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Economic Value of Shrublands' Ecosystem Services in mainland Portugal

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Master in Economics

Supervisor:
Ph. D. Catarina Roseta-Palma, Associate Professor
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October, 2023

BUSINESS SCHOOL

Department of Economics

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ACKNOWLEDGEMENTS

I would like to express my deepest gratitude to my family and colleagues for all of their support and encouragement throughout my academic path. Without them I would not be able finish my MSc dissertation. I would also like to thank Professor Catarina Roseta-Palma for her mentorship, expertise and guidance in the execution of this paper. I would also like to thank Henrique Cerqueira for his technical advice and support regarding GIS data, and Gerard Najem for sharing all his time and expertise every time I felt the need. I deeply thank you all for your support.

RESUMO

Apesar de serem uma componente integral da paisagem portuguesa, os matos são muito subvalorizados e geralmente considerados territórios estéreis e improdutivos, em grande parte devido à falta de conhecimento do papel do bioma e dos serviços do ecossistema que presta. Neste documento, abordamos a tarefa inédita de fazer uma avaliação económica dos matos de Portugal Continental, quantificando o valor dos seus serviços ecossistémicos em termos monetários, tornando-os assim mais visíveis e acessíveis aos decisores e responsáveis políticos, bem como ao público em geral.

Os dados sobre os matos de Portugal Continental foram recolhidos do conjunto de dados SIG COS Portugal Continental 2018 e analisados através da ferramenta QGIS, mostrando que representam 12,4% do total do território continental. Através dos métodos de avaliação de transferência de benefícios e dados, foram obtidos valores monetários para os serviços do ecossistema de retenção de carbono, conservação da biodiversidade e prevenção da degradação do solo. Esses valores foram posteriormente convertidos para os mesmos valores em USD de 2022 e somados, perfazendo um valor total que varia entre 1 225 milhões e 7 918 milhões de USD/ano, consoante o modelo específico de avaliação utilizado. Este valor representa entre 0,5% e 1,25% do PIB português e entre 12% e 78% da produção do sector agrícola do país em 2022.

Palavra-chave: Ecossistema, Matos, Portugal, Valor dos Serviços do Ecossistema.

Classificação JEL: Q56, Q57

ABSTRACT

In spite of being an integral component of the Portuguese landscape, shrublands are greatly undervalued and generally deemed barren, unproductive territories, largely because of the lack of knowledge of the biome's role and the ecosystem services it provides. In this paper, we tackle the unprecedented task of making an economic valuation of the mainland Portugal shrublands, quantifying the value of its ecosystem services in monetary terms, and therefore rendering it more visible and accessible for decision-makers and policymakers, as well as the wider public.

Mainland Portugal shrubland data was collected from COS mainland Portugal 2018 GIS dataset and analyzed through QGIS tool, showing that it represents 12.4% of the total mainland territory. Through benefit and data transfer methods of valuation, monetary values were obtained for the ecosystem services of carbon sequestration, biodiversity conservation, and land degradation prevention. Those values were later converted to the same USD 2022 currency values and combined, amounting to a total value that ranges between 1 225 million and 7 918 million USD/yr, depending on the specific model of evaluation used. This value that represents between 0.5% and 1.25% of Portugal's GDP and 12% to 78% of the output of the country's agricultural industry in 2022.

Keyword: Ecosystem, Ecosystem Service Value, Portugal, Shrublands.

JEL Classification: Q56, Q57

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GLOSSARY OF ACRONYMS AND ABBREVIATIONS

CVM	Contingent valuation method
ES	Ecosystem service
ESV	Ecosystem service values
EU	European Union
EU ETS	European Union Emissions Trading System
GDP	Gross domestic product
GIS	Geographic Information Systems
HANPP	Human appropriation of net primary productivity
NPESV	Net present value of ecosystem service
NPP	Net primary production
NPV	Net present value
TEV	Total Economic Value
USD	United States Dollar

1. INTRODUCTION

Shrublands, an integral component of mainland Portugal's landscape, are often perceived as barren and unproductive terrains. They are frequently associated with abandoned lands and, as such, are largely undervalued. However, beneath this perception lies valuable and various of ecosystem services that are vital to our environment. This master's thesis aims to bring out the economic valuation of shrublands' ecosystem service (ES) in mainland Portugal, thereby unveiling the latent benefits that they bring to the ecosystem and society at large. Shrublands, in fact, serve as a nursery for forest regeneration, act as a crucial carbon sink, prevent land degradation, and contribute significantly to biodiversity preservation, among other functions (Castro et al., 2004; Duponnois et al., 2011; Goberna et al., 2007; Gratani et al., 2013).

The primary goal of this research is to assess the ecosystem service value (ESV) of the shrublands in mainland Portugal. The investigation lies a fundamental query: How do we value shrublands? To put it differently, what is the most suitable means of capturing value to shrublands? By quantifying their often-overlooked economic value, the objective is to render their economic value more transparent to foster greater awareness of their essential role in the ecosystem, to protect, conserve, and optimize the use of shrublands. By highlighting their economic worth, we hope to reshape the narrative around these landscapes, foster efforts to safeguard and restore these integral components of a balanced ecosystem.

The study of the economic valuation of ecosystem services has been widely debated in the past two decades. Economists and researchers have come up with different methods to reflect its value, yet the valuation is complex and different types of valuation methods can be used depending on data availability and research goals. Therefore, the ESV of the Portuguese shrublands will be assessed through findings from existing reports and studies, by exploring their ecosystem services, employing appropriate valuation methods, by conducting focus on the determination of appropriate discount rates and the calculation of net present values under a range of time horizons. By taking into account different time frames, we aim to provide a nuanced understanding of how the economic value of shrublands evolves over time, acknowledging both short-term and long-term benefits.

2. LITERATURE REVIEW

In this section, some of the reports and studies related to ecosystem service valuation will be presented. Since the economic valuation of shrublands has received scant attention in previous scientific papers, studies on the ESV of different types of ecosystems will be covered. The framework of ES valuation, previous studies on ecosystem services and, environmental interaction of shrublands and ES valuation methods used in past studies will be discussed and assessed to decide whether they can be applied in this specific case.

2.1. The framework of ecosystem services valuation

According to Millennium Ecosystem Assessment (2005) and Pascual et al., (2012) the four categories of ecosystem services (ES) that contribute to human well-being are provisioning, regulating, cultural and supporting services. Provisioning services are material benefits that people can extract from nature such as food, water, wood, among other goods, and therefore are normally visible. Regulating services in the other hand operate in the background, not easily recognizable by people, and include regulating ecosystem services such as pollination and climate regulation. Cultural services are non-material benefits that can be used for purposes of recreation, spiritual and aesthetic values. Supporting services are basic services that maintain the ecosystem's function, such as soil formation, photosynthesis and nutrient cycling. In the IPBES report (IPBES, 2019), the framework of Nature's Contribution to People (NCP) was built on the concept of ecosystem services (ES), with aims to incorporate more symmetric consideration of diverse stakeholders and world views. NCP refers to all the contributions that humanity obtains from nature, both benefits and detriments of living nature to people's quality of life. It is subdivided into material, regulating and non-material contributions.

As stated in The Economics of Ecosystems & Biodiversity Report (TEEB, 2010), it can be complex and controversial to value environmental services and biodiversity. However, simple invisibility of natural services' values can lead to the degradation of biodiversity, the ecosystem and its services. Although it is still unclear how to best evaluate nature, making its services visible through pricing and valuation measurement can help decision-makers take into account the cost and benefits of natural resources, managing them more effectively, encouraging a better consideration of the consequences of certain actions and the choice of more effective conservation techniques.

The TEEB approach (TEEB, 2010) proposes three steps for the economic valuation of ecosystem services. The first is recognizing the full range of stakeholders influenced by affected ecosystem services and biodiversity (e.g., the contribution of forests and other ecosystems to the livelihoods of poor rural households). Secondly the value of ecosystem services should be estimated and demonstrated through appropriate methods. For instance, various studies indicate that two thirds of the total economic value (TEV) of tropical forests come from regulating services, but the economic importance of forests is often based on provisioning services which account for a relatively small share of forest TEV. The third step aims to capture the value of ecosystem services and overcome their undervaluation by means of economically-informed policy instruments. For example, since 2003 in Mexico, landowners may apply for public payments in exchange for commitments to preserve forest land and forgo certain uses, such as agriculture and cattle raising.

The Economic of Biodiversity: The Dasgupta Review (Dasgupta, 2021), sees biodiversity loss as an asset-management problem. Nature as an asset and biodiversity as a diversity of natural assets. As our economies are embedded within Nature, we are affected by the consequences of how we use it. This review suggests Inclusive Wealth as the most appropriate measure of sustainable economic prosperity. The 'true' social values, called 'accounting prices' by Dasgupta, cannot be found in any market. The report makes clear that only social cost-benefit analysis, using the same accounting prices as those estimated for sustainability assessment, would tell the social evaluator which investment projects are socially desirable. The accounting price of an asset or service is the sum of its market price and the tax that ought to be imposed on it. Put simply, if taking q_i as the market price of asset *i* and e_i as the value of the externalities generated by the deployment of a marginal unit of *i*, then the asset's accounting price is: $p_i = q_i + e_i$. The gap between accounting prices and market prices is therefore a measure of inefficiency in the allocation of goods and services: it will reflect waste in our use of resources. But unlike food waste, the gap is not visible. For instance, open-access resources such as groundwater and ocean fisheries have no market price, so their accounting prices ought to be the taxes imposed on their use to reflect resource scarcity. Effective institutions and systems are required to help fix the problems of widespread institutional failure and pervasive externalities; this is similar to Step three of the TEEB approach in the earlier part of this section, capturing economic value.

2.2. Shrublands' ecosystem services and environmental interaction

In the IPBES report (IPBES, 2019), the aggregate impact of the direct drivers of nature degradation, i.e., direct human influence upon nature is classified into five categories, which are: land-use/ sea-use change; resource extraction; pollution; invasive and alien species; and climate change. According to the report, today's 75% of the total land surface and 40% of the ocean area are severely altered. More than half of the Earth's land surface is under anthropogenic cover types, including cropland, pasture and rangeland, and cities (Hooke et al., 2012). Agricultural expansion is by far the most widespread form of land cover change, with more than a third of the terrestrial land surface currently used for crops or livestock at the expense of forests, wetlands, grasslands and many other natural land cover types (FAO-ITPS, 2015; Foley et al., 2005). Between 1980 and 2000, tropical agriculture expanded by more than 100 million hectares, more than half at the expense of tropical forests (Gibbs et al., 2010). The largest agricultural land expansion in Latin America was due to pasture for cattle. City areas doubled between 1992 and 2015. The most severe increases were in tropical and subtropical savannas, grasslands, deserts, and xeric shrublands, where the urban areas tripled. Land cover changes have led to increasing fragmentation of the remaining forests. Technological advances in agriculture, fisheries and aquaculture, and forestry have led to sometimes irreversible shifts in ecosystems and natural services. These are exacerbated by higher livestock densities, changes in fire regimes and intensification, leading to accelerated soil and water pollution. Soil degradation - including erosion, acidification and salinization - has increased globally. Moreover, the IPBES Land Degradation Assessment (IPBES, 2018), showed that intensive land use can lead to progressive changes in ecosystem functions and, in some cases, irreversible changes then lead to land abandonment.

In the IPBES Report (IPBES, 2019) it is noted that Mediterranean biome has the second-lowest land protection level among terrestrial biomes and is at risk of substantial biodiversity loss in the future. The Mediterranean plant species in regions around the Mediterranean Sea, are uniquely adapted to cope with extended periods of hot and dry summers. Most of the vegetation is adapted to fire and in fact, depends on this disturbance for its sustainability. However, these ecosystems are highly sensitive to habitat fragmentation, grazing, and changes in fire patterns. Native species face threats from invasive species and are vulnerable due to the arid climate and shallow limestone soils. Transformation of pastures poses significant challenges in many countries, resulting in issues such as erosion, organic carbon loss, and decline in biodiversity. Soil salinity may also restrict agricultural use in certain regions (FAO-ITPS, 2015).

In a study of Riera et al., (2007) regarding climate change effects on shrublands in Catalonia (northeastern Iberian Peninsula), it is noted that the welfare of the individuals in the region is expected to drop in line with the changes in its shrublands, costing each person on the estimated average cost of 2.9 euros per year in terms of lost welfare for an increase of 1% in the shrubland area affected by erosion.

Shrublands, primarily offer non-material or regulating Nature’s Contribution to People (NCP) rather than the more visible material NCP. While the Mediterranean shrublands do not play any significant role in agriculture, they act as biodiversity hotspots, nurturing plants for reforestation, (Castro et al., 2004; Duponnois et al., 2011) and providing regulating services like pollination, supporting biodiversity habitats, climate regulation, and supporting service such as soil conservation. In the Iberian Peninsula, they are rich in diverse plant species, small mammals, birds, reptiles, and amphibians, (Myers et al., 2000; Torre et al., 2022; Wessel et al., 2004). with 4.3% of global plant species endemism in the Mediterranean. Additionally, their vegetation cover safeguards against soil erosion, especially important in the Mediterranean's intense spring and autumn precipitation (Wessel et al., 2004). Despite their vital role, these ecosystem services often go unnoticed and are undervalued due to their subtle nature and being beyond people’s perception.

As mentioned before, one of their valuable ecosystem services is acting as nursery plants for reforestation. An increase in the early growth of *Cupressus atlantica* seedlings is observed when shrub cover increases, by enhancing soil properties such as infiltration, retention capacity, and soil enzyme activity (Duponnois et al., 2011; Sardans & Peñuelas, 2013). In the driest Mediterranean areas with sparse vegetation, soil patches under vegetation present higher levels of water infiltrability, Soil Organic Matter (SOM), microbial biomass, and enzyme activity than bare soil (Goberna et al., 2007). In addition, a 4-year study of using shrubs as nurse plants for reforestation in the Mediterranean mountains (the Sierra Nevada mountains, SE Spain) noticed that shrubs enhanced seedling establishment. The survival for P. sylvestris and P. nigra was 2.6- and 1.8-fold respectively higher than the value reported under traditional technique in the bare soil (Castro et al., 2004).

When we measure the value of an ecosystem service, carbon sequestration is one of the most common factors that is taken into account due to the carbon market existence. Carbon sequestration of the Mediterranean shrublands along the Latium coast (Capocotta, Italy) is observed at 80 tonne of $\mathrm{CO_2 ha^{-1} year^{-1}}$ in 2011 and a nationwide estimated annual benefit of roughly USD 500 million(Gratani et al., 2013). The mentioned study obtained the primary data on $\mathrm{CO_2}$ sequestration rate by measuring shrub density, classifying shrub into Small (24%), Medium (50%), Large (26%) categories according to their volume and leaf area index. $\mathrm{CO_2}$ sequestration rate is decreased by 30% in M compared to L, 80% in S compared to L. Regarding the seasonal variation, Spring has the highest CO2 sequestration capacity, decreasing 64% in winter and 67% in drought.

2.3. Different types of economic valuation of ecosystems in previous literature

Ecosystem services can be assessed through three approaches: qualitative analysis, quantitative analysis, and monetary analysis (TEEB, 2010). Qualitative analysis considers non-numerical indicators, such as mental, physical health and social benefits from recreation. Quantitative analysis focuses on numerical data like carbon sequestration and water quality, etc. Monetary analysis involves converting both qualitative and quantitative aspects into a specific currency (Stolton & Dudley, 2015). Monetary valuation is the most common approach, as it can often provide a comprehensible measure for decision-making and facilitating direct comparisons between the costs and benefits of biodiversity conservation and other development goals (Christie et al., 2012).

In estimating the value of ecosystem services, various valuation methods exist and have been applied. An important distinction is between market-based and non-market-based valuation methods. Market-based valuation relays on existing market behavior and transactions. Direct market valuation methods include market price-based, cost-based and production function-based methods. Unfortunately, for many ecosystem goods and services, direct markets do not exist and direct market prices are not available. In these cases, indirect market valuation methods, also known as revealed preference (RP) methods, are often used. The main RP methods are the travel cost and the hedonic pricing methods. When market prices are not available, when RP methods are inapplicable, or changes in ecosystem services are hypothetical, stated preference (SP) methods are used. The main SP methods are contingent valuation, choice modelling and group deliberation. Additionally meta-analysis and value transfer approaches are two alternative approaches which do not belong to any of the previous method categories. These methods, while not valuation methods themselves, are often used to derive values for ecosystem services (Koetse et al., 2015).

A meta-analysis conducted on the studies between 1994 and 2017 regarding ESV of forest suggests that a range of monetary and non-monetary, or a combination of methods, have been used to estimate the ESV of forest. Monetary valuation methods were predominant during 2006-2014, but after 2014, non-monetary approaches gained prominence until 2017. The use of combined methods (monetary and non-monetary) was limited from 1994 to 2014 but increased slightly thereafter. Contingent valuation, market price, benefit transfer, mapping/modeling, and social survey methods were similarly utilized between 1994 and 2005. However, from 2014 to 2017, forest ecosystems were primarily valued using modeling/mapping, followed by contingent valuation and social surveys, while hedonic, cost-based (replacement/damage avoided), and market price methods remained less popular. Interestingly, only 21% of the research considered all three services for valuation, with regulating services being the most prioritized, followed by provisioning and cultural services (Acharya et al., 2019).

Benefit (value) transfer is an approach used to transfer estimates from valuation studies to policy-relevant sites. It is gaining popularity due to higher demand for economic information on environmental goods, and limited time and resource to conduce new environmental valuation studies. Even though the term “benefit transfer” is commonly used, “value transfer” would be a more general term since it encompasses not only benefits but also damage estimates. However, this is most reliable when the environmental good and the population are closely matched between the study and policy sites. Benefit transfer has two primary approaches: Unit Value Transfer (Simple Unit Transfer) and Function Transfer (Benefit Function from one study). Unit Transfer involves transferring a single point estimate from study summaries, while Function Transfer conveys a model describing how benefit measures change with variations in population or resource characteristics. Simple unit transfer is the most straightforward method, directly transferring the mean estimate from the study site to the policy site. In cases where primary data is unavailable, data transfer studies are often used. The first consideration in data transfer is data applicability, which assesses how closely the original data aligns with the problem in the transferred area. The second consideration is the incorporation of imprecise physical, natural, and behavioral information transfers as primary components of the analysis. The third consideration is the impact of non-behavioral information transfers on the value transfer process from an economic perspective. The introduction of data that does not directly pertain to human behavior or economic factors can affect the accuracy and reliability of the value transfer, potentially reducing its precision. (Navrud & Ready, 2007).

Boyer & Polasky (2004) conducted a review of literatures on non-market valuation applied to wetlands with focus on urban wetlands. Hedonic studies of the value of wetlands in urban areas showed that proximity to wetlands increases nearby property values. The same in rural areas showed a more mixed response and negative effects on rural land values were found on forested and emergent palustrine wetlands in Florida. The main drawback of this method according to this review is that it only measures the perceived value of wetlands by nearby property owners, regulating and supporting services (e.g., flood control, water-quality improvement) may be largely invisible and not accurately capture the full value of wetlands. The travel cost approach is primarily applied to assess the recreational value of wetlands by analyzing the number of trips taken to specific sites and their associated costs. However, its applicability to the valuation of urban wetlands is limited. Like the hedonic method, travel cost studies only evaluate partial aspect of TEV of wetlands.

Survey (stated preference) methods are used when there is no observable behavior to generate value estimates, such as market price or travel costs, direct production linkages or substitutes. The most commonly used stated preference method is contingent valuation, in which respondents are asked to express their willingness to pay (WTP) for a potential environmental benefit or for the avoidance of its loss. In the case of sensitive environmental issues, respondents may be reluctant to make a trade-off between environmental quality and money. Conjoint analysis can be used to avoid difficult WTP questions by asking people what trade-offs they are willing to make between different choice attributes. Notably, the results from these survey-based methods are sensitive to various factors, including the information provided, the focus of valuation, and the survey's methodology. For instance, the value of WTP tends to be higher when respondents believe that no mitigation will occur compared to when mitigation is anticipated. Furthermore, using the "double bounded format" can result in significantly higher mean household WTP than direct questioning about the amount they are willing to pay (Boyer & Polasky, 2004).

In a study of Chinese netizens' WTP through contingent valuation method (CVM) for protecting grassland ecosystem services in Inner Mongolia, it is learned that there is a significant spatial difference in respondents' WTP depending on their age, income, region and knowledge about the grassland ecosystem services. While respondents' concern about grassland ecosystem protection is positively related to their WTP, the distance from the grassland has a significant and negative impact (Ning et al., 2019). Hence the validity and reliability of survey research, and contingent valuation in particular, has long been debated. Critics of contingent valuation also point out that because of the hypothetical nature of the choices, as they do not involve actual payments, potentially can lead to overestimation of WTP. Advocates of survey-based methods claim that useful empirical results can be obtained from carefully designed surveys. These debates underscore the need for robust survey design and data interpretation when employing contingent valuation and related methods for ecosystem service valuation. (Boyer & Polasky, 2004).

In The Value of Land report by ELD Initiative (2015), the loss of ecosystem services due to land degradation was estimated using human appropriation of net primary productivity (HANPP). A map of land degradation showing the loss of net primary productivity (NPP) has been produced by the Food and Agriculture Organization of the United Nations (FAO). As a proxy for land degradation, NPP is calculated using a Normalized Difference Vegetation Index (NDVI) derived from MODIS satellites and adjusted for Rainfall Use Efficiency (RUE). There are a number of difficulties in using satellite data of NDVI as a proxy for NPP due to rainfall variability and regional differences in agricultural and pastoral practices. Two models, Imhoff and Herberl, used in the report gives different estimates of HANPP. The Imhoff representation spatially assigns HANPP to the location of its consumption. The Haberl representation spatially allocates the degradation primarily to the agricultural and grazing areas where the degradation actually occurs. The estimated values from the two models represent fractions of 10-17% of world GDP in 2010, which is significantly larger than the total agriculture representing 2.8% of world GDP. The estimates highlight the importance of the economic impact of land degradation beyond the market value of agricultural products.

Chen et al., (2009) compared the net value of ecosystem services (NVES) of three wetlands; Beijing constructed wetland, human-interfered wetland in Wenzhou, China and, natural wetlands around the world as a mean by using cost-benefit analysis with different discount rates and time horizons, to determine engineering and management of wetland. Services such as waste treatment, disturbance and water regulations are calculated with the replacement cost method. Market price method was used for service values of food and material production, water supply and gas regulation. For the value of habitat and refugia provision which attribute to biodiversity, the result from a previous study was adopted and constructed according to the current case. Ecosystem service cost (ESC) includes construction cost, operation and maintenance cost, and virtual cost. Even though the net present value of ecosystem services (NPVES) under different discount rates are calculated, the assumption here is that the reasonable discount rate is lower than 10%. For the time horizon, a lifetime of 20 years and an infinite time horizon were used. A lifetime of 20 years came from the expected lifetime of the Beijing constructed wetland in this study. Results show that in both finite and infinite time horizons considered, the constructed wetland has the largest net services value at a reasonable discount rate.

As we have seen in above studies, even for widely studied ecosystems and biomes, ecosystem services valuation methods are still highly diverse and straggling to reflect resource scarcity and the total value of an ecosystem services. The concept of 'true' social values, called "accounting prices" by Dasgupta, is still challenging to incorporate. Since shrublands are a successional plant community that can evolve into a different landscape in the future, depending on climate, weather, and people's involvement, some of the methods e.g., discount factor needs to be applied very carefully. Therefore, it is important to note that even though Mediterranean shrublands provide very important ecosystem services in the Mediterranean biome, they face the risk of under-protection and under valuation due to their nature.

3. METHODOLOGY

In the section, the valuation process of ESV of shrublands in mainland Portugal will be described. A comprehensive dataset pertaining to the total shrubland area across mainland Portugal is obtained in Geographic Information Systems (GIS) format. As the research is centered on the monetary assessment of ecosystem services offered by these shrublands, we will be considering both regulating and supporting services. From a thorough analysis of existing literature, it has been established that shrublands play a vital role in the prevention of land degradation, carbon sequestration, and the preservation of biodiversity. These pivotal ecosystem services have been selected for valuation within this study. Notably, the estimation of ecosystem service losses stemming from land degradation encompasses all ecosystem services associated with degradation, thus constituting an alternative Ecosystem Service Value (ESV) for shrublands.

3.1. Area of Study

Portugal's landscape is characterized by a remarkable diversity of terrestrial, marine, and coastal habitats. This diversity is, in part, a consequence of its unique geographical location, situated at the crossroads between the Atlantic and Mediterranean bioclimatic regions (ILTER & LTER Europe, 2023). The focus of this research takes mainland Portugal as its geographical area of investigation.

3.2. Collecting Shrublands area data for mainland Portugal

In the initial phase of the methodology, the total area of shrublands in mainland Portugal has been acquired from the COS mainland Portugal 2018 dataset (Carta de Uso e Ocupação do Solo Continental Portugal 2018) provided by Direção-Geral do Território in 2019. This dataset is presented in vector format as a polygon-based GIS resource. The dataset's Minimum Cartographic Unit (MCU) is 1 hectare, with a minimum distance of 20 meters between lines, and it is designed at an equivalent scale of 1:25 000. The dataset comprises 83 distinct land use classifications, from which the shrubland area has been extracted. The open-source QGIS application has been employed for data handling and analysis. Following the extraction of the shrubland classification, the total shrubland area in mainland Portugal has been determined to be 1 109 437 hectares, which corresponds to 11 094,37 square kilometers. This area constitutes 12,4% of the total land area of mainland Portugal (Statistics Portugal, 2023a).

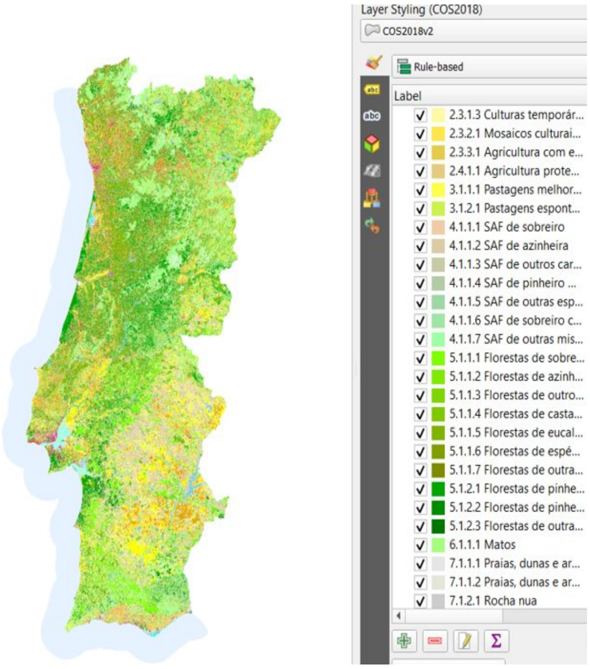

Figure 3.1: Land Use and Occupancy Map – 2018 Mainland Portugal – With all classifications

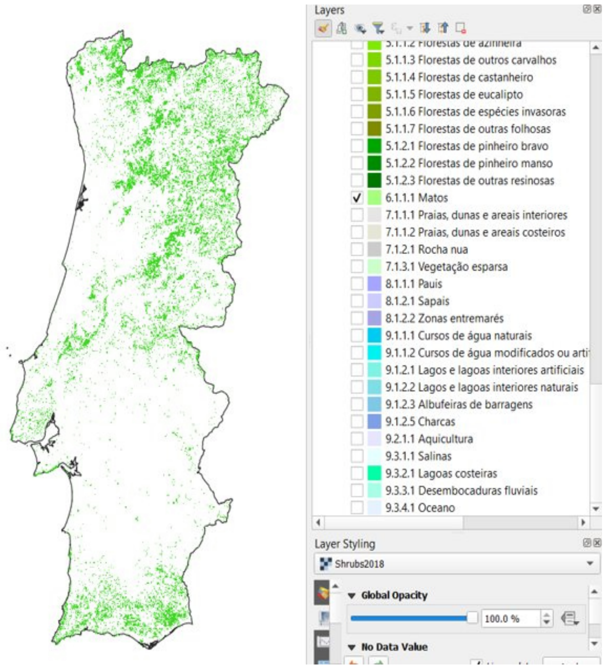

Figure 3.2: Land Use and Occupancy Map 2018 Mainland Portugal - Shrubland area

3.3. Carbon sequestration

Regarding Carbon sequestration capacity of shrublands in mainland Portugal, the benefit transfer method will be used, considering the values from a related study in a similar region. Gratani et al., (2013) conducted a study about Mediterranean shrublands along the Latium coast in Capocotta, Italy, in which they observed a total annual $\mathrm{CO_2}$ sequestration of 80 tonne of $\mathrm{CO_2 ha^{-1} year^{-1}}$ in 2011. Total annual $\mathrm{CO_2}$ sequestration is estimated using below Equation (1). By applying the value, the total $\mathrm{CO_2}$ sequestration of shrublands in Portugal is 88 754 960 tonne per year (Table 3-1).

$$\text{Total annual } CO_2\text{sequestration} = \text{shrubland total area (ha)} \times \text{annual } CO_2\text{sequestration /ha} \qquad (1)$$

The assessment of the value of carbon sequestration, in terms of a metric tonne, draws upon a few references. Firstly, we consider the carbon price within the European Union Emissions Trading System (EU ETS), which stood at USD 86,53 in 2022 (The World Bank, 2023a). Additionally, the social cost of carbon dioxide ($\mathrm{CO_2}$) estimated by Interagency Working Group on Social Cost of Greenhouse Gases (IWG), 2016), presented value at three discount rates: 2,5%, 3%, and 5%. The value at 3% discount rate which is USD 57,01 will be used as it closely aligns with the 3,5% discount rate recommended by HM Treasury's Green Book for projects primarily focused on environmental and social impacts (HM Treasury, 2018)(Table

3-2). All the values have been converted to 2022 price using GDP deflator by The World Bank, as indicated in Equation (2).

$$Value_{target\ year} = \frac{GDP\ Deflator_{target\ year}}{GDP\ Deflator_{year\ of\ study}} \times Value_{year\ of\ study} \tag{2}$$

Table 3-1: Total annual CO_2 sequestration of shrublands in Portugal

	Total area (ha)	CO2 Tonne (ha/year)	Total tonne/ year
Mediterrenean Shrublands in Portugal	1 109 437	80	88 754 960

Table 3-2: Adjusted EU ETS Price of Carbon and Social Cost of Carbon (USD/tonne) (The World Bank, 2023b).

	Year	Value/tonne	Currency	GDP Deflator in the year of study	GDP Deflator (2022)	2022 Value/tonne
Carbon Sequestration						
EU ETS Price	2022	86,53	USD	121,5	121,5	86,53
Social Cost of Carbon (IWG 2016)	2020	51,00	USD	108,7	121,5	57,01

3.4. Biodiversity

For the assessment of ESV associated with biodiversity conservation in Mediterranean shrublands, we turn to the benefit transfer method, referencing a study conducted in the Sierra y Cañones de Guara Natural Park (SCGNP). This study estimated the Willingness To Pay (WTP) for Total Economic Value (TEV) concerning Mediterranean mountain agroecosystems among livestock farmers using pastures within the park and local citizens. The estimation relies on stated-preference methods, specifically employing choice modeling techniques, and deliberative processes such as focus groups. The ESV for biodiversity is established at €22,20 per person per year in 2014 (Bernués et al., 2014). This value is adjusted to 2022 prices using Equation (2) and converted into USD based on the exchange rate indicator from OECD (Table 3-3).

Table 3-3: Adjusted WTP for biodiversity preservation (USD/person) (OECD, 2023; The World Bank, 2023b)

	Year	Value/person/yr	Currency	GDP Deflator in the year of study	GDP Deflator (2022)	2022 Value /person/yr	in USD
Biodiversity preservation							
WTP for biodiversity preservation	2014	22,20	€	99,5	112,7	25,15	23,89

3.5. Land degradation

To assess the value related to land degradation, the Value of Land: ELD Main Report (ELD Initiative, 2015) is referred. This report discusses various approaches to characterizing land degradation, including the estimation of human appropriation of the Earth's net primary productivity (HANPP). Two models are used to determine HANPP: the Imhoff model, which employs models derived from satellite observations and statistical data, while the Haberl model uses process models and agricultural statistics, consistent with Imhoff's estimates. Based on the assessments from two models, ecosystem service value losses due to land degradation are summarized in ELD Main Report, featuring global, regional, per capita, and per square kilometer assessments presented visually in Table 3-4. Notably, the two employed models exhibit significant differences. The key distinction lies in their spatial allocation: Imhoff focuses on consumption locations, serving as a demand-based proxy, while Haberl allocates degradation to agricultural and grazing areas, functioning as a supply-based measure informed by agricultural statistics. The Imhoff model emphasizes geographic representation, highlighting land degradation drivers with a focus on factors like population and consumption. In contrast, the Haberl model provides a more detailed spatial analysis of land degradation, particularly within agricultural areas, although it gives a narrower scope concerning non-agricultural land degradation assessment. (ELD Initiative, 2015).

Given that the ELD Initiative (2015) framework serves as a guide for estimating the TEV of land and its associated ecosystem services to society, these values are considered alternative estimates for shrublands. Regional values per square kilometer for Southern Europe, derived from the Imhoff and Haberl models in 2015, are therefore applied to Portugal. These values amount to 160 916 USD/sq/km and 90 862 USD/sq km, respectively. Notably, the Imhoff model's value is 77% higher than that of the Haberl model. These values are adjusted to 2022 prices using Equation (2) in Table 3-5.

Table 3-4: Regional ecosystem services annual losses from land degradation based on Imhoff and Haberl models. Source: ELD Initiative (2015)

	per person	per sq km
Africa	1,164	43,826
	1,517	57,092
Eastern Africa	928	51,996
	1,553	87,015
Middle Africa	1,455	31,658
	1,393	30,319
Northern Africa	1,074	28,323
	935	24,640
Southern Africa	2,208	50,830
	1,240	28,554
Western Africa	1,160	66,516
	1,945	111,551

	per person	per sq km
Americas	1,686	39,634
	2,126	49,981
Caribbean	863	165,422
	1,200	229,948
Central America	854	57,883
	1,067	72,308
South America	2,198	51,438
	1,891	44,256
Northern America	1,581	26,428
	3,007	50,267
Latin America and the Caribbean**	1,746	53,462
	1,622	49,682

	per person	per sq km
Asia	908	124,191
	1,641	224,434
Central Asia	1,847	29,888
	3,734	60,424
Eastern Asia	155	21,208
	992	135,481
South-eastern Asia	836	118,738
	1,203	170,746
Southern Asia	248	65,490
	998	263,406
Western Asia	10,213	561,088
	10,775	592,016

	per person	per sq km
Europe	2,211	72,206
	2,570	83,934
Eastern Europe	4,500	71,050
	3,085	48,719
Northern Europe	1,763	102,393
	5,305	308,156
Southern Europe	766	90,862
	1,356	160,916
Western Europe	120	21,087
	1,306	229,989

	per person	per sq km
Oceania	6,616	29,623
	3,740	16,746
Australia and New Zealand	8,087	28,899
	3,312	11,835
Melanesia	2,232	39,881
	4,847	86,620
Micronesia	2,227	851,024
	13,972	5,340,272

	per person	per sq km
World	867	46,365
	1,438	76,910

Haberl model
Imhoff model

Table 3-5: Regional ecosystem service losses from land degradation adjusted into 2022 price (USD/sq km) (The World Bank, 2023b)

	Year	Value/sq km	Currency	GDP Deflator in the year of study	GDP Deflator (2022)	2022 Value/sq km
Land Degradation Prevention						
Haberl Model	2015	90 862	USD	100	121,5	110 397,33
Imhoff Model	2015	160 916	USD	100	121,5	195 512,94

3.6. Uncertainty: Fire risk in Portugal

Over recent decades, Portugal has struggled with a growing wildfire concern. According to Statistics Portugal (2023d), a total of 110 099 099 hectares of mainland Portugal were burn, with shrubland and grassland accounting for 1.58% of this area (Statistics Portugal, 2023c). When a shrubland area is affected by fire, the associated ESV is lost. Hence, it is imperative to assess the likelihood of wildfires impacting shrubland areas in Portugal. According to Beighley & Hyde (2018) in the upcoming decade, there is an alarming risk of fire affecting an estimated 600 000 to 750 000 hectares or more in Portugal, encompassing both forest, shrub and agriculture areas. The probability of experiencing high and extreme fire years has been on the rise, with high fire years posting the greatest risk, representing 45% of the occurrences. Between 2000 and 2017, an annual average of 100 000 to 200 000 hectares burned during high fire years. In the worst-case scenario, known as the "Black Skies" scenario, nearly 750 000 hectares could be affected, which equates to almost 10% of Portugal's combined forest, shrubland, and agriculture areas (Beighley & Hyde, 2018) (Table 3-6). Therefore, if the findings from the Black Skies scenario are extrapolated to High Fire Year scenario, it can be estimated that the fires would burn between 1,33% to 2,67% of Portugal's forest, shrubland and agriculture area annually. For ESV calculation of shrublands, the probability from Hight Fire Year scenario will be adopted since as it presents the highest risk factors. Since the data from Beighley & Hyde (2018) does not distinguish between forest, shrubland and agriculture area, the rage of 1,33% to 2,67% will be applied specifically to shrublands.

Table 3-6: Integrating fire risk: scenarios projecting Portugal's future (Source: *Beighley & Hyde, 2018)*

Fire Risk Scenarios	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
Descriptor	**Low Fire Year**	**Moderate Fire Year**	**High Fire Year**	**Extreme Fire Year**	**Black Skies**
Annual hectares burned in thousands	0-50	50-100	100-200	200-500	Approaching 750
Number of times it occurred in 18 years (2000-2017)	3 in 18	4 in 18	8 in 18	3 in 18	0 in 18
Historical Risk Factor, (percent of actual occurrence)	17%	22%	44%	17%	0%
Weather/Climate adjustment factor	Reduced chance	Reduced Chance	Increased chance	Increased chance	Increased chance
Future Risk Factor	**12%**	**18%**	**45%**	**20%**	**5%**

3.7. Discount Rates, Discount Factors and Time Scale

The discount rate serves as the interest rate used to calculate the present value of future cash flows, reflecting the rate of return expected by investors (Khan & Greene, 2013). Social discount rate is where an interest rate used in discounting future cost and benefit of social or public projects (Harrison, 2010). The discount rate facilitates the comparison of cash flows occurring at different points in time, making it a crucial tool for assessing the value of future benefits in present terms. Ecosystem services generate benefits that extend into the future, necessitating the use of discount rates to convert these future benefits into their present value. This conversion allows for informed decision-making by enabling the evaluation and comparison of these benefits against current costs or alternative investments (Khan & Greene, 2013).

The discount factor is a mathematical component representing the present value of future cash flows, calculated based on the discount rate and the time frame during which the cash flows are anticipated. The discount factor transforms the value of future cash flows into their present worth, akin to a multiplier that determines the present value of a future cash flow. In the context of annuities, the discount factor is computed using the annuity formula as follow.

$$Discount\ Factor_{Annuity} = \frac{1-(1+r)^{-t}}{r} \tag{3}$$

Where r = discount rate, t = number of years.

In this regard, there are a few discount rates that are to be considered. HM Treasury's Green Book recommended discount rate of 3,5% for projects with primarily environmental and social impacts (HM Treasury, 2018). The World Bank, in reports such as "Where is the Wealth of Nations" (2006) and "The Changing Wealth of the Nations" (2011), applies a 4% social discount rate to estimate natural capital. An application of Net Present Value (NPV) analysis in woodland valuation by Eurostat suggests an acceptable range of discount rates ranging from 0,5% to 3,5%, deemed appropriate for valuing woodland assets (Khan & Greene, 2013). Even though some study suggests to use zero or negative discount rates, they would not be considered here since it is not compatible with actual decision making. Determining the time horizon is critical as well when calculating the NPV. For successional plantations like shrublands, a shorter time horizon is typically the focus. However, discount factors for different discount rates at different time horizons will be calculated to conduct the sensitivity analysis to see the incremental changes as discount factors and the time horizon change.

3.7.1. Incremental Changes in Discount Factors with Varying Time Horizons

The Appendix 1 illustrates the percentage variation resulting from increases in the time horizon while holding the discount rate constant. Time horizons of 20, 30, 50, 100 years, and infinity are commonly employed in the literature and are considered for sensitivity analysis. Notably, the discount factor value nearly doubles as the time horizon increases from 10 to 20 years (e.g., from a discount factor of 9,47 to 18,05 at a 1% discount rate). However, the magnitude of this increase diminishes as additional years are considered. Consequently, it can be inferred that the discount factor value demonstrates minimal change over long time horizons, and an exceedingly high discount rate renders neglecting future actions due to its economically inconsequential. This insight carries significant implications for valuing ecosystem services, particularly in scenarios where discounting may substantially devalue the economic impact of severe environmental events occurring beyond a 50-year time horizon or more.

3.8. Limitation and Assumption

This study relies on certain underlying assumptions in the valuation of ecosystem services within shrublands. First, it assumes that shrublands offer consistent ecosystem services, regardless of their location and size. Additionally, it assumes that the values of biodiversity and carbon sequestration from various Mediterranean areas, as well as the regional value of land degradation for Southern Europe, are equally applicable to mainland Portugal. Third, it supposes that the preferences and Willingness To Pay (WTP) of surveyed individuals from different areas are similar to the broader population in the study. Finally, the fire risk for total area of forest, shrubland and agriculture area is assumed equally for shrublands. Consequently, the primary limitation of this study is the representativeness of the data, as it relies on the value (benefit) transfer method due to the absence of primary and secondary data. It is crucial to acknowledge that while value transfer may serve as a practical tool in some policy contexts, its effectiveness can be enhanced as our understanding of ecosystem service valuation evolves. Recognizing the situations where value transfer is most appropriate is a valuable step towards refining its application. Further research and ongoing analysis will be instrumental in defining its boundaries and improving its utility in the broader landscape of ecosystem service valuation methodologies (Navrud & Ready, 2007).

4. RESULTS AND DISCUSSION

4.1. Ecosystem Service Value of Shrublands

Drawing upon the findings of previous literature and research, this study has adopted four distinct approaches to determine the ESV of shrublands in mainland Portugal. In this section, the results from these four approaches will be presented.

Approach 1:

$$\text{ESV} = \text{ESV of Carbon sequestration by EU ETS price} + \text{ESV of biodiversity preservation} \quad (4)$$

Approach 2:

$$\text{ESV} = \text{ESV of Carbon sequestration by social cost of carbon} + \text{ESV of biodiversity preservation} \quad (5)$$

Approach 3:

$$\text{ESV} = \text{ES losses from land degradation/ sq km(Haberl model)} \times \text{Total shrubland area(sq km)} \quad (6)$$

Approach 4:

$$\text{ESV} = \text{ES losses from land degradation/ sq km (Imhoff model)} \times \text{Total shrubland area(sq km)} \quad (7)$$

4.1.1. Approach 1 and 2

In Approach 1 and 2, the ESV for the Carbon sequestration service of Portugal shrublands was calculated using both the EU ETS price and the Social Cost of Carbon for the purpose of comparison (Table 4-1). The price per tonne of CO_2 from both representations is multiplied by the total annual tonnage of carbon sequestration. This comparison revealed a significant difference, with the ESV calculated using the Social Cost of Carbon being USD 5 060 million, representing a 34.12% decrease compared to the ESV derived from the EU ETS price, which stood at USD 7680 million. This substantial difference highlights the impact of the choice of valuation method. ESV for biodiversity preservation is computed by multiplying WTP per person per year by the total population of mainland Portugal in 2022 (Statistics Portugal, 2023b) (Table 4-2). Table 4-3 presents the ESV from approach 1 and 2 with USD 713 716 and USD 477 520 per sq km per year respectively.

Table 4-1: ESV of mainland Portugal shrublands for Carbon sequestration service per year

	Value/tonne (USD)	Total tonne/ year	ESV (USD/year)	ESV (USD in million/year)	% in variation
EU ETS Price	86,53	88 754 960	7 679 966 689	7680 Million	Base
Social Cost of Carbon	57,01	88 754 960	5 059 522 628	5060 Million	-34,12%

Table 4-2: ESV of mainland Portugal shrublands for biodiversity preservation service per year

	WTP/person/year (USD)	Total population Portugal	ESV (USD/year)	ESV (USD in million/year)
Biodiversity preservation	23,89	9 974 165	238 261 550	238 Million

Table 4-3: ESV per year of mainland Portugal shrublands (Approach 1 and 2)

	Carbon sequestration	Biodiversity preservation	ESV (USD/year)	ESV (USD in million/year)	ESV (USD/sq km/year)
Approach 1	7 679 966 689	238 261 550	7 918 228 239	7918 Million	713 716
Approach 2	5 059 522 628	238 261 550	5 297 784 178	5298 Million	477 520

4.1.2. Approach 3 and 4

Approaches 3 and 4 introduced an alternative value, focusing on the ESV of shrublands from land degradation preservation services. Imhoff model gave ESV of USD 2 169 million per year which is 77,1% higher than the ESV gave by Harberl model (Table 4-4).

Table 4-4: ESV per year of mainland Portugal shrublands (Approach 3 and 4)

	Value/sq/km	Total shrubland area (sq km)	ESV (USD/year)	ESV (USD in million/year)	% Variation
Approach 3: Haberl Model	110 397	11 094	1 224 788 826	1225 Million	Base
Approach 4: Imhoff Model	195 513	11 094	2 169 092 896	2169 Million	77,10%

The ESV generated by these four approaches yields values that significantly differ from one another (

Table *4-3* & Table 4-4). Among them, Approach 1 produces the highest ESV for shrublands in Portugal, amounting almost USD 8 billion (USD 7 918 million) for the entire area and USD 713 716 per square kilometer per year. In contrast, approaches 3 and 4 provide the lowest ESV, with Approach 3 being the least at USD 1,2 billion (USD 1 225 million). This disparity underscores the sensitivity of ecosystem services valuation methods and the potential for diverse outcomes based on the selected approach, perceptions, and methodologies.

4.2. ESV of shrublands with fire risk

Considering the High Fire Year scenario where 1,33% to 2,67% of forest, shrubland and agriculture area in Portugal could potentially burn annually, the monetary loss of shrublands' ESV was estimated. The loss would range between USD 16 million to USD 211 million annually, depending on the approach and probability. (Table 4-5).

Table 4-5 : ESV per year of mainland Portugal shrublands after adjusting fire risk probability

	ESV (USD in million/yr)	Annual economic loss from High Fire Year		ESV after adjusting fire risk (USD in million/yr)	
Probability of Fire (% of total value)		1,33%	2,67%	1,33%	2,67%
Approach 1	7918 Million	105 Million	211 Million	7813 Million	7707 Million
Approach 2	5298 Million	70 Million	141 Million	5227 Million	5156 Million
Approach 3	1225 Million	16 Million	33 Million	1208 Million	1192 Million
Approach 4	2169 Million	29 Million	58 Million	2140 Million	2111 Million

4.3. Net Present Value of Ecosystem Services (NPVES) of Shrublands

In this section, the Net Present Value of Ecosystem Services (NPVES) was discounted to year 2022 using a 3% discount rate. NPVES was calculated for various time horizons, including 10 years, 20 years, and infinite time, for without fire risk and 10 years with fire risk incorporated. The choice of 10 and 20 years as time horizons is influenced by the successional nature of shrublands, which can experience degradation or transition into different plant communities (Castro et al., 2004; Sardans & Peñuelas, 2013). While infinite time horizon may not be directly relevant to shrublands, it is considered for the purpose of comprehensive NPVES comparison.

The NPVES of shrublands was highest with Approach 1, amounting to about USD 68 billion at a 10-year horizon, and approximately USD 10 billion for the approach with the lowest value. However, The increase in NPVES from a 10-year to a 20-year horizon was only 74%, and the NPVES for an infinite time horizon was nearly four times that of the 10-year horizon. When fire risk was factored into NPVES calculations, the NPVES of shrublands ranged from USD 10,27 million to 66,65 million (Table 4-6 & Table 4-7).

Table 4-6: NPVES at 3% discount rate and different time horizons

	ESV (USD in million/yr)	Discount factor at 3% discount rate			NPVES (USD in million)		
		10 years	20 years	Infinte horizon	10 years	20 years	Infinte horizon
Approach 1	7918 Million	8,53	14,88	33,33	67544 Million	117803 Million	263941 Million
Approach 2	5298 Million	8,53	14,88	33,33	45191 Million	78818 Million	176593 Million
Approach 3	1225 Million	8,53	14,88	33,33	10448 Million	18222 Million	40826 Million
Approach 4	2169 Million	8,53	14,88	33,33	18503 Million	32271 Million	72303 Million

Table 4-7: NPVES (adjusted for fire risk) at 3% discount rate and 10-year time horizon

	ESV after adjusting fire risk (USD in million/yr)		Discount factor at 3% discount rate	NPVES with fire risk (USD in million)	
Probability of Fire (% of total value)	1,33%	2,67%	10 years	1,33%	2,67%
Approach 1	7813 Million	7707 Million	8,53	66646 Million	65741 Million
Approach 2	5227 Million	5156 Million	8,53	44590 Million	43985 Million
Approach 3	1208 Million	1192 Million	8,53	10309 Million	10169 Million
Approach 4	2140 Million	2111 Million	8,53	18257 Million	18009 Million

4.4. CONCLUSION

According to The World Bank (2023c), Portugal’s GDP for the year 2022 is estimated around USD 252 billion. The findings presented in this paper, underscore the substantial economic significance of shrublands. The annual ecosystem service value (ESV) of shrublands ranges from 0.49% to 3.14% of Portugal's GDP, which contrary to the prevailing perception of these lands as unproductive. The various approaches and fire risk scenarios reveal that shrublands contribute substantially to society by providing valuable ecosystem services. signifying their economic importance. In addition, ESV of shrublands equates to 12% to 78% of the total output of the agriculture industry in Portugal in 2022, which stands at USD 10,11 billion according to PORDATA (2023). While shrublands may not primarily offer important provisioning services, the value from regulating and supporting services is substantial.

However, it is imperative to acknowledge the limitations of this study, stemming from the assumptions and extrapolations made regarding the value and data transfers, and relevancies of the available (mostly foreign) data to the Portuguese case. Even though value (benefit) transfer is widely recognized practice to estimate ESV, can only provide accurate estimates when the characteristics of study and target area are identical (Navrud & Ready, 2007). Additionally, the scarcity of accurate environmental data specific to an ecosystem in a specific geographic location, in this case shrublands in mainland Portugal, is one of the biggest challenges in this study. To enhance the relevance and reliability of economic valuations of ecosystem services, there is an urgent need for more accurate and region-specific data. Governments should address this urgent need and invest in collecting precise environmental data as a priority for the correct assessment of their natural capital.

While the evaluation of nature's worth remains a subject of debate, rendering ecosystem services visible through pricing and valuation metrics can facilitate the assessment of the benefits and costs of such projects or policies. It gives a tangible way for the wider public to recognize the importance of the ecosystems. At the very least, the paper highlights the potential impact of shrublands in Portugal’s economy. Therefore, the pursuit of better, more accurate economic valuations is urgent and only possible if more studies about this country’s shrublands are made. With enhanced ESV estimates, we hope to see the inclusion of such valuations as a standard practice in policymaking and in the approval of development projects that imply changes to the landscape and impact affected ecosystems.

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Appendices

Appendix 1: Incremental Percentage Changes in Discount Factors with Varying Time Horizons

Discount Rate =	1%		2%		3%		3,5%	
Time Scale (Year)	Discount Factor	Incremental increase in %	Discount Factor	Incremental increase in %	Discount Factor	Incremental increase in %	Discount Factor	Incremental increase in %
10	9,47	-	8,98	-	8,53	-	8,32	-
20	18,05	91%	16,35	82%	14,88	74%	14,21	71%
30	25,81	43%	22,40	37%	19,60	32%	18,39	29%
40	32,83	27%	27,36	22%	23,11	18%	21,36	16%
50	39,20	19%	31,42	15%	25,73	11%	23,46	10%
100	63,03	61%	43,10	37%	31,60	23%	27,66	18%
Infinite	100,00	59%	50,00	16%	33,33	5%	28,57	3%