An economic valuation of the provisioning ecosystem services in the south-west of Europe

Noelia García-Rubio, Beatriz Larraz, Matías Gámez, Mélanie Raimonet, Roxelane Cakir, Sabine Sauvage, José Miguel Sánchez Pérez

HAL Id: hal-04567238
https://hal.univ-brest.fr/hal-04567238

Submitted on 6 May 2024

An economic valuation of the provisioning ecosystem services in the south-west of Europe

Noelia García-Rubio[1] · Beatriz Larraz[1,5] · Matías Gámez[1] · Mélanie Raimonet[2] · Roxelane Cakir[3] · Sabine Sauvage[4] · José Miguel Sánchez Pérez[4]

Abstract

A methodology has been developed to quantify the economic value of the provisioning ecosystem services in the South-West of Europe (SUDOE) at any scale. This article describes the methodology that allows the quantification of nutritional and non-nutritional materials and their economic valuation related to agriculture activities at the regional scale. In a complementary way, quantification and valuation of freshwater consumption for domestic uses is described using regional official statistics converted into basin statistics by a geographical scale change matrix. The results show important differences depending on the regions at a sub-system scale. The highest values of agricultural production and its economic valuation are observed in the southeastern part of the SUDOE territory (South France, Spain, and Portugal), both in absolute and relative terms. In the Tagus basin, where the two capitals (Madrid and Lisbon) are located, the water supply and economic value are the highest in the SUDOE region. Provisioning ES is valued at 56.7 billion euros in SUDOE territory per year (current 2015 euros). In conclusion, this work makes it possible to identify those systems with higher economic valuation of provisioning ecosystem services, which, together with the expected increase in temperature and decrease/irregular rainfall, makes these areas particularly vulnerable. The proposed methodology can be applied to any river basin in the world.

✉ Beatriz Larraz
beatriz.larraz@uclm.es

[1] Instituto de Desarrollo Regional, Universidad de Castilla-La Mancha, Paseo de los Estudiantes, S/N, 02071 Albacete, Spain

[2] Univ Brest, CNRS, IRD, Ifremer, LEMAR, 29280 Plouzane, France

[3] HETWA, 31400 Toulouse, France

[4] Laboratory of Functional Ecology and Environment, University of Toulouse, CNRS, Auzeville Tolosane, 31326 Castanet Tolosan, France

[5] Faculty of Law and Social Sciences, University of Castilla-La Mancha, Cobertizo de San Pedro Martir, S/N, 45071 Toledo, Spain

Graphical Abstract
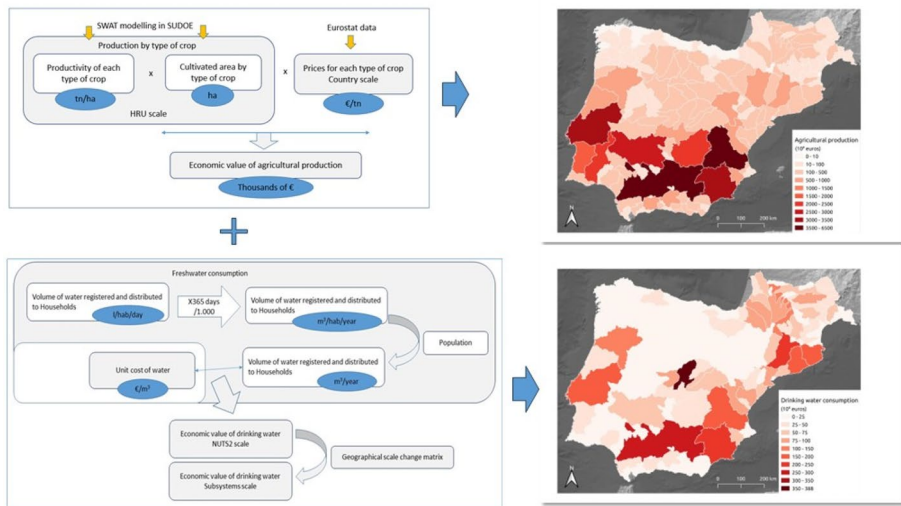
Keywords Agriculture · Economic valuation · Freshwater · Provisioning ecosystem services · SUDOE territory · Sub-system scale

1 Introduction

Ecosystem services (ES) are humans' benefits from natural ecosystems. *Millennium Ecosystem Assessment* (UN, 2005) defines ES as direct contributions to human well-being from ecosystems' biotic and geological structure. Research on the concept of ES has gained importance in recent decades, allowing the quantification of ecosystems' value to society (Bateman et al., 2011; Costanza et al., 1997; de Groot et al., 2002). The ES concept has even become more than a research topic. It has started to be included in policy agendas, becoming instruments of implementation, such as conservation, incentives, investment guidelines and stakeholder participation (Meli et al., 2023), favoured by the increasing political attention to conservation (Gómez-Baggethun et al., 2010).

Three different types of ES, all vital to human health and well-being, have been identified: provisioning, regulating and cultural (Costanza et al., 1997; de Groot et al., 2002; La Notte et al., 2017). The commonly accepted, extensive, and detailed classification is compiled in the Common International Classification of Ecosystem Services (CICES, 2022). These services include, among others, providing food, water, clean air, wood, and other natural resources, as well as regulating climate, crop pollination, waste decomposition, water purification, and flood protection. They also provide cultural and recreational benefits, such as tourism and aesthetic enjoyment of nature.

Among these services, provisioning ES, which encompass the provision of essential resources, have gained increasing attention in recent decades (Acharya et al., 2019; Ala-Hulkko et al., 2019; Corona-López et al., 2021; Karabulut et al., 2016; Kibria et al., 2018). Numerous studies have addressed the provision of food, water, wood, and other natural

resources from an ecosystem perspective (Palacios-Agundez et al., 2015). Approaches and methodologies have also been developed to assess the availability and quality of these services in different landscapes and regions (Guerbois & Fritz, 2017). Quantifying provisioning ecosystem services is thus crucial to (i) provide a solid foundation for understanding and assessing the state and capacity of ecosystems to provide essential natural resources, and (ii) enable informed decision-making and more effective resource management, ensuring long-term sustainability.

The economic valuation of provisioning ES is important due to their intrinsic relationship with human well-being and the economy (Ashagre et al., 2018; Brouwer et al., 2018; Krause et al., 2017; Lara-Pulido et al., 2018; Marre & Billé, 2019; Nannan et al., 2022). This economic valuation can help to efficiently allocate resources and conservation efforts by considering the costs and benefits associated with their provision and use. It also provides a solid foundation for implementing incentive mechanisms and compensations, such as payments for ecosystem services, which promote ecosystem conservation and the sustainability of resource provisioning. Economic valuation of these services allows for their integration into economic and policy decision-making processes and the assessment of development projects and policies. When it comes to the economic valuation of provisioning ecosystem services, although many of them are traded in markets, it should be borne in mind that their value may be much higher in cases where households are directly dependent on them for their livelihoods (FAO, 2022). Economic valuation is still significant in allowing society's value on ecosystem services to grow (TEEB, 2023).

Within ES, the water-related ES are particularly relevant in the current climate change scenario. Grizzetti et al. (2019) quantify the main ES related to aquatic ecosystems at the European scale, but the analysis is not exhaustive; it is based on the availability of data at the European scale and the possibility of calculating quantitative indicators. They conclude that the use of provisioning services can cause pressures on the ecosystem. This suggests the importance of maintaining good ecological conditions of ecosystems to ensure the provision of ES in the future. Also, Bennett et al. (2009) stated that ecosystem management that attempts to maximise the production of one ecosystem service results in substantial decreases in the provision of other ES. In the case of provisioning ES, focusing on maximising agricultural yields or drinking water supply has downstream consequences (Gordon et al., 2010). Among others, water quality declines as a consequence of excess nutrients from agriculture and wastewater, coupled with a decrease in clean headwater flows capable of diluting, impacting on an increased cost of obtaining drinking water downstream and impacting on recreational purposes; also, a reduction in water quantity leads to loss of wetlands and coastal ecosystems which influences nutrient retention and the conservation of migratory bird ecosystems.

Despite advances, challenges and gaps exist in understanding and accurately quantifying and valuing provisioning ES. One of the gaps is the quantification and economic valuation of provisioning ES at small spatial scales. Most mapping has been done at large spatial scales (regional and provincial), with limited attention to understanding supply and demand across multiple scales (Burkhard et al., 2013). Aziz et al. (2023) compare different spatial resolutions of datasets (local, regional, global) to quantify ecosystem services in a watershed, and show that using local values is associated with lower uncertainties and provides different estimates for several services.

Albert et al. (2016) highlight that it is important to consider what kind of aggregated, national-level information on ecosystem services would be useful, for which type of decisions, and for whom. Nuissi et al. (2009) argue that environmental issues have different properties regarding their spatial implications. For example, a national indicator cannot

simply be applied in assessments of areas at lower spatial scales but needs to be adapted to the specific context and decision-makers' needs. It follows that there is a need for a methodology suitable for quantification and economic valuation at any spatial scale. Malinga et al. (2015) provide a systematic review of papers that have spatially mapped (only from the perspective of quantification without approaching an economic valuation) ES, where we can conclude that none works at hydrological scales. In a SCOPUS analysis in 2002–2020, just 2.4% of the published documents are about Water–Food nexus, which includes its interactions with land use, security, agriculture, climate, economy, and health, which evidences the need for further research (Corona-López et al., 2021). Moreover, none have been made at a small hydrological scale, being the most common basin or management scale. However, there are now models able to simulate large-scale territories to have agro-hydro-ecological information at high spatial resolution on large domains to quantify ecological functions (e.g. Cakir et al., 2020) that could be used to overcome this limitation.

In recent decades, there has been a growing interest among the research community in the monetary value of ES. Most economic valuation methods are based on estimating the willingness-to-pay by individuals for ES (Costanza et al., 1997). The applicable economic valuation methods depend, first, on whether market prices are available for the ecosystem goods and services which are being assessed. If these market values are not available, there are alternative methods, such as the Travel Cost Method (Clawson, 1959; Trice & Wood, 1958) or Random Utility Models (Herriges & Kling, 1999), which have been mainly used to estimate the in situ value of visiting a recreational site that is related to an aquatic ecosystem (National Research Council, 2005). Other methodologies include the Hedonic approach to consider environmental quality in real estate prices (Palmquist, 1991; Sander & Haight, 2012); Contingent valuation (Bateman et al., 2002), as a variant of the Stated-preference methods; or Replacement cost method (Shabman & Batie, 1978), among others. In other cases, a market exists, and therefore, there is commercial value for these services. Other methods estimate gross expenditures in using goods or services, economic impact, etc. However, these methods do not measure the economic value of the ES in terms of consumer surplus. In the case of water supply, the ES has significant market and non-market characteristics (Krieger, 2001). More specifically, regarding drinking water, a direct method based on market prices can be applied, while in the case of agriculture, since the ecosystem service provides goods that can be traded, the market value provides a gauge of the value, although it is only a partial measure since it incorporates only part of the value (Goulder & Kennedy, 1997).

The specific objective of this article is to develop a methodology to provide quantification and economic valuation of provisioning ecosystem services, particularly materials from agriculture and water supply, at the hydrological scale, bridging the disparity of spatial scales at which the data necessary for both calculations are available. The case study chosen for its application is in the south-west of Europe. The challenge of quantifying and economically valuing the ES in a territory that includes four different countries is based on the necessary adjustment of official statistics, both in terms of their methodologies and the periods with available information. This paper provides first a description of a methodology to quantify provisioning ES for nutritional (biomass (cultivated crops) and water) and non-nutritional materials (fibres and other materials from plants) from agriculture and freshwater consumption for domestic uses, a list of the statistical and modelled data used, before exploring and discussing the results obtained at the scale of the South-West of Europe.

Through our proposed methodology, this article aims to contribute to advanced knowledge and practical tools for sustainable resource management at the South-west of Europe

scale, where water scarcity conditions are worrying since the impacts of climate change on ecosystems are increasing (Maes et al., 2020) and anthropogenic activities are high, which means a high pressure on the resources. By enhancing the understanding of provisioning ecosystem services and their economic importance, we strive to support evidence-based decision-making and the implementation of effective conservation and resource management strategies.

The link between ecosystems and aspects of human well-being, including health, has been less well understood among the social science community, making attention to the quantification and valuation of ecosystem services of vital importance. Understanding the quantification and economic valuation of provisioning ES is pivotal for effective decision-making and informed management of natural resources. Quantifying these services also facilitates the identification of critical or vulnerable areas where conservation and restoration actions are needed. It allows for assessing the impacts of human activities, such as intensive agriculture or forestry exploitation, on resource provision and establishing appropriate mitigation or compensation measures. In rural regions, it can also be the main source of livelihood for their inhabitants (LifeAdaptamed, 2020). Following Butler and Oluoch-Kosura (2006), insufficient access to the ecosystem service of food (and water) provisioning is a significant factor in the loss of human well-being, which requires maintaining healthy ecosystems.

2 Materials and method

2.1 Study site

SUDOE territory considered in this study spans 773,191 km^2 in the south-western part of Europe (Fig. 1). SUDOE includes the south-west regions of France, Andorra, and the Iberian Peninsula (Spain, Portugal, and Gibraltar). This area serves as the primary agricultural region of the European Union, with approximately half of its land dedicated, as per the Corine Land Cover 2012 (CLC) map (Fig. 2). The remaining portion of the region consists of 46% forests and semi-natural areas, 3% artificial surfaces, and 1% wetlands and water bodies. Notably, a significant portion of the agricultural land is irrigated, with around 13.5% permanently irrigated in Spain and Portugal (Eurostat, 2019). Over 60 million people reside in the SUDOE region, which comprises some of the largest European cities (Fig. 1). The two main climates in the area are (i) temperate oceanic, extending from the northern coast of Spain to the western and inland parts of south-western France (Rivas-Martinez et al., 2019), and (ii) Mediterranean, covering most parts of the Iberian Peninsula and the Mediterranean coast of France. 785 mm is the average annual precipitation (Aquastat, 2019).

2.2 Quantification and economic valuation of provisioning ES

2.2.1 Nutritional and non-nutritional materials from agriculture

The first step consists of quantifying the resource. Since the information on production at the farm system level is not available, we estimate it using a model that provides us with the surface area (number of hectares) dedicated to each crop type and an estimate of their productivity (tons per hectare). Subsequently, the resource is valued by multiplying the

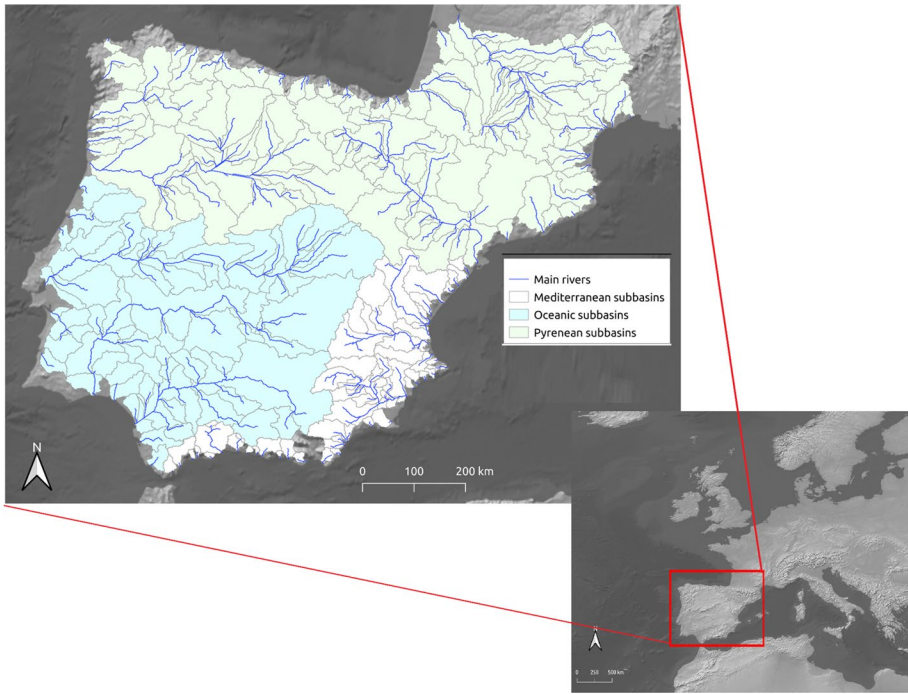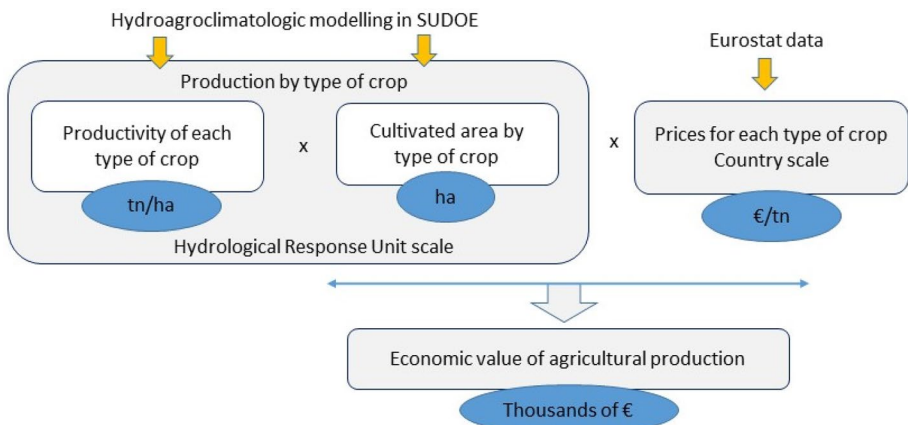**Fig. 1** SUDOE territory and their hydrological basins

Fig. 2 Quantification and assessment methodology for ES Nutritional and other non-nutritional materials from agriculture

production obtained by its market price (euros per ton). The methodological scheme can be seen in Fig. 2.

The methodology followed for estimating agricultural production is explained in more detail below.

2.2.1.1 Agro-hydro-ecological modelling The agro-hydro-ecological model SWAT is a valuable numerical model for simulated hydric resources in agricultural areas and agricultural management practices, including irrigation supplies. SWAT concentrates on evaluating irrigation systems in agriculture and simulating production at the field scale. The primary objective is to achieve high yields in crop production, enhance water productivity, and ensure the sustainable distribution of water resources. SWAT (Soil and Water Assessment Tool) is a publicly available model that proves invaluable in assessing the Water-Food Nexus (Corona-López et al., 2021). In the SUDOE territory, where water stress is particularly severe, the quantity of irrigation volume holds significant importance (EEA Report, 2018). SWAT has been used for over 20 years in different countries worldwide (Tan et al., 2020).

Three modelling projects have been established to portray three distinct regions—the Pyrenean, the Mediterranean, and the Oceanic zones (Fig. 3). The delineation, warm-up, and calibration/validation steps for hydrology in SWAT were executed using a consistent procedure for each project, as described in Cakir et al. (2020). Hydrological Response Units (HRUs), which consist of the same slope, soil, and land cover characteristics, were
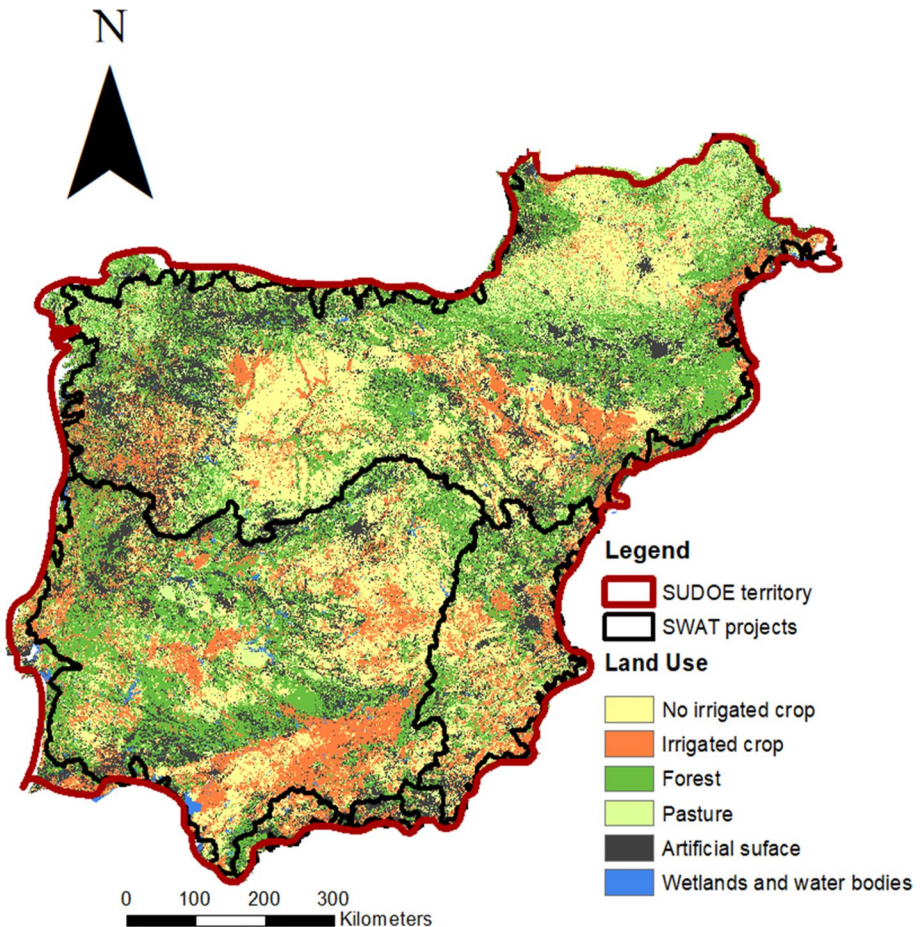

Fig. 3 Land use from Corinne Land Cover 2012. Source: adapted from Cakir et al. (2020)

defined. A total of 24 land classes, 34 soil classes, and 5 slope classes were employed in this delineation process, and 263 sub-basins with 22,997 HRUs were identified (Cakir et al., 2020). All the details on hydrological calibration and validation of these three projects are detailed in Cakir et al. (2020).

According to national statistics, crop management was applied to all HRUs with intensive production. The primary crop and its corresponding technical management were chosen for integration into the model for each region. To streamline the model, correlations were established between the Corine Land Cover classification and the dominant crop in each area (Table 1). For example, cotton is the dominant crop in irrigation in the South-East of Spain.

According to the bibliography, the model implemented plant growth functions with heat unit values (Battude et al., 2017; Evans & Poorter, 2001; Gilmore & Rogers, 1958; Wilson, 1981). The "heat unit" concept is utilised to indicate the stage of plant development and is typically represented on a scale from 0 to 1. In this scale, 0 signifies the sowing time, while 1 represents the optimal plant harvest moment. In the SUDOE territory, we apply the same crop management procedure. To simulate irrigation in the model, we set up the auto-irrigation module. Auto-irrigation activation terms are defined according to the crop type and the area's climatic condition. The auto-irrigation system can be activated based on two selected conditions established during calibration: (1) plant water demand or (2) soil water content. Within the model, the crops identified for irrigation include rice, olive, vineyard, fruit trees, berry plantations, maise, and almonds. The nearest dams or main rivers supplied irrigation for these crops. The annual irrigation volumes (Table 1) were used to calibrate the auto-irrigation and soil characteristics.

The beginning of plant growth is set up to 0.15 heat units. Auto-irrigation begins at the same heat unit, meaning every time the plant or the soil needs water, a fixed amount of water is applied to HRU. Fertilisers are applied automatically according to plant needs before starting at a heating unit equal to 0.001 or manually with repeating crop management at a precise plant development stage defined by fertiliser manufacture recommendations for specific crops. In conclusion, the harvest in this study is set at 1.2 heat units. Choosing a value exceeding 1 is deliberate, acknowledging a time lag between the optimised harvest time and the actual harvest. In the interval between 1 and 1.2 heat units, the plant consumes energy without significantly increasing its production. Leaf development, light interception, and the light conversion to biomass ratio are simulated to estimate the potential biomass according to the SWAT model method (Monteith, 1977). This potential biomass is calculated under ideal growing conditions, assuming sufficient water, nutrient supply, and a favourable climate.

During a harvest or kill operation, a portion of the plant biomass is removed from the system as yield. Crop yield is determined by biomass production, harvest efficiency, and the fraction of above-ground plant dry biomass removed as dry yield, also known as the harvest index. The daily calculation of the harvest index considers the plant's stage of development and the Hydrological Response Unit (HRU) water deficit.

Crop management practices were applied to HRUs based on recommendations from authoritative agricultural bodies such as the French Chambers of Agriculture (FCA), the French and Spanish Directorate-General for Agriculture and Rural Development (DGADR), and the Spanish Confederations (SC). These recommendations guided the selection and implementation of dominant crops, as determined by national agriculture statistics databases, along with crop technical management. Specific technical management details, including fertiliser application, were derived from the FCA, DGADR, and

Table 1 Crops and fertiliser inputs integrated into the SWAT model according to seed companies' recommendations (Cakir et al., 2020)

Territory	Country	Classification description from CLC 2012	CLC code	Selected crops	Fertilisers type	Quantity applied (kg ha^{-1} year-1)
Pyrenean	France	Permanently irrigated land	2.21.212	Maise	15–15–15[1]	400
		Non-irrigated arable land	2.21.211	Wheat	15–30–0[2]	200–150
		Land principally occupied by agriculture, significant areas of natural vegetation	2.24.243	Maise	15–15–15[1]	400
		Vineyard	2.22.221	Vine	N	30
					P	20
		Fruit trees and berry plantation	2.22.222		18–46–0[3]	100
		Rice fields	2.21.213	Rice	N	50
					P	60–80
					K	60–100
		Non-irrigated arable land	2.21.211	Wheat	N	240
					P	96
					K	144
		Land principally occupied by agriculture, significant areas of natural vegetation	2.24.243	Sunflower	15–15–15[1]	400–600
		Permanently irrigated land	2.21.212	Cotton	N	150
					K	33

Table 1 (continued)

Territory	Country	Classification description from CLC 2012	CLC code	Selected crops	Fertilisers type	Quantity applied (kg ha^{-1} year-1)
	Spain, Portugal	Vineyard	2.22.221	Grape	N	30
					P	20
		Fruit trees and berry plantation	2.22.222		N	190
					P	120
					K	360
		Rice fields	2.21.213	Rice	N	190
					P	60
					K	240
		Almonds	2.22.222	Almonds	N	245
					P	112
					K	156

Table 1 (continued)

Territory	Country	Classification description from CLC 2012	CLC code	Selected crops	Fertilisers type	Quantity applied (kg ha^{-1} year-1)
Oceanic	Spain, Portugal	Non-irrigated arable land	2.21.211	Wheat	N	290
					P	126
					K	144
		Land principally occupied by agriculture, significant areas of natural vegetation	2.24.243	Cotton	N	150
					P	33
		Permanently irrigated land	2.21.212	Cotton	N	150
					P	33
		Vineyard	2.22.221	Grape	N	60
					P	40
		Olive groves	2.22.223	Olive	N	133
					P	52
					K	195
		Rice fields	2.21.213	Rice	N	190
					P	60
					K	240
		Almonds	2.22.222	Almonds	N	245
					P	112
					K	156

Table 1 (continued)

Territory	Country	Classification description from CLC 2012	CLC code	Selected crops	Fertilisers type	Quantity applied (kg ha^{-1} year-1)
Mediterranean	Spain, Portugal	Non-irrigated arable land	2.21.211	Melon	N	85
					P	45
					K	95
		Land principally occupied by agriculture, significant areas of natural vegetation	2.24.243	Lemon	21–0–0 + 24S[4]	610
		Permanently irrigated land	2.21.212	Almonds	N	245
					P	112
					K	156
		Vineyard	2.22.221	Grape	N	60
					P	40
		Olive groves	2.22.223	Olive	N	133
					P	52
					K	195
		Rice fields	2.21.213	Rice	N	190
					P	60
					K	240
		Fruit trees and berry plantation	2.22.222	Almonds	N	210
					P	49
					K	190

(1) 15–15–15, NPK fertilise, is a combination of nitrogen (N), phosphorus (P) and potassium (K) designed to maximise crop yield and quality. It contains 15%N, 15% P, and 15% K. (2) 15–30–0 is a high phosphate fertiliser. It contains 15%N, 30% P and 0% K. (3) 18–46–0, Diammonium phosphate, is a fertiliser, and when applied as food for plants, the soil pH is increased temporarily. It contains 18%N, 46% P2O5, and 0% K. (4) 21–0–0 + 24S, and the Ammonium Sulfate Fertiliser contains 21%N and 24% of sulfur
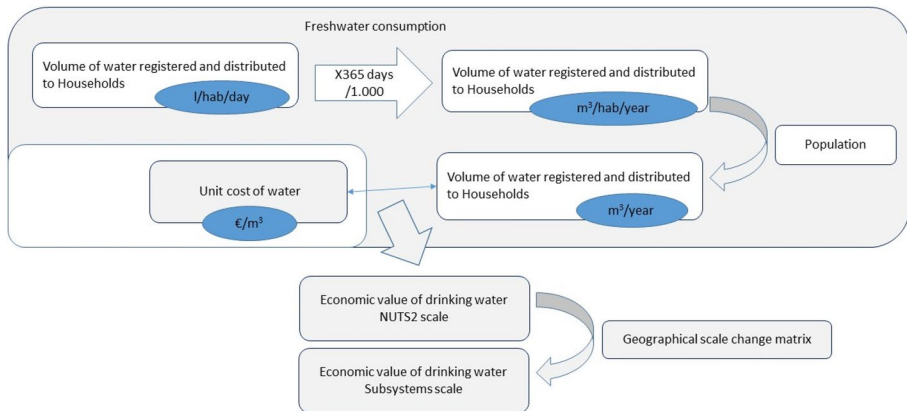
Fig. 4 Quantification and economic assessment methodology for ES Freshwater consumption for domestic uses at sub-system scale

SC guidelines. For instance, the predominant irrigated crops are cotton in the south-east of Spain and corn and wheat in France.

To evaluate how well the simulated and observed mean crop yields align, a per cent bias value (PBIAS) is calculated using the method introduced by Moriasi et al. (2007). PBIAS acts as a metric, indicating whether the simulation tends to underestimate (positive value) or overestimate (negative value) the observed values on a global scale.

2.2.2 Freshwater consumption for domestic uses

Regarding the valuation of the ES of freshwater supply for domestic uses, the proposed methodology, considering its feasibility based on available data, consists of estimation at the NUTS-2 level (Eurostat, 2023) and then converted to sub-system level employing a change of scale matrix. The first step, therefore, consists of obtaining the volume of registered water distributed to households annually in each NUTS-2, starting from the average consumption data in litres per inhabitant and day and multiplying by 365 and by the population of each population unit. Once the ES has been quantified, its valuation is followed by a market valuation methodology. The economic value of the domestic water supply ecosystem service at the NUTS-2 scale is obtained by multiplying the volume of water (in m^3) consumed by the price per cubic meter of water billed to households (€/m^3). Finally, the economic value of the ES of domestic water supply at the sub-system and river basin level is obtained through a geographical scale change matrix based on the percentage of the population of each NUTS-2 that belongs to each sub-system or district (Fig. 4).

The results have been calculated and represented at the sub-system scale. This transformation implies that, in the interpretation of the results, the size of each area must be taken into account: note that the sub-systems of the northern SUDOE (those of the Cantabrian coast and the French basins) are significantly smaller than those of the central-south-western SUDOE, the latter including the exploitation systems of Portugal. To avoid this issue, the relative results per area are also shown.

2.3 Datasets

For the quantification and assessment of ES for nutritional (biomass (cultivated crops) and water) and non-nutritional materials (fibres and other materials from plants) from agriculture, the statistical information produced by the SWAT model was used together with the official statistical information shown in Table 2. The *Institut National de Recherche en Sciences et Technologies pour l'Environnement et l'Agriculture* (IRSTEA) (Campardon et al., 2012; Gleyses & Rieu, 2004; Loubier et al., 2013), the *Compagnie d'Aménagement des Coteaux de Gascogne* (CACG) and the Spanish Hydrological National Plan (Ministerio de la Presidencia, 2001; Ministerio de Medio Ambiente, 2002) provided information on irrigation volumes to validate interannual monthly model estimation. At the SUDOE scale, 7,914,000 hectares are irrigated with an annual water amount of 16,917 hm^3. To validate crop management routing, crop yield and biomass amount given by the model were calibrated at the model outputs. Production data for each country is provided on the national statistics website (Table 2). More details about the calibration and validation procedure are in Cakir et al. (2020). Then, the evaluation phase requires information on the selling prices of crop products. Finally, the distribution of the irrigated area of each NUTS-2 in each subsystem is needed to elaborate the scale change matrix.

In the case of freshwater consumption for domestic uses, the quantification of the resource is based on the total volume of water registered and distributed to households in the SUDOE territory. For Spain and France, the average consumption per inhabitant per day in each of the NUTS2 regions has been used as a starting point. Regarding Portugal, the data on consumption per inhabitant at the regional level is available per year, so it is only necessary to multiply it by the population of each NUTS2 to quantify the total resource in each Portuguese region. The statistics used in each of the processes are shown in Table 2.

3 Results

3.1 Nutritional and non-nutritional materials from agriculture

ES quantification: Annual average yields obtained with SWAT modelling were first compared to national statistics for each country (Table 3). The PBIAS absolute average of 8% indicates a good adjustment between simulated and observed mean crop yield. PBIAS values range from -35% to $+40\%$ depending on crops (except -70% for olives in Portugal), which is particularly good considering that mean statistical yields are characterised by a range of variability (indicated in Table 3). This study considers that the nitrogen input in fields is as close as possible to real value thanks to a good agreement between crop yield and national statistical yield.

Agricultural production's highest values are observed in the South-East of the SUDOE territory: it is above 16 million tn/year on average in the Guadalquivir River—mainly because it is the biggest sub-system—and it is above 13 million tn/year in Jucar sub-system, followed by Segura and Upper Guadiana, which produce more than 10 million tn/year on average each, while Central Guadiana produces around 7.5 million tn/year (Fig. 5a). On the Atlantic side, Tagus and Ribeiras do Oeste, belonging to the Portuguese demarcation, show an average production of more than 5 million tn/

Table 2 Statistical information

	Source	Organism
Nutritional and other non-nutritional materials from agriculture		
Quantification		
France	Agricultural accounts	Agreste (2018)
	Annual agricultural statistics (SAA)	Ministère de l'Agriculture et de la Souveraineté Alimentaire (2018)
Portugal	Regional Economic Accounts for Agriculture and Crop Production Statistics	Instituto Nacional de Estatística (INE-Portugal, 2018)
Spain	Crop surface area and yields survey (ESYRCE)	Ministerio de Agricultura, Pesca y Alimentación (2018)
Valuation		
All	Selling prices of crop products	Eurostat (2018a)
Scale change		
France	Own elaboration	
Portugal	Own elaboration	
Spain	Conversion coefficients for irrigated areas between NUTS-2 and River Basin, 2011–2014	Instituto Nacional de Estadística (INE-Spain, 2018a)
Freshwater consumption for domestic uses		
Quantification		
France	Water Agency data	Sispea-Services Eau France (Sispea, 2018)
Portugal	Domestic use of water	PORDATA (2018)
Spain	Statistics on water supply and sanitation	Instituto Nacional de Estadística (INE-Spain, 2018b)
Valuation		
France	Water Agency data	Sispea-Services Eau France (Sispea, 2018)
Portugal	Service fees	Entidade Reguladora dos Serviços de Águas e Resíduos (2018) (ERSAR)
Spain	Water economic indicators	Instituto Nacional de Estadística (INE-Spain, 2018c)
Scale change		
All	Population and social conditions	Eurostat (2018b)

Table 3 National data and simulated annual crop yield values averaged over the simulation time series by country for the period 2000–2010

Crops	Country	National Yield (tn/ha)	Simulated Yield (tn/ha)	PBIAS (%)
Almonds	Spain	2	2.2	− 11
Lemon tree		25.4 (± 12.2)	25.0	29
Vineyard		17.5 (± 12.5)	10.9	37
Olive		9	9.1	− 1
Fruit tree		18 (± 7)	15.3	15
Melon		13	12.1	7
Rice fields		7	6.0	14
Wheat		3.75 (± 1.25)	3.5	7
Cotton		4	4.3	− 8
Sunflower		2 (± 1.5)	2.7	− 35
Maise		9.9 (± 2.4)	7.4	26
Almonds	France	1	1.3	− 30
Vineyard		7	8.3	− 19
Fruit tree		18	16.2	10
Rice fields		5	3.3	34
Wheat		5 (± 2)	3.5	30
Maise		9	6.4	29
Melon	Portugal	8 (± 3)	5.9	26
Cotton		2.5 (± 1)	2.9	− 16
Almonds		0.3 (± 0.2)	0.4	− 33
Vineyard		4 (± 3)	8.0	− 14
Olive		4 (± 1)	7.1	− 78
Fruit tree		12 (± 6)	11.8	2
Rice fields		7 (± 3)	4.2	40
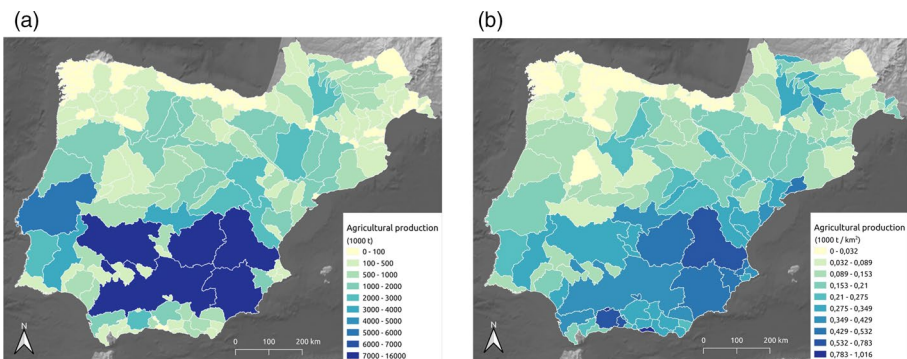
Fig. 5 **a** Quantification in absolute terms (thousands of tons per year) and **b** in relative terms (thousands of tons per km^2) of agricultural production in the SUDOE territory on average during the period 2000–2015

year. Low-production areas are mainly in the North of SUDOE territory, including some French areas and the North of Spain. In relative terms (Fig. 5b), all the basins of the
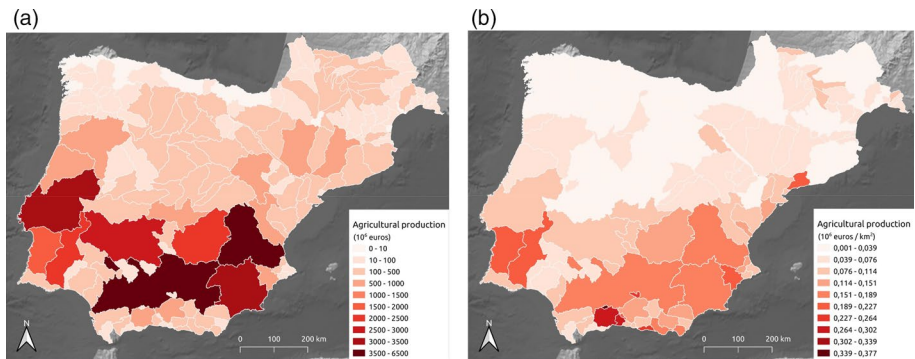
Fig. 6 **a** Economic valuation in absolute terms (millions of euros) and **b** in relative terms (millions of euros per km^2) of agricultural production in the SUDOE territory at sub-system scale on average during the period 2000–2015

Cantabrian slope, the Atlantic part of the Minho basin and some French sub-systems (Neste, Truyère and Lot amont) have lower productions by area compared to the others. The most productive zone by km^2 is, inside de SUDOE territory, the south-east: Júcar, Segura, Upper Guadiana, the South of the inner Catalonia basin, and some of the Mediterranean basins of Andalusia. Also, the Portuguese part of the Guadiana basin can be considered to have a high production by km^2 and the French sub-systems of Hers Mort Girou, Rivière de Gascones and Tarn Aval zone.

ES economic valuation: Once the number of tons per km^2 produced by each type of crop and the number of hectares in each water body of the SUDOE territory is available, the market valuation method is applied to obtain an economic value of this production. Table ***4 (Appendix 1) shows the absolute selling prices of crop products in 2015. The assessment of the ES provided by the sub-system as a whole (€) and concerning the area (€/km^2) are important.

The economic valuation of this production depends to a large extent, as seen in Table ***4, on whether the crop is rainfed or irrigated and, within this, on the type of crop. In the south of the SUDOE area, a large part of the Guadalquivir basin produces above 6500 million euros annually on average. In the Guadalquivir basin, 70% of its hectares correspond to rainfed crops, generating 34% of the economic valuation of production. It is followed by Júcar, where 27% of the cultivated hectares are irrigated, providing them 70% of the total economic value of production, with more than 4000 million euros per year; Segura basin and Tagus and Ribeiras do Oeste (Portugal) with more than 3000 million euros per year each. The Guadiana basin (Central and Upper parts, in Spain, and the Portuguese zones) has reached more than 2000 million euros for each sub-system (Fig. 6a). The Douro basin in the Spanish demarcation has mainly rainfed production, which generates between 100 and 500 million euros per sub-system.

On the other hand, the Ebro basin, in the south of the Pyrenees, produces between 1000–1500 million euros. The North of Spain has a low economic value in its agricultural production. Looking at the map of production divided by sub-system surface area (Fig. 6b), we can see that it is not due to the small size of these sub-systems because the value is still lower than other ones: The North (especially the North-West) of Iberian Peninsula presents lower production per unit area. At the same time, the South (and
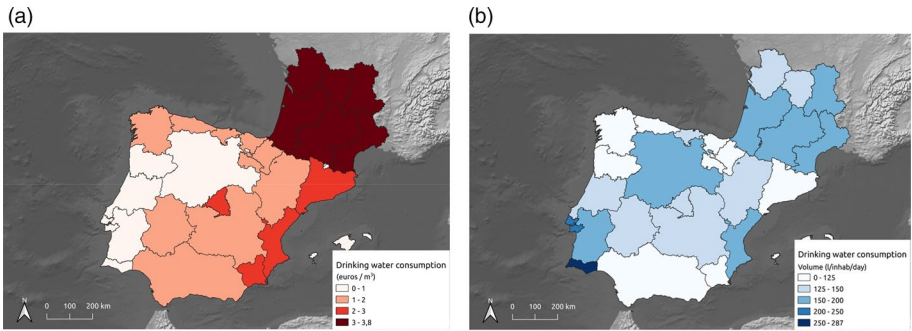
(a) (b)

Fig. 7 **a** Water consumption per inhabitant per day at regional level (NUTS2) and **b** Cost per m³ of drinking water at regional level (NUTS2), annual average 2008–2013. Source: **a** EUROSTAT, **b** Sispea-Services Eau France, Water and Waste Services Regulation Authority (ERSAR-Portugal) and National Institute for Statistics (INE-Spain)
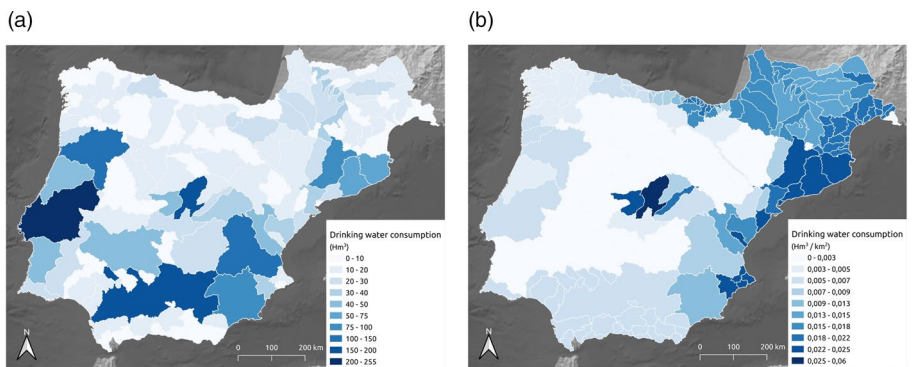
(a) (b)

Fig. 8 **a** Quantification in absolute terms (hm³) and **b** in relative terms (hm³ per km²) of drinking water supply ES in the SUDOE territory annual average during the period 2008–2013

part of the East), in general, can be considered the most productive per unit area zone in SUDOE territory.

3.2 Freshwater consumption for domestic use

ES quantification: Regarding the quantification of water used for human supply, from the official data on water consumption (in litres/inhabitant/day) (Fig. 7b) at SUDOE scale, together with the population data, we arrive at the map of consumption in each of the sub-systems at SUDOE scale (Fig. 8b). Here we can see the influence of the two capitals located in the Tagus basin, Madrid (Spain) and Lisbon (Portugal), with their sub-systems having the highest water consumption due to the effect of the population, in the case of Madrid (since consumption per inhabitant is low), and due to the joint effect of the population and the medium–high consumption per inhabitant in the case of Lisbon. Most of the Guadalquivir basin has high values for the ecosystem service of drinking water supply as a consequence of the high population in the sub-system due to its large size. Drinking water consumption is also high in the Segura basin, in a
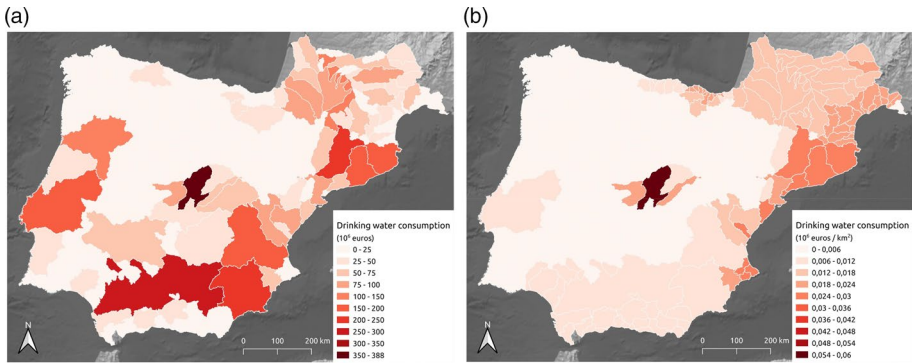
(a)　　　　　　　　　　　　　　　(b)

Fig. 9 a Economic valuation in absolute terms (millions of euros) and **b** in relative terms (millions of euros per km2) of drinking water supply ES in the SUDOE territory annual average during the period 2008–2013

large part of the Júcar basin and the internal basins of Catalonia and part of the Ebro sub-systems. The North of Portugal also presents high values for the drinking water supply ecosystem service. In the rest of the SUDOE territory, deficient consumption of drinking water is observed as a consequence of either the small size of the sub-basin or a sparse population (as is the case of the inland-northern part of Spain, which corresponds to areas of lower population density) in conjunction with low consumption per inhabitant per day (in comparison with other locations). This includes the sub-basins of France, where despite the medium–high consumption per inhabitant per day, the low population and the small size of the sub-systems lead to areas with low water supply consumption compared to other areas. In relative terms, per unit area (Fig. 8b), Madrid and Lisbon continue to appear as areas with the highest consumption of drinking water due to the population concentration (higher in Madrid and its metropolitan area than in Lisbon). However, the picture changes: the Catalonia area, Vasque country (Eastern Cantabrian), the coastal Jucar sub-systems, and the Mediterranean coast in France present higher drinking water supply values than the rest of the SUDOE territory.

ES economic valuation: Based on the cost per m^3 of water supply (Fig. 7a) and drinking water consumption (Fig. 7b), its economic valuation is obtained at the sub-system level (Fig. 9a). Note that the Jarama-Guadarrama sub-system, in which Madrid is located, continues presenting the highest value of the ecosystem service of supply as a consequence of the combination of high consumption (by quantity of population, not by individual consumption) and the relatively high price paid per m^3 of drinking water. In the sub-system of Madrid, the ES for water supply reaches the value of 388 million € per year. In the case of the Lisbon and northern Portugal area, the high valuation is a consequence of the high water service provided and not because of the price per m^3, as this is one of the lowest in the SUDOE territory. It is essential to note the high value of the ecosystem service of drinking water supply in the Segura basin, the Júcar basin, the internal basins of Catalonia and part of the Garonne because of the high cost per m^3 of water.

Due to its small sub-basins and despite having a unit price around the average, the Cantabrian coast obtains a low economic valuation in each sub-system compared to the values of the rest of the SUDOE territory. The same occurs in areas of low population density in the centre-north of Spain. Finally, note that the basins in the south of France

obtain a value for the ecosystem service of drinking water supply higher than the average, mainly due to the relatively high price per m^3.

In relative terms, per km^2, the sub-system with the highest ES value of drinking water supply continues to be Jarama-Guadarrama (Madrid), followed by the Segre basin and Catalonia internal basins. The French SUDOE area reaches medium–high ES drinking water supply values (Fig. 9b).

4 Discussion

4.1 Improving the spatial resolution of provisioning ES quantification and economic valuation at large transboundary scales

Implementing a hydro-agro-environmental model with consistent spatial resolution across Europe is essential to fulfilling the European Union's need for standardised tools amid global changes (Cakir et al., 2020). The model's effectiveness in conducting cross-boundary analyses relies on three key elements: (i) integrating climate variables to accommodate changing conditions, (ii) accurately representing water quantity at appropriate spatial and temporal scales, and (iii) incorporating anthropogenic pressures, including agricultural, industrial, and urban inputs. This comprehensive approach ensures a holistic understanding of the territory and aligns with the EU's objective of harmonising transboundary tools for assessing ecosystem services.

Allocating modelling methodology at a large scale with high spatial resolution is also important to quantify ES linked to the water cycle, crop management and watershed dynamics. The large-scale modelling allows us to consider multiple watersheds in the same project and their interactions in the case of highly anthropised watersheds (with canals, water transfers and dams). The modelling methodology can be applied to any other approach when the model fulfils large-scale criteria (Moriasi et al., 2007).

The methodology to estimate and value the provisioning ES proposed in this article allows us to compare the current situation for basins of the transboundary SUDOE territory from the point of view of the quantity of agricultural production and freshwater for domestic uses assets and their economic valuation. It allows us to obtain the results at hydrological scales, though most of the information in France, Spain and Portugal is available at an administrative scale (national, regional or local). This was possible by utilising a change of scale matrix based on irrigated and non-irrigated areas for agriculture production and population for drinking water use.

The quantification was done at the scale of sub-basins because economic data was unavailable at a smaller scale. However, using SWAT modelling, ES could be quantified at a much smaller scale and constitute a perspective for future work.

4.2 A new framework for the quantification and economic valuation of provisioning ES (agricultural products and drinking water) in South-Western Europe

The proposed methodology allowed for identifying vulnerable areas in terms of agricultural products. There are sub-systems within the SUDOE region, that compared with the rest of the sub-systems, they provide a higher number of tonnes of agricultural products. These are the Guadalquivir (General Regulation area), followed by Júcar and Segura

basins, and the Upper and Central Guadiana areas in Spain. The first three also provide the highest ES in monetary units due to their large size, followed by Tejo e Ribeiras do Oeste in Portugal. However, in relative terms, some internal basins of Andalusia and Catalonia and some areas in the Guadalquivir basin stand out as regions with a high value of agricultural production despite their small size. These sub-systems would be identified as critical due to the high value of the provisioning of agriculture SE that they provide. Specifically, the Guadalquivir sub-system and the East basins in Spain can also be identified as vulnerable due to the foreseeable increase in temperatures and decrease/irregularity in rainfall. Nevertheless, as De Groot et al. (2012) point out, it is imperative to be very careful when interpreting monetary values per surface area as often the provision of services is not proportional across the whole system being valued due to factors such as spatial heterogeneity or decreasing returns to scale, among others.

This methodology also allowed the identification of two critical areas in terms of water supply to the population: the sub-systems to which Madrid and Lisbon belong, as well as the South of France and the East coast of Spain. The economic value and quantification of water supply are the highest in these regions, facing increasing water scarcity associated with climate change. The problem of water scarcity is even more substantial as moving southward of SUDOE, as the climate is becoming more arid with lower latitudes.

This study provides a methodology to compare environmental and economic indicators at the same scale. For example, in the sub-systems to which Madrid belongs, the drinking water supply is important, unlike agricultural production. The area around Madrid is characterised by low production pastoralism.

4.3 Towards a systemic approach to help decision-making and reach socio-ecosystem sustainability

Following Butler and Oluoch-Kosura (2006), while inadequate access to the ecosystem service of food provisioning is a critical factor in the loss of human well-being, attention must nevertheless be paid to all ecosystem services (not just provisioning), as they contribute in some way to well-being. Although perhaps evident to ecologists, the links between ecosystems and aspects of human well-being, including health, have been less well understood among the social science community. As Jorge-García et al. (2023) state, prioritising ecosystem services is essential to start a decision-making process focused on environmental policies, highlighting the necessity of equilibrium between sustainability and human well-being.

The causality between ecosystem services and human well-being is bidirectional: functioning societies with some level of well-being may protect or enhance ecosystem services, while societies with declining well-being may also experience a decline in ecosystem services, in particular, provisioning services through reduced population engagement in extensive agriculture, and recreation and access to clean water through reduced engagement in purification.

In the last decade, with climate change and climate disasters, the question of the well-being of our ecosystems is becoming central for more and more people, particularly younger generations. For that purpose, environmental issues are central to the quest for well-being. Nowadays, considering economic services regardless of the environmental impact might not be relevant. This study suggests evaluating several ES to have a global overview of one management rule.

The chosen conceptual framework to evaluate provisioning ES in this work also has the potential to be improved. Thanks to the SWAT model, productivity and water supply assessments over the territory have been possible, but the SWAT simulation depends on the initial assumption during the model implementation. The land cover used as model data input is static and does not evolve in time and space. This is the main limitation of this study. There are possibilities to make the landscape evolve through the SWAT model. Further studies can focus on that for future research.

In the same way, the evaluation of freshwater consumption does not evolve in space and time. Further studies should try to understand whether water savings made by one or more communes significantly impact ES on a sub-basin scale. Likewise, considering information relating to three countries and not being available in joint databases (Eurostat) with the same methodology implies an added difficulty. It has been necessary to look for common ground in each of the situations, being aware that the most up-to-date information possible is not being used in each case, but rather than that of neighbouring countries conditions this.

These results can be interesting for stakeholders as they highlight the ecosystem services. However, these results on provisioning ES valuation should now be studied in parallel with the capacity of ecosystems to provide these services in the long term, especially in the context of climate change. In fact, in many regions, there is an overexploitation of water resources that leads to groundwater level decreases that are not sustainable (Lopez et al., 2022). Valuing nature is necessary to ensure that decision-making and policy design are made to recognise these values (Termansen et al., 2022). However, the valuation of nature is relevant but insufficient for public decision-making in the framework of social and environmental policies. Over the last 30 years, studies evaluating ecosystem services have increased significantly. However, incorporating valuations into public decision-making has been hampered by some valuation methods' lack of robustness and reliability, a lack of resources to commission studies, or insufficient technical capacity (Barton et al., 2022). In this sense, the market-based values of nature must be balanced with the relational, intrinsic and non-market instrumental values that are also part and parcel of the reasons nature matters to people (Pascual et al., 2023).

5 Conclusion

In conclusion, the ES of drinking water supply is valued at 5.7 billion euros, and the ES of supplying products from agriculture is valued at 51 billion euros per year in the SUDOE territory. In relative terms, drinking water is valued at 85 €/ha per year on average, while the products from agriculture are considered to reach 750 €/ha per year on average (current 2015 euros). Just to give a reference, though comparison makes no sense, there are no studies in SUDOE territory, and comparison cannot be done because of spatial heterogeneity reasons. These estimates are included in the ranges De Groot et al. (2002) provided: 3–4600 $/ha per year for water supply and 6–1014 $/ha per year for food. In this paper, a higher value per hectare was obtained for agricultural ES than for drinking water supply ES, results that go in the opposite direction to the results of De Groot et al. (). This may be related to the important participation of the primary sector in the SUDOE territory. Also, Costanza et al. (1997) estimated the average global value of annual ES per biome and the type of ES to be compared.

Table 4 Selling prices of crop products (absolute prices)—2015 annual Price

€/Tn	Spain	France	Portugal
Soft wheat	181.2	223.0	183.9
Durum wheat	292.3	494.0	280.0
Barley	174.0	244.0	196.0
Maise	172.9	201.0	172.4
Rice	276.8	235.9*	274.4
Cotton	501.7		
Sunflowers	364.4	483.0	375.0
Early potatoes	354.4	1396.0	275.3
Main crop potatoes	198.3	437.0	195.6
Peaches	561.0	1532.5*	981.8
Dessert apples	377.3	412.6*	575.9
Dessert pears	547.8	666.1*	621.9
Cherries	1459.1		1759.9
Plums	611.2		836.9
Almonds	1854.8		1015.6
Oranges	185.6		397.6
Mandarins	278.7		613.0
Lemons	530.9		573.0
Dessert grapes	701.7	1515.5*	1373.0
Grapes for wine production	316.0	551.0*	371.1
Table olives	716.8	1545.3*	491.3
Other olives	560.6		348.4

Source EUROSTAT

*Data for 2011

This article highlights the level of provisioning ES provided by the ecosystem and shows the significant differences in their valuation according to spatial location. This should be studied with other regulating and cultural ES to provide sustainable resource uses, ecosystems, and human well-being (Costanza, 2020).

In further studies, it could be interesting to evaluate provisioning ES and their evolutions regarding climate change. Specific trends in ES could be identified across the SUDOE, a vast and diverse territory which cannot be seen on a regional scale (Uniyal et al., 2023).

Appendix

See Table 4.

Development of a water resource management platform during low water periods in the SUDOE region (SOE1/P5/F0026 project, http://www.aguamod-sudoe.eu) and funded by ERDF.

Data availability Data sources are included in Table 2.

Declarations

Conflict of interest There is no conflict of interest.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by/4.0/.

References

Acharya, R. P., Maraseni, T., & Cockfield, G. (2019). Global trend of forest ecosystem services valuation—An analysis of publications. *Ecosystem Services, 39*, 100979.

Agreste. (2018). La estatistique, l'evaluation et la prospectivedu Ministère de l'Agriculture et de la Souveraineté Alimentaire. Ministériel de l'agriculture et de la Souveraineté alimentaire. https://www.agreste.agriculture.gouv.fr/agreste-web/disaron/SAAAR_DEVELOPPE_2/detail/

Ala-Hulkko, T., Kotavaara, O., Alahuhta, J., & Hjort, J. (2019). Mapping supply and demand of a provisioning ecosystem service across Europe. *Ecological Indicators, 103*, 520–529.

Albert, C., Bonn, A., Burkhard, B., Daube, S., Dietrich, K., Engels, B., Frommer, J., Götzl, M., Grêt-Regamey, A., Job-Hoben, B., Koellner, T., Marzelli, S., Moning, C., Müller, F., Rabe, S.-E., Ring, I., Schwaiger, E., Schweppe-Kraft, B., & Wüstemann, H. (2016). Towards a national set of ecosystem service indicators: Insights from Germany. *Ecological Indicators, 61*, 38–48.

Aquastat. (2019). FAO, AQUASTAT database [WWW Document]. Retrieved July 30, 2019, from http://www.fao.org/nr/water/aquastat/data/query/index.html?lang=en

Ashagre, B. B., Platts, P. J., Njana, M., Burgess, N. D., Balmford, A., Turner, R. K., & Schaafsma, M. (2018). Integrated modelling for economic valuation of the role of forests and woodlands in drinking water provision to two African cities. *Ecosystem Services, 32*, 50–61.

Aziz, T., Nimubona, A.-D., & Van Capellen, P. (2023). Comparative valuation of three ecosystem services in a Canadian watershed using global, regional, and local unit values. *Sustainability, 15*(4), 11024.

Barton, D.N., Chaplin-Kramer, R., Lazos, E., … & González-Jiménez, D. (2022). Chapter 4: Value expression in decision-making. In P. Balvanera, U. Pascual, M. Christie, B. Baptiste, & D. González-Jiménez (Eds.), *Methodological assessment report on the diverse values and valuation of nature of the intergovernmental science-policy platform on biodiversity and ecosystem services*. IPBES secretariat. https://doi.org/10.5281/zenodo.6522261

Bateman, I. J., Carson, R. C., Day, B., Hanemann, M., Hanley, N., Hett, T., Lee, M. J., Loomes, G., Mourato, S., Özdemiroğlu, E., Pearce, D., Sugden, R., & Swanson, J. (2002). *Economic valuation with stated preference techniques: A manual*. Edward Elger.

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G., & Turner, K. (2011). Economic analysis for ecosystem service assessments. *Environmental and Resources Economics, 48*(2), 177–218.

Battude, M., Al Bitar, A., Brut, A., Tallec, T., Huc, M., Cros, J., Weber, J.-J., Lhuissier, L., Simonneaux, V., & Demarez, V. (2017). Modeling water needs and total irrigation depths of maize crop in the south west of France using high spatial and temporal resolution satellite imagery. *Agricultural Water Management, 189*, 123–136. https://doi.org/10.1016/j.agwat.2017.04.018

Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters, 12*(12), 1394–1404.

Brouwer, R., Ordens, C. M., Pinto, R., & de Melo, M. T. C. (2018). Economic valuation of groundwater protection using a groundwater quality ladder based on chemical threshold levels. *Ecological Indicators, 88*, 292–304.

Burkhard, B., Crossman, N., Nedcov, S., Petz, K., & Alkemade, R. (2013). Mapping and modelling ecosystem services for science, policy and practice. *Ecosystem Services, 4*, 1–3. https://doi.org/10.1016/j.ecoser.2013.04.005

Butler, C. D., & Oluoch-Kosura, W. (2006). Linking future ecosystem services and future human well-being. *Ecology and Society, 11*(1), 30.

Cakir, R., Raimonet, M., Sauvage, S., Paredes-Arquiola, J., Grusson, Y., Roset, L., Meaurio, M., Navarro, E., Sevilla-Callejo, M., Lechuga-Crespo, J. L., Gomiz Pascual, J. J., Bodoque, J. M., & Sánchez-Pérez, J. M. (2020). Hydrological alteration index as an indicator of the calibration complexity of water quantity and quality modeling in the context of global change. *Water, 12*, 115. https://doi.org/10.3390/w12010115

Campardon, M.; Loubier, S.; Morardet, S. (2012). The current situation of irrigation in France and its evolutions since 2000—Rapport final convention BSE—IRSTEA 2011–2012; CemOA: Antony, France p. 58. Water 2020, 12, 115 31 of 33.

CICES (2022). Common international classification of ecosystem services. European Environment Agency. https://cices.eu

Clawson, M. (1959). *Methods of measuring the demand for and value of outdoor recreation. Reprint no. 10*. Resources for the Future.

Corona-López, E., Román-Gutiérrez, A. D., Otazo-Sánchez, E. M., Guzmán-Ortiz, F. A., & Acevedo-Sandoval, O. A. (2021). Water-food nexus assessment in agriculture: A systematic review. *International Journal of Environmental Research and Public Health, 18*(9), 4983.

Costanza, R. (2020). Valuing natural capital and ecosystem services toward the goals of efficiency, fairness, and sustainability. *Ecosystem Services, 43*, 101096.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature, 387*, 253–260.

de Groota, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L. C., ten Brink, P., & van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services, 1*(1), 50–61.

De Groot, R., Wilson, M. A., & Boumans, R. M. J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics, 41*(3), 393–408.

EEA Report (2018). European waters—Assessment of status and pressures 2018 [WWW Document]. European Environment Agency. Retrieved March 22, 2019, from https://www.eea.europa.eu/publications/state-of-water

Eurostat (2018a). Selling prices of crop products https://ec.europa.eu/eurostat/databrowser/view/APRI_AP_CRPOUTA/default/table?lang=en

Eurostat (2018b). Population and social conditions. https://ec.europa.eu/eurostat/databrowser/view/TGS00096/default/table?lang=en

Eurostat (2019). Water use by NUTS 2 regions—Eurostat [WWW Document]. The Statistical Office of the European Union. Retrieved May 14, 2019, 2019 https://ec.europa.eu/eurostat/web/products-datasets/-/env_watuse_r2

Eurostat (2023). Nomenclature of territorial units for statistics—Eurostat. https://ec.europa.eu/eurostat/web/nuts/background

Evans, J. R., & Poorter, H. (2001). Photosynthetic acclimation of plants to growth irradiance: The relative importance of specific leaf area and nitrogen partitioning in maximizing carbon gain. *Plant, Cell & Environment, 24*, 755–767. https://doi.org/10.1046/j.1365-3040.2001.00724.x

FAO (2022). Ecosystem services & biodiversity. Provisioning services. Food and Agriculture Organization of the United Nations (FAO). http://www.fao.org/ecosystem-services-biodiversity/background/provisioningservices/en/

Gilmore, E. C., & Rogers, J. S. (1958). Heat units as a method of measuring maturity in corn 1. *Agronomy Journal, 50*, 611–615. https://doi.org/10.2134/agronj1958.00021962005000100014x

Gleyses, G., & Rieu, T. (2004). *L'irrigation en France: État des lieux 2000 et évolution*. Editions Quae.

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics, 69*(6), 1209–1218.

Gordon, L. J., Finlayson, C. M., & Falkenmark, M. (2010). Managing water in agriculture for food production and other ecosystem services. *Agricultural Water Management, 97*(4), 512–519.

Goulder, L. H., & Kennedy, D. (1997). Valuing ecosystem services: Philosophical bases and empirical methods. In In. Daily (Ed.), *Nature's services: Societal dependence on natural ecosystems* (pp. 23–47). Island Press.

Grizzetti, B., Liquete, C., Pistocchi, A., Vigiak, O., Zulian, G., Bouraoui, F., De Roo, A., & Cardoso, A. C. (2019). Relationship between ecological condition and ecosystem services in European rivers, lakes and coastal waters. *Science of the Total Environment, 671*, 452–465.

Guerbois, C., & Fritz, H. (2017). Patterns and perceived sustainability of provisioning ecosystem services on the edge of a protected area in times of crisis. *Ecosystem Services, 28*, 196–206.

Herriges, J. A., & Kling, C. L. (Eds.). (1999). *Valuing recreation and the environment: revealed preference methods in theory and practice*. Edward Elgar.

INE-Portugal (2018a). Regional economic accounts for agriculture and crop production statistics. Instituto Nacional de Estatística. https://www.ine.pt/xportal/xmain?xpgid=ine_tema&xpid=INE&tema_cod=1510

INE-Portugal (2018b). Service fees. Entidade Reguladora dos Serviços de Águas e Resíduos. https://www.ersar.pt/pt/consumidor/tarifas-dos-servicos/encargos-tarifarios/pesquisa-por-concelho

INE-Spain (2018a). Conversion coefficients for irrigated areas between NUTS-2 and River Basin, 2011–2014. Instituto Nacional de Estadística. https://www.ine.es/jaxi/Tabla.htm?path=/t26/p067/p05/&file=02001.px

INE-Spain (2018b). Statistics on water supply and sanitation. Instituto Nacional de Estadística. https://www.ine.es/dynt3/inebase/en/index.htm?padre=8709&capsel=8711

INE-Spain (2018c). Water economic indicators. Instituto Nacional de Estadística. https://ine.es/jaxi/Tabla.htm?tpx=53456&L=0

Jorge-García, D., Estruch-Guitart, E., & Aragonés-Beltrán, P. (2023). How geographical factors and decision-makers' perceptions influence the spriroritisation of ecosystem services: Analysis in the Spanish rice field areas in RAMSAR Mediterranean wetlands. *Science of The Total Environment*, 869.

Karabulut, A., Egoh, B. N., Lanzanova, D., Grizzetti, B., Bidoglio, G., Pagliero, L., Bouraoui, F., Aloe, A., Reynaud, A., Maes, J., Vandecasteele, I., & Mubareka, S. (2016). Mapping water provisioning services to support the ecosystem–water–food–energy nexus in the Danube River basin. *Ecosystem Services, 17*, 278–292.

Kibria, A. S., Costanza, R., Groves, C., & Behie, A. M. (2018). The interactions between livelihood capitals and access of local communities to the forest provisioning services of the Sundarbans Mangrove Forest, Bangladesh. *Ecosystem Services, 32*, 41–49.

Krause, M. S., Nkonya, E., & Griess, V. C. (2017). An economic valuation of ecosystem services based on perceptions of rural Ethiopian communities. *Ecosystem Services, 26*, 37–44.

Krieger, D. J. (2001). *Economic value of forest ecosystem services: A review*. The Wilderness Society.

La Notte, A., D'Amato, D., Mäkinen, H., Paracchini, M. L., Liquete, C., Egoh, B., Geneletti, D., & Crossman, N. D. (2017). Ecosystem services classification: A systems ecology perspective of the cascade framework. *Ecological Indicators, 74*, 392–402.

Lara-Pulido, J. A., Guevara-Sanginés, A., & Martelo, C. A. (2018). A meta-analysis of economic valuation of ecosystem services in Mexico. *Ecosystem Services, 31*, 126–141.

LifeAdaptamed (2020). LIFE Adaptamed project. Protecting key ecosystem services through the adaptive management of Mediterranean ecosystems threatened by climate change. LIFE14 CCA/ES/000612. www.lifeadaptamed.eu/

Lopez, J. R., Winter, J. M., Elliott, J., Ruane, A. C., Porter, C., Hoogenboom, G., Anderson, M., & Hain, C. (2022). Sustainable use of groundwater may dramatically reduce irrigated production of maise, soybean, and wheat. *Earth's Future*, 10(1), e2021EF002018.

Loubier, S., Campardon, M., & Morardet, S. (2013). L'irrigation diminue-t-elle en France? Premiers enseignements du recensement agricole de 2010. *Sciences, Eaux & Territoires*, 12–19.

Maes, J., Teller, A., Erhard, M., Conde, S., Vallecillo Rodriguez, S., Barredo Cano, J. I.,…, & Santos-Martín, F. (2020). Mapping and assessment of ecosystems and their services: An EU ecosystem assessment, EUR 30161 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-76-17833-0. https://doi.org/10.2760/757183, JRC120383

Malinga, R., Gordon, L. J., Jewitt, G., & Lindborg, R. (2015). Mapping ecosystem services across scales and continents—a review. *Ecosystem Services, 13*, 57–63.

Marre, J.-B., & Rl, B. (2019). A demand-driven approach to ecosystem services economic valuation: Lessons from Pacific island countries and territories. *Ecosystem Services, 3*, 100975.

Meli, P., Vieli, L., Spirito, F., Reyes-Riveros, R., Gonzalez-Suhr, C., & Altamirano, A. (2023). The importance of considering human well-being to understand social preferences of ecosystem services. *Journal for Nature Conservation, 72*, 126344.

Ministère de l'Agriculture et de la Souveraineté Alimentaire (2018). Annual agricultural statistics (SAA). https://agriculture.gouv.fr/french-ministry-agriculture-and-food

Ministerio de Agricultura, Pesca y Alimentación (2018). Crop surface area and yields survey (ESYRCE). https://www.mapa.gob.es/en/estadistica/temas/estadisticas-agrarias/agricultura/esyrce/default.aspx

Ministerio de la Presidencia (2001). Ley 10/2001, de 5 de julio, del Plan Hidrológico Nacional; BOE-A-2001-13042. Madrid, Spain.

Ministerio de Medio Ambiente (2002). Plan Hidrológico Nacional; Secretaría de Estado de Aguas y Costas: Madrid, Spain.

Monteith, J. L. (1977). Climate and the efficiency of crop production in Britain. *Philosophical Transactions of the Royal Society of London b. Biological Sciences, 281*, 277–294. https://doi.org/10.1098/rstb.1977.0140

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., & Veith, T. L. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE, 50*(3), 885–900.

Kang, N., Hou, L., Huang, J., & Liu, H. (2022). Ecosystem services valuation in China: A meta-analysis. *Science of the Total Environment, 809*, 151122.

National Research Council (2005). *Valuing ecosystem services: toward better environmental decision-making*. The National Academies Press. https://doi.org/10.17226/11139

Nuissi, H., Haase, D., Lanzendorf, M., & Wittmer, H. (2009). Environmental impact assessment of urban land use transitions—a context-sensitive approach. *Land Use Policy, 26*, 414–424.

Palacios-Agundez, I., Onaindia, M., Barraqueta, P., & Madariaga, I. (2015). Provisioning ecosystem services supply and demand: The role of landscape management to reinforce supply and promote synergies with other ecosystem services. *Land Use Policy, 47*, 145–155.

Palmquist, R. B. (1991). Hedonic methods. In J. B. Braden, & C. D. Kolstad (Eds.), *Measuring the demand for environmental quality* (pp. 77–120). North Holland.

Pascual, U., Balvanera, P., Anderson, C. B., Chaplin-Kramer, R., Christie, M., González-Jiménez, D., et al. (2023). Diverse values of nature for sustainability. *Nature*, 1–11.

PORDATA (2018) Domestic use of water. Estatísticas sobre Portugal e Europa. https://www.pordata.pt/DB/Municipios/Ambiente+de+Consulta/Tabela

Sander, H. A., & Haight, R. G. (2012). Estimating the economic value of cultural ecosystem services in an urbanizing area using hedonic pricing. *Journal of Environmental Management, 113*, 194–205.

Shabman, L. A., & Batie, S. S. (1978). Economic value of natural coastal wetlands: A critique. *Coastal Zone Management Journal, 4*(3), 231–247.

Sispea (2018). Données Agences de l'eau. Services Eau France. Observatoire national des services d'eau et assainissement. https://www.services.eaufrance.fr/

Tan, M. L., Gassman, P. W., Yang, X., & Haywoo, J. (2020). A review of SWAT applications, performance and future needs for simulation of hydro-climatic extremes. *Advances in Water Resources*, *143*. https://doi.org/10.1016/j.advwatres.2020.103662

TEEB (2023). The economics of ecosystems and biodiversity. https://teebweb.org/

Termansen, M., Jacobs, S., Mwampamba, T. H., & González-Jiménez, D. (2022). Chapter 3: The potential of valuation. In P. Balvanera, U. Pascual, M. Christie, B. Baptiste, & D. González-Jiménez (Eds.), *Methodological assessment report on the diverse values and valuation of nature of the intergovernmental science-policy platform on biodiversity and ecosystem services*. IPBES secretariat. https://doi.org/10.5281/zenodo.6521298

Trice, A. H., & Wood, S. E. (1958). Measurement of recreation benefits. *Land Economics, 34*(3), 195–207.

UN (2005). Millennium ecosystem assessment. Global Assessment reports. https://www.millenniumassessment.org/en/index.html

Wilson, J. W. (1981). Analysis of growth, photosynthesis and light interception for single plants and stands. *Annals of Botany, 48*, 507–512.