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Carbon stock assessment in two Mediterranean forest habitats in Montesinho Natural Park

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“Eu vejo o futuro repetir o passado,

Eu vejo um museu de grandes novidades
O tempo não para…”

- Cazuza

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ABSTRACT

This study quantifies carbon stocks in two Mediterranean oak forest habitats—*Quercus pyrenaica* (habitat 9230) and *Quercus rotundifolia* (habitat 9340)—within Montesinho Natural Park in northeastern Portugal. Carbon was assessed across major ecosystem compartments: above-ground biomass (tree, shrub, and herbaceous layers), below-ground biomass, litter, and soil organic carbon (SOC). Field-based measurements and species-specific allometric equations were used to estimate biomass, while laboratory analyses determined dry mass and SOC content. Results revealed significantly higher carbon values in *Q. pyrenaica* across all compartments. Above-ground biomass (AGB) reached 104.80 Mg ha^{-1} in *Q. pyrenaica*, compared to 32.39 Mg ha^{-1} in *Q. rotundifolia*. Below-ground biomass (BGB) followed a similar trend, with 31.44 Mg ha^{-1} and 9.72 Mg ha^{-1}, respectively. Soil organic carbon (SOC, 0–30 cm) was also higher in *Q. pyrenaica* (128.56 Mg ha^{-1}) than in *Q. rotundifolia* (82.28 Mg ha^{-1}). These differences resulted in total carbon stocks (TC) of 193.92 Mg C ha^{-1} for *Q. pyrenaica* and 103.61 Mg C ha^{-1} for *Q. rotundifolia*, highlighting the superior carbon sequestration potential of the former. Variations in understory composition and forest structure also contributed to differences in carbon distribution and accumulation. These findings underscore the importance of habitat-specific assessments in Mediterranean landscapes and support the inclusion of native oak forests in carbon offset strategies. The integration of biodiversity, structural complexity, and soil health further highlights their multifunctional value for climate mitigation and ecosystem resilience.

Key words: *Quercus pyrenaica*, *Quercus rotundifolia*, Ecosystem Services.

RESUMO

Este estudo quantifica os estoques de carbono em dois habitats florestais mediterrânicos de carvalhos — *Quercus pyrenaica* (habitat 9230) e *Quercus rotundifolia* (habitat 9340) — no Parque Natural de Montesinho, no nordeste de Portugal. O carbono foi avaliado nos principais compartimentos do ecossistema: biomassa acima do solo (camadas de árvores, arbustos e herbáceas), biomassa abaixo do solo, manta morta e carbono orgânico do solo (COS). Foram utilizadas medições de campo e equações alométricas específicas para cada espécie para estimar a biomassa, enquanto análises laboratoriais determinaram a massa seca e o teor de COS. Os resultados revelaram valores de carbono significativamente mais elevados em *Q. pyrenaica* em todos os compartimentos. A biomassa acima do solo (AGB) atingiu 104,80 Mg ha^{-1} em *Q. pyrenaica*, em comparação com 32,39 Mg ha^{-1} em *Q. rotundifolia*. A biomassa abaixo do solo (BGB) seguiu uma tendência semelhante, com 31,44 Mg ha^{-1} e 9,72 Mg ha^{-1}, respectivamente. O carbono orgânico do solo (SOC, 0–30 cm) também foi maior em *Q. pyrenaica* (128,56 Mg ha^{-1}) do que em *Q. rotundifolia* (82,28 Mg ha^{-1}). Essas diferenças resultaram em estoques totais de carbono (TC) de 193,92 Mg C ha^{-1} para *Q. pyrenaica* e 103,61 Mg C ha^{-1} *para Q. rotundifolia*, destacando o potencial superior de sequestro de carbono da primeira. As variações na composição do sub-bosque e na estrutura florestal também contribuíram para as diferenças na distribuição e acumulação de carbono. Essas descobertas ressaltam a importância das avaliações específicas do habitat nas paisagens mediterrâneas e apoiam a inclusão das florestas nativas de carvalho nas estratégias de compensação de carbono. A integração da biodiversidade, da complexidade estrutural e da saúde do solo destaca ainda mais seu valor multifuncional para a mitigação climática e a resiliência do ecossistema.

Palavras-chave: *Quercus pyrenaica*, *Quercus rotundifolia*, Serviços Ecossistêmicos.

SUMMARY

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1. INTRODUCTION

In recent decades, the rising concentration of greenhouse gases (GHGs) in the atmosphere and their severe environmental consequences have drawn global attention, becoming one of the central topics of international debate due to their direct impact on the intensification of the greenhouse effect and, consequently, global warming (IPCC, 2023). Among these gases, carbon dioxide (CO_2) stands out as the most significant, given its high atmospheric concentration and long residence time (Lal, 2009). Deforestation, the combustion of fossil fuels, and, increasingly, wildfires, are key contributors to rising CO_2 emissions. These processes intensify global warming and drive climate change, contributing to the increased frequency and severity of environmental disturbances observed in recent years (Kabir et al., 2023).

Natural carbon sinks are essential in mitigating atmospheric CO_2 accumulation and in slowing the progression of global warming (Friedlingstein et al., 2023). Forests, oceans, and soils act as major reservoirs capable of absorbing and storing significant amounts of carbon over time (Mitchard, 2018; Macreadie, 2021). Oceans absorb nearly one-quarter of anthropogenic CO_2 emissions, functioning as a vast and dynamic carbon sink (Lee and Lee, 2025). Similarly, terrestrial ecosystems, particularly forests, sequester approximately 20% of anthropogenic CO_2 emissions each year through photosynthesis, incorporating atmospheric CO_2 into plant biomass and soil organic matter (Tang et al., 2018; Friedlingstein et al., 2020; Sha et al., 2022; Friedlingstein et al., 2025).

In forest ecosystems, carbon is distributed among several pools, as defined by the Intergovernmental Panel on Climate Change (IPCC, 2006). The main carbon pools include above-ground biomass (AGB), below-ground biomass (BGB), litter, and soil organic carbon (SOC). AGB comprises all vegetation above the soil surface, such as trees, shrubs and herbaceous understory (Wheeler et al., 2016; Malaga et al., 2022; Wan et al., 2024). BGB refers primarily to the root system, including fine and coarse roots, which are closely associated with the soil (Huang et al., 2021). Litter consists of decomposing organic material as leaves, twigs or bark on the soil surface, playing a key role in nutrient cycling and contributing to soil carbon inputs (Krishna and Mohan, 2017; Liu et al., 2023). Among these, soil is recognized as the largest and most stable carbon reservoir in forest ecosystems (Xue, 2024; Gong et al 2025).

Considering the complexity and variability of these carbon pools, accurate carbon stock assessments are therefore essential for the credibility of mitigation strategies and for reducing uncertainty in greenhouse gas inventories (UNDP, 2023). To ensure accurate assessments, region-specific approaches should be implemented, taking into account the ecological heterogeneity among habitats, including variations in forest structure, floristic composition, climatic conditions, and soil characteristics (Lamsal et al., 2012; Doblas-Miranda et al., 2013; Illarionova, 2024; Joshi and Garkoti, 2025).

Given this context, quantifying carbon stocks across diverse habitats enables the conversion of this ecosystem service into a measurable and potentially tradable asset (Mishra and Agarwal, 2024). This supports the development of environmental offset markets, such as carbon credit trading and Payments of Ecosystem Services (PES) (Sierra and Russman, 2006; Ingram et al., 2014). Beyond their contribution to ecological stability and climate change mitigation, carbon storage initiatives can also generate significant social and economic benefits, particularly for communities involved in conservation, reforestation, and sustainable land management (Montagnini and Finney, 2011).

Mediterranean ecosystems hold significant ecological and socio-cultural value (Torras and Saura, 2008; Rota et al., 2025). They are characterized by high levels of biodiversity, including a significant number of endemic species, and by a marked resilience to environmental stressors such as drought and fire (Salesa et al., 2022; Santibáñez, 2025). Although these forests are generally low in timber productivity, they provide a wide range of ecosystem services, such as carbon sequestration, water regulation, food provision, and the preservation of cultural identity (Bangash et al., 2013; Castro et al., 2021; Castellano, 2022). These attributes ensure the continuity of key ecological functions while sustaining human well-being in surrounding rural and peri-urban landscapes.

Within the Mediterranean context, Portugal occupies a biogeographically diverse position, extending into northern zones where temperate species predominate (Amigo et al., 2017). These areas are generally marked by low population density and widespread rural abandonment (Lasanta et al., 2016; Dolton-Thornton, 2021; Castro et al., 2021). Despite demographic challenges such as rural abandonment, these regions may retain or regain high potential for ecosystem service provision. These landscapes can recover and enhance regulating, cultural, and provisioning services, especially water retention and carbon sequestration, through natural regeneration, mixed-species woodlands, and low-intensity forestry, thus achieving both ecological and socio-economic revitalization and strengthening their carbon sink function (Fonseca et al., 2012; Fonseca et al., 2022; Campos et al., 2024; de Souza et al., 2025).

This study aims to quantify carbon stocks in two protected Mediterranean forest habitats within Montesinho Natural Park in northeastern Portugal: habitat 9340 (*Quercus rotundifolia* woodlands) and habitat 9230 (*Quercus pyrenaica* forests), as defined by the EU Habitats Directive. Carbon stocks were evaluated across the main ecosystem compartments: arboreal biomass, understory vegetation, litter, and soil, thereby enabling a more accurate characterization of total ecosystem carbon storage. These results could support the consideration of these ecosystems as viable areas for participation in future carbon credit schemes, based on their capacity to sequester and store atmospheric carbon. The study combines field-based measurements with established estimation methods to ensure reliable quantification.

To achieve this main objective, the following specific objectives were established to guide the development of the study.

- Evaluate the above-ground biomass across forest stands, encompassing both tree and understory vegetation (herbaceous and shrub layers).
- Evaluate the below-ground biomass of the stands.
- Evaluate the amount of dead organic matter present in the ecosystem.
- Estimate the carbon content in these compartments.
- Evaluate the soil organic carbon content as a measure of soil organic matter.
- Calculate the total carbon stock across each ecosystem.

2. LITERATURE REVIEW

2.1 Carbon pools in forests: stocks and quantification methods

According to the Intergovernmental Panel on Climate Change (IPCC, 2019), carbon in forest ecosystems is stored both above and below ground, in living and non-living biomass, and is distributed across five principal components, referred to as carbon pools. These pools include above-ground biomass, below-ground biomass, soil organic carbon (SOC), litter, and deadwood (Figure 1). These compartments together form the basis for understanding how forests act as carbon sinks, contributing to climate regulation through the processes of carbon sequestration, storage, and release.

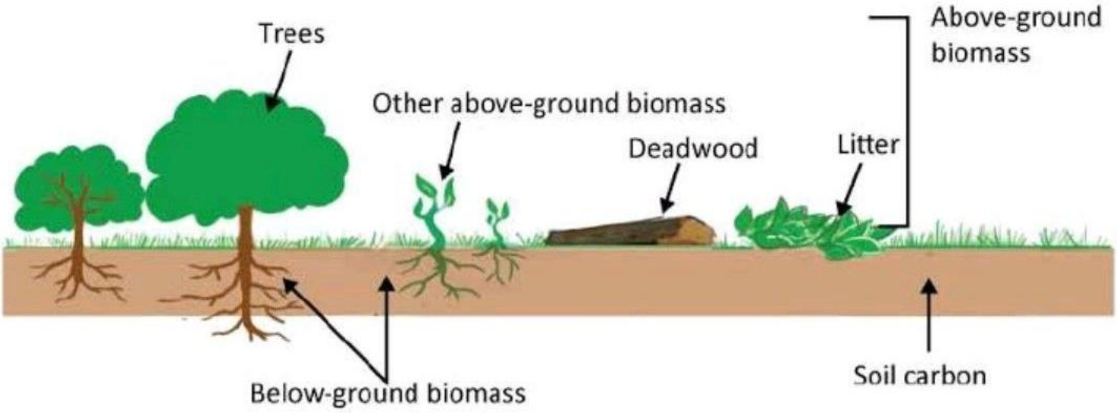

Figure 1 - Main carbon pools in forests (IPCC, 2019).

Above-ground biomass (AGB):

Above-ground biomass includes the total mass of living vegetation above the soil surface, encompassing all components of the individual plant such as leaves, branches, stems, and reproductive structures (Ali et al., 2023). It primarily refers to trees, shrubs, and herbaceous vegetation, representing a dynamic and measurable pool that plays a central role in forest carbon sequestration and monitoring. The proportion of carbon in each part varies, depending on the species, as well as the age and growth pattern of individual specimens (Angkahad et al., 2024; Imran et al., 2025).

In this compartment, biomass estimation is typically conducted through standard forest inventories and understory assessments, which aim to characterise vegetation structure across all vertical layers (Ali et al., 2023). Tree-based surveys are the most widely used method and tend to yield the highest above-ground biomass values among field-based techniques (Sullivan et al., 2018). Quantification approaches may include destructive methods, such as tree felling and direct weighing, which are highly accurate but ecologically invasive (Lewis et al., 2020). Alternatively, non-destructive techniques, particularly the use of allometric equations, are commonly applied. These equations may be generic or specifically developed for certain species or regions (Kuyah et al., 2015). Among them, the most widely used approach in commercial forest inventories combines species-specific allometric equations with wood density values to improve accuracy (Jilo, 2025).

Recent advances in biomass estimation have been driven by the integration of remote sensing technologies such as terrestrial and aerial LiDAR, drones, and satellite imagery. Terrestrial LiDAR offers high accuracy in estimating individual tree and deadwood volumes, though it remains costly and less accessible (Xu et al., 2021). Aerial and drone-based LiDAR, when combined with satellite data, field inventories, and machine learning algorithms, enhance canopy-level and regional biomass mapping (Zhu et al., 2020; Bruening, et al, 2023). Geospatial models and GIS-based regressions further improve the spatial precision of biomass estimates (Zhu et al., 2020). Despite these technological advances, challenges like sensor resolution limits, data saturation, and model assumptions persist, requiring validation through ground-truth data (Song et al., 2023). Innovative tools, such as NASA's GEDI LiDAR and small area estimation models, have recently improved national-scale biomass assessments by reducing bias and increasing precision (Bruening et al., 2023).

Below-ground biomass (BGB):

Below-ground biomass refers to the living mass of plant material located beneath the soil surface, primarily composed of roots, including both fine roots and larger structural roots (Annighöfer et al., 2022). Estimating BGB is particularly challenging due to its inaccessibility and the difficulty in accurately quantifying fine root biomass (Park et al., 2007). The most direct and precise method involves destructive sampling through root system excavation, which is highly labour-intensive and time-consuming, making it impractical for large-scale applications. Consequently, indirect estimation methods are more commonly employed (Neumann et al., 2020). These typically rely on root-to-shoot

(R/S) ratios, which estimate BGB as a proportion of AGB, or on allometric equations that use easily measurable variables such as diameter at breast height (DBH) and tree height to predict BGB.

Litter:

Litter comprises all non-living organic material accumulated on the forest floor, including fallen leaves, twigs, bark, and other plant debris in various stages of decomposition (Cao et al., 2018). Sampling is generally carried out using quadrats or plots of known area (typically between 0.25 and 1 m^2), after which the collected material is dried and weighed in the laboratory to determine the dry mass per hectare. This allows for standardized comparisons of litter biomass across different sites and environmental conditions (Lee et al., 2020).

Soil Organic Carbon (SOC):

Soil organic carbon refers to the carbon contained within the organic matter in soil and often constitutes the largest and most stable long-term carbon pool (Zhang et al., 2017). It is typically quantified through the collection of soil samples at defined depth intervals (Raffeld et al., 2024). These samples are processed in the laboratory to determine their bulk density and carbon concentration (% C)**,** commonly via elemental analysis. SOC stock is then calculated using a standard formula that integrates carbon concentration, bulk density, and sampling depth (Fenton et al., 2024; Agaba et al., 2024).

Deadwood (DW):

Deadwood includes non-living woody material such as fallen logs, standing dead trees, branches, and other woody debris in various stages of decay (Wijas et al., 2023). While several methods exist for estimating deadwood biomass, the most widely applied approach involves calculating the volume of the material and applying species-specific wood density values to convert volume into biomass (Bitunjac et al., 2023). This compartment plays a vital role in comprehensive forest carbon accounting (Sun et al, 2021).

Biomass-to-Carbon Conversion:

To convert biomass into carbon, the Intergovernmental Panel on Climate Change (IPCC) provides a globally recognised framework applicable across diverse ecosystem types.

These standardized guidelines (IPCC, 2006; IPCC, 2019) are widely adopted in the absence of region-specific studies or locally calibrated conversion factors (Herold et al., 2019). In contexts where detailed empirical data are lacking, such as certain forest types, soil conditions, or vegetation classes, the default values and methodologies recommended by the IPCC offer a scientifically accepted alternative, ensuring consistency and comparability in carbon assessments (Requena-Suarez et al., 2019). Although elemental analysis can also be used for direct carbon quantification, it entails higher costs and resource demands.

2.2 Factors that interfere with carbon accumulation

2.2.1 Biotic and abiotic factors

Forest carbon storage is influenced by a broad range of biotic and abiotic factors that influence the amount and distribution of carbon among ecosystem compartments (Li et al., 2020). Among these, forest age and management practices play a critical role. Older and less-disturbed forests tend to accumulate greater carbon stocks, across all pools including soil and deadwood, due to long-term organic matter inputs and minimal anthropogenic disturbance (Pardos et al., 2025). While silvicultural interventions such as thinning or reforestation may increase carbon uptake in younger stands, the conservation of old-growth forests remains essential for maximizing long-term carbon sequestration, particularly in stable reservoirs like soil organic carbon (Ameray et al., 2021; Roebroek et al., 2023).

Structural complexity and species richness further enhance carbon storage potential. Forests with greater biodiversity tend to show higher productivity and more efficient resource utilization, translating into increased biomass accumulation (Li et al., 2020; Mildrexler et al., 2020). Large trees, in particular, contribute disproportionately to above-ground biomass and total carbon stocks, underscoring their ecological importance.

Environmental conditions such as climate, soil texture and fertility, and the frequency of disturbances, e.g., wildfires, droughts, and pest outbreaks, also exert a strong influence, especially on below-ground and soil carbon pools (Hofhansl et al., 2020). These external stressors can both enhance and reduce carbon storage, depending on their intensity and frequency.

Forest management plays a pivotal role in shaping carbon dynamics. Sustainable practices, including extended rotation cycles, afforestation, selective thinning, and the promotion of mixed-species stands, not only improve carbon sequestration but also support biodiversity and other ecosystem services (Ruiz-Peinado et al., 2017; Chiavetta, 2023). Less intensive approaches, such as maintaining uneven-aged structures and longer harvesting intervals, are generally associated with higher carbon retention, although they may involve trade-offs with wood production and economic returns (Bottalico et al., 2016).

2.2.2 Land-use and land cover changes

Land-use and land-cover (LULC) changes exert a profound influence on carbon dynamics in Mediterranean ecosystems, particularly through their effects on soil organic carbon (SOC) stocks (Nadal-Romero et al., 2021). The conversion of natural or semi-natural land into agricultural areas typically results in significant SOC losses, with cultivated soils often retaining only about half the SOC content found in uncultivated soils (Kılıç et al., 2006). This depletion is especially pronounced in vineyards, which are characterized by low organic matter input and frequent soil disturbance, limiting their carbon storage potential (Eldon et al., 2015). Conversely, the abandonment of agricultural land, followed by natural revegetation, facilitates the gradual recovery of SOC stocks. This process is strongly linked to ecological succession, whereby systems evolve from herbaceous cover to shrublands and ultimately to forested landscapes, enhancing both above-ground and below-ground carbon accumulation over time (Lasanta et al., 2019).

Land-use changes, particularly the conversion of agricultural land to forest, have historically enhanced carbon sequestration in Mediterranean regions, underscoring the critical role of woodland conservation in sustaining ecosystem services (Padilla et al., 2010). In parallel, innovative land-use practices, such as short-rotation coppice plantations and mixed orchards, present opportunities to integrate carbon capture with biodiversity goals, reinforcing the multifunctionality of Mediterranean landscapes (Oliveira et al., 2025; Ioannidou et al., 2025)

Despite these benefits, Mediterranean ecosystems remain vulnerable due to climate variability, increased wildfire frequency, and a legacy of land degradation, necessitating adaptive and evidence-based management approaches (Roces-Diaz et al., 2021). Carbon stocks in these forests are highly variable. For example, riparian forests dominated by

Alnus glutinosa can store up to 162 Mg C ha^{-1}, primarily in woody biomass (Fernandes et al., 2020). Likewise, old-growth Mediterranean island forests may accumulate around 200 Mg C ha^{-1} across above-ground biomass, litter, and soil compartments, substantially exceeding values observed in early-successional systems such as agricultural systems (Badalamenti et al., 2019).

Historical data from Spain illustrate a marked decline in forest carbon stocks due to deforestation until the mid-20th century, followed by significant recovery. By 2010, national forest carbon stocks reached 844 Tg C, largely attributed to forest expansion and increased carbon density (Infante-Amate, 2023). Forest management has been a key factor in this trend; mixed-species stands, especially combinations of beech with fir or other broadleaves, can store 25–317% more carbon than monocultures. Furthermore, management intensity and strategy are decisive for future carbon storage under changing climatic conditions (Testolin et al., 2022; Murphy et al., 2025).

In Mediterranean pine forests, practices such as extended rotation periods and lower management intensity can enhance on-site carbon stocks. At the same time, harvested wood products contribute to off-site carbon storage, strengthening the role of sustainable forestry in climate change mitigation (Testolin et al., 2022).

Overall, the combination of structural complexity, adaptive functional traits, and ecological diversity supports the long-term stability of carbon stocks in Mediterranean forests, emphasizing the urgency of their conservation amidst accelerating environmental change.

2.3 Carbon sequestration in Mediterranean forest ecosystems

2.3.1 Structural complexity and resilience

Carbon stocks in these ecosystems vary widely depending on forest type, management practices, species composition, and successional stage. In some regions of Spain, the average stand-level carbon stock is approximately 45.1 Mg C ha^{-1}, with total living biomass carbon stocks reaching 621 Tg C. Forests with greater structural complexity, such as uneven-aged and broadleaf-dominated stands, generally store more carbon than even-aged or conifer-dominated ones (Vayreda et al., 2012).

The effectiveness of these forests as carbon sinks is closely linked to their ecological traits, which support sustained biomass accumulation and carbon retention. A key factor is resilience to environmental stressors, particularly recurrent drought and wildfires, which are common in Mediterranean climates (Blanco-Rodriguez, 2023). This resilience is underpinned by a persistent and well-structured understory that contributes to both above and below-ground carbon storage and enhances post-disturbance recovery (Gavilàn et al., 2018).

The vertical complexity of Mediterranean forests, with multiple vegetation strata, enhances light-use efficiency and promotes biomass accumulation across layers (Marziliano et al., 2021). The dominance of sclerophyllous species, characterised by tough, long-lived leaves adapted to water scarcity and fire, further supports ecosystem stability and prolongs carbon residence time in biomass (Bussotti and Pollastrini, 2020). Functional diversity also enhances carbon sequestration by promoting resource-use complementarity and ecosystem productivity (Wu et al., 2025).

As recognised biodiversity hotspots, Mediterranean forests support many endemic and specialized species (Mèdail and Quézel, 1999). This rich biodiversity reinforces ecosystem functioning and resilience, contributing to the long-term stability of carbon stocks and strengthening the conservation value of these landscapes.

2.4 Ecosystem services and compensation policies

2.4.1 Concept and valuation of ecosystem services

Ecosystem services refer to the multiple benefits that humans obtain, directly or indirectly, from natural ecosystems (Schmidt et al., 2016). The concept gained international prominence following the publication of the Millennium Ecosystem Assessment (MEA, 2005), which categorised ecosystem services into four main groups: provisioning (e.g., food, timber, freshwater), regulating (e.g., climate regulation, carbon sequestration, flood control), cultural (e.g., recreation, spiritual value), and supporting services (e.g., soil formation, nutrient cycling). This framework highlights the critical role ecosystems play not only in sustaining biodiversity and ecological functions, but also in underpinning human well-being and economic development (Naeem et al., 2009; Hernandez-Blanco et al., 2022).

Valuing ecosystem services involves quantifying their contribution to society in monetary or non-monetary terms to inform policy-making, resource management, and land-use planning more effectively (Costanza et al., 2017; Brander et al., 2024). Economic valuation aims to internalise the externalities of ecosystem functions by assigning monetary value to services such as carbon sequestration, water purification, pollination, and other ecological processes. (Tinch et al., 2019). Common valuation methods include market pricing, replacement cost, avoided cost, contingent valuation, and choice modelling, each with strengths and limitations depending on the ecosystem service under assessment (Gunton et al., 2017; Tinch et al., 2019). For example, the value of carbon sequestration can be estimated using carbon market prices or the social cost of carbon emissions.

2.4.2 Payments for Ecosystem Services (PES) and Reducing Emissions from Deforestation and Forest Degradation (REDD+) mechanisms

The valuation of ecosystem services plays a strategic role in environmental economics and conservation planning by supporting mechanisms such as PES, REDD+, and natural capital accounting (Hejnowicz et al., 2014). These instruments aim to incentivise landholders and stakeholders to conserve or restore ecosystems by compensating them for the public goods their land provides (Wegner, 2016).

Recognizing the multifunctionality of forest systems supports more integrated and sustainable land-use strategies, particularly in areas under pressure from agricultural intensification, urbanisation, or climate change (Simons et al., 2021). Integrating ecosystem service valuation into policy and decision-making enhances the visibility of nature's contributions to people and enables more cost-effective conservation interventions (Başkent, 2020). In this context, accurate quantification of services such as carbon sequestration is relevant not only for climate policy (e.g., national GHG inventories, carbon markets), but also for supporting holistic land management approaches that reconcile conservation goals with socioeconomic development (Schwaiger et al., 2019).

2.4.3 European Union (EU) policy and financial tools

The EU recognises ecosystem services, defined as the direct and indirect benefits ecosystems provide to society, as fundamental to its environmental and climate policies.

Under the European Green Deal, the EU Biodiversity Strategy 2030 sets ambitious targets: protecting 30% of terrestrial and marine areas and restoring 20% of EU land and sea by 2030 through the Nature Restoration Law. Expanding the Natura 2000 network is central to this strategy. Technically, the MAES initiative supports the mapping and assessment of ecosystems and their services, informing national restoration efforts (European Environment Agency, 2015). Complementary policies such as the Water Framework Directive (2000/60/EC) aim to achieve the good ecological status of water bodies and maintain hydrological services (European Commission, 2000; European Commission, 1992; European Commission, 2009).

Policy instruments such as the EU's 2030 Biodiversity Strategy and the Common Agricultural Policy (PAC) have gradually incorporated the PES approach, aiming to reward landowners and communities for maintaining or enhancing ecosystem functions. Although PES remains more consolidated in global initiatives such as REDD+ or voluntary carbon markets, it is gaining traction in EU contexts (European Commission, 2020; European Commission, 2023).

The EU has developed several financial and policy instruments to support ecosystem service-based strategies. The LIFE Programme funds initiatives related to habitat restoration, pollinator protection, and carbon sequestration, while Horizon Europe promotes research on Nature-Based Solutions and ecosystem services assessment in climate change contexts. Under the CAP 2023–2027, 25% of direct payments are allocated to eco-schemes that incentivize sustainable agricultural practices, such as agroecology, agroforestry, and carbon farming, in exchange for benefits like soil conservation and carbon storage (European Commission, 2021).

2.4.4 Natura 2000 Network

The Natura 2000 network is a European conservation initiative established in 1992 under the Habitats Directive and the Birds Directive. It aims to protect natural habitats and species of European importance, ensuring the preservation of biodiversity across the EU (European Commission, 2020). The network consists of Special Areas of Conservation (SACs) and Special Protection Areas (SPAs), covering around 18% of the EU’s land area and 6% of its marine territory. Natura 2000’s primary objective is to safeguard both species and habitats while maintaining the ecosystem functions that support essential services such as climate regulation, water purification, and pollination.

PES schemes play a crucial role in supporting the management and conservation of protected areas. These financial mechanisms provide compensation to landowners and local communities for adopting conservation-friendly practices, which help maintain biodiversity and offset the economic costs associated with land use restrictions (Kaiser et al., 2021). By promoting the sustainable management of natural resources, PES schemes contribute to long-term ecological benefits. Linking conservation efforts to economic incentives, they enhance the effectiveness of Natura 2000, facilitating the protection and restoration of habitats that are essential for both ecosystem functionality and human well-being (Santos et al., 2019).

Many Natura 2000 sites, established under the EU Habitats and Birds Directives, provide critical regulating services (Kokkoris et al., 2020). The protection status of these areas creates a strong institutional foundation for implementing PES and carbon projects (Kati et al., 2015). Furthermore, abandoned rural landscapes, particularly in southern and interior Europe, offer unique opportunities for ecological restoration, often leading to natural reforestation, biomass accumulation, and increased soil organic carbon (Campos et al., 2024). In regions like northeast Portugal and central Spain, demographic decline and agricultural abandonment have inadvertently fostered climate benefits, making carbon-based PES schemes a cost-effective approach to conservation (García-Ruiz et al., 2020).

Within this ecological and policy landscape, certain habitats stand out for their carbon storage potential and eligibility for market participation. Habitat 9230 (Galician-Portuguese oak forests of *Quercus robur* and *Quercus pyrenaica*) and Habitat 9340 (*Quercus ilex* and *Quercus rotundifolia* forests), both protected under Annex I of the Habitats Directive, are ecologically valuable Mediterranean forest systems with high carbon sequestration capacity (ICNF, 2023). These habitats often feature structural complexity, long-lived species, and well-developed understory layers, which contribute to stable and resilient carbon pools. Their occurrence in low-density, rural landscapes enhances their suitability for PES integration and carbon offset initiatives, especially when combined with conservation-compatible land uses.

3. MATERIAL AND METHODS

3.1 Study area

The study was conducted in Montesinho Natural Park (PNM), a protected area of 75,000 ha located in the northeastern Portugal. Two forest stands of different oak species were established as experimental sites: one dominated by *Quercus pyrenaica* Willd. and the other by *Quercus rotundifolia* Lam (Figure 2).

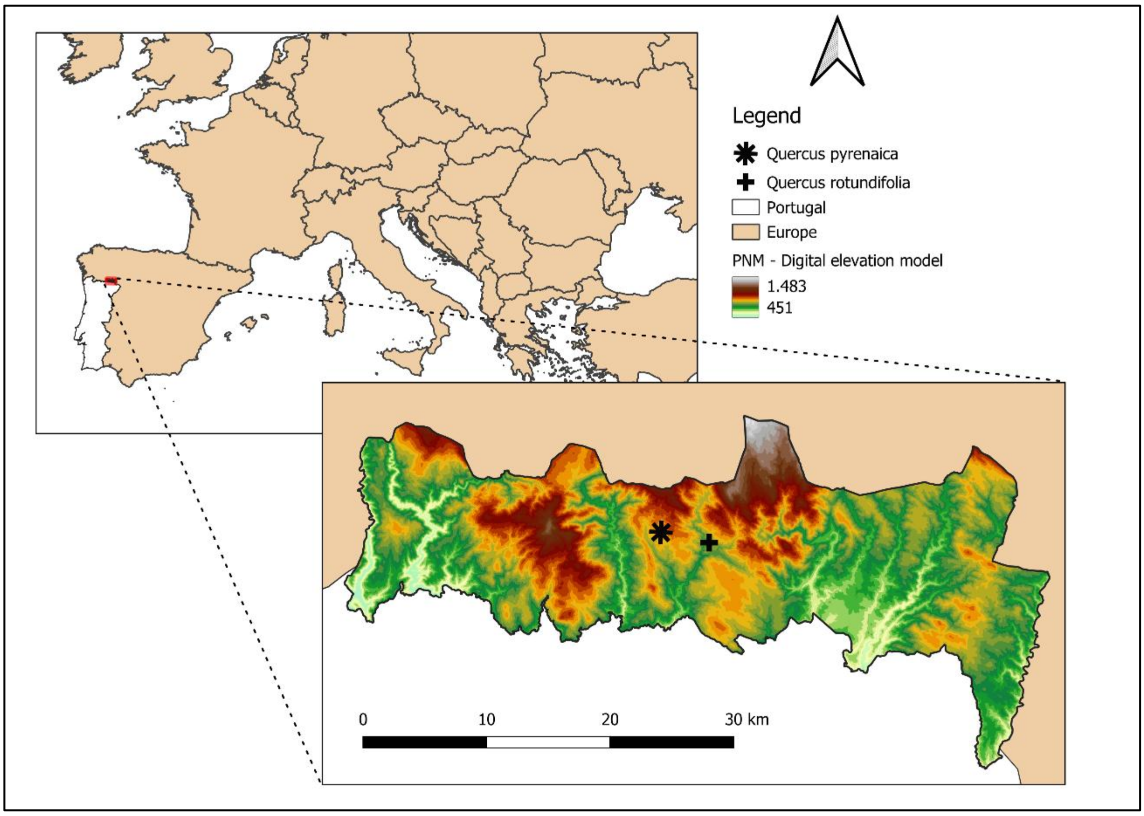

Figure 2- Location of the study area (DGT, 2019).

The territory of PNM exhibits a heterogeneous topography, featuring a plateau cut by deep valleys and mountain slopes ranging from flat to very steep. The elevation ranges from 438 (Mente River) to 1486 m asl (Montesinho peak). Mean annual precipitation ranges from 806 mm on the Lombada plateau to 1262 mm in the Montesinho mountain range, while mean annual temperatures vary from 8.5°C to 12.8°C (INMG 1991; Castro et al., 2021).

The soils are primarily classified with Leptosols (77.1%) and Cambisols (20.1%), while Luvisols and Alisols together account for less than 2% of the territory (Agroconsultores and Coba, 1991). Landscape heterogeneity reflects the wide diversity of land uses and cover types. The area comprises annual and perennial crops, pasture lands, natural woodlands dominated by *Q. pyrenaica* and *Q. rotundifolia*, riparian forests, pine woodlands, and an extensive cover of semi-natural shrubs. These forest environments support a high diversity of both plant and animal species (Sil et al., 2016; DGT, 2019).

The *Q. pyrenaica* experimental site is located near the village of Zeive (41° 54′ 30.68″ N, 6° 53′ 39.74″ W), covering an area of 2.72 hectares. The area is located at an altitude of 985 meters, with soil predominantly classified as Umbric Leptosol (Agroconsultores and Coba, 1991) and total annual rainfall is 772.7 mm (data from 1981 to 2010, IPMA). Its marcescent habit contributes to the formation of a dense litter layer, which plays a key role in enhancing soil moisture retention and nutrient cycling. Additionally, the irregular spatial distribution of trees fosters a structurally diverse understory, comprising species such as *Cytisus scoparius* (L.) Link, G*enista falcata* Brot., and *Rubus ulmifolius var. ulmifolius* Schott (Figure 3, left).

The *Q. rotundifolia* experimental site is situated near the village of Vilarinho de Cova de Lua (41° 54′ 0.31″ N, 6° 50′ 49.50″ W) and encompasses 2.67 hectares. The area is located at an altitude of 884 meters, with soil predominantly classified as Leptosols derived from basic rocks (Agroconsultores and Coba, 1991) and total annual rainfall is 772.7 mm (data from 1981 to 2010, IPMA). This evergreen forest is characterized by irregular tree spacing, which supports the development of a well-structured understory, predominantly composed of herbaceous plants and shrub species such as *Pterospartum tridentatum* L., *Cistus spp*., and *Erica spp, Lavandula pedunculata* (Mill.) Cav. The litter layer in Vilarinho is less abundant than that observed in Zeive. These two native *Quercus* species represent the region’s most characteristic forest ecosystems, offering contrasting litter dynamics and forest structural features, including understory composition, that support comparative assessments of carbon stocks (Figure 3, right).

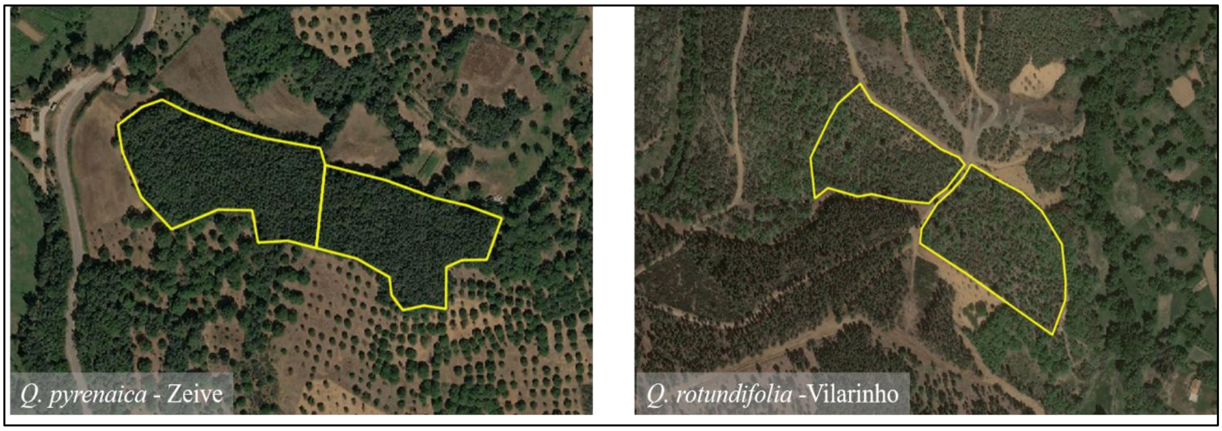

Figure 3 - Aerial images of both experimental forest stands: on the left, the stand of *Q. pyrenaica* near the village of Zeive (coordinates 41° 54′ 30.68″ N, 6° 53′ 39.74″ W), and on the right, the stand of *Q. rotundifolia* near the village of Vilarinho (coordinates 41° 54′ 0.31″ N, 6° 50′ 49.50″ W).

3.2 Field measurements and sampling

Fieldwork was conducted between January and June 2024 to quantify carbon stocks across the main ecosystem compartments in both forest stands. The assessment covered four principal carbon pools: above-ground biomass (AGB), below-ground biomass (BGB), litter, and soil organic carbon (SOC).

Above-ground biomass was quantified through direct field measurements, while BGB was estimated indirectly using allometric models based on AGB values, given the impracticality of destructive root sampling. Litter and soil samples were collected and processed in the laboratory to determine dry mass and organic carbon content.

To estimate AGB, vegetation was stratified into three layers: arboreal (AGBtree), shrub (AGBshrub), and herbaceous (AGBherb). Each stratum was assessed independently to account the structural and compositional differences in biomass contribution. The total above-ground biomass (Mg ha^{-1}) was calculated by summing the dry biomass per hectare from all strata, providing an integrated estimate.

- Above-ground tree biomass sampling (AGBtree)

For this layer, AGBtree, was considered only as trunk biomass, disregarding the branches and leaves of the trees in this evaluation. Four circular plots of 500 m² were randomly established in each oak stand. Plot centres were georeferenced using a Real-Time

Kinematic (RTK) GNSS system, ensuring centimeter-level positional accuracy. The nominal radius of each plot was corrected for slope by dividing by the cosine of the maximum incline angle, measured with a SUUNTO clinometer. Slope aspect was recorded using a magnetic compass.

Within each corrected plot, all trees with diameter at breast height (DBH, measured at 1.3 m above ground) ≥ 7.5 cm were tagged and numbered clockwise starting from geographic north. Diameters were measured with a Haglöf caliper (Figure 4). Trees were grouped into 5 cm DBH classes (e.g. 7.5–12.5 cm, 12.6–17.5 cm, 17.6–22.5 cm, 22.6–27.5 cm, 27.6–32.5 cm, 32.6–37.5 cm). Following the systematic sampling method of Draudt (1917), every 5th tree in the ordered list (i.e., the 1st, 6th, 11th, …) was selected as a sample tree, and five additional dominant trees were selected based on total DBHs. For each sample and dominant tree, the following biometric variables were measured:

- Stem height (h_s): from stem base to base of live crown using a Vertex hypsometer (Haglöf).
- Total height (h_t): from stem base to the crown apex.
- Cross diameter (d): a second DBH measurement perpendicular to the first.

This protocol follows the standard forest inventory methods currently adopted in Portugal (ICNF, 2019).

Figure 4 - Forest inventory procedures conducted within sample plots.

- Above-ground shrub biomass sampling (AGBshrub)

Prior to shrub sampling, a preliminary survey identified the dominant shrub species within the study area. Thereafter, their population density was quantified through systematic plot sampling. Finally, individual shrubs were harvested and placed in sealed plastic bags to prevent moisture loss prior to laboratory processing.

Shrub layer characterization began with an exploratory inventory following the method of Cerrillo and Oyonarte, (2006). For this purpose, ten 20 m transects were established at each site. At 1 m intervals along each transect, the nearest shrub to both the left and right of the tape was identified, and its height recorded (Figure 5). This systematic approach resulted in the identification and measurement of 393 shrubs in the *Q. rotundifolia* stand and 336 shrubs in the *Q. pyrenaica* stand. Dominant species and their most frequent height classes were identified. A “morphotype” was defined as a unique combination of species and height class (0.0-0.5m, 0.51-1.0m, 1.01-1.5m, or 1.51-2.0 m) representing at least 5 % of total individuals sampled. Only morphotypes with at least two individuals were retained for density estimation and biomass sampling.

Figure 5 - Exploratory shrub inventory using line-intercept transects.

Shrub density was then estimated using ten circular plots of 25 m² (5.64 m diameter), randomly placed within each stand. Each plot was divided into four quadrants to facilitate counting (Figure 6). Individuals of each morphotype were recorded in each quadrant, and the total per plot calculated.

Figure 6 - Assessment of morphotype density within quadrants.

For each morphotype, three representative individuals were randomly selected. Total height and crown dimensions were measured, and individuals were harvested at ground level (Figure 7). For example, if *G. falcata* occurred in all four height classes, so then 12 individuals were collected (3 per class), ensuring morphometric and vertical structure were captured.

Figure 7 - Destructive sampling of representative shrub morphotypes.

- Above-ground herbaceous biomass sampling (AGBherb)

Herbaceous biomass was assessed using the harvest (cut-and-weigh) method with 0.25 m^2 metallic quadrat. Ten samples were randomly collected from each stand (figure 8), and the material was placed in sealed plastic bags to prevent moisture loss prior to laboratory processing.

Figure 8 - Herbaceous biomass collection.

- Litter sampling

Litter was quantified by collecting ten random samples of organic matter at each stand using a 0.25 m² metallic frame (figure 9). All dead plant material above the mineral soil, such as leaves, twigs, bark, and fragments, was included. Samples were sealed in plastic bags for transport and moisture preservation.

Figure 9 - Collection of surface litter.

- Soil sampling

Soil organic carbon (SOC) was assessed using ten composite samples per stand at two depths: 0–10 cm and 10–30 cm. Each composite consisted of three sub-samples collected using a soil auger (Figure 10), which were then homogenised to form a representative sample for each depth and location. Soil samples were collected near the collection point for both herbaceous and litter samples.

Figure 10 - Soil sample collection using auger.

3.3 Treatment of Collected Samples

Laboratory Processing of Soil and Plant Samples

The collected shrub morphotypes were placed in a laboratory oven and dried at 105 °C for 48 hours to determine their dry mass (DM), following the protocol described by Tommila et al., (2024). In contrast, herbaceous and litter samples were oven-dried at 70 °C for 48 hours, ensuring complete moisture removal. After the drying period, the dry mass of each sample was weighed using a precision analytical balance (Figure 11).

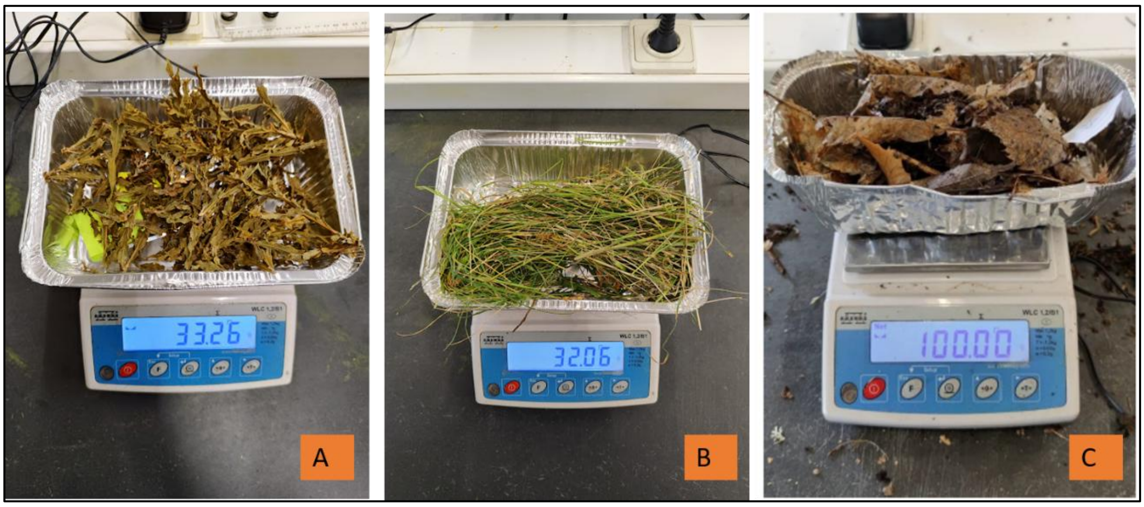

Figure 11 - Dry mass (DM, g) of (A) shrub, (B) herbaceous, and (C) litter samples.

To evaluate soil bulk density, disturbed soil samples were collected at different depth intervals, following the methodology described by Howlett et al., (2011). Initially, the samples were air-dried at room temperature (approximately 20–25 °C) until a constant weight was achieved. Subsequently, the samples were oven-dried at 60 °C for 24 hours in a forced-air oven. After drying, they were passed through a 2-mm mesh sieve and later analysed to determine their organic carbon content (%C) (Figure 12).

Figure 12 - Sieved soil sample for carbon content analysis.

3.4 Biomass estimates

- AGBtree assessment.

Individual tree volumes were estimated using species-specific allometric models standardized for Portuguese forest inventories. For *Q. rotundifolia*, the single-entry equation of Paulo and Tomé, (2006), based on diameter at breast height (d), was applied (Eq. 1). For *Q. pyrenaica*, the double-entry equation of Carvalho, (2000), incorporating both diameter (d) and total tree height (ht), was used (Eq. 2).

$$V_{Qr} = 0.000452 \times d^{1.9783} \qquad \text{(Eq. 1)}$$

$$V_{Qp} = \frac{0.08011 \times (d^2 h)^{0.922}}{1000} \quad \text{(Eq. 2)}$$

Where:

- VQr and VQp represent the estimated tree volume (m^3) of *Q. rotundifolia* and *Q. pyrenaica*, respectively;
- d is the diameter at breast height (cm);
- ht is the total height (m).

In addition to volume, the following stand variables were calculated for each site: mean dominant height (Hd), mean total height (Ht), basal area per hectare (G), and tree density (N).

Above-ground biomass of trees per hectare (AGBtree, Mg ha^{-1}) was estimated by multiplying the calculated tree volume by species-specific wood densities, 600 kg m^{-3} for *Q. pyrenaica* (Castaño-Santamaría and Bravo, 2013) and 900 kg m^{-3} for *Q. rotundifolia* (Sousa et al., 2021). The resulting biomass values were then extrapolated to a per-hectare basis, accounting for the size of the sample plot and tree density.

- AGBshrub assessment.

Shrub individuals selected for destructive sampling had their biovolume calculated using field measurements of height and canopy cover. For each morphotype, the average biovolume of the three sampled individuals was computed and referred to as the mean morphotype biovolume (BVMm, m^3). These same individuals were oven-dried to determine their mean dry biomass (BMm, g).

To estimate above-ground shrub biomass per hectare (AGBshrub, kg ha^{-1}), the dry mass for each morphotype (DM_morphotype, kg ha^{-1}) was calculated using the Eq. 3 and the following parameters:

- VPM – Relative proportion of each morphotype within the species (%), based on morphotype density assessment;
- BVM – Mean biovolume per species (m^3 ha^{-1}), based on field inventory data (see attachments).
- BMm – Mean dry biomass per morphotype (kg);

- BVMm – Mean biovolume per morphotype (m^3)

The AGBshrub was then estimated by summing the dry biomass of all morphotypes. Since different morphotypes (e.g., height classes) contribute differently to total biomass, their specific densities (individuals ha^{-1}) and proportions were used to adjust the calculations. These values were derived from field inventory data, ensuring a realistic representation of biomass distribution.

$$DM_{morphotype} = \frac{VPM \times BVM \times BM_m}{BVM_m} \qquad \text{(Eq. 3)}$$

- BGB estimates.

Below-ground biomass (BGB) of the tree stratum was estimated indirectly from the AGB using a root-to-shoot ratio (R) as recommended by the IPCC (2006). A ratio of R = 0.30 was adopted, within the reported range of 0.24 to 0.37 for temperate broadleaf and coniferous forests (IPCC, 2006, Chapter 4 – Forest Land, Table 4.4). This value was deemed appropriate for the ecological context of *Q. pyrenaica* and *Q. rotundifolia* stands in Mediterranean-temperate regions of Portugal, where direct measurement of root biomass is logistically challenging. The BGB (Mg ha^{-1}) was calculated using the following equation:

$$BGB = AGB \times 0.30 \qquad \text{(Eq. 4)}$$

- AGBherb and litter assessment.

The dry mass (DM) of herbaceous vegetation and litter, determined as described in section 3.3, was extrapolated to a per-hectare basis based on the area sampled, and expressed in kg ha^{-1} and Mg ha^{-1}, respectively.

3.5 Soil Organic Carbon (SOC)

Soil organic carbon (SOC, Mg C ha^{-1}) were estimated following the methodology described by Seboko et al., (2021), using Eq. 5. This approach incorporates the carbon concentration in the fine soil fraction, bulk density, and sampling depth to calculate the organic carbon stock per hectare.

$$\mathrm{SOC} = C_{Soil} \times \mathrm{BD} \times \mathrm{t} \times 0.01 \qquad \text{(Eq. 5)}$$

Where:

- SOC stocks = soil carbon stock (Mg C ha^{-1});
- C_{Soil} = soil carbon concentration (%);
- BD = soil bulk density (kg m^{-3});
- t = soil depth (cm);
- 0.01 = unit conversion factor.

3.6 Total carbon storage (TC)

Total carbon in biomass and litter (C_{BL}, Mg ha^{-1}) was calculated using Eq. 6:

$$C_{BL} = CF \times (AGB + BGB + Litter) \qquad \text{(Eq. 6)}$$

Where:

- CF is the carbon fraction of dry organic matter, expressed as tonnes of carbon per tonne of dry matter (Mg C Mg^{-1} DM).
- A default value of 0.47 Mg C Mg^{-1} DM was applied, in line with the IPCC (2006) Tier 1 and Tier 2 guidelines, indicating that 47% of the dry mass of biomass and litter is composed of carbon.
- This factor was consistently applied across all biomass compartments, above-ground biomass (AGB), below-ground biomass (BGB) and litter, to estimate their respective carbon stocks (Doraisami et al., 2024).

The total carbon storage (TC, Mg C ha^{-1}) reflects a comprehensive estimate of the ecosystem's carbon stock by integrating carbon stored in vegetation and litter (CBL) with soil organic carbon (SOC), as expressed in Eq. 7 (Fonseca et al., 2012; Das and Mondal, 2025):

$$\mathrm{TC} = C_{BL} + SOC \qquad \text{(Eq. 7)}$$

Where:

- C_{BL} represents the total carbon stored in vegetation (AGB, BGB, and litter);
- SOC corresponds to the soil organic carbon stock.

3.7 Statistical analysis

The statistical analysis was performed using one-way analysis of variance ANOVA Type III SS, with stands as sources of variation. Tukey's honestly significant difference test was used for pairwise comparisons between groups ($p < 0.05$). Means and standard deviations by compartiment were calculated. All statistical analyses were performed using the SYSTAT® 13.2 software package.

3.8 Diagram summarising the methodology.

To summarize the methodology and enhance comprehension, the following flowchart was created to depict the study's main steps: field sampling of biomass in carbon pools, litter and soil and calculation of total carbon stocks in each compartment (Figure 13).

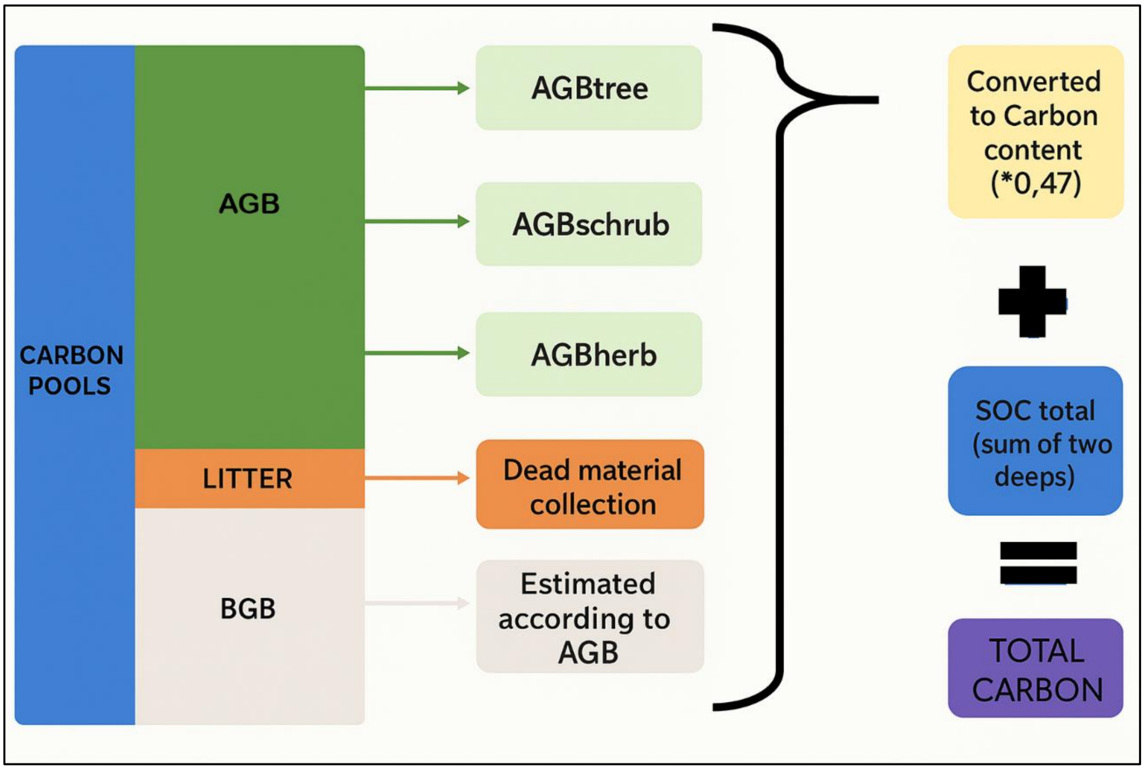

Figure 13 - Diagram summarising the methodology employed. (Source: author)

4. RESULTS AND DISCUSSION

Initially, the biomass results for each compartment are presented, including those of the litter layer. These results are subsequently converted into carbon stocks, followed by a detailed discussion of the findings.

4.1 Above-ground biomass (AGB) compartiments

- AGBtree

Q. pyrenaica and *Q. rotundifolia* stands exhibited marked differences in tree-level structural attributes (Table 1). *Q. pyrenaica* showed a higher number of individuals per plot than *Q. rotundifolia* (77.25 ± 5.38 vs 48.25 ± 7.80 trees plot^{-1}) and its individuals are significantly larger, with greater total height (11.44 ± 0.16 m vs. 6.22 ± 0.89 m), dominant height (12.55 ± 0.77 m vs. 7.09 ± 1.67 m), stem height (7.68 ± 0.74 m vs. 2.45 ± 0.44 m), and larger mean diameter at breast height (15.33 ± 0.44 cm vs. 12.43 ± 1.51 cm).

These differences likely result from a combination of factors, including site quality and different environmental conditions, management history, and species-specific growth strategies. The contrasting structural characteristics between these oak forest types have direct implications for carbon sequestration (Moreno-Fernández et al., 2020; Sousa et al., 2021; Carvalho, 2023).

Table 1 - Tree and forest level stand structural attributes of the two oak stands.

Stand	Np (trees plot^{-1})	d (cm)	Hd (m)	Ht (m)	Hs (m)
Q. pyrenaica	77.25 ± 5.38	15.33 ± 0.44	12.55 ± 0.77	11.44 ± 0.16	7.68 ± 0.74
Q. rotundifolia	48.25 ± 7.80	12.43 ± 1.51	7.09 ± 1.67	6.22 ± 0.89	2.45 ± 0.44

Note: NP: trees per plot; d: mean diameter at breast height, Hd: mean dominant height, Ht: mean total height, Hs: mean stem height.

At the stand scale, *Q. pyrenaica* again exceeds *Q. rotundifolia* in all key metrics, recording higher tree density (1,545 ± 107.55 vs. 965 ± 161.97 plants ha-^{1}), basal area

(8.69 ± 0.67 m² ha⁻¹ vs. 6.10 ± 1.28 m² ha⁻¹), tree volume (171.64 ± 6.40 m³ ha⁻¹ vs. 34.13 ± 2.99 m³ ha⁻¹) and tree AGB (102.98 ± 3.82 Mg ha⁻¹ vs. 30.72 ± 3.10 Mg ha⁻¹) (Table 2).

Table 2 - Forest stand structural attributes and biomass.

Stand	N (trees ha^{-1})	G (m² ha^{-1})	Volume (m³ ha^{-1})	AGBtree (Mg ha^{-1})
Q. pyrenaica	1,545 ± 107.55	8.69 ± 0.67	171.64 ± 6.40	102.98 ± 3.82
Q. rotundifolia	965 ± 161.97	6.10 ± 1.28	34.13 ± 2.99	30.72 ± 3.10

Note: N: tree density; G: basal area; volume derived from Eqs. 1-2; AGBtree calculated using species-specific wood density.

These patterns are consistent with previous studies showing that *Q. pyrenaica* forests on favourable sites tend to accumulate a greater amount of biomass (Castaño-Santamaría et al., 2013; Carvalho and Parresol, 2003). In contrast, the lower biomass of *Q. rotundifolia* reflects its physiological adaptation to Mediterranean constraints of water and nutrients affecting the tree distribution in the stand (Macedo et al., 2018), which are further accentuated in this region by the presence of ultrabasic soils. Figure 14 shows the tree biomass values (AGBtree) in both stands.

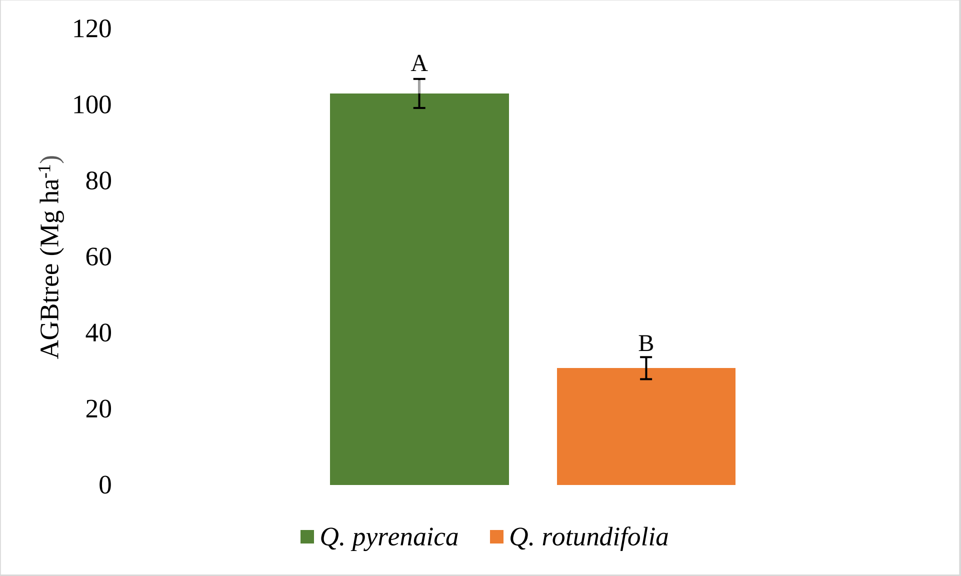

Figure 14 - Average AGBtree for each stand, with standard deviations represented by error bars. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

- Shrub layer (AGBshrub)

Visually, the two stands exhibit clear structural differences in the shrub layer. However, statistical analysis of the mean shrub density revealed no significant difference between them (*Q. pyrenaica*: 659.70 ± 978.40 plants ha^{-1} vs. *Q. rotundifolia*: 1,129.51 ± 1,251.72 plants ha^{-1}). This result can be attributed to the high variability observed within and between plots, as reflected by the large standard deviations, likely due to the heterogeneous distribution of shrub species across the sampled area. The patchy occurrence of dominant shrubs and localised patterns of regeneration may have masked overall differences in mean density, despite evident visual contrasts in structure and cover.

Shrub above-ground biomass also differed markedly between the two stands, with *Q. rotundifolia* reaching 857.47 kg ha^{-1} and *Q. pyrenaica* only 106.30 kg ha^{-1}**.** However, statistical analysis indicated no significant difference between the means. Although not statistically significant, these structural contrasts remain evident when analysing the detailed shrub layer attributes of each stand (Tables 3 and 4), which highlight distinct spatial organisation and biomass allocation patterns.

Table 3 - Structural and biomass attributes of shrub species recorded in *Q. pyrenaica*.

Shrub species	Class height (m)	Density (plants ha^{-1})	Biovolume (m³) - BVMm	Biomass (kg ha^{-1}) BMm
Rosa spp.	0 - 0.5	63.16 ± 29.77	0.01 ± 0.00	0.13
	0.51 - 1.0	231.56 ± 109.16	0.01 ± 0.00	0.57
	1.01 - 1.5	84.2 ± 39.69	0.05 ± 0.02	0.15
R. ulmifolius	0 - 0.5	3,648.26 ± 1270.44	0.00 ± 0.00	12.41
	0.51 - 1.0	2,345.45 ± 816.76	0.03 ± 0.03	2.88
	1.01 - 1.5	223.47 ± 77.48	0.04 ± 0.00	0.37
	1.51 - 2.0	223.47 ± 77.48	0.99 ± 0.24	0.27
C. scoparius	0 - 0.5	111.65 ± 54.6	0.02 ± 0.00	4.13
	0.51 - 1.0	379.49 ± 185.6	0.06 ± 0.03	3.95
	1.01 - 1.5	223.3 ± 109.21	0.34 ± 0.12	1.22
	1.51 - 2.0	223.3 ± 109.21	0.99 ± 0.24	1.58
Erica spp.	0 - 0.5	547.36 ± 182.54	0.20 ± 0.04	2.21
	0.51 - 1.0	210.56 ± 70.19	0.64 ± 0.12	0.99
G. falcata	0 - 0.5	447.78 ± 191.69	0.00 ± 0.00	64.80
	0.51 - 1.0	1169.33 ± 500.58	0.21 ± 0.03	7.72
	1.01 - 1.5	422.89 ± 181.04	0.60 ± 0.09	2.91

In the *Q. pyrenaica* stand, *R. ulmifolius* dominates the lowest height class (0-0.5 m), displaying high shrub density but relatively low above-ground biomass. In contrast, *C. scoparius* is more prevalent in the upper shrub strata, contributing more substantially to biomass at those levels.

Erica spp. were well represented in the lower shrub strata, with their biomass (2.21 kg ha^{-1}) recorded in the 0-0.5 m height class, indicating their dominance near the soil surface and their potential role in early-stage ecosystem processes. *G. falcata* showed a wider vertical distribution, with a density maximum of 1,169.33 ± 500.58 individuals ha^{-1} in the 0.51-1.0 m class, and contributed substantially to the shrub layer, with a notable above-ground biomass of 7.72 kg ha^{-1}.

In contrast, *Rosa spp.* were generally scarce and contributed minimally to vegetation biomass. Their highest recorded density (231.56 ± 109.16 individuals ha^{-1}) and biomass (0.57 kg ha^{-1}) occurred in the 0.51–1.0 m height class, these values remained low compared to the dominant shrub species.

Table 4 - Structural and biomass attributes of shrub species recorded in *Q. rotundifolia*.

Shrub species	Class height (m)	Density (plants/ha)	Biovolume (m³) BVMm	Biomass (kg ha^{-1}) BMm
P. tridentatum	0 - 0.5	4,595.94 ± 1,722.88	0.05 ± 0.02	305.30
	0.51 - 1.0	804.06 ± 301.42	0.27 ± 0.02	69.37
Cistus spp.	0 - 0.5	1,920.05 ± 572.02	0.01 ± 0.00	95.77
	0.51 - 1.0	159.95 ± 47.65	0.03 ± 0.01	4.54
Erica spp.	0 - 0.5	1,590.41 ± 710.65	0.04 ± 0.01	105.64
	0.51 - 1.0	662.74 ± 296.14	0.13 ± 0.06	14.71
	1.01 - 1.5	574.24 ± 256.59	0.96 ± 0.16	21.93
	1.51 - 2.0	132.61 ± 59.25	2.53 ± 1.09	8.42
Cytisus multiflorus (L'Hér.) Sweet	0 - 0.5	226.71 ± 105.20	0.03 ± 0.00	20.04
	0.51 - 1.0	793.29 ± 368.09	0.13 ± 0.04	28.43
	1.01 - 1.5	283.29 ± 131.45	0.57 ± 0.27	23.02
	1.51 - 2.0	56.71 ± 26.31	1.29 ± 0.65	1.13
G. falcata	0 - 0.5	2,995.27 ± 761.69	0.01 ± 0.00	76.19
	0.51 - 1.0	1,996.85 ± 507.80	0.04 ± 0.02	60.80
L. pedunculata	0 - 0.5	1,176.19 ± 514.00	0.01 ± 0.01	20.04
	0.51 - 1.0	103.81 ± 45.36	0.03 ± 0.01	2.15

In the *Q. rotundifolia* stand, *P. tridentatum* emerged as the most abundant shrub specie, especially in the 0-0.5 m height class, where it exhibited the highest density (4,595.94 ± 1,722.88 individuals ha^{-1}) and a notable contribution to above-ground biomass (305.3 kg ha^{-1}). This distribution pattern persisted in the 0.51-1.0 m layer, with significant values for density and biomass, further confirms its central role in shaping the shrub stratum. This dominance is consistent with ecological trends observed in Mediterranean *Q. rotundifolia* woodlands, where species such as *P. tridentatum* flourish under open canopies and are well adapted to the specific edaphic and climatic conditions (Canas et al. 2021).

G. falcata was the second most abundant in the lowest stratum (2,995.27 ± 761.69 individuals ha^{-1}), with moderate biomass, and remained well represented in the 0.51-1.0 m class, contributing to the structural complexity of the understorey. *Cistus* spp. displayed a more restricted vertical distribution, with their peak density (1,920.05 ± 572.02 plants ha^{-1}) and biomass (95.77 kg ha^{-1}) concentrated in the 0-0.5 m height class. *L. pedunculata*, although present in the lower stratum (1,176.19 ± 546.41 plants/ha), made a minor contribution to above-ground biomass, indicating a less significant structural role compared to *C. multiflorus* and *G. falcata*. Despite this limited vertical range, *Cistus spp.* can contribute significantly to carbon storage in the ecosystem due to their relatively high biomass and carbon content in above and below-ground tissues (Carrión-Prieto et al., 2017; Pehlivan and Özturna, 2023).

By contrast, *Erica* spp., exhibited a broader vertical distribution, occurring across all height classes and maintaining substantial biomass throughout. They showed considerable values in the 0-0.5 m class (1,590.41 ± 710.65 individuals ha^{-1}, 105.64 kg ha-1 biomass), with even higher biomass recorded in the 1.01-1.5 m class (21.93 Mg ha^{-1}). *C. multiflorus* also exhibited a wide vertical distribution, contributing substantially to shrub density and biomass across strata, particularly in the 0.51-1.0 m class (793.29 ± 368.09 individuals ha^{-1}, 28.43 kg ha^{-1} of biomass). It maintained a presence in both lower and upper layers, underlining its ecological versatility.

The lower biomass observed in the understory of *Q. pyrenaica* reflects its sparser and less developed shrub community, while the *Q. rotundifolia* stand supports a denser and more structurally complex understorey, dominated by woody species with greater potential for biomass accumulation. These patterns are consistent with findings from remote sensing and field-based studies carried out in southern Spain and Portugal, which indicate that *Q. rotundifolia* stands generally support higher above-ground biomass due to their denser canopy and more productive shrub layer, as estimated through vegetation indices and canopy cover data (Andivia, 2017; Macedo et al., 2018). Differences in shrub structure between *Q. pyrenaica* and *Q. rotundifolia* stands may also be related to variations in canopy density, light availability, and soil moisture, all of which influence understory vegetation patterns and their ecological roles (Plieninger et al., 2010; Fuentes et al., 2019).

Beyond structural differences, the ecological roles of dominant shrub species further influence ecosystem functioning in Mediterranean oak systems. Research indicates that the presence and abundance of specific shrubs, such as *C. scoparius*, play a key role in woodland–grassland mosaics by providing microhabitats that help protect seedlings from abiotic stress and herbivory (Costa et al., 2017; Díaz-Hernández et al., 2021). This species not only contributes to overall shrub biomass, but also facilitates the survival and emergence of oak seedlings improving soil organic matter and offering protective cover (Costa et al., 2017; Díaz-Hernández et al., 2021). Similarly, the high biomass and cover contributed by *P. tridentatum* in *Q. rotundifolia* stands shape understorey structure, influence nutrient cycling and microhabitat availability, and may facilitate oak regeneration by shielding seedlings from grazing and environmental stress (Nunes et al., 2005; Canas et al., 2021). The capacity of *Erica spp.* to persist and accumulate biomass across multiple vertical strata highlighting their ecological importance in Mediterranean shrubland ecosystems, where they contribute to carbon sequestration and enhance structural complexity (Fonseca et al., 2012; Carrión-Prieto et al., 2017;). Research in Mediterranean oak systems has also demonstrated that certain shrub species, particularly *Erica* and *G. falcata*, are more effective in facilitating the establishment of oak seedlings due to their physical structure and lower palatability to herbivores, making them key components for ecosystem management and restoration (Perea et al., 2016).

However, intra- and inter-stand variability, along with factors such as species composition, site conditions, and management history, can influence the statistical significance of observed differences (Vinué-Visús et al., 2022). The structural diversity and composition of the shrub layer are vital to ecosystem functioning, influencing not only plant recruitment but also carbon sequestration and habitat complexity (Plieninger et al., 2010; Van Den Bor et al., 2023). These findings highlight the importance of integrating shrub layer structure and composition into evaluations of ecosystem functioning and carbon dynamics in Mediterranean oak woodlands (Plieninger et al., 2010; Andivia, 2017; Fuentes et al., 2019; Vinué-Visús et al., 2022).

- AGBherb

A clear contrast in herbaceous layer biomass (AGBherb) was observed between the two stands, with *Q. pyrenaica* presenting significantly higher herbaceous biomass (1,713.72 ± 0.23 kg ha^{-1}) than *Q. rotundifolia* (808.16 ± 0.10 kg ha^{-1}) (Figure 14).

These findings are consistent with previous studies in Mediterranean oak woodlands, which have shown that evergreen oak stands (such as *Q. rotundifolia*) generally support less herbaceous biomass due to increased shading and competition with woody vegetation, while deciduous stands (such as *Q. pyrenaica*) promote a more abundant herbaceous layer (Baraza et al., 2004).

This reflects differences in canopy structure and light availability, as *Q. pyrenaica* stands are typically leafless until May. As a result, the herbaceous layer benefits from higher light availability during the early spring, when its vegetative cycle is already in progress. This prolonged exposure to light promotes greater growth and productivity of herbaceous species (Baraza et al., 2004).

In contrast, the evergreen and more developed shrub layer in *Q. rotundifolia* stands reduce light availability in the understorey, constraining the accumulation of herbaceous biomass (Macedo et al., 2018). Furthermore, the process of ecological succession suggests that as the shrub layer expands, there is a reduction in the herbaceous layer (Díaz-Villa et al., 2003; Castro et al., 2010; de Souza et al., 2025).

The higher herbaceous biomass observed in *Q. pyrenaica* stands may enhance nutrient cycling and provide important resources for grazing and wildlife, highlighting the ecological importance of stand structure in influencing understorey dynamics and ecosystem processes (Baraza et al., 2004). Figure 15 below presents the AGBshrub and ABGherb values for the two stands.

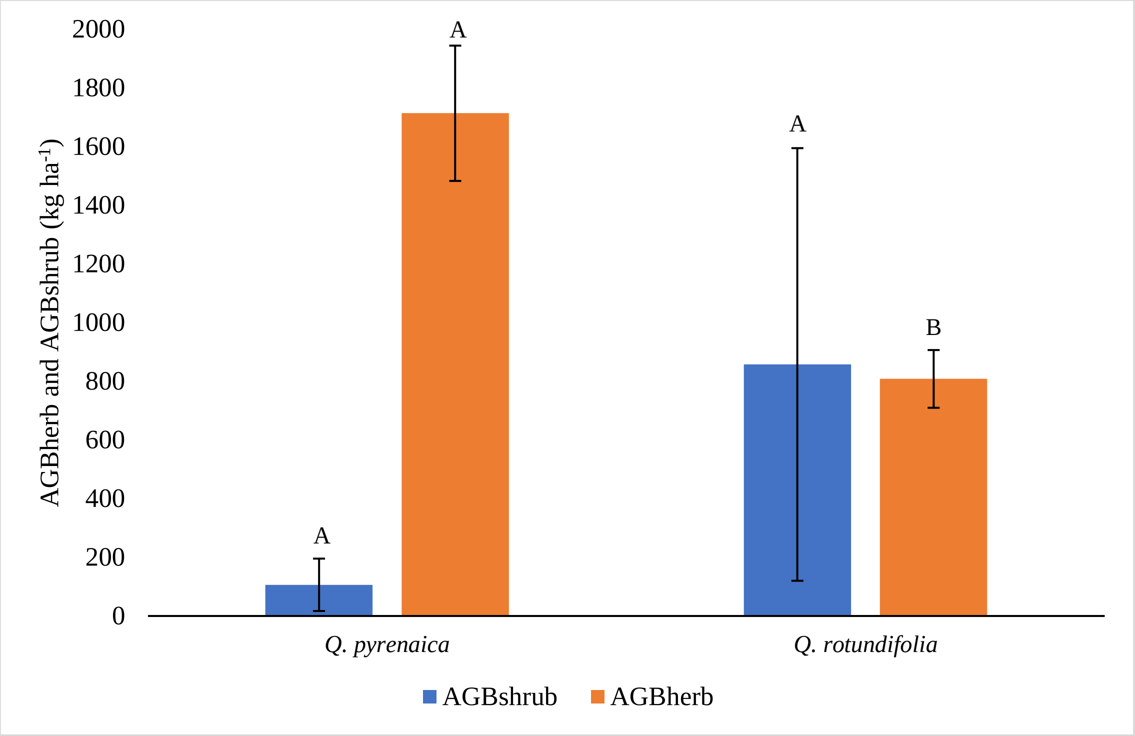

Figure 15 - AGBshrub and AGBherb for each stand, with standard deviations represented by error bars. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

4.2 Total above-ground biomass

Total above-ground biomass (AGB) was considerably higher in the *Q. pyrenaica* stand (104.80 ± 3.83 Mg ha^{-1}) compared to the *Q. rotundifolia* stand (32.39 ± 2.99 Mg ha^{-1}), as shown in Figure 16. This substantial difference reflects the larger size, density and evergreen woody material (trunks and branches) typically found in *Q. pyrenaica* forests, where evergreen components can represent up to 95% of the total above-ground biomass (Montero et al., 2004).

In contrast, although *Q. rotundifolia* stands are capable of supporting significant shrub and tree biomass, they generally have lower total above-ground biomass. Reported values vary considerably, but often remain below those recorded in *Q. pyrenaica* stands, depending on factors such as stand structure, age and management practices (Ispikoudis et al., 2024).

These differences are largely influenced by species-specific growth strategies, site conditions, and the relative dominance of woody versus herbaceous layers. While *Q. pyrenaica* stands tend to support taller and denser trees, *Q. rotundifolia* stands are generally more open and shrub-dominated (Macedo et al., 2018). Such variation in total AGB has significant implications for carbon storage and overall ecosystem functioning in Mediterranean oak woodlands (Ispikoudis et al., 2024).

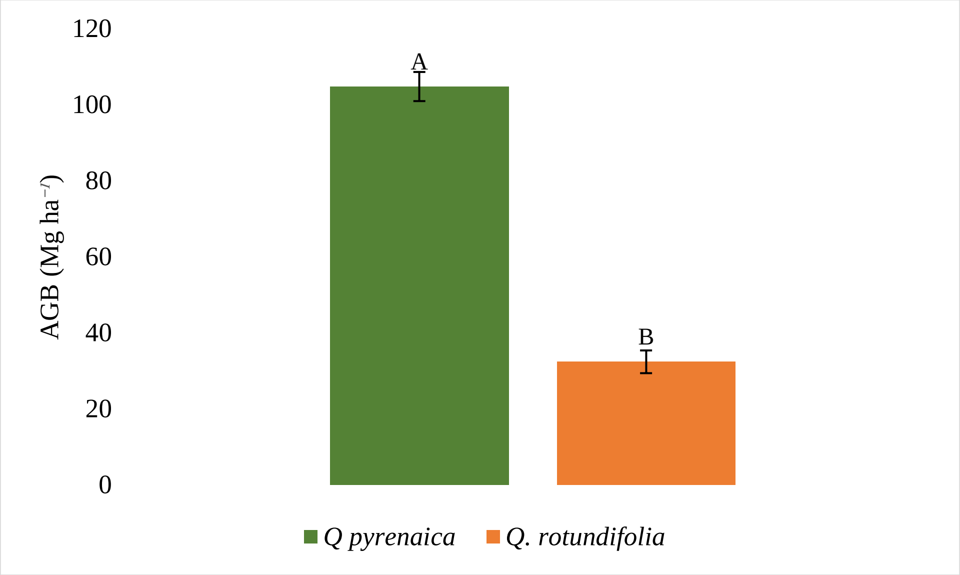

Figure 16 - Total above-ground biomass (Mg ha^{-1}) recorded in *Q. pyrenaica* and *Q. rotundifolia* stands. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

4.3 Below-ground biomass (BGB)

Below-ground biomass was significantly higher in the *Q. pyrenaica* stand (31.44 ± 1.15 Mg ha-^{1}) than in the *Q. rotundifolia* stand (9.72 ± 0.90 Mg ha-^{1}), reflecting differences in root development and carbon allocation strategies between these oak species (Figure 17). Studies have shown that native *Quercus* forests, particularly those dominated by *Q. pyrenaica*, tend to allocate a greater proportion of carbon below ground, within roots and soil, compared to other forest types, with up to 60% of total carbon stored underground (Van Den Bor et al., 2023)

This substantial root biomass enhances long-term carbon storage, as below-ground carbon is more resistant to disturbance, while also contributing to ecosystem resilience and nutrient cycling. The variation in BGB between *Q. pyrenaica* and *Q. rotundifolia* may be influenced by species-specific adaptations, environmental conditions and phylogenetic constraints, factors that shape how each species invests in root structures and responds to environmental pressures (Perkovich and Ward., 2021).

In summary, the higher BGB observed in *Q. pyrenaica* stands supports the view of Van Den Bor et al., (2023) regarding the crucial role of below-ground biomass in long-term carbon sequestration and ecosystem stability in Mediterranean woodlands.

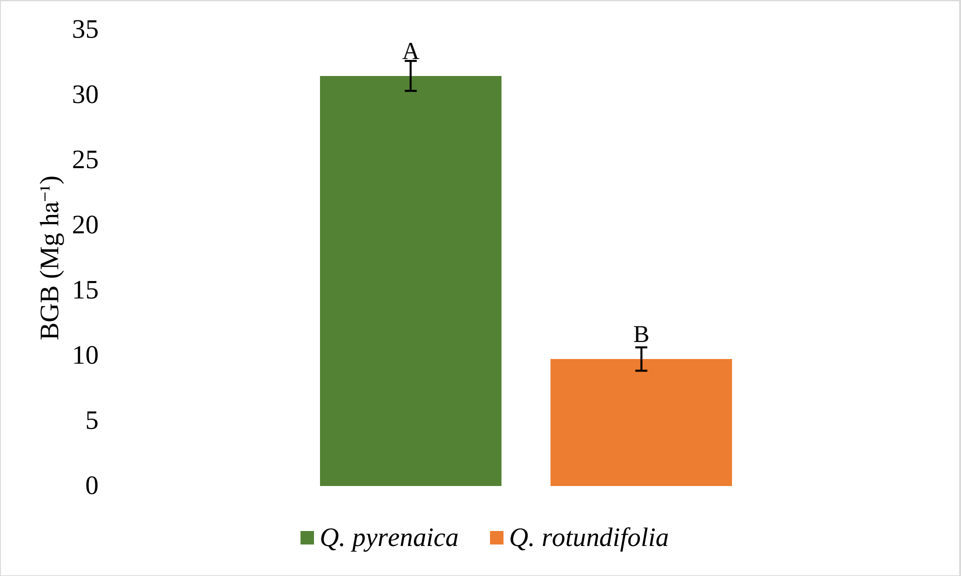

Figure 17 - Below-ground biomass (Mg ha^{-1}) recorded in *Q. pyrenaica* and *Q. rotundifolia* stands, with standard deviations represented by error bars. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

4.4 Litter

Litter accumulation was similar between the *Q. pyrenaica* and *Q. rotundifolia* stands, with mean values of 2.82 ± 0.20 Mg ha^{-1} and 3.28 ± 0.27 Mg ha^{-1}, respectively, and no statistically significant difference observed (Figure 18). This finding aligns with previous research indicating that, although deciduous (*Q. pyrenaica*) and evergreen (*Q. rotundifolia* or related species) oak forests may differ in nutrient content and decomposition rates, the overall quantity of litter accumulated on the forest floor often does not vary markedly under comparable site conditions (Fernández-Alonso et al., 2022).

Nevertheless, the composition and quality of the litter can have important implications for nutrient cycling and microbial community structure. For instance, *Q. pyrenaica* litter is frequently richer in phosphorus, magnesium, iron, and manganese, whereas evergreen oaks tend to produce litter with higher nitrogen and calcium concentrations (Rapp et al., 1999).

These chemical differences in litter, combined with variations in decomposition dynamics, can influence soil fertility and ecosystem functioning, even when total litter biomass remains similar (Fernández-Alonso et al., 2022).

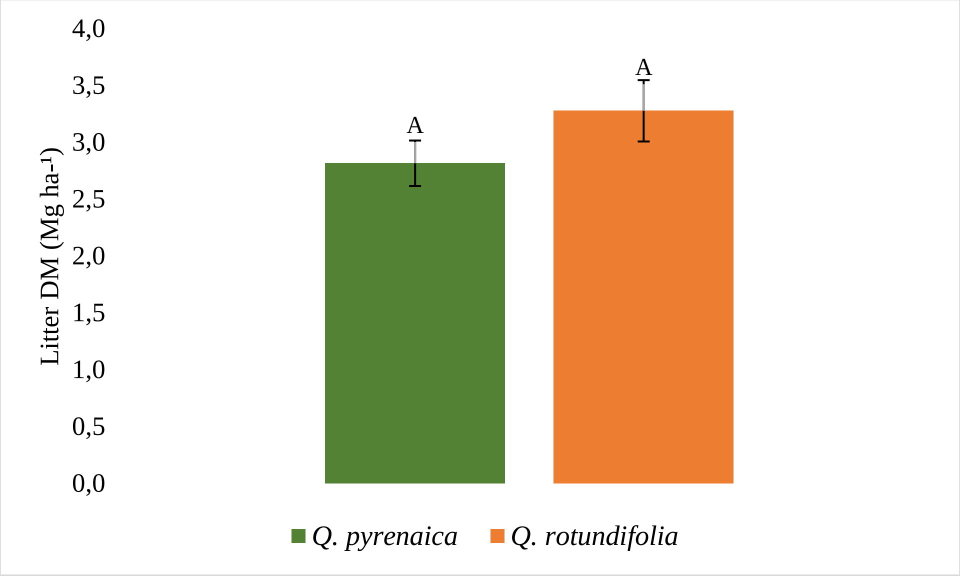

Figure 18 - Average litter dry matter (DM) for each stand, with standard deviations represented by error bars. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

4.5 Soil organic carbon (SOC)

In *Q. pyrenaica* stands, soil organic carbon (SOC) values at the 0–10 cm depth averaged 48.11 ± 14.53 Mg/ha, increasing to 80.45 ± 22.22 Mg/ha at the 10–30 cm layer. In *Q. rotundifolia*, SOC values were lower, with 32.78 ± 5.91 Mg/ha in the upper layer and 49.50 ± 16.89 Mg/ha in the deeper layer. Although *Q. pyrenaica* exhibited higher SOC stocks at both depths, no statistically significant differences were observed between the two forest types.

For both species, the 10–30 cm depth stored more carbon than the 0–10 cm layer, suggesting greater accumulation in subsurface horizons. These results indicate a higher potential of *Q. pyrenaica* to accumulate and retain carbon within the soil profile, particularly in deeper layers. Also, in the deeper soil layer (10–30 cm), the SOC was higher, which can be attributed to the greater thickness of this horizon, twice that of the superficial layer (0–10 cm). Furthermore, the larger standard deviation recorded at this depth indicates a higher variability in SOC distribution, suggesting that carbon is heterogeneously distributed within this deeper soil stratum (Figure 19).

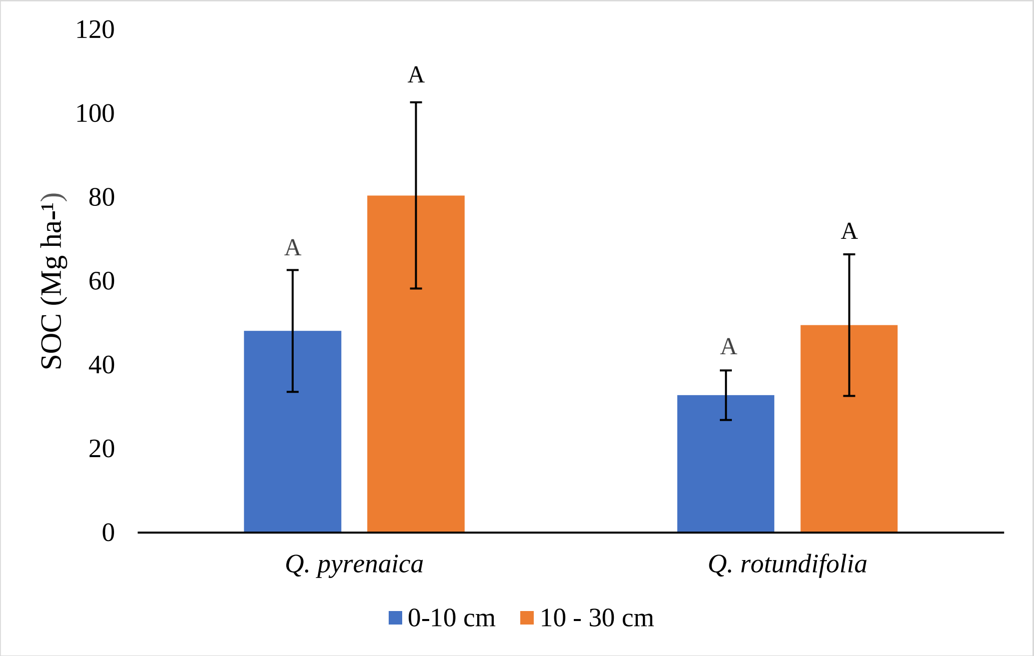

Figure 19 - Average SOC for each stand, with standard deviations represented by error bars. Means were compared using Tukey’s test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

The higher SOC content observed in *Q. pyrenaica* suggests relatively more efficient mechanisms for transferring carbon from the litter and roots to deeper horizons, along with a greater capacity for retention throughout the soil profile, compared with evergreen oak species such as *Q. rotundifolia*, which typically exhibit lower SOC levels under similar environmental conditions (Rapp et al., 1999; Fernández-Getino et al., 2018).

Previous studies have demonstrated that in *Q. pyrenaica* ecosystems, factors such as soil type, slope, and degree of humification significantly influence SOC accumulation, with gentler slopes and greater stabilization of humic substances being particularly conducive to higher levels of carbon storage (Fernández-Getino et al., 2018).

When compared with other Mediterranean and temperate ecosystems, the SOC values quantified in this study are lower than those reported for silvopastoral systems in northwest Spain, where Fernández-Núñez et al., (2010) recorded 130.7 Mg C ha^{-1} (up to 25 cm depth) in *Pinus radiata* stands and 143.4 Mg C ha^{-1} in *Betula alba* ten years after establishment. These differences largely reflect the greater structural complexity and significantly higher productivity of those vegetation strata, as well as more favourable climatic conditions that promote rapid biomass accumulation. Similarly, Nunes et al., (2010) reported values ranging from 90.2 to 123.9 Mg C ha^{-1} in the top 30 cm of soil under *Pinus pinaster* in north-western Portugal, reinforcing the observation that fast-growing species tend to be associated with higher biomass and carbon inputs.

These findings emphasize the importance of stand species composition and site-specific factors in determining the soil carbon sequestration potential in Mediterranean ecosystems oak woodlands.

4.6 Carbon total storage (TC)

The results revealed marked differences in carbon stocks between *Q. pyrenaica* and *Q. rotundifolia* habitats. *Q. pyrenaica* stands exhibited significantly higher total AGB, reaching 49.26 Mg ha^{-1}, while *Q. rotundifolia* recorded 15.22 Mg ha^{-1}. Similarly, below-ground biomass (BGB) carbon was higher in *Q. pyrenaica* (14.78 Mg ha^{-1}) than in *Q. rotundifolia* (4.57 Mg ha-^{1}). In contrast, litter carbon stocks were relatively similar between the stands, with *Q. pyrenaica* presenting 1.32 Mg ha^{-1} and *Q. rotundifolia* 1.54 Mg ha^{-1}. With regard to total soil organic carbon (SOC), *Q. pyrenaica* again demonstrated higher values (128.56 Mg ha-^{1}) compared to *Q. rotundifolia* (82.28 Mg ha-^{1}) (Table 5).

Table 5 - Distribution of carbon among components of each stand.

	C in AGB (Mg ha^{-1})	C in BGB (Mg ha^{-1})	Clitter (Mg ha^{-1})	SOC total (Mg ha^{-1})
Q. pyrenaica	49.26 ± 1.80	14.78 ± 0.54	1.32 ± 0.09	128.56 ± 26.55
Q. rotundifolia	15.22 ± 1.41	4.57 ± 0.42	1.54 ± 0.13	82.28 ± 17.89

The distribution of carbon stocks between biomass (AGB, BGB) + Litter (C_{BL}) and SOC also highlights substantial differences between the two species. *Q. pyrenaica* exhibited higher carbon values in both components: 65.4 Mg/ha in C_{BL} and 128.6 Mg ha^{-1} in SOC.

In contrast, *Q. rotundifolia* presented lower values in both compartments, with 21.3 Mg/ha in C_{BL} and 74.6 Mg ha⁻¹ in SOC, as shown in Figure 20. In both stands, soil carbon accounts for approximately 70% of the total carbon stock.

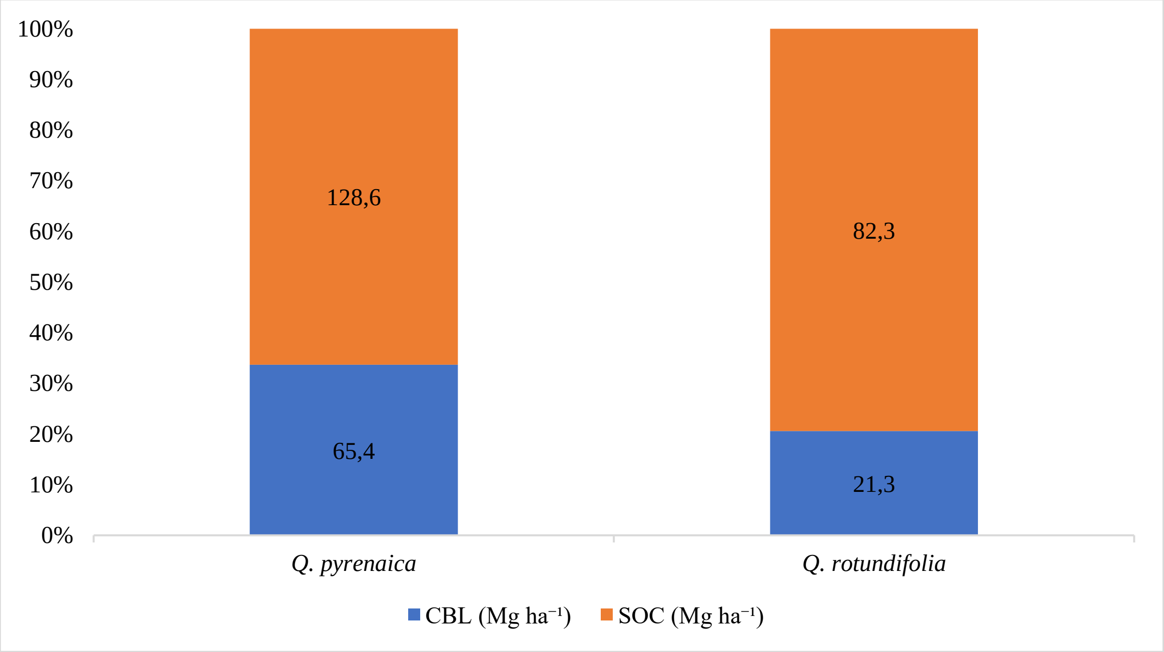

Figure 20 - C_{BL} and SOC values in the two stands and their respective contributions to total carbon.

The considerable difference in total carbon (TC) stock between *Q. pyrenaica* (193.92 ± 26.63Mg ha⁻¹) and *Q. rotundifolia* (103.61 ± 17.95 Mg ha⁻¹) reflects the greater above- and below-ground biomass and higher soil organic carbon content associated with *Q. pyrenaica* stands (Figure 21).

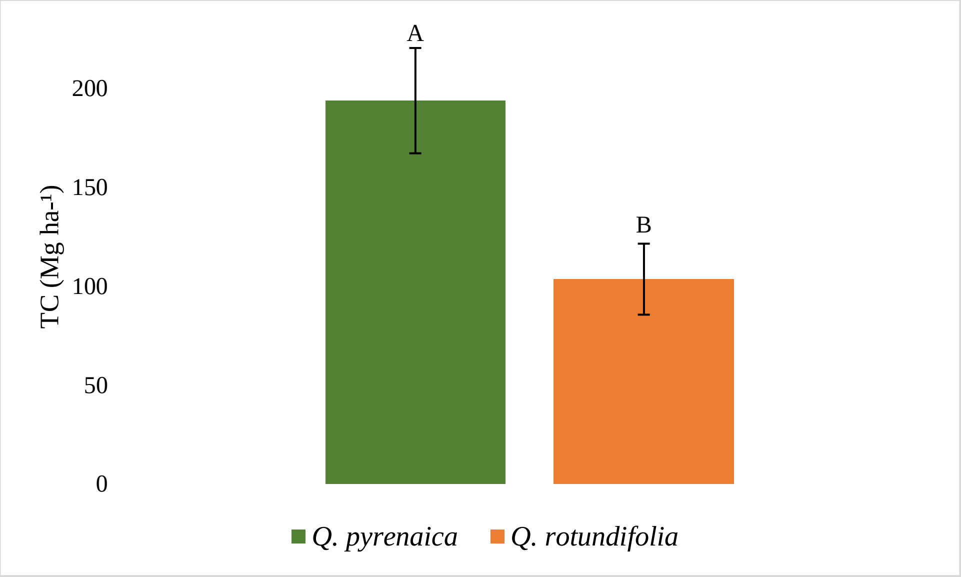

Figure 21 - Total carbon (TC) in each stand. Means were compared using Tukey's test at a 5% significance level. Different letters indicate statistically significant differences between the forest stands.

Total carbon (TC) stocks show a pronounced difference between the two stands: *Q. pyrenaica* accumulated approximately 87 % of the value recorded in the *Q. rotundifolia*. This disparity is mainly due to the greater live biomass in *Q. pyrenaica* (64.0 Mg C ha^{-1} in AGB + BGB, compared to 19.8 Mg C ha^{-1} in *Q. rotundifolia*) and the significant edaphic pool (128.6 Mg C ha^{-1} vs. 82.3 Mg Mg C ha^{-1}). The higher aerial and root biomass, combined with the marcescent leaf phenology of *Q. pyrenaica*, characterized by the seasonal retention and abrupt winter shedding of senescent foliage, results in substantial litter deposition. These dynamics contribute to a markedly greater carbon sequestration capacity in *Q. pyrenaica* stands compared to *Q. rotundifolia* (Ruiz-Peinado et al., 2012).

In contrast, the *Q. rotundifolia* stand shows a higher proportion of carbon stored in the litter layer (1,49% vs. 0,68%) and in the SOC (79,41% vs. 66,30%). This pattern reflects the evergreen phenology of the species, which maintains a stable canopy throughout the year and continuously recycles its foliage. As a result, litterfall occurs gradually, promoting a steady input of organic matter to the forest floor. This continuous deposition may enhance nutrient cycling dynamics and contribute to the formation and stabilization of soil organic carbon (SOC) over time.

5. CONCLUSION

This study provides a comprehensive quantification of carbon stocks across multiple ecosystem compartments in an integrated manner, namely above-ground biomass (ABG), below-ground biomass (BGB), litter, and soil (SOC), in two protected Mediterranean forest habitats within Montesinho Natural Park: *Q. pyrenaica* and *Q. rotundifolia* woodlands. The results revealed a markedly higher total carbon stock in *Q. pyrenaica* stands, with approximately 193.9 Mg C ha^{-1}, compared to 103.6 Mg C ha^{-1} in *Q. rotundifolia* stands. Although stand age was not quantified precisely, both woodlands can be characterised as stand with adult age but still in a phase of active biomass accumulation, and have not yet reached a late-successional or climax stage.

This difference was primarily attributed to greater live biomass and higher soil carbon accumulation in *Q. pyrenaica,* as opposed to the proportionally higher litter and soil carbon stocks observed in *Q. rotundifolia*. These findings highlight the high carbon sequestration potential of Mediterranean ecosystems and reinforce the importance of including all carbon pools, trees, shrubs, herbaceous layer, litter, and especially soil, for a comprehensive assessment of carbon forest dynamics. In particular, soil constitutes a fundamental component of ecosystem carbon storage, acting as a long-term and stable reservoir that often accounts for the largest share of total carbon.

Beyond advancing scientific understanding of carbon dynamics in Mediterranean woodlands, this research provides robust parameters for modelling emission mitigation scenarios. The superior carbon storage capacity of *Q. pyrenaica* suggests that low-intensity silvicultural practices and the protection of key stands could enhance atmospheric CO_2 sequestration. In contrast, the performance of *Q. rotundifolia*, with a higher proportion of carbon stored in the litter and soil, underlines the key role of these compartments in ensuring long-term below-ground carbon stability. This highlights the need for conservation strategies focused on preserving litter layer quality and continuity.

Finally, the carbon stock values reported in this study position Montesinho Natural Park as a potential site for future inclusion in carbon credit schemes. The carbon stock values identified highlight the potential for local producers in the park to enhance their income by participating in emerging carbon credit markets. The high per-hectare carbon density, particularly in *Q. pyrenaica* stands, along with the relative permanence of below-ground carbon pools, meets essential criteria such as additionality and permanence, which are required by carbon offset protocols. Therefore, beyond supporting biodiversity conservation and ecosystem service provision, the integration of these habitats into voluntary or regulated carbon markets could generate sustainable financial resources for adaptive forest management, long-term monitoring, and the expansion of protection efforts in these emblematic Mediterranean forests.

Looking ahead, the growing recognition of soil carbon within voluntary carbon markets represents a promising opportunity to expand the scope of creditable carbon pools. As soil often constitutes the largest and most stable carbon reservoir, its formal inclusion in future carbon offset mechanisms could significantly enhance the climate mitigation potential of these native ecosystems.

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7. ATTACHMENTS

The following data were provided by Júlio de Souza (pers. comm., 2025) and were used to calculate the dry mass per hectare (DM ha-1) for each shrub morphotype.

Table 6 – Mean biovolume of schrub species in both stands.

Q. rotundifolia		*Q. pyrenaica*	
Shrub species	Mean biovolume (m^3 ha^{-1})	Shrub species	Mean biovolume (m^3 ha^{-1})
P. tridentatum	476,20	*Rosa spp.*	5,02
Cistus spp.	64,45	*R. ulmifolius*	89,74
Erica spp.	360,65	*C. scoparius*	135,72
C. multiflorus	105,95	*Erica spp.*	14,85
G. falcata	203,67	*G. falcata*	139,75
L. pedunculata	26,60	-	-