly from those reported for Spain and W Mediterranean (Croitoru and Merlo 2005), where market ES are 47–56%; the difference is due to the absence of timber production in the studied forests.

Discussion and Conclusions

Mediterranean forests have low timber productivity due to the climate, with a vegetative period split in two by winter cold and summer drought. Pasture production is also poor and grazing often conflicts with forest regeneration. Hunting is a regressing activity, not very profitable except in some large game properties. Other non-wood products are of little financial importance, except for the cork and mushroom production in some regions of Spain (cork in Andalusia, Extremadura and Catalonia, and mushrooms especially in some areas of Catalonia, Aragon and Castile-Leon);

Table 12 Valuation of recreational use

Forest name	Visits/ha·yr			Value €/ha·yr
	Local visitors	External visitors	Owners' self-consumption	
Boadilla-Encinas	142.5	7.5	0	180.75
Cuerda Herrera	0	0	10.0	62.40
Jurisdicción	70.0	30.0	0	253.00
Monte Agudillo	1	1	0	7.18
Ventilla-Vinatea	0	0	0.1	0.62

Table 13 Forests' budget accounting main ecosystem services and opportunity cost of the capital (€/ha)

Concept	Group		Boadilla—Encinas	Cuerda Herrera	Jurisdicción	Monte Agudillo	Ventilla—Vinatea
Expenses	Silviculture		−13.88	−106.79	−59.58	−2.91	−139.24
	Reforestation and vegetation improvement		−28.89	−115.92	−52.72	−9.91	−86.08
	Forest fire prevention		−10.92	−16.76	−12.64	−22.47	−25.24
	Pest control and wildlife protection		−2.00	−23.14	−16.91	−3.70	−15.54
	Infrastructure maintenance		–	−14.29	−4.20	−0.39	−22.07
	Cleaning and recreation regulation		−34.24	–	−0.12	–	–
	Management and protection		−63.12	−81.48	−65.22	−45.06	−44.16
	Total		−153.06	−358.38	−211.39	−84.44	−332.23
Income	Current	Timber	–	2.60	11.99	–	1.33
		Non timber products (honey[a]. pine nuts[b])	–	–	0.20	2.29	–
		Grazing (concession)	2.96	–	1.69	8.00	–
		Hunting (concession)	–	–	3.61	11.82	–
	New	Carbon sequestration	27.47	64.35	64.85	29.95	51.23
		Erosion control	16.13	95.87	166.02	85.12	122.24
		Watershed protection	27.16	55.09	178.37	58.26	25.83
		Biodiversity conservation	202.93	202.93	385.79	278.75	207.39
		Landscape protection	50.46	53.69	83.75	86.97	53.69
		Recreation	180.75	62.40	253.00	7.18	0.62
	Total		507.86	536.93	1.149.27	568.34	462.33
Annual balance			354.80	178.55	937.88	483.90	129.99

Table 14 Value of ecosystem services in forests (€/ha)

Area	Forest value (€/ha·yr)*	References
Central Spain	987.40	Caparrós et al. (2001)
Central Spain (agroforestry systems)	164.05	Mesa et al. (2016)
Central Spain (mean)	536.86	This study
Central Spain (range)	358.62–981.24	This study
France	410.84	Croitoru and Merlo (2005)
Italy	357.38	Croitoru and Merlo (2005)
North Italy	842.14	Häyhä et al. (2015)
North Mediterranean (including Spain)	247.63	Croitoru and Merlo (2005)
North Mediterranean (including Spain)	244.45	Croitoru (2007)
Portugal	484.01	Croitoru and Merlo (2005)
Spain	126.63	Croitoru and Merlo (2005)
Spain (undefined ecosystems)	540.53–946.88	MAGRAMA (2012)
South Spain (conifer farms)	295.05	Campos et al. (2021)
Temperate forests. Worldwide	530.58	Krieger (2001)
Temperate forests. Worldwide	2,500.88–3,464.43	De Groot et al. (2012)
Temperate forests. Worldwide	1,106.56	Hussain et al. (2012)

*Annual rent or annual value including ES

however, they may have economic importance, for example for foraging or local consumption.

Forests require investment for management, surveillance, silviculture (particularly in reforestations and regenerated stands of resprouting species), and preventive measures due to the extreme fire risk in the region. This greatly unbalances the economic result, with excessive expenses for the income they produce; for this reason they are often defined as "loss-making forests". As a result, the private income from Mediterranean forests is usually very low, although at a community level benefit is net positive.

Social benefits are recognized in public forest, receiving public funds for conservation, although often without a clear idea of what the benefits produced are. However, in private forests social benefits are less frequently recognized (although exists, such as a biodiversity, landscape o water conservation); negative economic balances may lead to consider them as environmental liabilities, requiring investment for conservation, but without benefits. Private forests are frequently sustained by public subsidies, a recognition of social benefits; however, subsides quantification is difficult when ES have not been valued. If no income is produced and no subsidies are received, private owners rarely invest in forest conservation.

Currently the studied forests have annual losses of 62–356 €/ha·yr but valuing the main ES they would have a positive balance of 162–992 €/ha·yr. Taking into account the average value of the forest land in the region, these results imply an average opportunity cost of using the land in forestry of 2.9%. In the two private forests opportunity cost are 0.9–1.1%. These forests have a historical deficit of investments in silviculture and reforestation, which makes that the costs associated with these items in the management plans were exceptionally high. Once these investments were made, future needs will be much lower, and opportunity cost could rise to 1.9–2.2%. In the three public forests the opportunity cost would be 2.2–5.4%. Consequently, the forests are currently loss-making because the ES are not adequately valued, but if this were done, in an objective way, they would be socially profitable. The results indicate that it is worth considering conservation measures to continue this flow of benefits. Conservation measures may include some form of payment to landholders, but the appropriateness of that would need to undergo further, more detailed analysis into the net benefits of such measures.

Currently the conservation of the studied forests, public and private, is mainly financed through the general budgets of the regional government. The politicization of these budgets and the lack of knowledge on the benefits provided by forests imply that investment may vary from year to year, and may even be insufficient to maintain the flow of benefits. Effects of investment reduction are not immediate: delaying regeneration will not be a problem until existing trees start to decay, and reduce fire prevention does not mean that the forest is going to burn, although the risk increases. It would be necessary to compare a scenario of investment reduction with another of conservation measures to know the real influence on the flow of benefits generated; this is, a valuation of changes in flows (Pagiola et al. 2004). However, a valuation of ES would make it possible to incorporate indirect ways to finance forests, for example through carbon or water taxes. This may increase the economic independence of the forests, reducing their dependence on the government budgets.

As a first step, forest management plans should include the valuation of at least the main ES, so that a more objective and realistic planning can be made. The next challenge is to convert this economic valuation into real income, through PES schemes, which allow a proper forest management, and even a more ambitious management, investing more and achieving greater ES. PES provides a robust framework for forest owners, public or private, as it implies recognition of ES provided and the right to be remunerated or compensated for them, while other schemes, such as subsidies, imply the option (at the subsidizer's discretion) to reward for such services. This step requires dedicated leadership willing to invest political capital and real resources (Polasky et al. 2015), or at least to promote and facilitate private initiative.

The role of government is important in balancing economic, social and environmental objectives in forestry (Freer-Smith et al. 2019). There is a disparity between social benefits and private ones; socially beneficial land uses are at risk because private landholders are only able to perceive a limited range of benefits, mostly circumscribed to direct-use values such as timber or some non-timber forest products for which there are markets, and may transform them into income; PES schemes may help reduce this disparity.

Efficient and equitable valuation of ecosystem benefits require a variety of institutional arrangements (Turner et al. 2003); PES could be part of a broader strategy, but should not be the sole instrument, and they must also incorporate the notions of legitimacy, justice and empowerment (Mauerhofer et al. 2013). Possible government actions related to PES are: amending regulations to make ES more marketable (Do et al. 2018); using forest funds as intermediary tools for PES (Liagre et al. 2021); carbon taxes or credits to offset forest's sequestration (e.g. Kerchner and

Keeton 2015; Enríquez-de-Salamanca et al. 2017a; Van Kooten 2017); water payment schemes (e.g. Muñoz-Piña et al. 2008; Abildtrup et al. 2013; FAO-UNECE 2018); or promoting integrated biodiversity and carbon offsets schemes (Enríquez-de-Salamanca 2017b).

Mediterranean forests provide countless ecosystem services, but there is often little investment in their maintenance and improvement, especially when the production of market-valued goods is low. It is necessary to understand at a social and political level the benefits of these ecosystems, and the need to maintain them; the valuation of ecosystem services is a very useful tool to achieve this.

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