WORKING PAPER

#656 | 2023

WP

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The Economic Value of Land-Based Ecosystem Services in Portugal: A Spatially Explicit Approach[1]

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Abstract: This study estimates the economic value of seven land based ecosystem services for mainland Portugal in 2018. The estimated services are Climate Regulation, Drought Regulation, Erosion Prevention, using the market price methodology, and Food Supply, Pollination, Recreation and Water Purification using a meta-analytic benefit transfer function. By estimating a unique meta-analytic benefit transfer function for each service, the commodity consistency condition is addressed. Different welfare measures were not pooled together and methodological variables are not included in the vector of explanatory variables. The results are spatially explicit at the hectare level providing the benchmark to which the consequences of land-use changes to the value of ecosystem services and, therefore, to the welfare of local populations can be adequately assessed.

Keywords: Ecosystem Services; Meta-analysis; Benefit Transfer; Economic Valuation; Portugal

[1] **Acknowledgements:** This study was supported by national funds through FCT under the ASEBIO project (PTDC/CTA-AMB/28438/2017). CVS partially acknowledges support from FCT (UIDB/00124/2020, UIDP/00124/2020, and Social Sciences DataLab PINFRA/22209/2016), POR Lisboa and POR Norte (Social Sciences DataLab PINFRA/22209/2016).

1. Introduction

Ecosystem services (ES), an intrinsically anthropocentric concept (Fisher & Kerry Turner, 2008), are the direct and indirect benefits that people obtain from ecosystems (e.g., MEA, 2005). As life-support systems, they are key to human well-being and development (e.g., de Groot et al., 2012; Hufnagel et al., 2018; MEA, 2005). They are typically classified into Provisioning, Regulation and Maintenance, and Cultural Services (Haines-Young & Potschin, 2018).

Due to their public good features, ES are mostly nonmarketed, originating market failures. Hence, they are typically neglected in public policy decisions (Costanza et al., 1997; TEEB, 2010), resulting in degradation and overexploitation of the environmental asset. Increasing human pressure affects negatively ecosystems' conditions compromising their ability to provide benefits. Therefore, not only economic assessment of ES has been recognized by decision and policymakers as a useful tool to assess ES's contribution to well-being, but it is also acknowledged as a key element to integrate biodiversity conservation policies in the EU and worldwide (EU, 2011, 2020).

Building upon a spatially explicit approach, this paper aims to assess the economic value of seven land-based ES in Portugal. The selected ES are Climate Regulation (*ClimReg*), Drought Regulation (*DrouReg*), Erosion Prevention (*EroPrev*), Food Supply (*FoodSup*), Pollination (*Polli*), Recreation (*Rec*), and Water Purification (*WaterPur*). The results are spatially explicit and adjusted to account for the biophysical supply of the services.

Whenever possible and available, market prices are used to estimate the economic value of the selected ES. In the absence of market prices, a meta-analytic benefit transfer function, under a refined framework, is used to value the selected ES.[2] This research contributes to the existing literature (Boyle & Wooldridge, 2018; Nelson & Kennedy, 2009; and Vedogbeton & Johnston, 2020) by proposing a refinement to the meta-analytical benefit transfer function's framework in what concerns the "commodity consistency condition", the "welfare consistency condition", and the exclusion of methodological explanatory variables in the meta-regression model.

Different methods for non-market economic valuation have been developed in the literature, using revealed and/or stated preference data, conditional on the different types of value at stake (Nelson, 2015; Segerson, 2017). In fact, these methods assess both direct and indirect use values as well as non-use values (Goulder & Kennedy, 1997; Nelson, 2015; Segerson, 2017). However, when time and

[2]Economic values will be estimated in €/ha/y, and then multiplied by the supply values provided by the ASEBIO project. For clarifications regarding the measurement of the supply of the selected ES please see (Cabral et al., 2016; Chaplin-Kramer et al., 2019; Marques et al., 2021; Sharp et al., 2020; Vallecillo et al., 2019; Wentling et al., 2021)

financial constraints prevent the implementation of primary valuation studies, Benefit Transfer has been largely adopted in the literature to overcome the lack of location-specific studies (de Groot et al., 2012; Nelson, 2015). Benefit transfer relies on the estimates of past studies to obtain the monetary values for similar ecosystems in new policy and geographical contexts after making some adjustments (Johnston & Rosenberger, 2010). There are different types of possible adjustments, and despite some methodological issues (Nelson, 2015; Nelson & Kennedy, 2009), a meta-analytic benefit transfer function can show significant improvements when compared to other adjustments (e.g., unit transfer) (Moeltner et al., 2007; Nelson, 2015). This method has been applied in multiple contexts, such as to value the world's ES (Costanza et al., 1997), the value of ES provided by green infrastructure (Barton et al., 2015), in Lakes (Reynaud & Lanzanova, 2017), and, more recently, to ES provided by green and blue spaces in cities (Bockarjova et al., 2020).

Commodity and welfare inconsistency has been prevalent in many applications as dissimilar services, and different welfare measures are pooled in the same model. Many authors include both Marshallian and Hicksian welfare measures in the same model without adjustments. Examples in the literature include Costanza *et al*., (1997), Moeltner *et al*. (2007), Quintas-Soriano *et al*. (2016), and Zhou *et al*. (2020). While Quintas-Soriano *et al*. (2016) and Zhou *et al*. (2020) pooled different welfare measures, in Moeltner *et al*. (2007) the welfare commodity consistency was addressed by assuming that the income effect is negligible, implying that the full set of estimates refers to the same welfare measure, that is, consumer surplus. However, different welfare measures cannot be directly comparable without adjustments (Bergstrom & Taylor, 2006). Thus, Johnston et al. (2017) only pooled similar welfare measures when estimating the willingness to pay (WTP) for water quality improvements. In a literature review on environmental economics meta-analysis, Nelson & Kennedy (2009) found that 26 studies combined both Stated and Revealed Preference data out of 130 studies. Typically, authors control for the differences in welfare measures using vectors of dummy variables for each valuation method. However, using dummy variables to measure additive differences in welfare measures may not be enough as these scalar differences do not consider non-linear relations between Hicksian and Marshallian welfare measures deriving from income and quantity effects (Bergstrom & Taylor, 2006). Similarly, meta-regression usually considers the dissimilarities between commodities by introducing a vector of dummy variables, each controlling a specific service. For non-transformed WTP, this implies that commodities differ up to an additive scalar. A log transformation of the WTP would imply that they differ by a multiplicative scalar (Vedogbeton & Johnston, 2020). This excludes any sort of non-linear relation between different commodities.

Furthermore, Moeltner *et al*. (2007), in the "N x K" dilemma, suggested that explanatory variables should only be included in a model if they are common to all primary studies, i.e., if they can be collected in every primary study site. As a consequence, only variables such as GDP (per capita or per area) are included, as in Quintas-Soriano et al. (2016), Reynaud & Lanzanova (2017), or Bockarjova *et al*. (2020). This results in the omission of relevant variables for some services albeit irrelevant for others (Vedogbeton & Johnston, 2020), which may generate an omitted variable bias. This type of bias is typically controlled by including methodological variables in the models.

Though contentious, including methodological variables in the prevision model has been standard practice in the literature. Moeltner *et al*. (2007), proposing a refinement to the literature, suggested a Bayesian approach that assigned probabilities to each combination of methodological attributes in contrast to the standard approach, i.e., assigning the average value of the meta-database to the methodological attributes followed by Nelson (2005). Yet, the true economic value of the service should be independent of these variables (Boyle & Wooldridge, 2018), and thus their exclusion may be relevant. Therefore, the contribution of this paper is twofold. First, by pooling in the same model only similar commodities and second by pooling similar welfare measures. Despite the methodological differences in the present study, when compared with previous studies, it is still possible to track symmetries between the estimated values and those in the literature.

The results obtained are value maps estimated for different scenarios[3]. In particular, the evidence suggests that forested areas present higher economic values for *ClimReg*, Polli, and *EroPrev* as carbon sequestering values are typically larger in these areas, pollinators abundance is higher, and more sediments, on average, are retained, respectively. Estimates for *WaterPur'*s WTP are higher in regions with higher population density[4] and lower baseline water quality levels, such as the *Lisbon Metropolitan Area* (AML) and *Porto Metropolitan Area* (AMP). This result is not surprising as respondents are willing to pay less when the baseline water quality is good (Meyerhoff *et al*., (2014). High-income areas typically show higher economic value as WTP for *WaterPur* increases with income as in Bonnichsen & Olsen (2016). The recreational value increases with the distance from the Equator[5] in the meta-database, and with population density, whenever there is no crowding-out effect from increased population density in primary study sites, as the relationship between the economic values and population density, in the dataset, is linear and positive. Spatial value distributions show that the *Alentejo* region, in the southern part of Portugal, presents lower economic values, except for *ClimReg* and *FoodSup*. *ClimReg* shows higher economic values in non-urban areas compared to urban areas,

[3] A complete list and detailed explanation of each scenario is present in the Methodology section.
[4] From hereafter Population Density refers to both population density from individuals living in a determined study site and to visitation density from individuals to a determined study site.
[5] Distance to Equator measures latitude in absolute values.

and *FoodSup* reflects agricultural land use patterns in Portugal by showing high economic values in *Alentejo*, littoral *Centro* and interior *Norte* NUTs II regions.

The remainder of this study is organized as follows. Section 2 describes the methodological approach followed in the ES valuation at a national scale, including the meta-database construction and the econometric estimation. Section 3 presents the estimated results, value maps, and general comments. Finally, section 4 discusses the results and addresses the limitations of the meta-analytical benefit transfer literature, and the challenges underlying the methodology proposed in this paper. Policy remarks regarding land-use and climate change policy management conclude the paper. In the supplementary materials the search strategy, the primary study list, GIS sources, external dataset references, spatial references for Portugal, and ES averages per group are presented.

2. Methodology

This section presents the methodology followed for the economic valuation of the selected services in Portugal. While *ClimReg, DrouReg,* and EroPrev, are valued using market prices, for the remainder, that is, *FoodSup*[6]*, Polli, Rec, and WaterPur,* a meta-analytic benefit transfer function was used instead. Sections 2.1 and 2.2 relate only to the services estimated through meta-analytic benefit transfer while Section 2.3 includes all the services.

2.1. MetaDatabase Construction

A clear and well-defined protocol for primary studies selection is paramount to constructing the meta-database (Nelson, 2015). To this end, the protocol suggested in Boyle & Wooldridge (2018) was followed. This protocol sets five steps: 1) The identification of the measure to be analyzed and the definition of the value; 2) Identification of the relevant primary studies; 3) Selection of value estimates from studies; 4) Conversion of value estimates to a common metric and 5) The choice and coding of the estimated model. More recent studies also followed the same protocol such as Bockarjova et al., (2020) and Vedogbeton & Johnston (2020).

Primary studies were recovered from three online sources: Scopus, Web of Science, and Environmental Valuation Reference Inventory (EVRI). A Boolean search was applied separately to each ES and source. A detailed description of the search strategy can be found in Supplementary Materials S.1. To account for publication bias, both peer-reviewed and grey literature studies were included (Nelson, 2015).

[6] We did not use market prices for Food Supply as there is no available information on prices and quantities at the national scale.

Primary studies were filtered at five stages. Table 1 shows the number of primary studies at each stage. In the first stage, studies were recovered from the online repositories. In the second stage, they were filtered by publication year, title and abstract. Studies published before 2000 were discarded as the commodity to be valued may have changed significantly over time. Studies without clear information on whether there exists a reference to an economic valuation of a specific ES in either the title or the abstract were discarded, as well as duplicate studies. In the third stage, studies without a clear location-specific identification were not considered as the location is a cornerstone of the approach followed in this study.

In the fourth stage, centroid coordinates were recovered for each study site using Arcmap[7] and cross-referenced with the RESOLVE 2017 EcoRegions map. For a study to be considered, the centroid coordinates of the study site should overlap with any of the two biomes present in Portugal: "Mediterranean Forests, Woodlands & Shrubs", and "Temperate Broadleaf & Mixed Forest". This requirement was imposed to ensure methodological consistency, as some study sites may not be fully contained within any of the two biomes considered. Land-cover was measured using Corine Land Cover (CLC) 2018 Version 2020_20u1 levels 2 and 3.

Table 1: Primary Studies selection stages

Service	Stage 1	Stage 2	Stage 3	Stage 4	Stage 5
Food Supply	1525	183	154	70	18
Pollination	112	58	41	22	12
Recreation	2016	446	401	152	33
Water Purification	1149	255	214	100	35
Total	**4802**	**942**	**810**	**344**	**98**

The final stage[8] involved a detailed screening of each remaining study. The studies were then selected based on the valuation method, the quality of the estimates, and their relevance to the research question. Studies with estimates for more than one service were considered more than once and their estimates were included in the respective ES to increase the set of observations. Our final sample of studies consists of 18 for *FoodSup*, 12 for *Polli*, 33 for *Rec*, and 35 for *WaterPur*.

[7] The reference for the full set of maps used in this stage can be found in Supplementary Materials S.3. Shapefiles were not available for all locations. Observations in which the centroid coordinates were recovered using other data sources are identified in the appendix.
[8]The complete list of primary studies included in the meta-database can be found in Supplementary Materials S.2.

2.2. Standardization of estimates

There are different formats under which the estimates in the primary studies can be reported. Value estimates should be converted to the same monetary, temporal, and spatial units while explaining all the assumptions followed in the process before the value transfer. Albeit controversial (Ghermandi *et al*., 2010; Londoño & Johnston, 2012; Reynaud & Lanzanova, 2017), this transformation is fundamental. At the primary study level, we use market prices for *FoodSup,* and both market prices and restoration costs for *Polli,* while for *Rec* and *WaterPur* WTP is used. As services are measured differently, it is key to distinguish between the different value measures. Hereinafter, monetary estimates, value estimates or estimated values refer to the final estimated values, regardless of using market prices, restoration costs, or WTP.

WTP estimates were not aggregated at the representative population level as we do not know it and because WTP can vary with distance and frequency of visits to the study site (Bateman *et al*., 2006).[9] Estimates measured in Marginal Willingness to Pay (MWTP) are multiplied by the average number of units of the commodity supplied in the valued scenario in the primary study. We assume MWTP is constant for all supply levels of the service implying that MWTP is equal to the mean WTP. Typically, WTP estimates are measured yearly. When this is not the case, the estimates were adjusted to elicit the WTP/year. Finally, WTP estimates were converted to the same spatial unit by dividing the individual WTP estimate by the study-site area in hectares.

Then, all primary studies' estimates, data on income, and market prices for the remaining services were converted to PT2018€ following a two-stage procedure. First, the values were deflated to the 2018=100 base. Then, the obtained values were converted to INT$ and then from INT$ to PPT (Purchasing Power Parity) PT€. Monetary estimates are in PT2018€ per hectare per year allowing for a direct comparison to other studies.

2.3. Econometric Estimation

In this section, the econometric estimation procedure is explained in detail. The different methods used are presented in the different sections. While section 2.3.1 considers market prices, section 2.3.2 explores the meta-analytic benefit transfer function method.

2.3.1 Market Prices

Estimating the value of ecosystem services using market prices requires information about the unit price of the service being valued and the quantity of the service supplied. The quantity of the service

[9] While this is not the best solution, given data restrictions at the primary study level this approach was followed. This issue is further explored in Section 4.

supplied was given by Cabral *et al*. (2021). Note that study site refers to the primary study valuation site while policy site refers to the site being valued in this paper. Each price, defined by equation (1), was recovered from official estimates and the literature.

$$p_j = \bar{p}_k + r_k \tag{1}$$

In (1), p_j stands for the unit price of the service, at the policy site to be valued (j), $\bar{p}_k$ is an average of unit prices of service at different sites k, and r_k is an error term defined by equation (2), accounting for heterogeneity at the price level. The measurement error term e_k in (2) is assumed to average out to zero given a large enough set of unit prices, as follows:

$$r_k = u_k + e_k \tag{2}$$

ClimReg unit price is set by the 2018 European Emission Trading Scheme (ETS) average permit price. This price is an average of the k-set of daily prices of CO2 allowances traded in the ETS. The measurement error and heterogeneity are assumed to average out to zero as information is assumed to be fully included in the price, and there is a large set of daily prices. This price is multiplied by the sequestered amount of Carbon Dioxide, CO_2 in t/ha, after adjusting from Carbon to Carbon Dioxide, using the atomic conversion rate of 3.67.

DrouReg unit price is set by the national average of the municipal water tariff in 2018 from *Entidade Reguladora dos Serviços de Águas e Resíduos* (ERSAR). This price was obtained by averaging the municipal tariff between all municipalities of Portugal. This price may not directly reflect the cost of surface water, and measurement errors are bound to occur at the observation level as they may also reflect the cost of groundwater, supply inefficiencies, and eventually some assessment of location-specific water scarcity. However, these non-negligible measurement errors are assumed to average out to zero given the large set of prices. These prices are multiplied by the supplied amount of water in m^3/ha.

EroPrev costs were obtained from the literature. Marta-Pedroso *et al*. (2007) estimated the restoration cost of replacing the nutrients (organic matter, Phosphorous and Potassium) lost from erosion and the cost of returning the eroded sediments to the farmland as a proxy. Replacement costs on nutrients were recovered from national firms and different on-site costs of erosion were estimated. The price per tonne was recovered by dividing the restoration cost per hectare by the level of erosion in the indicative scenario of 3.7tonnes of eroded soil per hectare. This estimated cost of erosion was adjusted to PT2018€ and multiplied by the service provision level in t/ha.

2.3.2 Meta-Analytic Benefit Transfer Function

As previously mentioned, services for which market prices are unavailable are valued using a Meta-analytic Benefit Transfer Function. Each service is valued individually, and methodological variables are not included in the regressors as discussed in the introduction since the true underlying value of the service should be independent of the valuation method. We argue that by estimating each service separately while including service contingent variables, both this issue and the "NxK" dilemma can be accommodated. Hence, the meta-analytic benefit transfer function is defined as follows:

$$y_j^* = \alpha + v_k\beta_1 + x_k\beta_2 + r_k \quad (3),$$

where y_j^* denotes the value of the service estimated at the policy site j and r_k is the error term defined by

$$r_k = u_k + e_k \quad (4)$$

that is, including an i.i.d. error term u_k and a measurement error e_k averaging out to zero for a large enough set of primary studies. The measurement error accounts for differences derived from different valuation methods at the primary study level, such as the choice of payment vehicle or elicitation format. The u_k term measures unobservables at the primary study level, uncorrelated with the explanatory vectors affecting y_j^*. In (3), v_k is a vector of spatial attributes common to both the study site and the policy site. Attributes in this vector include protected areas[10], continent, biome, latitude, and land cover using the CLC classification.[11] x_k is a vector of attributes common to both the study and policy sites, contingent on the service being valued. Attributes in this vector vary across services and include the share of Agriculture in the GDP, the share of organic farming in the total farming area, pollinator species, type of pollinator[12], income, and population density. u_k is assumed to follow a normal distribution, as in (5) below, and e_k is assumed to average out to zero given the careful selection of primary studies

$$u_k \sim \mathrm{N}(\mu, \sigma^2) \quad (5)$$

Equation (3) was estimated using ordinary least squares (OLS). We report robust standard errors as suggested by Nelson & Kennedy (2009) for conducting hypothesis testing in meta-analytic studies.

[10] A dummy variable that takes 1 if there are any protected areas in the study site and 0 otherwise. No distinction between study sites entirely or partially covered by a protected area was made.
[11] Corine only classifies land-cover in Europe. Studies outside of Europe were converted to Corine's classification using information available both in the primary study and online repositories. Level 2 classification was used at this stage when level 3 information was not available. *FoodSup*, *Polli* and *Rec* are estimated using level 2 information. *WaterPur* is estimated using level 3 information.
[12] Restored or naturally occurring in nature.

FoodSup estimates are multiplied by the supply map which values food provision as a dummy variable taking the value 1 if there is the provision of agriculture and 0 otherwise. *Polli*, *Rec*, and *WaterPur* supply maps are measured based on an index ranging from 0 to 1. *Polli* uses a pollinator's abundancy index, measuring the suitability to host bee's species (*Apis mellifera*). *Rec* is measured by a suitability index, considering the land's suitability for nature-based recreation, the presence of elements that provide nature-based recreational opportunities, and the presence of water elements. Finally, *WaterPur* measures nutrient retention (Chaplin-Kramer et al., 2019; Vallecillo et al., 2019; Wentling et al., 2021). These indexes were weighted by a factor obtained from equations (7), (8), and (9) below. This weight adjusts the supply levels so that the average of the transferred monetary estimates matches the average supply for each service in Portugal. Monetary estimates are adjusted according to vectors v_k and x_k and the adjusted values are multiplied by (6) which is the weighted supply index provided by the ASEBIO project, as follows:

$$Ajusted\ Service_{supply} = Service_{weight} * Service_{supply} \quad (6)$$

Polli's abundancy index is weighted by a factor from equation (7)

$$P_{weight} = \frac{Pollinator\ abundance\ index\ at\ policy\ site\ j{,}land\ cover\ i}{Average\ pollinator\ abundace\ index\ at\ policy\ site\ j} \quad (7)$$

In (7), the denominator measures the average for the pollinator abundance index in Portugal, while the numerator considers the pollinator abundance index at each pixel in Portugal. *Rec* is weighted by equation (8).

$$R_{weight} = \frac{Recreation\ propensity\ at\ policy\ site\ j{,}land\ cover\ i}{Average\ recreation\ propensity\ at\ policy\ site\ j} \quad (8)$$

and *WaterPur*'s weight is defined by (9), as follows:

$$WP_{weight} = \frac{Nutrient\ retention\ at\ policy\ site\ j{,}land\ cover\ i}{Average\ Nutrient\ retention\ at\ policy\ site\ j} \quad (9)$$

Rec and *WaterPur*'s values, when the land-cover attributes are missing, are estimated using the average for the available land covers under Corine level 1. When it is not possible to use these averages, i.e., when there is no land-cover attribute under that level 1 group, the average of all the land covers was used. Table 2 presents the different scenarios estimated for each service.

Table 2: Ecosystem Services Valuation Scenarios

Ecosystem Services	**Valuation Scenarios**
Climate Regulation	ETS price of 15€/ton
Drought Regulation	Municipal tariff averaged to the national level

Erosion Prevention	Cost of erosion as in Graves et al. (2011)
Food Supply	Meta-Analytic Benefit Transfer Function (CLC average)
Pollination	Meta-Analytic Benefit Transfer Function (CLC average)
Recreation	Meta-Analytic Benefit Transfer Function (Average CLC level 1)
Water Purification	Meta-Analytic Benefit Transfer Function (Average CLC level 1)

3. Results

This section presents descriptive statistics from the meta-database and the estimation results from ES valuation using Market Prices and Meta-analytic Benefit Transfer Function methodologies.

3.1. Descriptive Statistics

Preliminary descriptive statistics in Table 3 highlight the discrepancy between values in primary studies. Under *FoodSup*, the mean value is 90 times larger than the minimum value and over 7 times lower than the maximum value. For *Polli*, the differences are larger. The minimum value is about 1500 times lower than the mean value and the maximum value is 7 times higher than the mean value.

Under *Rec* and *WaterPur*, the mean WTP is more than 350,000 times and 140,000 times larger than the minimum value, respectively. Also, the maximum value is almost 7 times and 27 times larger than the mean value, respectively.

Table 3: Descriptive Statistics

Ecosystem Service	Mean - €/ha/y	Min - €/ha/y	Max - €/ha/y
FoodSup	891.7416	9.808881	6574.655
Polli	1500.039	0.9826286	10379.79
Rec	0.2029759	0.000000528	1.370611
WaterPur	0.1007165	0.000000703	2.598813

These values hide differences between countries, land cover, and protected areas. For example, while in *FoodSup* the mean value is 873.22€/ha/y, the mean value of this service in the United Kingdom (UK), representing 37.84% of the sample, is 1433.69€/ha/y[13]. On the other hand, Bulgaria with only one observation shows a mean value of 13.92€/ha/y. Under Polli, *Heterogeneous Agricultural Areas* have a mean value of 7.25€/ha/y while *Permanent Crops* have a mean value of 1969.68€/ha/y about 270 times larger. Under *Rec*, the mean WTP/ha/y for protected areas is almost 43 times lower than for non-protected areas. Thus, mean values hide striking differences between groups as they are contingent on the primary study site and its characteristics. This is relevant as most of the studies are conducted in the same regions, mainly the UK and USA, which may be causing a sampling problem unable to be addressed without increasing the number of primary studies available for systematic review. While this may impose a bias in our sample of primary studies, it is not possible to overcome it without dropping a large number of observations as the UK and USA were pioneers in the economic valuation of environmental goods and services, and therefore the number os studies available for those countries is much larger. A detailed distribution of the values between services and groups can be found in Supplementary Materials S.6.

3.2. Econometric Estimation

Table 4 presents the estimated results from the different meta-regressions. For *FoodSup*, differences in land cover show that *Permanent Crops* present, on average, lower monetary estimates for the service, relative to the baseline - *Arable Land*. When the share of agriculture in GDP decreases by 1%, the value of *FoodSup* decreases by 2.03% (significant at 10% level). This can be due to technological innovation in farming. Not only do agricultural intensive countries have a lower multiplier in the agro-business sector than developed countries (Arias, 2016)[14], but also the OECD[15] also acknowledges that technological innovation is key for increased food productivity. This result is then expected as countries with a lower share of agriculture in GDP are more industrialized, with more productive farming, resulting in higher economic output per hectare than countries with a higher share of agriculture in GDP.

For *Polli*, the results show that *Permanent Crops*' value is 500% of that in *Arable Lands*. This coefficient is not only large but also significant at 1% level. A possible explanation is the predominance of fruit

[13] We estimated a model for *FoodSup* without the UK observations. The signs of the coefficients are similar for all variables with the exception of biomes. Removing the observations from the UK improves the general fit of the model by increasing the R^2 and more variables become statistically significant at a 5% level. However, tests to the structure of the model point towards the presence of omitted variable bias in addition to the loss of a large number of observations. Therefore we keep the observations from the UK.

[14] See https://www.iica.int/en/press/news/contribution-agriculture-development

[15] See https://www.oecd.org/agriculture/topics/agricultural-productivity-and-innovation/

trees and berry plantations (Corine Level 3) in this land cover. In our dataset, this land cover shows a higher value, compared to the baseline. On the other hand, protected areas show an estimated value 300% higher than in non-protected areas, significant at a 1% level.

Table 4: Estimates of the meta-regression models

	FoodSup[1]		*Polli*[2]		*Rec*[3]		*WaterPur*[4]	
CLC Level 2[a]								
Heterogeneous Agricultural Areas	-1.153	(1.263)	0.904	(1.261)				
Pastures	-.107	(0.664)						
Permanent Crops	-1.273**	(0.483)	5.018***	(0.56)				
Artificial non-agriculture vegetated areas					-4.266	(2.447)		
Inland Waters					-5.086**	(-0.471)		
Marine Waters					-1.25	(5.782)		
Open Spaces with Little or no Vegetation					-6.727	(1.933)		
CLC Level 3[b]								
Estuaries							-0.24593	(1.17185)
Inland Marshes							-0.76978	(0.96562)
Sea and Ocean							0.62926	(1.47252)
Water Courses							-1.43333	(0.95052)
Geographic Attributes								
Protected Areas	1.019	(0.653)	3.444***	(0.819)	-6.022***	(1.397)	0.94862	(0.61719)
Temperate Broadleaf & Mixed Forests	-0.817	(1.299)		**c**	0.284**	(6.131)	0.09179	(1.01017)
Asia	1.516	(1.942)	2.152*	(1.085)	-5.189	(7.078)	3.50486*	(2.07211)
Europe	-0.631	(0.62)	3.66	(2.76)	-7.418	(2.001)	1.62223	(1.28717)
Oceania		**d**		**d**	-10.556	(1.961)	1.37766	(1.48271)
Latitude		**e**	-0.072	(0.094)	0.125	(0.088)		**e**
Ecosystem Service Specific Attributes								
Log of Agriculture Share of GDP	-2.028*	(1.041)						
Organic share of total farming area	-5.208	(4.805)						
Natural pollinators			-2.033	(2.303)				
Honeybee			-3.89***	(0.832)				
Native Bee			-.903	(1.907)				

Wild Insects			-4.96***	(0.75)				
Income							0.0002***	(0.00004)
Log Population Density					0.063**	(0.595)	0.843***	(0.12965)
Constant	-1.138	(3.279)	5.048	(4.188)	-19.774*	(1.457)	-14.03912***	(2.5037)

**** p<.01, ** p<.05, * p<.1*
Robust SEs are reported in parentheses
1: Baselines are Arable Land; Non-protected Areas; Mediterranean Forests, Woodlands & Shrubs; America.
2: Baselines are Arable Land; Non-protected Areas; America; Bumblebees.
3: Baselines are Forests; Non-protected Areas; Mediterranean Forests, Woodlands & Shrubs; America.
4: Baselines are Water Courses; Non-protected Areas; Mediterranean Forests, Woodlands & Shrubs; America.
a: FoodSup, Polli and Rec were estimated using CLC level 2 data.
b: WaterPur was estimated using CLC level 3 data.
c: The inclusion of this variable in the model generated collinearity problems.
d: Observations from Oceania were not available in the FoodSup and Polli services´ metadatabase.
e: The inclusion of this variable in the model generated misspecification problems.

Under *Rec*, inland waters show on average an economic value 500% lower than forested areas. While this suggests that users value recreation services in forests more than in rivers or lakes, this may also result from different scales. As WTP is measured in €/ha/y, the primary study site area may impose a scaling effect on this value, that is, the value may be biased downwards by primary studies with larger site areas and upwards by those with smaller study site areas. Due to the need of a common spatial metric for the value definition, it is not possible to circumvent all limitations. Moreover, aggregating the estimates based on the population of interest would result in stricter assumptions upon the value of the services. This explains the approach followed in this paper.

In protected areas, WTP for *Rec* is 600% lower than in non-protected areas, significant at a 1% level. As protected areas are typically far from urban areas, it could be the case that WTP is lower due to distance decay. However, Söderberg & Barton (2014) argue that distance decay observed in previous contingent valuation studies may come from the definition of 'protest' and 'true zero' respondents.[16]. Furthermore, despite the potential for delivering benefits, the provision of protected areas may not always generate recreation demand and the associated economic benefits as there may exist other local circumstances, such as changes in visitors' preferences, infrastructure degradation, and the economic situation at different scales (local, regional, national or international) that may negatively influence local recreational demand (Booth *et al*. 2010). We found evidence of statistically significant differences in biomes as *Temperate Broadleaf & Mixed Forests* have higher WTP compared to *Mediterranean Forests, Woodlands & Shrubs.* Magalhães Filho *et al*. (2021) also found statistically

[16] Söderberg & Barton (2014) extended the standard model of contingent valuation to allow for a sequential decision process. In the first stage respondents decide whether they are 'protesters', ´true zero' or have a WTP>0. In the second stage, conditional on a strikingly positive WTP, WTP is estimated. Results show that the likelihood of respondents being 'protester' or stating a 'true zero' decreases with proximity to the study site. No distance decay was observed for the positive WTP estimates. The authors argue that a correct specification of 'protest' and 'true zeros' responses is key to a correct interpretation of distance decay.

significant differences for cultural services between biomes in a global meta-analysis of ecosystem services values. Nonetheless, caution is advised as 84% of the sample is from study sites in *Temperate Broadleaf & Mixed Forests*[17]. Population density is statistically significant at a 5% level with a positive impact. Data suggests that this relationship is linear. As long as congestion does not occur, higher visitation rates reflect higher attractivity to a recreational site, generating larger monetary benefits. In contrast, Manning & Valliere (2001) found that changes in the demand for recreation sites is a common coping mechanism in response to overcrowding in outdoor recreation. In (Richardson *et al*., 2006), visitors closer to the recreational sites were found to accrue higher benefits from recreation as visits are more frequent. Thus, if those recreational sites are closer to regions with higher population density, a large number of short-distance recreational trips are expected, providing larger monetary estimates for *Rec.* Conversely, we expect a distance decay effect, driving visitors to the nearest recreational areas, particularly close to urban centers. While this distance decay is considered axiomatic there are cases in which it is not apparent, such as when visiting national parks (Hanink & White, 1999).

It is important to note that due to the predominance of primary studies from the UK and USA in the sample with, on average, higher WTPs than in other regions, in the northern hemisphere[18], the further north the higher the estimated value. This may be explained by different attributes that characterize the local populations, namely related to income, education level, preferences, among others.

There are no significant statistical differences in land cover, biome, and protected areas for *WaterPur* (Table 7). However, similarly to *Rec*, both the log of population density and a new covariate, Income, have a positive impact on WTP, significant at 1%. When income increases, typically WTP also increases as individuals can afford to allocate more income to restoration projects through tax payments or water bills. This is in line with the results obtained in primary studies (e.g., Meyerhoff *et al*. (2014), where higher WTPs are found in more populated areas and with higher income. Typically, these areas also present poor water quality. In this case, agents are willing to pay more for the maintenance or improvement of water quality rather than when quality levels are higher (Meyerhoff *et al*. 2014).

3.3. Spatial Distribution of Ecosystem Services' Economic Value

In this subsection, figures 1 to 7 show the distribution of the estimated values in the Portuguese mainland territory and spatial references for all of the sites mentioned throughout this work

[17] We estimated the models without data from Temperate Broadleaf & Mixed Forests. The loss in observations compromised the power of the results and problems of collinearity and misspecification arose in the different model specifications. Therefore, this data was included in the model.
[18] Our meta-database only includes studies valued in the northern hemisphere for this service.

For *ClimReg* a clear contrast between urban and non-urban areas is observed. This is particularly salient when comparing *AML* and *AMP* to the surrounding forested areas and the remaining areas of the country. *Sintra-Cascais Natural Park* and *Arrábida Natural Park*, for example, show values 10 times as large as those found in urban areas. *Continuous Urban Fabric's average estimated value is* 375€/ha/y while *Broad-leaved Forest's average estimated value is* 8 800€/ha/y. With an average value of 556€/ha/y, *Burnt Areas'* value is significantly lower than forested areas, which is indicative of the potential social losses caused by fires, namely due to release of sequestered carbon. In fact, the burnt area surrounding *Pedrogão Grande* is visible in the map. This area suffered from massive wildfires in 2017, resulting in 43,201 hectares of burnt forest (Gomes, 2017)[19].

In *DrouReg*, as expected, differences between regions are found. Areas in the central and northern littoral of Portugal are found to be valued 10 times higher than those in *Alentejo* and *Algarve*. This is relevant as the *Alentejo* is also the region with the highest value for *FoodSup*. We observe that in central Portugal, particularly in the *Serra da Estrela Natural Park* and *Peneda-Gerês National Park*, the monetary value for *DrouReg* is higher, reflecting precipitation patterns that typically characterize this region. Also, any difference found in the spatial distribution of monetary values is solely due to the difference in supply values of the service as we are using a national average water tariff to value *DrouReg.*

Erosion is not usually evaluated underwater. The provisioned amount, and thus the estimated value, of *EroPrev* is zero in aquatic areas. On the other hand, *Bare* Rock, shows an average value of 272.5€/ha/y. Urban areas also show lower estimated values, with *Continuous Urban Fabric* showing an average value of 45.7€/ha/y. This contrasts with seminatural and forested areas, particularly *Moors and Heathland* and *Sparsely Vegetated Areas*, with present an average value of 649.8€/ha/y and 454.6€/ha/y, respectively. While vegetation is very important to this service provision, land management practices and current policies in the Mediterranean region are failing to halt soil erosion prevention (Guerra *et al*. 2016), which is also expected to decrease with land abandonment. However, erosion rates can be significantly reduced with vegetation recovery and preservation of agricultural terraces (López-Vicente *et al*. 2013). That is, a sustainable land-use management may contribute positively to erosion prevention.

The spatial distribution of *FoodSup*'s estimated values reflects the agricultural development patterns in Portugal. Estimated values are higher in *Alentejo* hinterland and in specific areas along the coastline. These are associated with *Non-irrigated Arable Land,* predominant in the *Alentejo* hinterland with an

[19] See: https://observador.pt/2017/07/03/area-ardida-prejuizos-as-contas-todos-os-numeros-da-tragedia-de-pedrogao-grande-em-4-graficos/

average value of 626€/ha/y. *Permanently Irrigated Land,* with an average value of 487€/ha/y, is predominant in Littoral *Alentejo*, where estimated values are also high. Additionally, this is the region where *DrouReg* presents lower estimated values and water availability for hydro-agricultural use has been downclimbing since 2015 in both the *Sado* and *Guadiana* Rivers basins. For example, in one of the basins in the region (*Roxo's* reservoir), water availability decreased by 39% between 2013 and 2018 according to data from *Sistema de Informação de Regadio* (SIR). Moreover, in 2020, only one lake in the *Sado* river basin was subject to the contingency level of no hydro-agricultural water deficit. In a context of increased drought frequency and severity as climate change scenarios for the southern part of Portugal reveal, land-use change and water management practices should be seriously considered to better inform sustainable policy decision-making. In the *Sado* region, less precipitation volume and frequency is expected in winter, with negative consequences to aquifers´s sustainability. This in turn will affect agriculture, increasing the dependence on irrigation (Santos & Miranda, 2006).

Polli's value distribution reflects mostly the difference between protected and non-protected areas. As shown in Table 3, the monetary estimates of *Polli* in protected areas are, on average, three times higher than in non-protected areas. Moreover, non-urban areas show higher estimated values than urban areas, with *Moors and Heathland* presenting an average estimated value of 991€/ha/y compared to *Continuous Urban Fabric* with 4€/ha/y. Nonetheless, these results have to be put in context as one primary study represents more than one-third of the sample[20].

The spatial distribution of *Rec* is reflected in the econometric estimation results presented in Table 3. We observe higher WTPs near large urban centers as population density has a positive impact on economic benefits of recreation, as pointed out in section 3.2. Similarly, the impact of latitude is clear, as the further north we move, the higher the economic value. Also, a difference between the monetary values of protected and non-protected areas is salient, particularly when we compare the value at *Serra da Estrela Natural Park* with the surrounding areas. We can also see that *Water Courses* and *Water Bodies* show higher economic value as seen in the *Tagus River* and *Alqueva Dam*. These two land-covers show an average monetary value of 0.3644015€/ha/y and 0.169141€/ha/y, respectively, while *Agro-forestry* areas show an average monetary value of 0.0062936€/ha/y.

Under *WaterPur* the spatial distribution values reflect what we learnt before. We find higher economic value in *AML* and *AMP* as people are willing to pay more for improvements in water quality when water quality levels are lower as observed by the econometric estimation results. Urban areas show significantly higher average estimated monetary values than non-urban areas. For example,

[20] One primary study is responsible for 13 observations, from the same land-cover type and considered as protected area, with an average Market Value higher than the remaining observations.

Continuous Urban Fabric has an average monetary value of 0.001276€/ha/y, more than four-fold those in *Broad-Leaved* (0.0003089€/ha/y). Similarly, in Protected areas, such as *Serra da Estrela Natural Park*, the monetary value of *WaterPur* is higher than in non-protected areas, as shown by the results in Table 3. The differences found between the coastline and inland territory reflect population density and income distribution dynamics, as these two variables are key to WTP estimates for *WaterPur* improvements.

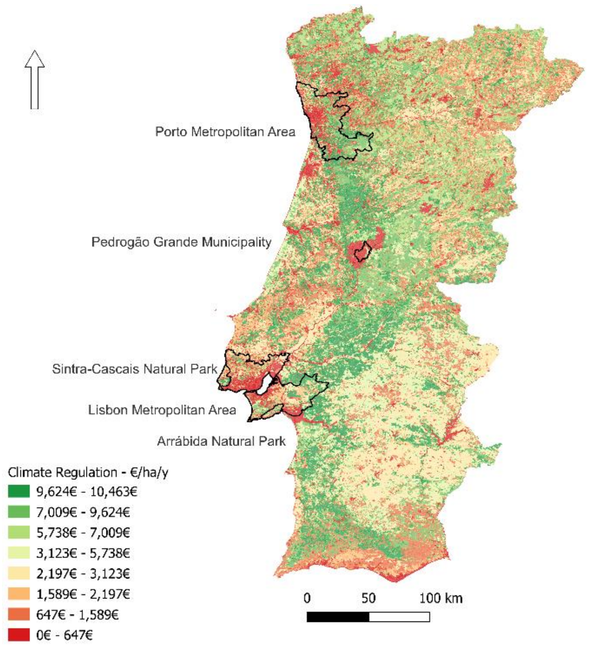

Figure 1 – ClimReg

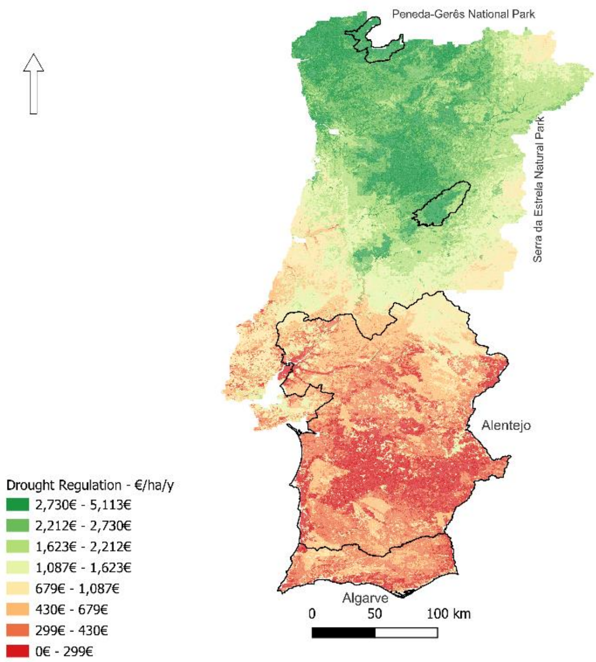

Figure 2 - DrouReg

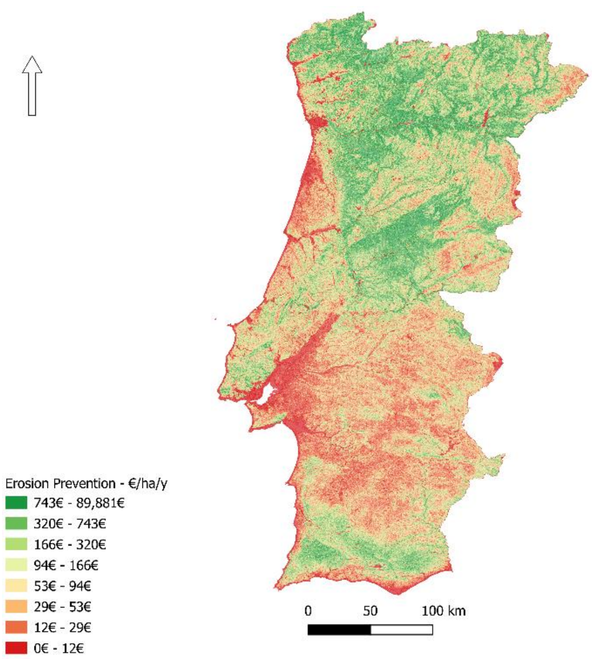

Figure 3 – EroPrev

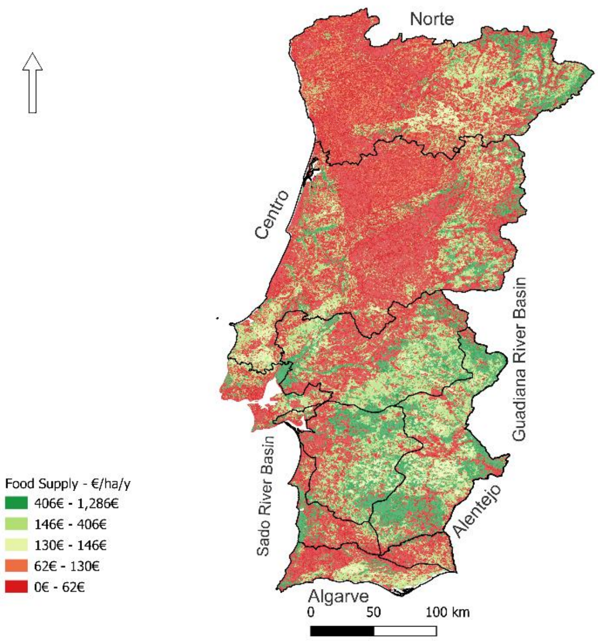

Figure 4 - FoodSup

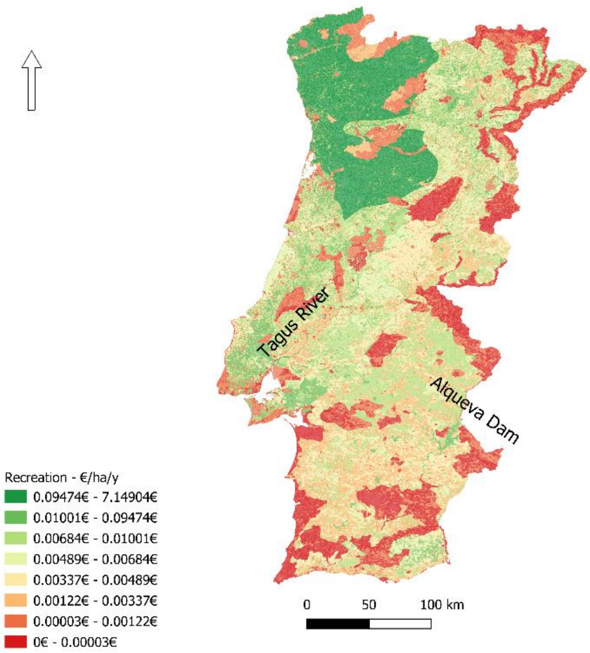

Figure 5 – Rec

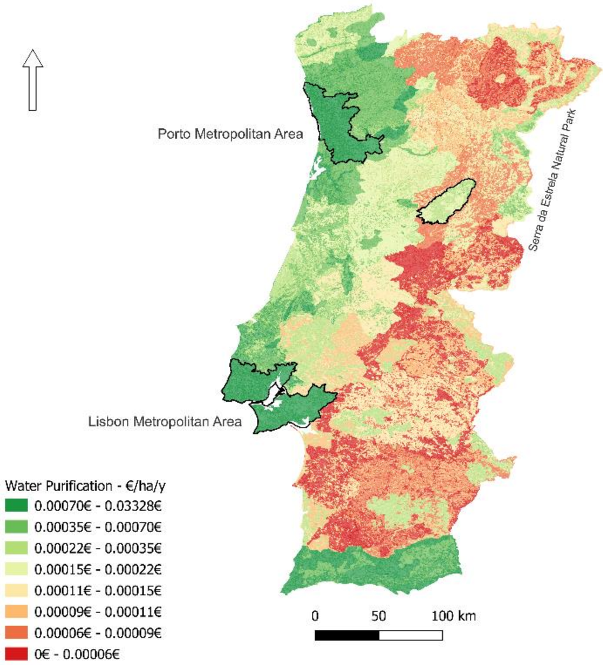

Figure 6 – WaterPur

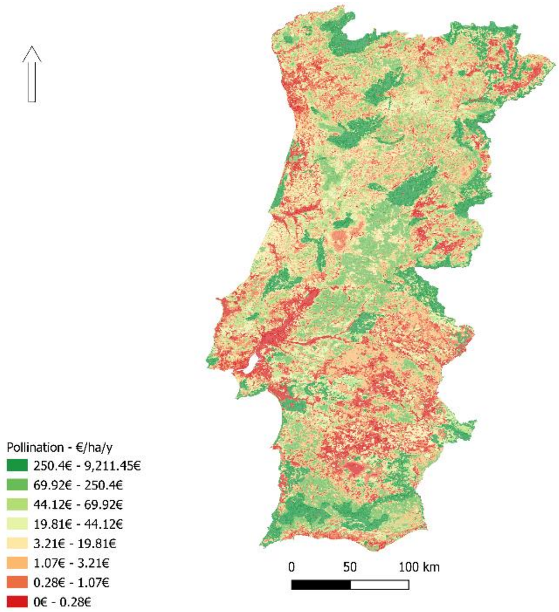

Figure 7 - Polli

4. Discussion

This paper contributes to the literature by proposing a spatially explicit approach to estimate the economic value of ecosystem services at a national scale. It combines the economic valuation of a supply unit for seven ES with supply maps provided by INVest and other methods (Cabral *et al*. 2021). Moreover, market prices are used whenever they are available, and a refined meta-analytic benefit transfer function is considered otherwise.

The proposed refinement for the meta-analytic benefit transfer function methodology incorporates recommendations from the literature. This method is not only replicable but also follows a protocol that can be easily implemented at a national, regional, or even local scale elsewhere. By incorporating both spatial and ES contingent attributes, it allows for a significant degree of detail, estimating over 8.9 million value points for Portugal for each service. The estimated meta-regression results can be transferred to any other study site as long as the biomes considered in the primary studies are similar to those present in Portugal and the covariates are adjusted accordingly. In case the study site is significantly different from Portugal, the meta-database can be adjusted as the protocol used in this paper can easily accommodate adjustments.

Moeltner *et al*. (2007) concluded that the omission of methodological variables typically generates omitted variable bias. However, after testing the individual and joint-significance of the squared and cubic predicted error in each of the models, we did not reject the null hypothesis for each model of absence of omitted variable bias.

Multicollinearity is not an issue as the VIF estimate in all models was always below the threshold of 10.0. Leverage and influential points were observed in each service database. Data points were then removed if, when plotted, significant visual signs of outlier behavior were found when compared to the remainder of the data distribution for each covariate. This has significantly improved the estimates and the predictive power of each model.

The defined protocol for the sampling of primary studies allows to appropriately address both the commodity and welfare consistency conditions. By estimating each model separately, dissimilar commodities were not mixed. Moreover, by restricting the sample of primary studies to those that are located in any of the two biomes present in Portugal, we assumed that not only the commodities were the same but they also show similar characteristics as biomes can be expected to reflect different characteristics for different services.

Usually, cluster robust standard errors at the study level are estimated (Nelson & Kennedy, 2009). However, this comes with a trade-off. While intra-study correlation is expected (Nelson, 2015; Nelson & Kennedy, 2009; Vedogbeton & Johnston, 2020), clustering at the study level will create groups with only one observation[21]. Moreover, this may not be the only instance of correlation as there could exist inter-study correlation as well, as, for example, study site effects can occur in multiple primary studies from the same site. Abadie *et al*. (2017) pointed out that clustering should only be done if there at the sampling and assignment levels, which should be investigated before implementing it. We decided not to use cluster-robust standard errors as standard errors were already conservative from the Huber/White estimators, and the estimator for the majority of the models would not be of sufficient rank under clustering to perform the modal tests.

The estimated results and value maps are tools that can help understanding some of the challenges faced by Portugal in the the context of climate change in implementing sustainable land-use management policies, particularly, adaptation policies. In the agricultural sector, there is a clear

[21] This limits Stata's ability to estimate the F-statistic for each model. The estimators may not of have a sufficient rank to perform the modal test due to the large number of clusters and low number of observations.

contrast between traditional land-use management and more modern water and land-intensive agriculture. For example, *Montados,* a class characteristic of the Mediterranean region, have been acknowledged as land use systems of high natural and social values, providing important services, and promoting biodiversity conservation (Ferraz-de-Oliveira *et al*. 2016; Pinto-Correia *et al*. 2011). However, land-use policies in the last decades, particularly with the increase of irrigation in the region, have contributed to the abandonment of the traditional *Montado* and to an increase in intensive olive oil, almonds plantations and fruit trees (Palma *et al*. 2020). This monocultures trend resulted in soil erosion, depleted soil fertility, and increased fire risk and loss of ES. Restoration policies should revolve around the plantation of native trees and shrubs, as a mosaic landscape with diversified tree cover and vegetation generates ecological and socioeconomic values (Martins-Loução, 2021; Ritsche *et al*. 2021).

In the Alentejo region, where typically Montado is predominant and intensive agriculture is being developed, the recent increase in the irrigation area is pressing the already strained water systems. November 2021 was the third driest month of the last 90 years (Volta e Pinto, 2021)[22]. As mentioned before, land-use policies in the agricultural sector, in face of Climate Change, cannot ignore the increase in frequency and intensity of such events. Hence, the adoption of sustainable solutions for the maintenance and protection of ecosystems, namely by promoting the appropriate incentives to induce adaptation to the new circumstances, is a matter of urgent need. Among others, payments for ecosystem services (PES) can be an interesting market instrument to be considered. Based on a voluntary agreement betwen the parts involved, these arrangements provide compensation to local populations conditional on the delivery of previously set conservation outcomes. The incurred opportunity costs have to be assessed, thereby ensuring service provision. The implemented scheme should reflect the spatial diversity of the selected site (e.g., ecological, socioeconomic, and cultural characteristics) and should also be designed to allow for a rigorous evaluation of its environmental and socioeconomic impacts.

By creating a payment scheme, indexed to the EU ETS market, in which landowners are incentivized to sequester carbon, land-use changes that benefit land-covers with higher sequestration potential may be induced.[23] For example, as *Agroforestry* areas, on average, sequester twice as much carbon as *Fruit Trees and Berry Plantations,* a well-designed PES could improve the relative returns to the traditional *Montado* systems, benefiting landowners. Moreover, as *Montado* systems also preserve

[22] See: Mês de Novembro em Portugal foi “muito frio” e o terceiro mais seco em 90 anos | Meteorologia | PÚBLICO (publico.pt)

[23] For example, the ETS market prices have increased significantly in 2021, reaching 88.88€/tonne in December 8th of 2021, while the values estimated in this paper use a price of 15.5€/tonne, as of 2018. This trend is expected to continue.

the good condition of watersheds, a more sustainable use of those particularly vulnerable lands would contribute to the welfare of populations. A recent study by Palma *et al*. (2020) provides evidence that the agro-silvopastoral land cover (*Montado*), has an important positive effect on water quality, acting as a buffer to the contamination of the streams. In contrast, the intensive agricultural and urban LULC areas may contribute to the decline of water quality.

When choosing the appropriate price for *DrouReg* is important to have in mind that prices per m3 of water should reflect scarcity. However, these prices often don't internalize scarcity. This sub-optimal pricing policy may persist over long periods due to the monopolistic nature of water providers at the local level (Zetland, 2021). In a context of increased water scarcity, water prices at the municipal level are not uniform, reflecting location-specific conditions both on supply and demand sides. That is, the prices may not reflect the geographical conditions, the relative abundance of the resource, as well as management choices, among others, at the local level. For example, it is worth noting that in Alentejo, where supply and demand levels are lower, the municipal tariffs are also lower, raising the question of whether water scarcity is included in the price. On the other hand, AML and AMP show higher tariffs, which is consistent with the fact that higher consumption would induce higher tariffs despite a higher availability of water supply in these regions compared with Alentejo, for example. Thus, the municipal price was averaged at the national level to eliminate price differences, allowing for a direct comparison between the different regions with different caracteristics. In this case, the differences reflect the supply side.

Similarly, the choice of the price for erosion was not obvious. While studies, such as Ghaley *et al*. (2014) used an average price of soil for vegetable gardens, it is our understanding that a more appropriate measure of the service's unit price should be used. Marta-Pedroso *et al*. (2007) estimated a price that reflects the costs per ton of lost soil from losses in Phosphorus, and Potassium. Despite having been estimated for a specific region of Portugal in 2007, since they refer to the Portuguese territory, we believe they provide a more appropriate measure of the unit value of erosion.

Nonetheless, the meta-regressions also present limitations. For example, most of *FoodSup*'s observations originate in the same country and land cover. With the UK representing 37.5% of the sample, estimates may be biased upwards due to the higher average value at the primary study level that characterizes UK observations. Ideally, the estimation of this service should be conducted using the information on agricultural output and prices practiced at the policy site. As this information was not available, we had to rely on benefit transfer. A further improvement of the estimates presented here should include a new set of attributes containing information on location-specific prices and output as the geographic location of the agricultural areas plays a key role to obtain a more realistic spatial distribution of value.

Between 1990 and 2018, there have been contrasting Pollinator's abundancy and supply trends in different regions of Portugal (Wentling *et al*. 2021). While tipping points of pollinators decline have not been identified yet, 87% of all plants depend on pollinators, and pollinator losses may induce ecosystem service collapse and consequent losses for human well-being (Christmann, 2019). As primary studies typically measure pollination either as the share of agriculture for which pollinators contribute, that is, between 20% and 35% (Breeze *et al*. 2011; Winfree *et al*. 2011), or as the cost of restoring pollinators hives, these estimates do not account for the full social value of pollinators as there is no enough information on the risk of pollinators' loss and ecosystem's collapse. Consequently, the estimates are conservative, reflecting only lower bounds of the potential losses generated by the collapse of the service, particularly in the context of potential climate change-induced tipping points.

Typically, WTP estimates are aggregated by multiplying the WTP sample mean by a measure of population density. While this may not necessarily lead to biased estimates as long as a representative sample is drawn from the entire economic jurisdiction, in some cases it may introduce biases. For example, aggregating sub-areas into larger areas may not account for the distance decay (Bateman *et al*. 2006). Besides, the population estimates can be very different between primary studies. While some studies offer estimates of visitation rates, others do not, implying that they should not be pooled together. While Bateman *et al*. (2006) suggest an approach based on the estimation of a spatially sensitive valuation function, due to data availability constraints we do not follow this approach. We divide the WTP estimates by the total site area, thus choosing not to aggregate values. By following this approach, we avoid distortions related to the primary study estimates. This does not solve all of the issues as distance decay is still a problem. In fact, by using the average WTP for a study site we necessarily end upunderestimating the WTP closer to the site centroids while overestimating the WTP of the areas further away from the study site centroid. This is not possible to overcome.

Moreover, the estimated values of *Rec* and *WaterPur*, without any type of aggregation, have no real meaning as they are measured in WTP/ha/y. These results can become more intuitive using one of two possible aggregations. Either we aggregate the individual WTP estimates at the whole site scale, such as the *Arrábida Natural Park* or we aggregate each WTP/ha/y estimate based on the representative population, such as the number of visitors in a natural park. Only then we can estimate the total ES value for a given site.

Despite the limitations, this study contributes to the literature by providing a more solid framework to estimate the value of ecosystem services at a national scale. This framework is not only replicable but adjustable to any reality, contributing to research transparency and better informing the design of sustainable public policies.

5. Conclusion

This study estimates the economic value of seven land-based ecosystem services in Portugal using market prices and meta-analytic benefit transfer functions, applied directly to ecosystem service provision maps provided under the ASEBIO project. It contributes to the strand of literature in ecosystem service valuation as the framework developed in this study consistently estimates the economic value addressing some of the issues raised by Boyle & Wooldridge (2018), Nelson & Kennedy (2009) and Vedogbeton & Johnston (2020).

The estimated values of land-based ecosystem services in Portugal are in line with the literature as, typically, non-urban areas show higher values for the majority of services, particularly, for protected areas. While the framework here proposed suffers from limitations, from our perspective it provides a tool that allows for researchers and policy makers to assess the value of ecosystem services at the national/sub-national level with significant spatial detail. The limitations and possible workarounds are well documented throughout this study.

Building upon this framework it is possible to estimate not only changes in economic value due to losses in the provision of ecosystem services over time and across space, but also to link those losses to individual households' heterogeneity, such as income levels, among others. This is an important instrument to better support equitable land-use management decision making in the presence of climate change mitigation and adaptation policies, in the context of the transition to carbon neutrality. For illustrative purposes, this can be undertaken by allowing to estimate the net benefits from irreversible land-use change, extreme weather events, and even public policy interventions such as those in the energy sector (e.g, deployment of utility-scale solar PV power plants). This is left for future research.

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7. Supplementary Materials

7.1. Supplementary Materials S.1: Primary Study Search Strategy

7.1.1. Food supply:

Scopus:

TITLE-ABS-KEY ("Agricultur*" OR "Crop" OR "Farm" OR "Agro*") AND TITLE-ABS-KEY ("economic valuation" OR "monetary valuation" OR valuation) AND TITLE-ABS-KEY ("ecosystem services") AND (LIMIT-TO (LANGUAGE, "English")

Web of Science:

(AB=((“Agricultur*” OR “Crop” OR “Farm” OR “Agro*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services") OR (TS=((“Agricultur*” OR “Crop” OR “Farm” OR “Agro*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services")))

EVRI:

Search restricted to primary studies with “land” as the environmental asset using the following key-words: “Agriculture”, “Agricultural”, “Crop”, “Farm”, and “Agro

7.1.2. Pollination:

Scopus

TITLE-ABS-KEY ("pollinat*”) AND TITLE-ABS-KEY ("economic valuation" OR "monetary valuation" OR valuation) AND TITLE-ABS-KEY ("ecosystem services") AND (LIMIT-TO (LANGUAGE, "English")

Web of Science

(AB=(("pollinat*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services") OR (TS=(("pollinat*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services")))

EVRI

Search restricted to primary studies using the following key-words: “pollination” and “pollinating”

7.1.3. Recreation:

Scopus:

TITLE-ABS-KEY (“Recreation*”) AND TITLE-ABS-KEY ("economic valuation" OR "monetary valuation" OR valuation) AND TITLE-ABS-KEY ("ecosystem services") AND (LIMIT-TO (LANGUAGE, "English")

Web of Science:

(AB=((“ Recreation*”) AND ("economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services") OR (TS=((“ Recreation*”) AND ("economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services")))

EVRI:

Search restricted to primary studies using the following key-words: “Recreation” and “Recreational”

7.1.4. Water purification:

Scopus:

TITLE-ABS-KEY (“Water purification” OR “Water Quality*”) AND TITLE-ABS-KEY ("economic valuation" OR "monetary valuation" OR valuation) AND TITLE-ABS-KEY ("ecosystem services") AND (LIMIT-TO (LANGUAGE, "English")

Web of Science:

(AB=((“Water purification” OR “Water Quality*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services") OR (TS=((“Water purification” OR “Water

Quality*”) AND (" economic valuation" OR "monetary valuation" OR valuation) AND "ecosystem services")))

EVRI:

Search restricted to primary studies “Water General” as the environmental asset using the following key-words: “Water Quality” and “Water purification”.

7.2. Supplementary Materials S.2: Primary Study List

Study Title	**Author-Year**
Integrated Assessment, Valuation And Mapping Of Ecosystem Services And Disservices From Upland Land Use In Wales	Ashley Hardaker, et al. (2020)
Economic Valuation And Mapping Of Ecosystem Services In The Context Of Protected Área Management (Natural Park Of Serra De São Mamede, Portugal)	Cristina Marta-Pedroso, et al. (2018)
Unraveling Local Preferences And Willingness To Pay For Different Management Scenarios: A Choice Experiment To Biosphere Reserve Management	Nekane Castillo-Eguskitza, et al. (2019)
Gis-Based Valuation Of Ecosystem Services In Mountain Regions: A Case Study Of The Chepelare Municipality In Bulgaria	Ekaterina Ivanova, et al. (2016)
Assessing, Valuing And Mapping Ecosystem Services At City Level: Thecase Of Uppsala (Sweden)	Natasha Nikodinoska, et al. (2017)
Quantification And Valuation Of Ecosystem Services In Diverse Production Systems For Informed Decision-Making	Bhim Bahadur Ghaley, et al. (2014)
Ecosystem Function And Servisse Quantification And Valuation In A Conventional Winter Wheat Production System With Daisy Model In Denmark	Bhim Bahadur Ghaley, John Roy Porter (2014)
Study On The Techniques Of Valuation Of Ecosystem Services Based On Remote Sensing In Anxin County	Hongyan Wang, et al. (2014)
Monetary Accounting Of Ecosystem Services: A Test Case For Limburg Province, The Netherlands	Roy Remme, et al. (2015)
Valuing The Provisioning Services Of Wetlands: Contrasting A Rural Wetland In Lesotho With A Peri-Urban Wetland In South Africa	Kathryn Lannas, Jane Turpie (2009)
Evaluating The Impact Of Regional Development Policies On Future Landscape Services	Louise Willemen, et al. (2010)
Forest Ecosystem Services And Their Values In Beijing	Gaodi Xie, et al. (2010)
Measuring Conflicts In The Management Of Anthropized Ecosystems: Evidence From A Choice Experiment In A Human-Created Mediterranean Wetland	Ángel Perni, José Miguel Martínez-Paz (2017)
Pollination Services Mapping And Economic Valuation From Insect Communities: A Case Study In The Azores (Terceira Island)	Ana Picanço, et al. (2017)
Courgette Production: Pollination Demand, Supply, And Value	Jessica Knapp, Juliet Osborne (2017)
What Is The Value of Wild Bee Pollination For Wild Blueberries And Cranberries, And Who Values It?	Aaron Hoshide, et al. (2018)

To Restore Or Not? A Valuation Of Social And Ecological Functions Of The Marais Des Baux Wetland In Southern France	Vanja Holmquist Westerberg, et al (2010)
Valuing Pollination Services To Agriculture	Rachael Winfree, et al. (2011)
Pollination Services In The Uk: How Important Are Honeybees?	Tom Breeze, et al. (2011)
Influence Of User Characteristics On Valuation Of Ecosystem Services In Doñana Natural Protected Area (South-West Spain)	Berta Martín-López, et al. (2007)
Lake Simcoe Basin'S Natural Capital: The Value Of The Watershed'S Ecosystem Services	Sara Wilson (2008)
Pollinators And Pollination Of Oilseed Rape Crops (Brassica Napus L.) In Ireland: Ecological And Economic Incentives For Pollinator Conservation	Dara Stanley, et al. (2013)
Avoiding A Bad Apple: Insect Pollination Enhances Fruit Quality Andeconomic Value	Michael Garratt, et al. (2014)
Synergistic Interactions Of Ecosystem Services: Florivorous Pest Control Boosts Crop Yield Increase Through Insect Pollination	Louis Sutter, Matthias Albrecht (2016)
Ecosystem Services Value: Case Of Pollination	Janusz Majewski (2018)
Opportunity Costs Of Alternative Management Options In A Protected Nature Park: The Case Of Ramat Hanadiv, Israel	Itai Divinski, et al. (2017)
Mapping The Recreational Value Of Coppices' Management Systems In Tuscany	Francesco Riccioli, et al. (2020)
Economic Valuation Of Street-Level Urban Greening: A Case Study From An Evolving Mixed-Use Area In Berlin	Erik Fruth, et al. (2019)
Recreational Values Of Forest Park Using The Contingent Valuation Method (Case Study: Saravan Forest Park, North Of Iran)	Soleiman Mohammadi Limaei, et al. (2016)
The Value Of Coastal Lagoons: Case Study Of Recreation At The Ria De Aveiro, Portugal In Comparison To The Coorong, Australia	Inês Clara, et al. (2017)
The Economic Value Of The Greater Montreal Blue Network (Quebec, Canada): A Contingent Choice Study Using Real Projects To Estimate Non-Market Aquatic Ecosystem Services Benefits	Thomas Poder, et al. (2016)
Latent Preferences Of Residents Regarding An Urban Forest Recreation Setting In Ljubljana, Slovenia	Anže Japelj, et al. (2015)
Valuing New Zealand Recreational Fishing And Na Assessment Of The Validity Of The Contingente Valuation Estimates	Sarah Wheeler, Richard Damania (2001)
Valuation Of The Services Provided By The Forest Recreational Reserve Of Valverde, Santa Maria, Azores	Fernando Páscoa, et al. (2014)
Valuing Recreation In The Coorong, Australia, With Travel Cost And Contingent Behaviour Models	John Rolfe, Brenda Dyack (2011)
Valuing Australian Botanic Collections: A Combined Travel-Cost And Contingent Valuation Study	Paul Mwebaze, Jeff Bennett (2012)
Valuation Of The Salini National Park	Luciano Pace Parascandalo (2010)
Identifying Societal Preferences For River Restoration In A Densely Populated Urban Environment: Evidence From A Discrete Choice Experiment In Central Brussels	Wendy Chen, et al. (2017)

British Tourists' Valuation Of A Turkish Beach Using Contingent Valuation And Travel Cost Methods	Frederick Blakemore, Allan Williams (2008)
Models For Sample Selection Bias In Contingent Valuation: Application To Forest Biodiversity	Serge Garcia, et al. (2009)
Estimation Of Median Willingness To Pay For A System Of Recreation Areas	Kimberly Rollins, Diana Dumitras (2005)
Estimating The Recreational Benefits Of Dibeen National Park In Jordan Using Contingent Valuation And Travel Cost Methods	Amer Jabarin, Said Damhoureyeh (2006)
Estimation Of Forest Values Using Choice Modeling: An Application To Spanish Forests	Raul Brey, et al. (2007)
Valuing Environmental Resources In The Context Of Flood And Coastal Defence Project Appraisal A Case-Study Of Poole Borough Council Seafront In The Uk	Serafeim Polyzos, Dionissios Minetos (2007)
Measuring The Net Economic Value Of Recreational Boating As Water Levels Fluctuate	Nancy Connelly, et al. (2007)
Valuing Forest Recreation On The National Level In A Transition Economy: The Case Of Poland	Anna Bartczak, et al. (2008)
Ecotourism Demand In North-East Italy	Tiziano Tempesta, et al. (2002)
The Valuation Of The Ijmeer Nature Reserve Using Conjoint Analysis	Barbara Baarsma (2003)
Valuing New South Wales Rivers For Use In Benefit Transfer	Mark Morrison, Jeff Bennett (2004)
Valuing The Recreational Benefits From The Creation Of Nature Reserves In Irish Forests	Riccardo Scarpa, et al. (2000)
The Application Of The Cost Benefit Method To Sea Defence And Coastal Protection Management In England	Ian Bateman, et al. (2001)
Valuing Ecosystem Services Across Water Bodies: Results From A Discrete Choice Experiment	Edel Doherty, et al. (2014)
Water Resource Management In New Zealand: Jobs Or Algal Blooms?	Dan Marsh (2012)
Estimating Benefits Of Improving Water Qualityin The Largest Remaining Tidal Flat In South Korea	Isao Endo, et al (2012)
Valuing Inland Blue Space: A Contingent Valuation Study Of Two Large Freshwater Lakes	Craig Mcdougall, et al. (2020)
Valuing Users’ Willingness To Pay For Improved Water Quality In The Context Of The Water Framework Directive	Dimitra Lazaridou, Anastasios Michailidis (2020)
River Water Quality: Who Cares, How Much And Why?	Danyel Hampson, et al. (2017)
“Adapted” Habitat Evaluation Procedure And Choice Experiment: Substitutes Or Complements?	Nathalie Dumax, et al. (2020)
Contingent Valuation Of Residents’ Attitudes And Willingness-To-Pay For Non-Point Source Pollution Control: A Case Study In Al-Prespa, Southeastern Albania	Dorina Grazhdani (2015)
Willingness To Pay For Achieving Good Status Across Rivers In The Republic Of Ireland	Cathal Buckley, et al. (2016)

Valuation And Transferability Of The Non-Market Benefits Of River Restoration In The Danube River Basin Using A Choice Experiment	Roy Brouwer, et al. (2016)
Increasing Beach Recreation Benefits By Using Wetlands To Reduce Contamination	Sebastain Awondo, et al. (2011)
The Conservation Against Development Paradigm In Protected Areas: Valuation Of Ecosystem Services In The Doñana Social–Ecological System (Southwestern Spain)	Berta Martín-López, et al. (2011)
Social Preferences And Economic Valuation For Water Quality And River Restoration: The Segura River, Spain	Ángel Pernin, et al. (2011)
Temporal Stability Of Individual Preferences For River Restoration In Austria Using A Choice Experiment	Markus Bliem, et al. (2012)
Valuation Of Environmental Improvements In A Specially Protected Marine Area: A Choice Experiment Approach In Göcek Bay, Turkey	Özge Can, Emre Alp (2012)
Directional Heterogeneity In Wtp Models For Environmental Valuation	Marije Schaafsma, et al. (2012)
Estimating The Non-Market Benefits Of Water Quality Improvement For A Case Study In Spain: A Contingent Valuation Approach	Julian Ramajo-Hernandez, Salvador del Saz-Salazar (2012)
Spatially Induced Disparities In Users' And Non-Users' Wtp For Water Quality Improvements—Testing The Effect Of Multiple Substitutes And Distance Decay	Sisse Liv Jørgensen, et al. (2013)
Public Demand For Remediating A Local Ecosystem: Comparing Wtp And Wta At Hongze Lake, China	Lei Huang, et al (2013)
The Value Of Water Quality Improvements In The Region Berlin–Brandenburg As A Function Of Distance And State Residency	Jürgen Meyerhoff, et al. (2014)
Cheaper Electricity Or A Better River? Estimating Fluvial Ecosystem Value In Southern France	Anna Créti, Frederico Pontoni (2014)
Correcting For Non-Response Bias In Contingent Valuation Surveys Concerning Environmental Non-Market Goods: An Empirical Investigation Using An Online Panel	Ole Bonnichsen, Søren Bøye Olsen (2016)
The Social Benefits Of Restoring Water Quality In The Context Of The Water Framework Directive: A Comparison Of Willingness To Pay And Willingness To Accept	Salvador Del Saz-Salazar, et al. (2009)
Spatial Preference Heterogeneity: A Choice Experiment	Roy Brouwer, et al. (2010)
Using The Choice Experiment Method For Valuing Improvements In Water Quality: A Simultaneous Application To Four Recreation Sites Of A River Basin	Julie Poirier, Aurore Fleuret (2010)
Property Owners' Willingness To Pay For Water Quality Improvements: Contingent Valuation Estimates In Two Central Minnesota Watersheds	Patrick Welle, James Hodgson (2011)
The Value Of Improved Water Quality In Guadiana Estuary—A Transborder Application Of Contingent Valuation Methodology	Maria Helena Guimarães, et al. (2011)
Estimating The Economic Value Of Water Quality Protection In The Catawba River Basin	Randall Kramer, Jonathan Hecht (2002)
Comparison Of The Contingent Valuation Method And The Stated Choice Model For Measuring Benefits Of Ecosystem Management:	Yuki Takatsuka (2004)

A Case Study Of The Clinch River Valley, Tennessee A Case Study Of The Clinch River Valley, Tennessee	
The Rotorua Lakes Evaluation Of Less Tangible Values	Brian Bell, Michael Yap (2004)
Comparing Contingent Valuation And Contingent Ranking: A Case Study Considering The Benefits Of Urban River Water Quality Improvements	Ian Bateman, et al. (2006)
The Impact Of Water Quality And Water Level On The Recreation Values Of Lake Hume	Lin Crase, Robert Gillespie (2012)

7.3. Supplementary Materials S.3: GIS maps references

1. Egirone. 2014. Armenia Lakes: ARM_Lakes [map]. Using: ArcMap
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7.4. Supplementary Materials S.4: Other relevant References

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7.6. Supplementary Materials S.6: Primary Studies Average Values per Group

Food Supply

Country	Average Value	% of observations
Bulgaria	13.91681	2.70
Canada	1471.643	2.70
China	225.6204	5.41
Denmark	922.9692	8.11
Netherlands	324.0199	10.81
Portugal	230.9348	10.81
Spain	754.5505	8.11
Sweden	224.2881	5.41
United Kingdom	1433.693	37.84
United States of America	1094.995	8.11
Non-Protected	352.2778	40.54
Protected	1259.558	59.46

Pollination

Country	Average Value	% of observations
Denmark	13.52122	3.23
Ireland	198.4351	6.45
Israel	275.8774	6.45
Poland	319.9826	6.45
Portugal	1.206343	6.45
Switzerland	54.06397	3.23
United Kingdom	2671.419	48.39
United States of America	795.2202	19.35
Non-Protected	1005.877	35.48
Protected	1771.828	64.52

Recreation

Country	Average Value	% of observations
Australia	0.0130813	15.52
Belgium	0.2976514	1.72
Canada	0.0010447	8.62
France	0.0000159	6.90
Germany	0.3000867	1.72
Iran	0.0018495	1.72
Ireland	0.5310004	22.41
Italy	0.0454314	6.90
Jordan	0.0017885	1.72
Malta	0.3813928	1.72
Netherlands	0.00000198	1.72

New Zealand	0.0000884	1.72
Poland	0.00000438	1.72
Portugal	0.2773115	3.45
Slovenia	0.0186041	1.72
Spain	0.0000617	3.45
Turkey	0.0085846	1.72
United Kingdom	0.3333393	15.52
Non-Protected	0.3060624	65.52
Protected	0.0071115	34.48

Water Purification

Country	Average Value	% of observations
Albania	0.00000255	1.75
Australia	0.0003043	3.51
Austria	0.030504	7.02
Belgium	0.3583726	3.51
Canada	0.0001572	1.75
China	0.0002168	1.75
Denmark	0.0012526	3.51
France	0.0517477	14.04
Germany	0.003546	5.26
Greece	0.0000662	1.75
Hungary	0.00000127	3.51
Ireland	0.0006174	5.26
Netherlands	0.1247307	3.51
New Zealand	0.0004589	3.51
Portugal	0.0003919	1.75
Spain	0.0004189	15.79
Turkey	0.0014509	1.75
United Kingdom	0.6993385	10.53
United States of America	0.0033376	10.53
Non-Protected	0.1678338	54.39
Protected	0.020692	45.61

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