EVALUATING THE EFFECTIVENESS OF POSTFIRE RESTORATION IN PORTUGAL WITH A FOCUS ON DECIDUOUS OAK FORESTS

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EFFICIENCY OF PUBLIC ENVIRONMENTAL MEASURES IN POST-FIRE RESTORATION OF OAK WOODLANDS IN PORTUGAL

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Nota prévia

De acordo com Regulamento Geral dos Ciclos de Estudos Conducentes ao Grau de Doutor no Instituto Superior de Agronomia (Despacho nº 9146/2017), e tendo por base o seu Artigo 2º, a elaboração de uma tese original é substituída pela compilação, devidamente enquadrada por uma introdução, revisão bibliográfica, discussão e conclusões gerais, de um conjunto coerente e relevante de trabalhos de investigação. Os trabalhos de investigação referidos estão atualmente publicados ou submetidos para publicação em revistas internacionais indexadas e com arbitragem científica. O candidato contribuiu de forma ativa em todas várias etapas de conceção dos mesmos, nomeadamente, conceção, análise e discussão dos resultados, assim como na estruturação e escrita dos documentos publicados.

Lista de artigos

Capítulo 2 – Lopes LF, Fernandes PM, Rego FC, Acácio V (2022) Public funding constrains effective postfire emergency restoration in Portugal. *Restoration Ecology*, 31. doi: 10.1111/rec.13769

Capítulo 3 - Lopes LF, Dias FS, Fernandes PM, Acácio V (2024) A remote sensing assessment of oak forest recovery after postfire restoration. *European Journal of Forest Research*, 143:1001–1014. https://doi.org/10.1007/s10342-024-01667-z

Capítulo 4 – Lopes LF, Santos E, Nunes L, Fernandes PM, Acácio V (*submitted*) Postfire oak afforestation in Portugal reveals trade-offs in the supply of ecosystem services. *Ecosystem Services.*

"The mountains are calling, and I must go." - John Muir

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Um bem-haja a todos.

Evaluating the effectiveness of postfire restoration in Portugal with a focus on deciduous oak forests

Abstract

Fire has long been a natural element of forest dynamics. However, changes in fire regimes in recent decades in Europe (e.g., more frequent and severe fires) have led to negative ecological, social, and economic impacts. Portugal, situated in Mediterranean Europe, exemplifies this scenario as a region highly susceptible to wildfires and presently undergoing shifts in fire regimes. Here, deciduous oak forest ecosystems, which have an important role in biodiversity conservation, soil and water provision, and climate regulation, are today reduced to small, fragmented areas and present a high fire incidence. Following fire events, postfire management strategies can be employed to reduce the negative fire effects, although such actions may also have consequences on the recovery of soil and vegetation. As such, the objective of this thesis was to examine the effectiveness of various public environmental measures employed in postfire restoration in Portugal over recent decades, with a focus on deciduous oak forests.

In this thesis, I identified three main factors within the public funding process that have been limiting the efficacy of postfire emergency stabilization in Portugal: the financing model, an oversimplified approach to eligible interventions, and slow decision-making and execution. I also observed a small positive impact of postfire interventions on the recovery of deciduous oak forests. Conversely, I found a negative impact of postfire drought events and frequent fires on this recovery. Finally, the quantification of multiple ecosystem services highlighted the existence of trade-offs due to management choices, indicating the necessity to outline long-term objectives for the burned area, since different postfire management approaches may favour distinct ecosystem services. Further studies are required to enhance the effectiveness (and consequently the efficacy) of the multiple stages of postfire restoration in Portugal.

Keywords:

Postfire restoration; Portugal; Vegetation recovery; Wildfires; Deciduous oaks

Avaliação da eficácia do restauro pós-fogo em Portugal com um foco nas florestas de carvalhos

Resumo

O fogo é, desde há muito, um elemento natural da dinâmica florestal. No entanto, as alterações nos regimes de fogo ocorridas nas últimas décadas na Europa (por exemplo, incêndios mais frequentes e mais severos) conduziram a impactos ecológicos, sociais e económicos negativos. Portugal, localizado na Europa Mediterrânica, exemplifica este cenário como uma região altamente suscetível a incêndios florestais e atualmente com alterações nos regimes de fogo. Em Portugal, os ecossistemas de carvalhais caducifólios, com um papel importante na conservação da biodiversidade, na qualidade do solo e da água e na regulação do clima, estão atualmente reduzidos a pequenas áreas fragmentadas e apresentam uma elevada incidência de fogo. Após a ocorrência de um incêndio, a implementação de técnicas de restauro pós-fogo permite mitigar os seus efeitos negativos. Porém, a implementação destas técnicas está também associada a um risco de efeitos negativos na recuperação do solo e da vegetação. Deste modo, o principal objetivo da presente tese consistiu em examinar a eficácia de medidas públicas utilizadas na recuperação pós-fogo de carvalhais caducifólios em Portugal nas últimas décadas.

No âmbito do trabalho desenvolvido nesta tese, identifiquei três fatores principais no processo de financiamento público que têm vindo a limitar a eficácia da estabilização de emergência pós-incêndio em Portugal: o modelo de financiamento, uma abordagem demasiado simplificada das intervenções elegíveis, e a lentidão na tomada de decisões e na execução. Adicionalmente, observei um pequeno impacto positivo das intervenções pós-incêndio na recuperação das florestas de carvalhais. Por outro lado, observei um impacto negativo dos fenómenos de seca no período pós-fogo e da frequência do fogo nesta recuperação. Por fim, a quantificação de múltiplos serviços de ecossistema destacou a existência de *trade-offs* devido às escolhas de gestão, indicando a necessidade de delinear objetivos a longo prazo para a área ardida, uma vez que diferentes abordagens de gestão pós-fogo podem favorecer serviços de ecossistema distintos. São necessários estudos adicionais para continuar este processo de melhoria da eficácia (e consequentemente da eficiência) das várias fases de restauro pós-fogo em Portugal.

Palavras-chave:

Restauro pós-fogo; Portugal; Recuperação da vegetação; Incêndios florestais; Carvalhos caducifólios

Avaliação da eficácia do restauro pós-fogo em Portugal com um foco nas florestas de carvalhos

Resumo alargado

A região Mediterrânica é uma área onde o fogo é um elemento natural e presente desde há muito. Nesta região, os fogos frequentes promoveram adaptações na vegetação nativa, através de estratégias de sobrevivência ao fogo, tais como a capacidade de rebrotar ou através da germinação de sementes no pós-fogo. No entanto, em resultado de diversos fatores como o abandono rural, as políticas de supressão do fogo ou as alterações climáticas, temos assistido, nas últimas décadas, a alterações nos regimes de fogo na Europa e na região Mediterrânica. O novo regime de fogo, caracterizado por incêndios mais frequentes e mais severos, origina impactos ecológicos, sociais e económicos negativos. Em termos de consequências ecológicas, esta alteração no regime de fogo poderá colocar em risco a integridade dos ecossistemas, por exemplo, impedindo a rebentação. Simultaneamente, estas alterações ao nível da estrutura e composição da floresta, tem como consequência uma homogeneização da paisagem, convertendo áreas de floresta em áreas de matos.

Portugal é um dos países do Sul da Europa mais afetados pelos incêndios florestais, com os maiores valores absolutos e relativos de área ardida, principalmente em consequência de uma densidade de ignição extremamente elevada e da ocorrência de grandes incêndios. Em Portugal, os ecossistemas de carvalhais caducifólios [grupo maioritariamente constituído pelo carvalho-negral (*Quercus pyrenaica* Willd.), carvalho-alvarinho (*Quercus robur* L.) e carvalho-português (*Quercus faginea* Lam.)], têm um papel importante na conservação da biodiversidade, na qualidade do solo e da água e na regulação do clima. A capacidade dos carvalhos de rebrotar após o fogo contribui significativamente para a recuperação do coberto vegetal, reduzindo assim os riscos de erosão e promovendo a retenção de nutrientes, ao mesmo tempo que cria refúgios para a fauna. Porém, os carvalhais estão atualmente reduzidos a pequenas áreas fragmentadas e apresentam uma elevada incidência de fogo. Caso o fogo seja demasiado frequente e/ou severo, os carvalhos poderão ver a sua capacidade de recuperação diminuída.

Nesse sentido, a recuperação pós-fogo é uma estratégia comum para minimizar os impactos negativos do fogo e facilitar a recuperação das funções dos ecossistemas afetados. A recuperação pós-incêndio pode ser dividida em cinco fases sequenciais. A primeira fase é a identificação de áreas potencialmente vulneráveis. A segunda fase ocorre imediatamente após o incêndio e inclui a identificação e avaliação dos efeitos do fogo, e a necessidade de intervenções pós-incêndio. As intervenções dividem-se em

intervenções de estabilização de emergência (fase 3), cujo objetivo é evitar uma maior degradação do ecossistema, mitigando os riscos para as pessoas e bens, e minimizando a perda significativa de cinzas e solo. A fase intermédia (fase 4) inclui a monitorização da resposta do ecossistema, e se necessário a aplicação de controlos fitossanitários e medidas de restauração biofísica. Por fim, a fase de recuperação (fase 5) implementa a visão a longo prazo para o território afetado e pode incluir a plantação ou a assistência à regeneração natural. Os efeitos do restauro pós-incêndio na recuperação do solo e da vegetação dependem de vários fatores, como o clima pós-incêndio, a topografia, a severidade do incêndio ou o historial de ocorrência de fogo no local.

Historicamente, a técnica de recuperação pós-incêndio mais utilizada tem sido a plantação e a sementeira, por vezes em resposta à forte pressão política e às expectativas sociais. Atualmente, e em consequência dos seus historiais de incêndios, os cinco países mediterrânicos (EUMED5, Portugal, Espanha, França, Grécia e Itália) têm a legislação mais completa para a gestão pós-fogo na Europa. Portugal, em particular, levou a cabo recentemente duas reformas estruturais. A primeira ocorreu após o período de 2003-2005, em resposta a áreas ardidas anómalas, e a segunda após o ano de 2017, marcado por incêndios catastróficos, e o qual enfatizou a importância da prevenção e de mudanças nos processos de recuperação pós-fogo. Contudo, frequentemente, os programas de recuperação pós-incêndio carecem de um plano de acompanhamento e monitorização que permita perceber os seus efeitos, principalmente no médio/longo prazo.

Tendo em consideração esta realidade, o principal objetivo da presente tese foi compreender a eficácia de várias medidas públicas empregues no restauro pós-fogo e, em particular, os efeitos desse restauro na recuperação de florestas de carvalhos caducifólios em Portugal. Como tal, várias questões de investigação foram analisadas ao longo dos diferentes capítulos, utilizando diversas abordagens e metodologias, e a diferentes escalas espaciais. Para atingir os objetivos propostos, a tese é composta por cinco capítulos, nomeadamente: a introdução (Capítulo 1), três capítulos que apresentam análises de três fases de restauro pós-fogo a diferentes escalas (Capítulos 2, 3 e 4), e a síntese (Capítulo 5).

O Capítulo 2 teve como objetivo compreender de que forma o processo de financiamento público tem afetado a eficácia da estabilização de emergência pós-fogo em Portugal, particularmente quais eram os principais fatores do processo de financiamento que limitavam a execução eficaz deste tipo de projetos. Com base em dados de relatórios públicos, concursos, e 517 projetos de estabilização de emergência pós-incêndio implementados em 2009-2018 e subsidiados por financiamento público, encontrei três limitações importantes para uma estabilização de emergência eficaz em áreas ardidas em Portugal, nomeadamente: (i) o modelo de financiamento, (ii) uma abordagem demasiado simplificada das intervenções elegíveis, e (iii) atrasos burocráticos na tomada de decisões e na execução. Esta execução tardia excede o prazo ótimo da estabilização de emergência pós-incêndio, minimizando a eficácia das intervenções implementadas.

O Capítulo 3 teve como objetivo compreender como a recuperação dos carvalhais caducifólios foi afetada pelas intervenções de estabilização de emergência pós-fogo, bem como pelas características do fogo, topografia e eventos de seca pós-fogo. Neste estudo, foram analisados projetos de estabilização de emergência pós-fogo implementados no Norte e Centro de Portugal, em áreas ocupadas por espécies de carvalho de folha caduca, nomeadamente, *Quercus faginea*, *Quercus pyrenaica* e *Quercus robur*. A recuperação das florestas de carvalhos foi analisada durante um período de quatro anos após o incêndio, em áreas ardidas com intervenções de restauro e em áreas próximas sem restauro (ou seja, de controlo). Esta análise utilizou técnicas de deteção remota, nomeadamente o índice NDVI como indicador da recuperação das florestas de carvalhos. Verifiquei que o restauro pós-fogo teve um efeito positivo marginal na recuperação das florestas de carvalhos. As possíveis razões para o pequeno efeito observado incluíram: a resiliência dos carvalhais de folha caduca após o incêndio; a impossibilidade em distinguir as intervenções analisadas; e a implementação de intervenções fora do prazo ótimo, conforme identificado no Capítulo 2. Para além disso, os eventos de seca pós-fogo (quantificados com base no índice de seca PDSI) foram identificados como um dos fatores mais importantes para uma menor recuperação dos carvalhais.

O Capítulo 4 visou compreender como o restauro pós-fogo com arborização de carvalho-negral (*Quercus pyrenaica*) afetou o fornecimento de serviços de ecossistema, e utilizar a quantificação desse fornecimento para avaliar a eficácia do restauro. Neste capítulo, foram recolhidos dados de campo de 15 projetos de arborização pós-incêndio, implementados com financiamento público no período 1994-2006, no nordeste e centro-leste de Portugal. Adicionalmente, foram recolhidos dados de campo de 15 áreas de controlo (sem arborização pós-fogo), próximas das áreas arborizadas. Os dados recolhidos incluíram topografia, características do povoamento, biometria florestal, vegetação do sob coberto, riqueza florística, regeneração natural, folhada e características do solo. As variáveis recolhidas foram utilizadas como indicadores para quantificar vários serviços de ecossistema. Os resultados mostraram que a arborização pós-fogo com carvalho negral teve impactos variáveis nos serviços de ecossistema, desde positivos a neutros ou negativos, dependendo do serviço analisado. Estes resultados revelam a existência de *trade-offs* entre serviços do ecossistema resultantes de opções de gestão, e sublinham a importância da monitorização das intervenções pós-incêndio para melhorar o conhecimento dos seus impactos e facilitar a gestão e a tomada de decisões.

Os resultados desta tese proporcionam uma compreensão da eficácia do restauro pós-fogo em carvalhais caducifólios portugueses, oferecendo novas informações para as comunidades académica e política. Em geral, os resultados mostram que é crucial realizar uma avaliação abrangente das opções de restauro pós-fogo para garantir o sucesso dos objetivos de restauro e a preservação das florestas de carvalhos. É imperativo avaliar minuciosamente as possíveis alternativas e os seus potenciais efeitos, considerando também a implementação eficiente das soluções escolhidas A análise cuidadosa das opções disponíveis é fundamental para tomar decisões informadas.

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Chapter 1 - General introduction

1.1. Fire in the Mediterranean region

The Mediterranean region is a highly fire-prone region due to its climate, characterized by the overlapping of the warm season with the year driest period (summer), which facilitates fire propagation in the dry vegetation (Keeley 2012; Castro Rego et al. 2021a). In the Mediterranean region, native vegetation evolved under a frequent fire regime that favoured species adapted to fire, which present different strategies of postfire recovery, such as resprouting (known as resprouter species) or germination (seeder species) (Pausas 1999a; Rundel et al. 2018). Resprouters frequently accomplish faster postfire ground cover in relation to the pre-fire situation, in comparison to seeders (Calvo et al. 2003; Pausas and Lloret 2007). In addition, resprouting individuals are able to persist, resulting in long lived specimens and more stable populations (Bond and Midgley 2001; Clarke et al. 2013). However, recurrent fires may drain storage reserves impeding further resprouting, while simultaneously eliminating saplings without sufficient reserves for resprouting (Burton et al. 2010; Monteiro-Henriques et al. 2018). In face of this, changes in fire regimes (e.g., frequency or severity) may have ecological consequences, posing a threat to the integrity of plant communities and ecosystems (Johnson and Miyanishi 2001; Pausas and Lloret 2007; McLauchlan et al. 2020).

Fire regimes in the Mediterranean region have changed in recent years, namely with an increase in the frequency of large fires, and an increase in fire severity (defined as the impact of fire on ecosystems, based on the loss or decomposition of organic matter), with large associated impacts (Keeley 2009; Brotons et al. 2013; Grünig et al. 2023). Changes in fire regimes and increased fire hazard in Mediterranean Europe can be attributed to rural land abandonment triggered by socio-economic factors, fire suppression policies, afforestation of former agricultural land, and climate change (Rigolot et al. 2011; Moreira et al. 2012b; Vallejo et al. 2012b; Regato et al. 2023). Climate change has led to more extreme and catastrophic events over a longer fire season, with great social and ecological impacts (Jolly et al. 2015; Pausas and Keeley 2021). Extreme fire events in Europe are expected to double by the end of the century, although climate influence on future fire danger and burned area is highly dependent on climate change projections (Dupuy et al. 2020; Grünig et al. 2023).

The years 2017, 2021, 2022 and 2023 were recorded as some of the worst ever in terms of burned areas and impacts, particularly in countries such as Portugal, Greece and Spain, resulting in loss of human lives and other negative impacts, such as loss of biodiversity, soil degradation, water contamination, economic costs, and carbon emissions (Dupuy et al. 2020; San-Miguel-Ayanz et al. 2022; JRC 2023; San-Miguel-Ayanz et al. 2023). Alterations in fire regimes have also caused changes in vegetation structure and composition that led to the homogenisation of forest landscapes and conversion of forest into shrubland throughout the Mediterranean region, within a self-perpetuating feedback cycle (Loepfe et al. 2010; Santana et al. 2014; Castro Rego et al. 2021a; Duane et al. 2021). Furthermore, increasing fire severity may compromise the provision of various ecosystem services such as water quality, erosion control and climate regulation (Warziniack et al. 2019; Roces-Díaz et al. 2022).

Hence, it is essential to address and mitigate the inevitable adverse impacts of these new fire regimes (European Commission 2021a; Harrison et al. 2021; San-Miguel-Ayanz et al. 2023). In this regard, fire-resilient landscapes have gained considerable attention and importance in the Mediterranean region. Fire-resilient landscapes are defined as a "*socio-ecological system that accepts the presence of fire, whilst preventing significant losses through landscape management, community engagement and effective recovery"* (Thacker et al. 2023). In this process, forests have emerged as crucial contributors to fire risk mitigation services (Rogers et al. 2020; Pausas and Keeley 2021). In parallel, fire-smart management of forest landscapes is necessary to minimise the socio-economic impacts of fire while maintaining and maximising its ecological benefits. Such management combines multiple strategies across a heterogeneous forest composition and structure, including species resistant to fire (low flammability) and/or resilient to fire (capable to recover after fire) (Fernandes 2013). Understanding the critical importance of forest management in strengthening landscape fire resilience, the European Union Forest Strategy for 2030 aims to promote multifunctional forest ecosystems, by supporting adaptation efforts and resilience-driven forest management practices, and by favouring mixed-species forests, especially those dominated by broadleaf and deciduous trees (European Commission 2021b).

1.2. Deciduous oak forests and fire

Oak species (*Quercus* sp. L., Fagaceae) began their ecological diversification during the Paleogene epoch (66 to 23 million years ago) and have continued to diversify into different environments to the present day (Barrón et al. 2017). Presently, oaks are broadly distributed throughout the northern hemisphere, inhabiting temperate, mediterranean, and tropical biomes, totalling around 435 recognized species (Vila-Viçosa et al. 2022). Deciduous oak forests in particular, expanded in the post-glacial period, reaching their largest extension in Europe around 6000 BP (Brewer et al. 2002; Zanon et al. 2018). Since then, the decline in the area occupied by deciduous oak forests has been linked to anthropogenic factors, such as deforestation, agricultural expansion or replacement of broadleaves by conifers (Brewer et al. 2002; Carrión et al. 2010; Githumbi et al. 2022).

Deciduous oak forest ecosystems have an important role in biodiversity conservation, in soil and water provision, in climate regulation through carbon sequestration and as source of timber and non-timber products (Silva 2007). In addition, *Quercus* species are capable of vegetative regeneration after major disturbances such as wildfires, with oaks being able to resprout, even at young ages (Calvo et al. 2003; Catry et al. 2013a). Oak capacity to resprout after wildfires significantly aids in the recovery of vegetative cover, thereby reducing erosion risks and promoting nutrient retention, while also creating a refuge for fauna (Moreira et al. 2012b). However, the occurrence of overly frequent or severe wildfires in Mediterranean Europe can reduce the resprouting ability of oaks (Catry et al. 2010b; Catry et al. 2013a) and may lead to reduced seed dispersal and viability (Marzano et al. 2012), resulting in lower oak natural regeneration, and degradation of oak forests (e.g. altered structure, reduced habitat quality or increased susceptibility to pests and diseases) (Álvarez et al. 2009). Ultimately, the increase of frequent or severe wildfires may lead to the replacement of burned oak forests by shrubland and to a decreased provision of ecosystem services (Mauri and Pons 2019; Nocentini et al. 2022). Furthermore, the timing of wildfires is critical, and those occurring in late summer may cause a higher decrease in oak resprouting capacity, as a consequence of nutrient distribution within the tree (Moreira et al. 2012b; Mauri and Pons 2019).

In Portugal, native deciduous oak forests persist. As of 2015, deciduous oak forests occupied 2.5% (81 700 ha) of the Portuguese forest land, showing a decrease in area in comparison with 1995 (92 000 ha) (ICNF 2016). Nevertheless, when present, these forests are highly modified in terms of structure and composition when compared to the pristine status (Silva 2007).

The Portuguese deciduous oak forest[1] is dominated by three species: Pyrenean oak (*Quercus pyrenaica* Willd.), English oak (*Quercus robur* L.) and Portuguese oak (*Quercus faginea* Lam.), with *Q. pyrenaica* occupying the largest extension. *Q. pyrenaica* occupies the bioclimatic transition zones between Atlantic and Mediterranean (northeast and centre Portugal) and grows predominantly on siliceous soils at mid-mountain

[1] Although some of the studied species, such as *Quercus pyrenaica* and *Quercus faginea*, may exhibit marcescence (the retention of dead leaves during winter), to ensure consistency and align with the nomenclature commonly adopted in standard databases used, the term 'deciduous' is used uniformly throughout this thesis.

elevations. *Q. robur* [2] occurs in regions with higher Atlantic influence (northwest Portugal), mostly in siliceous soils (Silva 2007). *Q. faginea* is reduced to small fragments along the coast of the Centre region of Portugal and upper-section of Douro valley. The subspecies *faginea* occurs in calcareous substrates of the Iberian plateau and the subspecies *broteroi* is restricted to the Portuguese Limestone Massif region (central and southern regions) (Bingre and Damasceno 2007). Figure 1.1 shows the natural range of the three dominant deciduous oak species in Portugal.

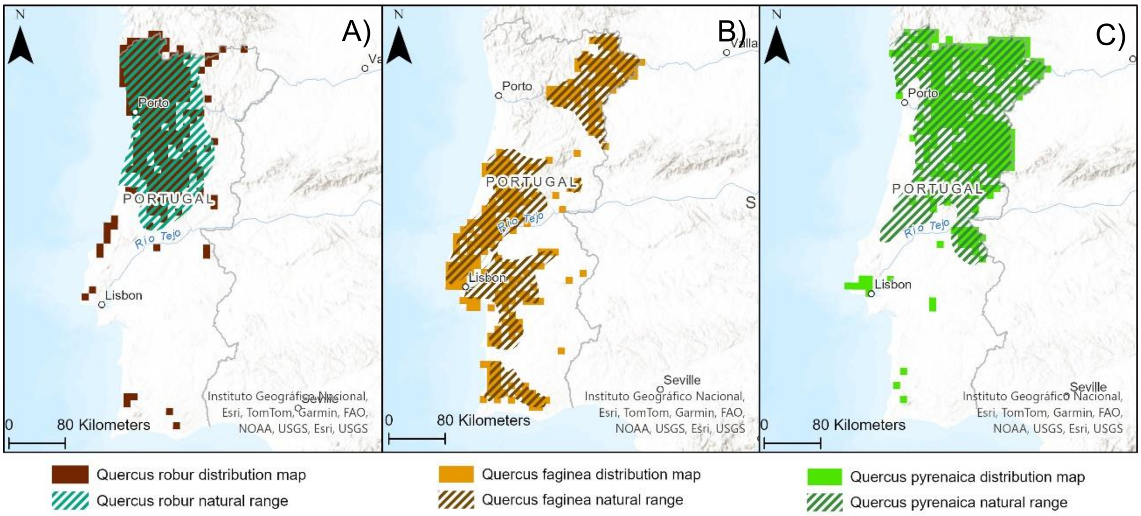

Figure 1.1 - Natural range and distribution maps of Portuguese deciduous oaks. a) *Quercus robur*; b) *Quercus faginea;* c) *Quercus pyrenaica*. Source: Natural range from Caudullo et al. (2017) and distribution maps from Araújo et al. (2024b); Araújo et al. (2024a); Porto et al. (2024).

Although deciduous oak forests occupy a small portion of the Portuguese landscape, they have one of the highest values of the fire selection index in Portugal, with about 44 000 ha of oak forests burned between 1996 and 2022, resulting in a very high fire incidence (Figure 1.2) (Fernandes and Guiomar 2017; ICNF 2024a). The proneness of deciduous oak forests to fire is mainly a consequence of their location within the country (northern and central regions are more prone to fire) and of their fragmentation, which means that these forests are often surrounded by land uses associated with a very high probability of burning (e.g. shrublands). In addition, Portuguese deciduous oak forests are mostly young forests, with absence of management, with a shrubby and flammable understory (Fernandes 2010; Fernandes et al. 2019). Conversely, mature deciduous oak forests ave a higher canopy cover (influencing the understory environment and regulating humidity levels), which results in lower burn probability, thus lower fire severity

[2] As of 2022, *Quercus robur* in Portugal is now referred taxonomically as *Quercus orocantabrica* (Vila-Viçosa et al. 2022).

and local limitation of wildfire spread (Fernandes and Guiomar 2017; Nunes et al. 2019). Consequently, depending on the deciduous oak forests' characteristics, their role in promoting landscape heterogeneity, with patches of oak forests intermixed with other vegetation types, may ultimately produce a more fire-resistant landscape (Fernandes et al. 2010; Fernandes 2010; Acácio et al. 2017).

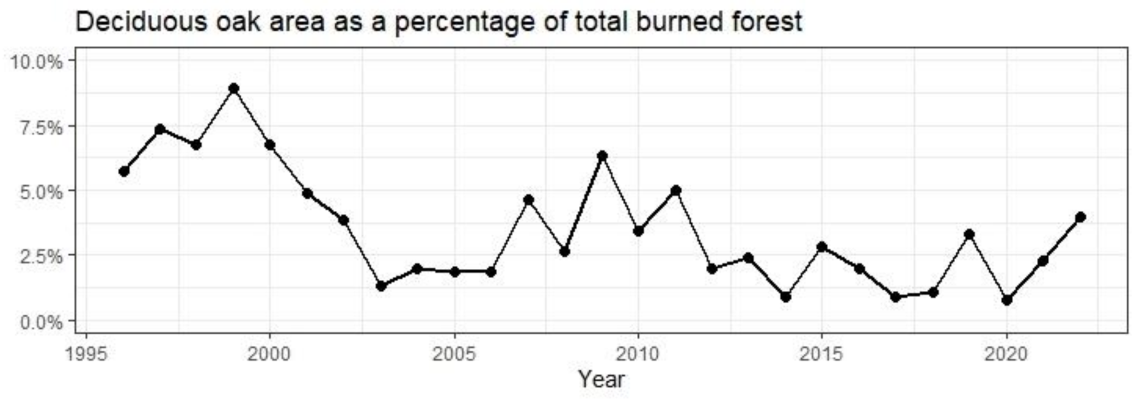

Figure 1.2 – Percentage of deciduous oaks area within the total burned forest. Source: ICNF (2024a)

1.3. Wildfires in Portugal

Portugal is one of the Southern European countries most affected by wildfires, with the highest absolute and relative values of burned area, mainly as a consequence of extremely high fire ignition density and occurrence of large fires (> 500 ha) (Mateus and Fernandes 2014; Tonini et al. 2017). The burnt area between 1980 and 2022 (49 532 km^2) was equivalent to almost 56% of mainland Portugal, with high annual variability (Figure 1.3) (ICNF 2023a).

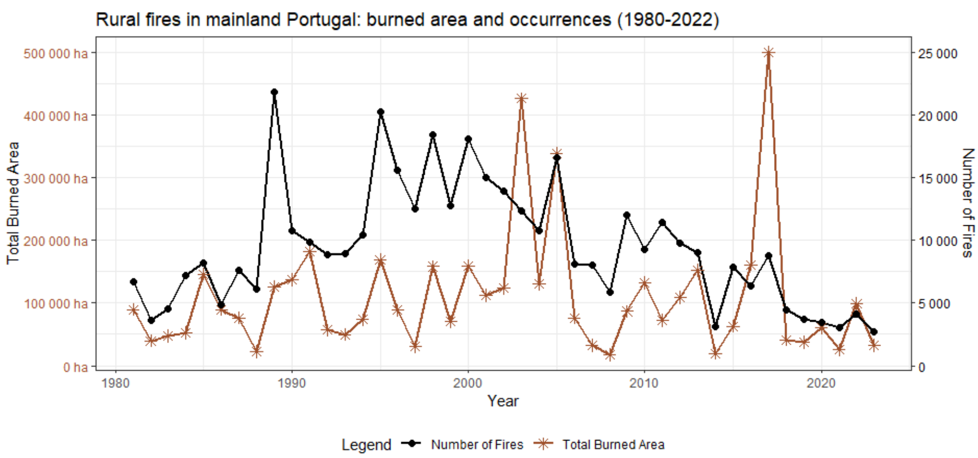

Figure 1.3 - Rural fires in mainland Portugal: burnt area and occurrences between 1980 and 2022. From 2001 onwards, fires in agricultural areas are accounted for. To ensure comparability over the time series, all fires of less than 0.1 ha were excluded. Source: ICNF (2024a)

In Portugal, large burned areas are associated to a small number of fires that were able to reach large magnitudes, with extraordinary negative environmental, social and economic impacts (Pereira et al. 2006; Marques et al. 2011; Tonini et al. 2017). The occurrence of wildfires in Portugal is concentrated in northern and central regions, and mountainous areas in the south. Figure 1.4 shows the geographical distribution of the burned area in Portugal from 1975 to 2022.

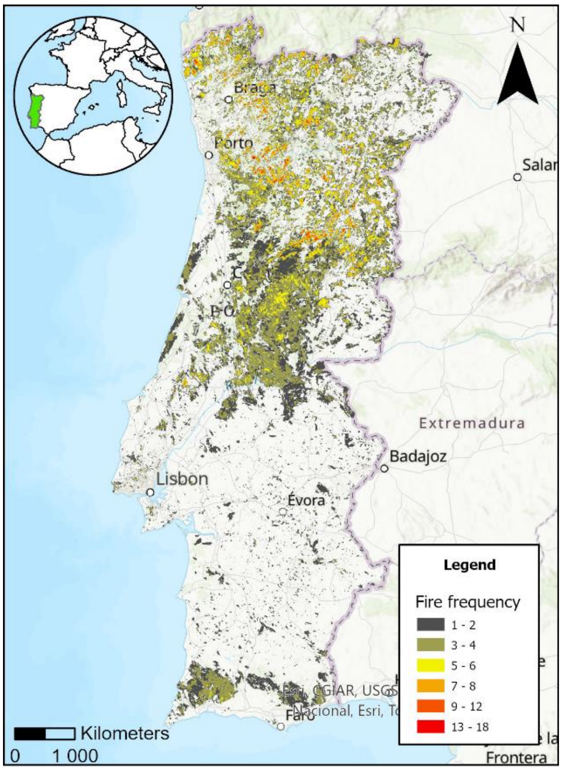

Figure 1.4 - Location of mainland Portugal in Europe and burned area in the period 1975-2022. Some of the areas burned more than once. Data source: (ICNF 2023a).

The large extent of wildfires in Portugal does not result solely from favourable weather conditions, but also from ineffective land management and planning (Beighley and Hyde 2018; Observatório Técnico Independente et al. 2018; Tedim 2018). In Portugal, the majority of forests are privately owned (97%), with the remaining 3% owned by the state, which represents the lowest percentage in Europe (Pulla et al. 2013; ICNF 2021a). In addition, most forest land holdings in Portugal (85%) are small (< 5 hectares) (Pereira et al. 2006), which halts effective forest management on a larger scale. This situation has been exacerbated by the conversion of classic forest owners into hobby or indifferent owners, and in many cases owners do not live on site (Feliciano et al. 2015; Valente et al. 2015). Additionally, there is a lack of spatial information on land registration for nearly half of the forest area (DGRF 2015; ICNF 2021a).

1.4. The importance of postfire ecological restoration

Ecological restoration is defined as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (SER 2004) and has been extensively applied in natural resource strategies at different levels and in many different ecosystems (Wortley et al. 2013). In order to be successful, ecological restoration practices should be effective, efficient and engaging (Mcdonald et al. 2016), with monitoring schemes aligned with clear objectives (Mansourian et al. 2017). The 2030 Agenda emphasises the ongoing importance and relevance of the topic (UN 2023) but there is a lack of knowledge regarding the quantification of the success of restoration programs (Suding 2011b; Nilsson et al. 2016; Lindenmayer 2020).

The Society for Ecological Restoration (SER) lists several attributes as appropriate indicators to measure restoration success (SER 2004; Wortley et al. 2013), which are divided in three general types: (1) species diversity and abundance, (2) vegetation structure, and (3) ecological processes (e.g. nutrient cycling and biological interactions) (Ruiz-jaen and Aide 2005). In recent years there has been a rising trend in employing ecosystem services (defined by Millennium Ecosystem Assessment (2005) as “*the benefits people obtain from ecosystems*”) as indicators to monitor and assess the advancements and achievements of restoration initiatives. Therefore, the provision of ecosystem services serves not only as an objective for restoration but also as a metric to evaluate the effectiveness of implemented practices (Jackson et al. 2022; Liu et al. 2023).

Postfire restoration is a common strategy to minimize fire negative impacts and to facilitate the recovery of burned forests and ecosystem functions (Moreira et al. 2012b; Lucas-Borja et al. 2021). Postfire restoration can be divided in five sequential phases (Figure 1.5). The first phase is the identification of potentially vulnerable areas. Phase 2 occurs immediately after the fire and includes the identification and evaluation of fire effects on the burned area, with collection of in-situ data if needed. Based on the assessments of fire effects, postfire interventions are applied where necessary, divided into emergency stabilization interventions (phase 3), the intermediate phase (phase 4) and the recovery phase (phase 5).

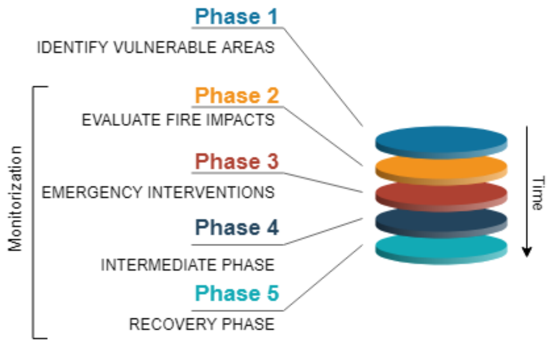

Figure 1.5 - Phases of post fire restoration. Adapted from Moreira et al. (2010).

Emergency stabilization interventions (phase 3) should be implemented shortly after the fire and, if possible, before the first autumn rains, to prevent further degradation, mitigate risks to people and assets, and minimize the significant loss of ashes and soil that typically occurs in the initial four months after the fire (Vallejo et al. 2012b; Ferreira et al. 2015; Girona-García et al. 2021). Emergency stabilization treatments are particularly crucial when there is an elevated risk of soil erosion, and the expected rate of plant regeneration is slow (Vallejo et al. 2012; Vega et al. 2013). Examples of emergency stabilization interventions include mulching, seeding, contour-felled logs and rock dams (see Figure 1.6) (Robichaud et al. 2000; Napper 2006; Girona-García et al. 2021).

Figure 1.6 – Emergency stabilization interventions: rock check dam (left) and contour-felled logs (right). Both aim to detain runoff and sediments, decreasing soil losses after fire. (Photos by Luís Lopes)

The intermediate phase (phase 4) should take place up to 2 years after the fire and includes damage assessment and monitoring of ecosystem response, implementation of phytosanitary controls and biophysical restoration measures, including analysis of the presence of natural regeneration or potential for the spread of invasive species (see Figure 1.7). Finally, the recovery phase (phase 5) takes place from the third year after the fire, which may include planting or assisting natural regeneration (see Figure 1.8) (Moreira et al. 2010).

Figure 1.7 – Intermediate phase: oak resprouting after fire (left) and seed dispersal of invasive species (*Hackea decurrens*) after fire (right) in central Portugal. (Photos by Luís Lopes)

Figure 1.8 – Afforestation project with Pyrenean oaks (*Quercus pyrenaica* Willd) (left) and Pyrenean oak resprouting (right), central Portugal. (Photos by Luís Lopes)

The effects of postfire restoration on soil and vegetation recovery depend on several factors, such as postfire weather, topography, fire severity or fire history (Pereira et al. 2018a). However, most restoration programs lack a follow-up assessment and postfire restoration is rarely monitored (Bautista et al. 2010a; Mansourian et al. 2017; Gann et al. 2019; Lindenmayer 2020). Understanding the historical trajectory of postfire restoration would provide a vital framework for the selection of restoration interventions, management practices and decision-making strategies.

1.5. Policies and funding for postfire restoration in Portugal

Postfire restoration techniques such as planting and seeding have been widely used in the Mediterranean region since the late 19th century due to strong political pressure and social expectations (Moreira et al. 2012b; Rego et al. 2018). In recent years, the European Union (EU) has placed greater emphasis on the importance of postfire restoration of forest ecosystems. This has been manifested in the promotion of cooperation frameworks, the provision of financial support for research efforts, and the development of postfire management strategies and technical recommendations (Rego et al. 2018). Subsequently, the policies are formulated by each country, considering the country wildfire history and the perceived level of threat. As a result, the five Mediterranean countries (the EUMED5, Portugal, Spain, France, Greece and Italy) have the most thorough legislation for postfire management within Europe (Mavsar et al. 2012a; Montiel-Molina 2013; Fernandez-Anez et al. 2021).

In Portugal, the increasing frequency of large wildfires over the last decades led to an increased importance of national policies regarding postfire management and restoration, resulting in numerous changes in national legislation and policies (Mateus and Fernandes 2014; Pinho and Mateus 2018; Ribeiro et al. 2018).

Public policies for postfire restoration in Portugal can be divided in four main time periods (Pinho 2017). Until 1960, those policies were almost inexistent, with the exception of afforestation within public programs of forest development. Between 1960 and 2003, reforestation was the priority action. In this period, and especially after 1975, there was an increment of specific policies (e.g., Forest Promotion Fund, Forest Defence against Fire legal regime, and Forest Law Policy) and a prioritization of burned area afforestation with the European Community financial programs in 1986-1993 (PAF - Forest Action Program) and 1995-2000 (PAMAF - Program to Support the Modernization of Portuguese Agriculture and Forests). Indeed, Portugal's accession to the European Community in 1986 marked a significant transformation in public policies. Public programs aimed at financing the recovery of burned areas began to receive substantial contributions from European funding (Mendes and Dias 2002).

After 2003, the Forest Sector Structural Reform was established in response to abnormal burnt areas. It included main strategic and planning instruments for the forest sector (DGRF 2005). Additionally, a compilation of guidelines for the recovery of burned areas was published (Conselho de Ministros 2006) and translated into main policy instruments. In 2004, the Permanent Forestry Fund was launched to fund postfire restoration, exclusively with national funds (Mendes and Dias 2002). Moreover, there was financial support from European programs for the restoration of burned areas, including emergency stabilization during 2007-2014 (program PRODER) and 2014-2020 (program PDR2020).

The several programs with European funding in the period 1986-2020 are shown in Table 1.1.

Table 1.1 - Programs with European funding (1986-2020). Letters a to d indicate entities involved in program management: a – AFN/ICNF; b – IFAP; c – PRODER managing authority; d – PDR managing authority. In PDF/PAMAF program, the responsible entity until 1996 was AFN/ICNF, later changing to IFAP. Adapted from DGRF 2006

	Support for afforestation and forest improvement	Support for afforestation of marginal agricultural areas
Program (period)	PAF / PEDAP (1986-1993) **a**	R2328/91 (1991-1993) **b**
	PDF / PAMAF (1995-2000) **ab**	R2080/92 (1994-1999) **b**
	AGRO (2000-2006) **b**	RURIS (2000-2006) **b**
	PRODER (2007-2013) **bc**	
	PDR2020 (2014-2020) **bd**	

The catastrophic fire season of 2017 in Portugal led to a transition from a policy centred on fire suppression to one that emphasizes the importance of prevention, while also promoting changes in postfire recovery processes. Specifically, the main objectives of the new National Plan for Integrated Management of Rural Fires were to incorporate faster and more flexible processes into the different postfire phases, not only in the emergency stabilization phase but also in the long-term, with the possibility of implementing transformative changes in the territory (Conselho de Ministros 2020a). In 2020, the National Plan for Integrated Fire Management of Rural Fires establishes economic and territorial policy instruments towards more fire-smart forests and landscapes at national level (Conselho de Ministros 2020a; Conselho de Ministros 2020b).

1.6. Thesis aims and outline

The objective of this thesis is to examine the effectiveness of various public environmental measures employed in postfire restoration in Portugal over recent decades, with a focus on deciduous oak forests.

In order to achieve the proposed objectives, the thesis consists of five chapters, including this introduction (Chapter 1), three chapters presenting analyses of three phases of postfire restoration at different scales (Chapters 2, 3 and 4) and the synthesis (Chapter 5). Chapter 2 analyses postfire emergency stabilization interventions at the national scale (Portugal); Chapter 3 analyses both the emergency stabilization interventions and the intermediate phase of postfire recovery at the macro-regional scale (north and central Portugal); and Chapter 4 analyses the recovery phase at the regional/local scale (northeast and central-east Portugal). Detailed justifications for selecting each spatial scale and the inclusion of specific oak species are provided in the respective chapters.

The organization of the thesis is guided by the conceptual model shown in Figure 1.9. This model represents the articulation between chapters, from fire occurrence to the implementation of short, medium and long term postfire management actions.

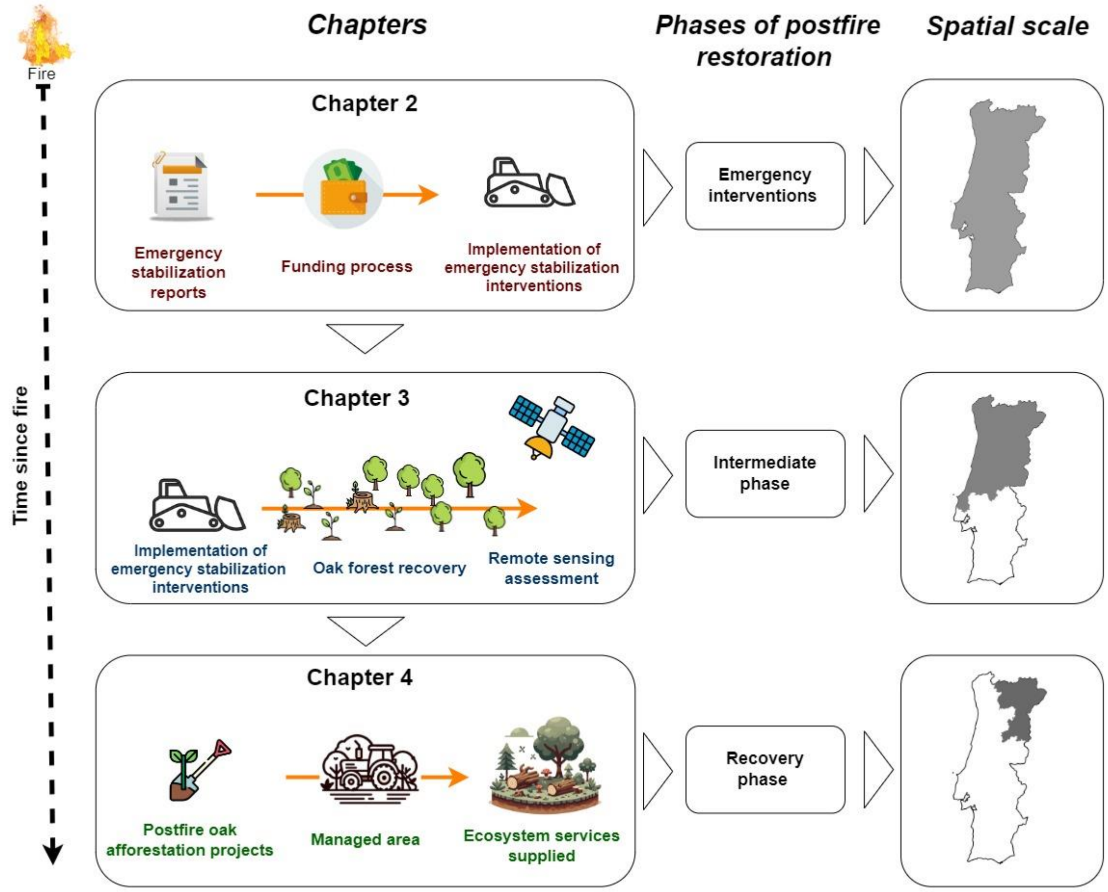

Figure 1.9 - Thesis conceptual model, divided by chapters.

Chapter 2 aims to understand which are the main factors of the public funding process that limit the effective execution of postfire emergency stabilization projects in Portugal. In this chapter, the process of public funding for postfire emergency stabilization in Portugal is analysed based on 517 implemented projects in 2009-2018.

Considering that postfire restoration treatments should be used strategically, since they can be expensive and may have adverse consequences (Fernández and Vega 2016; Bontrager et al. 2019), **Chapter 3** aims to understand if postfire emergency stabilization projects had a positive impact on the recovery of deciduous oak forests and how this recovery was affected by fire characteristics, topography and postfire drought events. It analyses postfire restoration projects implemented in North and Central Portugal with deciduous oak species, namely, *Quercus faginea*, *Quercus pyrenaica*, and *Quercus robur*. Restoration projects analysed include not only emergency stabilization interventions, but also sowing or planting (intermediate-stage interventions). In this chapter, remote sensing was used to assess the impact of postfire restoration interventions on the recovery of deciduous oak forests over a 4-year period following the fire event. Satellite images were used to quantify changes in vegetation indices (e.g., NDVI) over time in burned areas with restoration interventions and in close areas without restoration (control).

Chapter 4 aims to assess the impact of postfire afforestation with *Quercus pyrenaica* on the supply of ecosystem services (ES) and to use the observed ES supply to evaluate restoration effectiveness. In this chapter, field data was collected from 15 postfire afforestation projects implemented with public funding in the period 1994-2006, in northeast and central-east Portugal. Filed data was also collected from paired control areas without oak afforestation. Collected data included site conditions, stand characteristics, forest biometry, understory vegetation, floristic richness, natural regeneration, litter, and soil characteristics. Collected variables were used as proxies to quantify various ecosystem services.

Chapter 5 synthesizes and discusses the main results of the thesis. It is discussed how this thesis may contribute for the advancement of knowledge and improvement of the efficiency of postfire restoration in Portugal, including opportunities for future research.

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Chapter 2 - Public funding constrains effective postfire emergency restoration in Portugal

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Abstract

In the last decades, fire regimes in Europe have changed towards an increased occurrence of extreme fire events with large burned areas and associated impacts. Portugal is one of the countries most affected by wildfires, with extraordinary negative impacts. Postfire emergency stabilization is an important restoration practice to mitigate fire impacts in the short term. The present study aims to improve understanding of how public funding processes affect the efficiency of postfire emergency stabilization in Portugal. We analysed the process of public funding for postfire emergency stabilization in 2009-2018 using data from public reports assessing the needs for interventions (147), open calls resulting from those reports (12) and implementation projects subsidized by the calls (517). Our results show that available funding through calls for postfire emergency stabilization responded to the needs assessed by previous reports, but there was no available funding (reports and calls) for many (37%) of the large burned areas (> 1000 ha) in 2009-2018. Furthermore, the effectiveness of public funding for postfire emergency stabilization in Portugal is limited by the financial model, lack of eligibility of relevant treatments, the constant change of calls criteria, and slow decision-making and execution that exceeds the optimal timeframe. As a consequence, implementation of postfire emergency stabilization within an optimal timeframe is halted by insufficient financial capacity of private beneficiaries. We discuss effective funding mechanisms from other countries and suggest improvements for the funding process in Portugal.

Keywords

Postfire restoration; Postfire emergency stabilization; Public subsidies; Mediterranean climate; Portugal; Restoration funding

Implications for Practice

• Restoration through postfire emergency stabilization requires a promptly, dedicated, and simplified funding process, able to support the necessary interventions.

• The financial capacity of beneficiaries constrains the implementation of projects for postfire emergency stabilization.

• Involving land managers in the definition of public funding programs may help in their adhesion to those programs and their commitment with project implementation.

2.1. Introduction

Fire regimes have been changing in the last decades in many regions of the world, with temperate zones presenting a tendency towards larger burned areas and increasing negative impacts due to changes in climate and changes in land use that have modified forest structure and composition, including invasion by alien species (Castro Rego et al. 2021b). In Mediterranean Europe in particular, and despite a decreasing trend in annual burned area (Turco et al. 2016), extreme wildfire events with large burned area and increasing socioeconomic and ecological impacts have become more frequent during the last decades (Tedim et al. 2015; Doerr and Santín 2016; Fernandez-Anez et al. 2021) due to a complex interaction of increasing temperatures and dryness, and land abandonment and fire suppression policies that led to increased fuel load and connectivity (Fernandes 2013; Moreira et al. 2020).

Mediterranean Europe is a highly fire-prone region where the five European countries most affected by wildfires are located, namely, France, Greece, Italy, Portugal, and Spain (also known as EUMED5) (Dupuy et al. 2020). Here, wildfires are causing extraordinary socioeconomic and environmental negative impacts, such as income loss, human fatalities, changes in forest structure and composition, hydrological cycle disruption, biodiversity loss, and severe soil damage (Pereira et al. 2006; Catry et al. 2013b). Mediterranean-climate countries are especially susceptible to postfire soil impacts due to frequently intense autumn rainfall and summer thunderstorms that may intensify damages (Pereira et al. 2018b; Garrido-Ruiz et al. 2022), being soil erosion among the most damaging postfire processes (Vallejo et al. 2012b). Other postfire soil impacts include changes in soil structure, loss of organic matter, hydrophobicity, and volatilization and/or leaching of nutrients, and will depend on fire severity (defined as the degree of loss or decomposition of organic matter aboveground and belowground), topography and fire recurrence, among other factors (Keeley 2009; Caon et al. 2014; Pereira et al. 2018b).

Fire impacts on soil properties (biological, chemical, or physical) can be short-term, long-term or permanent (Caon et al. 2014). As such, and although Mediterranean vegetation is adapted to wildfires, severe and/or recurrent wildfires may limit and delay the vegetation recovery capacity (Caon et al. 2014; Pereira et al. 2018b; McLauchlan et al. 2020; Nolan et al. 2021). Restoring burned areas is therefore extremely important to minimize negative wildfire impacts, namely emergency stabilization treatments, which are the first stage of restoration, followed by rehabilitation and recovery (Vallejo and Moreira 2010). Emergency treatments are implemented after damage assessment and intend to stabilise the area and protect the soil in the shortest possible time, to avoid further degradation and diminish risks for people and assets. Emergency treatments are particularly needed when the risk of soil erosion is high and when plant regeneration is slow (Vallejo et al. 2012b; Vega et al. 2013a). The selection of treatments for emergency stabilization should consider diverse criteria such as effectiveness, cost, and speed of execution (Fernández et al. 2019). The latter is extremely important since early implementation maximizes success probability (Vega et al. 2013a). Hence, implementation should follow the wildfire as shortly as possible and, if feasible, preceding the first autumn rains, as most of the ashes and soil are lost in the first four months after fire (Moreira et al. 2012a; Ferreira et al. 2015).

Postfire restoration policies are heterogenous across European countries, depending on wildfire´s history and perceived level of threat (Mavsar et al. 2012b). For example, the legal postfire management framework may consider dedicated wildfire laws (e.g. Portugal and Spain), may be part of broader forest policies (e.g. Austria and Germany), or may be dispersed among other policies (e.g. Finland and Latvia) (Mavsar et al. 2012b; Montiel-Molina 2013). Within Europe, the EUMED5 countries present the most thorough postfire management legislation (Mavsar et al. 2012b).

Portugal registers the highest absolute and relative values of burned area in Europe, associated to a small number of particularly large fires (Keeley 2009; Mateus and Fernandes 2014; Fernandes et al. 2016a; Tonini et al. 2017). Current policy for postfire management and restoration in Portugal derives from consecutive changes in national laws and plans over the last decades (Mateus and Fernandes 2014; Ribeiro et al. 2018; Pinho and Mateus 2018b), as a response to the increased occurrence of large wildfires and to increased knowledge of the importance of mitigating fire impacts (Pinho 2017). In addition, after Portugal entered the European Union (EU) in 1986, a considerable external source of funds became available for national forest restoration policies and for postfire restoration in particular (Mendes and Dias 2002). As a consequence, postfire restoration incentives have gradually gained visibility on the Portuguese political agenda beyond burned areas reforestation (Ribeiro et al. 2018), especially in the beginning of the 21st century (DGRF 2005; DGRF 2006b).

Recent subsidies for postfire restoration, including emergency stabilization, have been provided by Portuguese Rural Development Programs (PRODER in 2007-2013 and PDR2020 in 2014-2020), funded by the European Agricultural Fund for Rural Development (EAFRD) (DGRF 2006b; Ribeiro et al. 2018). Public funding for postfire emergency stabilization in Portugal is a reactive mechanism available through specific calls that are launched based on reports that assess the need for emergency stabilization in response to wildfires, issued by the Institute for Nature Conservation and Forests (ICNF) (SEFDR 2017). The calls are the mechanism through which land managers can access the funding for emergency stabilization treatments.

Nevertheless, the financial support for postfire emergency stabilization in Portugal tends to be available belatedly. The beneficiaries often face lengthy procedures, in most cases surpassing the critical period of action (the first months after fire) and conditioning its timely application (Observatório Técnico Independente 2019a). In addition, the Portuguese forest is owned mostly by private landowners (around 91%) and 93% of forest properties are smaller than 10 ha (Pereira et al. 2006). Hence, postfire emergency restoration is unlikely to be implemented by small-scale owners unless funding is readily available (Mavsar et al. 2012b). Furthermore, about half of the forest area lacks spatial information on land registration (ICNF 2021a).

In this study, we aim to improve understanding of how the public funding process affects the efficiency of postfire emergency stabilization in Portugal, by analysing the related administrative, technical, and financial factors. In this context, efficiency is defined as accomplishing postfire emergency stabilization with the least resources consumption (Mcdonald et al. 2016). We ask the following questions: (i) is the number of emergency stabilization reports sufficient to address the restoration of large burned areas? (ii) do calls respond adequately to the costs estimated by emergency stabilization reports? (iii) what are the factors of the funding process that limit the effective execution of postfire emergency stabilization projects? To answer these questions, we analyse the process of public funding for postfire emergency stabilization in Portugal for the period 2009-2018 using data from public emergency stabilization reports, calls and subsidized projects. We compare the results found for Portugal with successful postfire emergency stabilization case studies with similar fire history or socio-ecological characteristics to understand differences in policy execution and measures implementation. Finally, we propose solutions that may improve the efficiency of the public funding process for postfire emergency stabilization in Portugal. Our study will contribute to the general understanding of how funding administration may affect work on the ground, which is a critical factor for successful restoration programs.

2.2. Methods

2.2.1. Study area

The study area is mainland Portugal (latitude 37º-42ºN, longitude 6º-10ºW), located at the extreme southwest of Europe, on the Iberian Peninsula, with an approximate area of 89,000 km^2. Climate is Mediterranean and there are considerable physiographic differences between the north and the south of the country (Tonini et al. 2017). In 2015, Portugal was predominantly covered by forests (36%), shrublands and pastures (31%), and agricultural land (23%) (ICNF 2019a).

Wildfires in Portugal occur mainly in the northern and central regions (Pereira et al. 2006; Tonini et al. 2017). Figure 2.1 shows the spatial distribution of the burned area (14,310 km^2) in the 2009-2018 period.

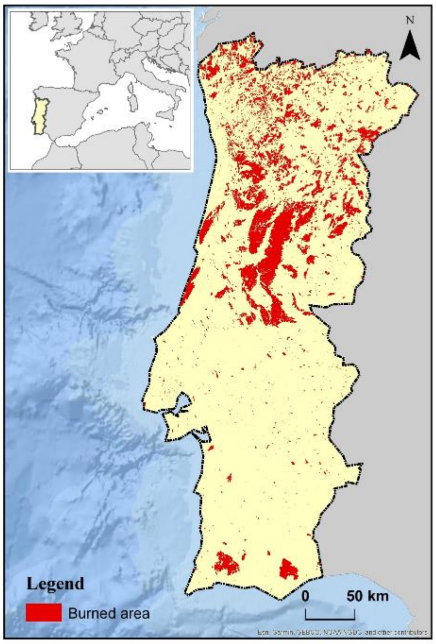


Figure 2.1 - Location of Portugal in Europe and burned area in the period 2009-2018. Data source ICNF (2021)

2.2.2. Data collection

We analysed the sequential stages of the process of public funding for postfire emergency stabilization in 2009-2018 (Figure 2.2), including administrative, technical and financial aspects, using data from: (i) emergency stabilization reports, (ii) open calls resulting from those reports, and (iii) projects subsidized by the calls.

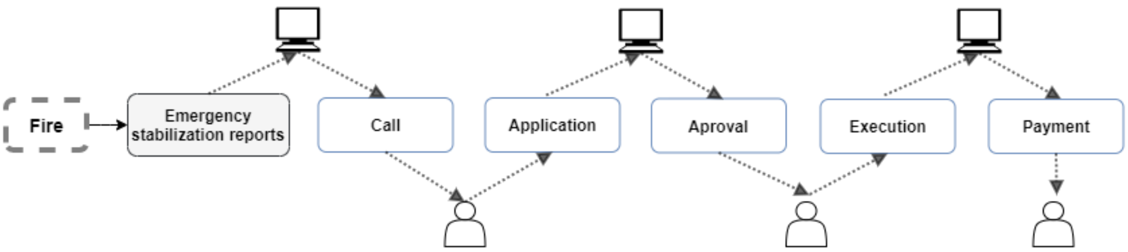


Figure 2.2 - Sequential stages of public funding for the implementation of postfire emergency stabilization projects since fire extinction and until the first payment. corresponds to “Beneficiary” and to “Program Managing Authority”.

Emergency stabilization reports (hereafter reports) are produced by ICNF since 2009. These reports assess the short-term emergency stabilization treatments required for large burned areas and the corresponding budgets. Their structure is standardized, containing systematized information following functional and objective criteria. Reports include 4 categories of treatments: 1) recovery of affected infrastructures (includes roads, fuel breaks, water points, fences and damaged signalling); 2) erosion control, treatment and slope protection (soil treatments, establishment of barriers and others); 3) prevention of contamination and recovery of watercourses (regularization of the hydrological regime and torrential correction); and 4) others, later renamed as biodiversity loss mitigation (such as protection of natural regeneration, reforestation, control of invasive species, or actions to support wildlife). For this study, we analysed a total of 134 reports produced between 2009 and 2018, of which 121 have been published online (available at www.icnf.pt/portal/florestas/dfci/relat/raa). The remaining 13 reports were obtained directly from ICNF services.

The formal process to operationalize the emergency stabilization reports is through the opening of a specialized call funded by the Portuguese Rural Development Program (PRODER program, in 2009-2013 and PDR2020 program, in 2014-2020). Each call corresponds to a specific region/fire or aggregates several regions/fires. Land managers who wish to execute emergency stabilization treatments in burned areas located within these areas can apply to the call, as long as they meet the eligibility criteria. In 2009-2018 there were 12 calls for postfire emergency stabilization, which are hereafter referred by their year (with an added letter if there is more than one call per year). We collected data from projects that were submitted, approved, and executed within these calls, in a total of 517 projects (114 approved in the first funding period and 403 approved in the second funding period).

Information was conditioned by data availability and type of data structure, which differs between funding programs. Data from emergency stabilization projects was provided by the respective program managing authority and by the Financing Institute for Agriculture and Fisheries (IFAP). When there were discrepancies in financial records between these two data sources, records from the program managing authority prevailed.

Data extracted and analysed from reports, calls and projects for 2009-2018 is shown in Table 2.1. Project total budget includes both eligible and non-eligible treatments. The former corresponds to the treatments considered eligible during the approval process and project financed amount corresponds to the project eligible expenses multiplied by the co-payment percentage.

Table 2.1 - Data extracted and analysed from emergency stabilization reports, calls, and projects for 2009-2018

Stage	Data used in the analysis
Emergency stabilization reports (n=134)	- Burned area - Dates of fire start and extinction - Costs assessed by type of treatment - Municipalities and parishes affected - Predicted area of treatment - Estimated cost of treatment
Calls (n=12)	- Application period - Available budget (not available for 2013-B call) - Eligible expenses - Number of reports per call - Geographical area covered
Projects (n=517)	- Corresponding fire and reports - Location at parish level - Date of decision/approval - Date of project submission - Date of request for 1st payment (request submission by the beneficiary and payment date) - Present operation status (as of 29/01/2020) - Predicted period (beginning and end) for project execution - Project total budget, eligible and financed amounts - Type of beneficiary (Common lands, Forest Producers Organizations, Private, Public Entities) (available for 499 projects)

In addition, to analyse report production as a response to large wildfire occurrence, we collected geospatial data on burned areas larger than 50 ha (vector format) between 2009 and 2018 (ICNF, 2021). Burned areas were divided in three groups: from 50 to 500 ha, from 500 to 1,000 ha and over 1,000 ha. Although wildfires in Europe are considered large above 500 ha, 50 ha has been used as the threshold for fire severity assessment by the EU wildfire information system (EFFIS) (European Commission, 2007).

2.2.3. Data analysis

To analyse if the number of reports and calls was adequate for the restoration of large burned areas, we compared selected burned areas with existing reports and funding calls, and tested with a two-sample t-test how wildfire size influenced the existence of a report. We also analysed administrative and financial aspects of reports, namely, report production over time, estimated report budget per area of treatment and budget distribution per treatment categories.

To analyse if calls budget responded to the costs assessed by the reports, we quantified the association between call budget and corresponding budget previously estimated by reports through correlation analysis (note that one call may correspond to several prior reports).

To identify the factors that may have limited the effective execution of projects for postfire emergency stabilization, we examined how changes in calls criteria over time may have influenced project conception and execution. In addition, we analysed changes over time in the proportion of types of beneficiaries of submitted projects. We also assessed the association of type of beneficiary and project total budget and eligible expenses with a Welch ANOVA, because the homogeneity of variance assumption was not met (Levene's test). Finally, we analysed if the projects monetary amounts (total budget and eligible expenses) varied in accordance with the necessary costs estimated by the reports and if the calls budget covered the projects financed amounts.

We also quantified time intervals since fire extinction and across the different stages of the funding process. Time intervals correspond to the difference, in number of days, between two different stages of the funding process.

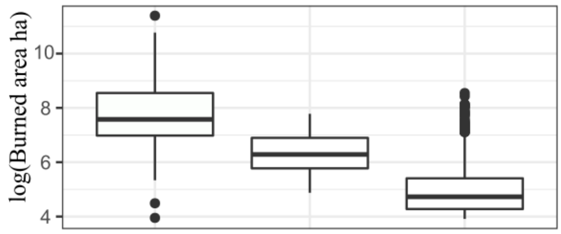
Burned areas between 2009 and 2018 were selected and analysed with ArcMAP 10.4 (Esri 2016). Statistical analyses were carried out with R (4.0.3) (R Core Team 2020). Statistical tests were performed with base R functions and the Hmisc package (Harrell Jr and Dupont 2020). Descriptive analyses for reports, calls and projects data were tabulated with the support of Dplyr and Tidyr packages (Wickham et al. 2020; Wickham 2020).

2.3. Results

2.3.1. Report production per size of burned area

In the period 2009-2018, a total of 2,475 fires larger than 50 ha occurred in mainland Portugal, of which 6.26% (n=155) exceeded 1000 ha. Interannual variation was high, both in the number of fires and their size-class distribution. In this period, reports were produced for 0.5%, 16.1% and 63% of respectively 50-500 ha fires, 500-1,000 ha fires and >1,000 ha fires. Still, reports were not produced for 37% of fires >1,000 ha. Considering all fires (> 50 ha), there were no reports for a total of 2,341 fires corresponding to 633 thousand hectares (around 7% of the Portuguese mainland) over 2009-2018. The smallest burned area with a report was 52 ha and the largest was 88,970 ha. There was a significant difference (t (133) = -5.606; $p < 0.001$) between burned areas depending on report existence: the mean size of burned areas was 5,380 ha and 215 ha, respectively for fires with and without reports. Likewise, the costs estimated by postfire emergency stabilization reports were positively correlated with the size of the burned area (r^2= 0.724). Despite the lack of reports for many large burned areas, the number of reports increased over time (43 reports in 2009-2013 and 91 reports in 2014-2018).

In addition, not all reports were followed by funding calls. For example, 11 reports were produced in 2011 but no calls were open. Most reports (78%) with funding calls corresponded to burned areas larger than 1,000 ha. The remaining reports with funding calls respected to burned areas of 500-1,000 ha (17.4%) and 50-500 ha (4.9%) (Figure 2.3).



		With report and call	*With report but without call*	*Without report*	*Total*
Burned area	50 to 500 ha	6	5	2148	2159
	500 to 1000 ha	21	5	135	161
	> 1000 ha	94	3	58	155
Total		121	13	2341	2475

Figure 2.3 - Number of fires with and without emergency stabilization reports and corresponding calls in 2009-2018 according to burned area size (> 50 ha).

2.3.2. Differences in treatments eligibility and budget among reports

During the period 2009-2018, the main categories of eligible treatments were distinct among reports. Only 43.3% of reports included all four main categories of treatments eligible for funding, while 38.1% included three categories, 14.2% considered two categories, and 4.5% (6 reports) considered just one category. The most represented category was “Recovery of affected infrastructures”, which was included in 95.5% of all reports, followed closely by “Prevention of contamination and recovery of watercourses” (91% of reports). The category “Erosion control, treatment and slope protection” was eligible for funding in 75.4% of reports and the category “Others/Decrease in biodiversity loss” was the least represented (58.2% of the reports). Nevertheless, the average number of categories included in the reports increased over time, with an average value below or around three until 2013 but progressing since then and achieving a value of four in 2018 (Figure S1).

The budget allocated to each treatment category was also highly variable among reports and distinct between the two funding programmes (Figure 2.4). During the first programme (2009-2013), treatments related to erosion/slope protection received an average of 65.8% of total funding, whereas biodiversity protection measures were nearly absent (only present in 2010 reports, with a residual value of 0.05% of total budget). In the 2014-2018 programme, erosion/slope protection and infrastructure treatments decreased in their relative importance in opposition to treatments for protection and recovery of watercourses and prevention of biodiversity loss, which attained their highest percentual values. However, considerable changes in the relative importance of each category occurred again in 2018, with a residual budget for the watercourses and biodiversity loss categories (1.45% and 0.29% of total budget for this year reports, respectively), with the remaining budget allocated almost in equal parts to the recovery of infrastructures and protection of erosion/slope.

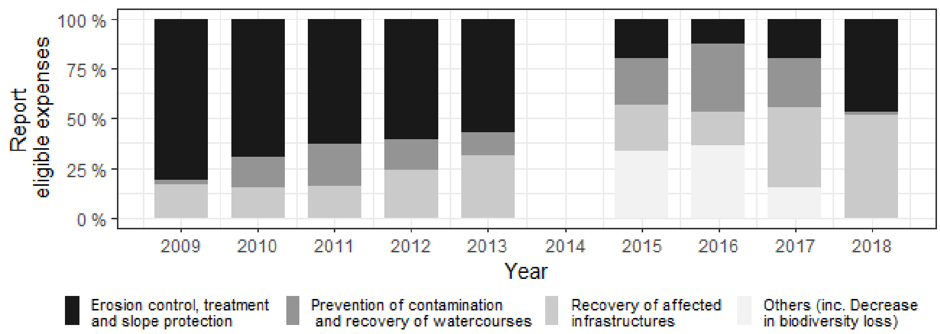


Figure 2.4 – Report eligible expenses per year for each category of emergency stabilization treatment in 2009-2018. Values are shown in percentage of the costs assessed by the reports.

2.3.3. Adequacy of calls budgets to reports and changes in calls announcements over time

Call budgets responded directly to the costs estimated in the reports for the implementation of emergency stabilization treatments (r^2= 0.981) (Figure 2.5). Nevertheless, there were numerous changes over the analysed period in key parameters of call's announcements, such as eligible expenses and timing of execution, project selection criteria and co-payment rates (Table S1).

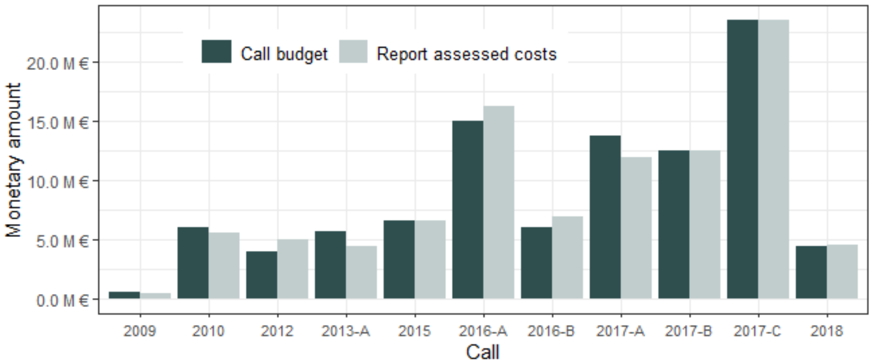


Figure 2.5 - Comparison of the costs estimated by the reports to implement postfire emergency stabilization measures with the corresponding call budget (M €).

Calls eligible expenses changed over time, with inclusion of additional treatments such as recovery of water points, fences, installation of shelters and feeders for wildlife, among others. Additionally, timing of execution also shifted. Initially, expenses were eligible if executed within the 12 months after fire occurrence (2009/2010) or until the end of the financing program (2012/2013). In 2015, expenses were eligible if executed after project submission; since 2015, if not fully executed before project submission.

The most important criteria for project selection between 2009 and 2013 were project location (such as natural classified areas or public forest regime), followed by beneficiary type and project treatment area. After 2014, burned area > 750 ha became the most important criterion for project implementation, followed by project location within areas with forest management in association or with protection status. Furthermore, in 2009-2013 the tiebreaker amid projects was project area, which was replaced after 2014 by fire area, percentage of forest stands affected, and location within protected areas.

Once again, co-payment rates also changed between funding programmes. In the first programme (2009-2013), eligible expenses were supported at 100%. In the second programme (2014-2020), reimbursement level became variable (50 to 100%) according to the type of beneficiary and expense.

2.3.4. Analysis of emergency stabilization projects

Project approval rate was 91% in 2009-2013 and 100% in 2014-2018. Approved projects were heterogeneously distributed among the 121 reports with open calls, with approximately 67% of reports with 3 projects or less. Additionally, 12 reports had no approved projects, despite available funding. Part of this discrepancy may result from lack of available information to link 11 projects to reports in the 2014-2018 period.

Public entities were the type of beneficiary with the highest number of approved projects in both programs (62% and 71% in the first and second program, respectively). Forest producers' organizations showed the second highest number of approved projects in the first funding program (22%) but were less represented in the second funding program (13%), being replaced by common lands (6% in the first program and 15% in the second). Lastly, the total number of approved projects from private landowners decreased from 10% to 1% from the first to the second funding program, becoming the beneficiaries with the lowest number of projects in the most recent period.

Accordingly, public sector organizations were the main investors in almost all calls. Forest producers' organizations and common lands showed considerable variations in their importance, ranging from total absence to a large percentage of projects total budgets. Finally, private beneficiaries represented a very small percentage of projects total budget, except for a few calls (Figure 2.6 – left). Regarding the distribution of eligible expenses among beneficiaries (Figure 2.6 - right), the public sector was still the main beneficiary but there was a large variation in expenses distribution among the remaining beneficiaries Welch ANOVA indicated a significant effect of type of beneficiary on project total budget [$F(3, 496) = 3.63$, $p = 0.018$], but a non-significant effect on eligible expenses [$F(3, 496) = 1.64$, $p = 0.19$].

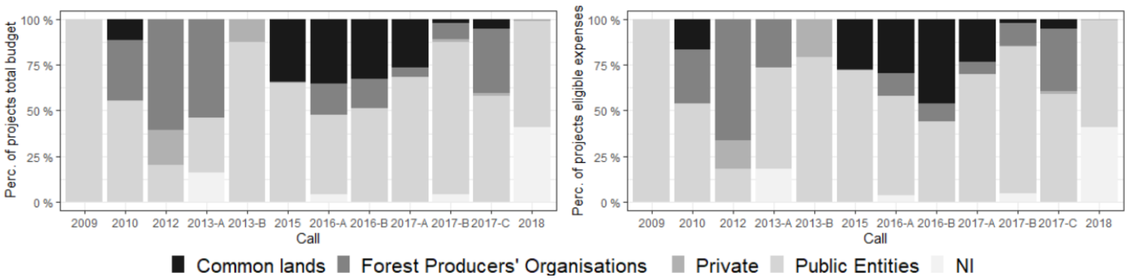


Figure 2.6 - Percentage of projects total budget by type of beneficiary, per call (left) and percentage of projects eligible expenses by type of beneficiary, per call (right).

Averaged project burned area within a report increased from 1,170 ha to 1,710 ha from the first (2009-2013) to the second funding period (2014-2018). On the contrary, there was a decrease in projects total budget per hectare of burned area from the first to the second funding period (299 € ha^{-1} to 190 € ha^{-1}).

Figure 2.7 shows the total costs estimated by each report for emergency stabilization (green line, equivalent to 100%) in comparison with total budget and eligible expenses of approved projects (aggregated by report), showing that projects total budget exceeded these costs for 58% of the reports. On the other hand, projects eligible expenses were lower than predicted costs in most situations (Figure 2.7).

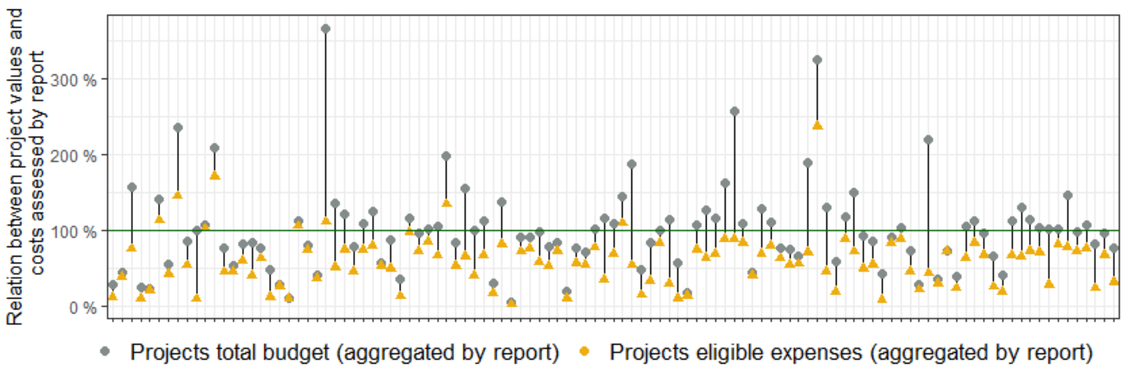


Figure 2.7 - Comparison of projects budget and eligible expenses (aggregated by report) with total costs assessed by report for emergency stabilization. Green line represents the total cost assessed by report (100%).

When comparing the projects financed amount with the respective call budget, a considerable amount of the calls budget supported by public funds was not used (<50% in 2009-2013 and < 72% from 2014 onwards) (Table 2.2).

Table 2.2 – Call budget execution rate based on project financed amount.

Call year	Call budget	Project financed amount	Budget execution
2009	0,60 M €	0,06 M €	9.8%
2010	6,00 M €	2,70 M €	45.0%
2012	4,00 M €	1,92 M €	48.1%
2013-A	5,70 M €	1,57 M €	27.6%
2013-B	-	2,58 M €	-
2015	6,65 M €	3,47 M €	52.2%
2016-A	15,00 M €	7,56 M €	50.4%
2016-B	6,00 M €	2,85 M €	47.6%
2017-A	13,74 M €	6,42 M €	46.7%
2017-B	12,50 M €	7,61 M €	60.9%
2017-C	23,50 M €	16,87 M €	71.8%
2018	4,50 M €	1,43 M €	31.7%

2.3.5. Chronological analysis of the funding process

Over the analysed period, the median time interval between fire extinction and project predicted starting date was 88 days, although this time was highly variable (interquartile range of 100). Note that this starting date is established by each applicant but may not correspond to the date when the project effectively began. The median time interval between fire extinction and project's decision of approval was 205 days (Figure 2.8).

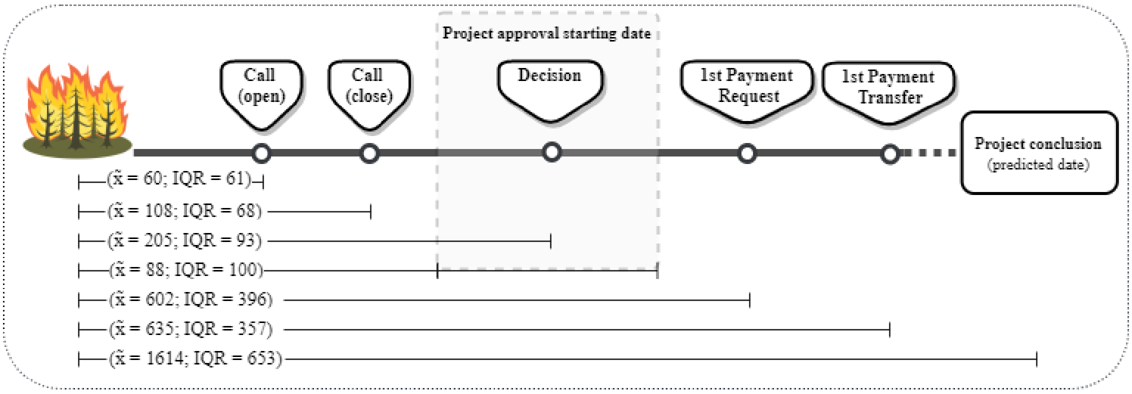


Figure 2.8 - Number of days (x̃ - median and IQR – interquartile range) between fire extinction and each stage of the funding process for postfire emergency stabilization in 2009-2018.

Furthermore, the median duration of the decision process between the end of the call and project approval was 110 days, being highly irregular among calls (Figure S2). The whole process lasted a median of 1,614 days (~4.5 years) between fire extinction and predicted project conclusion.

Nonetheless, calls started being launched sooner (between 25 to 50 days after fire extinction) and closing earlier (around 75 days after fire). Moreover, the time interval between fire extinction and the date predicted for the beginning of the project was reduced by 65% (from 222 days in 2009-2014 to 78 days in 2015-2018). Time intervals for other stages of the funding process are shown in Table S2.

2.4. Discussion

2.4.1. Major constrains of the public funding process for effective postfire restoration with emergency stabilization in Portugal

In our study, we identified three major constrains of the public funding process in Portugal that hinder an effective emergency stabilization in burned areas, namely: the financing model, an overly simplistic approach to eligible emergency treatments, and the slowness of decision-making and execution.

The financing model of the analysed funding programmes is based on non-refundable grants that reimburse eligible costs actually incurred and paid, and consequently demand beneficiaries to support the initial investment. This access scheme to financial resources has already been identified as an obstacle to forest management and restoration in Europe, leading to appeals to move from “degrading subsidies” to supportive financial assistance, while promoting innovative funding solutions (Fisher et al. 2019). This is particularly relevant, as rural development programmes (RDPs), financed by EAFRD, are the main public financial instruments to assist rural areas in terms of environmental aspects (Galiana et al. 2013; Colonico et al. 2022) and postfire restoration in Portugal is mostly implemented by the land owners/managers using public subsidies from European funding (Lamb 2009; Mavsar et al. 2012b; Thornton et al. 2020). In this sense, one of the aims of the new EU Forest Strategy for 2030 (European Commission 2021c) is to increase the interest of stakeholders' on restoration through more flexible processes that should be aligned with national specificities.

Regarding eligible treatments, there is a large variation of the costs assessed for different categories and constant changes in eligible categories across funding calls. As such, soil erosion mitigation is not eligible in 24.6% of the reports, despite being one of the most important treatments for emergency stabilization of burned areas since it can greatly reduce soil losses (estimated in 24 t ha^{-1} in the first postfire year) (Vallejo et al. 2012b; Fernández et al. 2019).

The slowness of decision-making and execution of the funding process was another major limitation for implementing restoration interventions within an optimal timeframe, which is a few months (usually 4) after the fire. We showed that the approval decision of the restoration project took a median of 7 months after fire extinction and that the whole funding process (until predicted project conclusion) lasted a median of 4.5 years, which is greatly beyond the period required for effective emergency stabilization actions.

In consequence, and considering the above-mentioned limitations, the financial capacity of the beneficiary will very likely condition the effective implementation of emergency stabilization actions. For example, the short-term intervention after fire that is necessary to attain a higher efficacy of treatments pressures the beneficiary to initiate execution immediately after the fire, but without any certainty of funding support since project approval usually occurs several months later, as our results show. In addition, the treatments needed for an effective restoration may not be eligible for funding and therefore need to be fully supported by the landowner/manager. In Portugal, the financial capacity of forest landowners to implement restoration projects is limited by small landholdings, landowners' absenteeism and landowner's resistance to associated management under a single responsible entity (Feliciano et al. 2015). In fact, our results show that available funding was predominantly used by public entities, despite 91% of private landownership in Portugal. Furthermore, recent changes in weighting criteria and levels of support by funding calls favour large-scale projects, usually managed by public entities or forest producers' organizations. Hence, if this financial capacity is lacking, it is most likely that landowners solely extract the most valuable wood as compensation for damages, leaving most of the burned areas to natural regeneration (Sousa 2011; Pereira et al. 2018b). However, salvage logging techniques may exacerbate the soil degradation process (Pereira et al. 2018b), especially on steep terrain or in soils with high erodibility (Vallejo et al. 2012b).

In the case of private and small-scale forest landowners in Portugal, the perception of a cycle of successive wildfires and consequent absence of return on investment, led to the believe that no interventions is the rational decision and the only economically viable option (Valente et al. 2015; Marques et al. 2020), based on a common risk avoidance strategy (Feliciano et al. 2015). To our knowledge, postfire emergency restoration projects without public funding are extremely rare in Portugal due to the high associated costs and the lack of financing capacity of small-scale landowners, although there is no available information on this topic specifically for Portugal (Thornton et al. 2020).

Nevertheless, there were positive improvements in the funding process over the analyzed period, namely: an increased production of postfire assessment reports; an increased diversity of restoration treatments eligible for funding; and a reduction of the time interval between fire extinction and calls announcement/closure and project predicted start date. In addition, there was an attempt in 2018 to expedite the implementation of postfire emergency stabilization through an exceptional regime by changes in public contracts and payments in advance; however, the application of such regime was still belated and without financial framework within the ongoing funding program (PDR2020) (Observatório Técnico Independente 2019b).

2.4.2. Necessary changes to improve the funding process for postfire emergency stabilization in Portugal

One of the most important changes needed to improve the funding process towards a more efective postfire emergency stabilization in Portugal is rapid reimbursement, to overcome the constrained investment capacity of private beneficiaries, who are the main landowners of Portuguese rural and forest land. One example of a funding program with fast approval process is the USA public funding program for postfire restoration in federal lands known as BAER (Burned Area Emergency Response), which has been largely used in the state of California, the region in the USA with Mediterranean climate and most affected by wildfires, and where 57% of forest lands are federal (Meyer et al. 2021). If emergency stabilization is necessary (through a pre-determined structure: the Burned-Area Emergency Assessment), an initial request for funding should be submitted within 7 days after total containment of the fire and the response is given within 3 days from submission (USDA Forest Service 2020). The costs are supported by agency's own funds, sometimes originating from federal government revenues from settlements or litigation for environmental damages caused by wildfires (USDA Forest Service 2020). There are also complementary programs such as the USA Emergency Forest Restoration Program (EFRP) (USDA 2022) that support owners of nonindustrial private forest to implement approved restoration practices (up to 75% of eligible expenses). EFRP is similar to the Portuguese funding process, as eligible owners must apply for funding, subject to approval. Another example is the process used in Galicia, Spain, a region with similarities with the portuguese reality in terms of human and physical factors. Here, damage assessment is made by a state agency in cooperation with a research center (Lourizán Forestry Research Centre), and a public company (SEAGA) implements the treatments, enabling restoration actions within days after the fire (Vega et al. 2013a; Fernández et al. 2019; Xunta de Galicia 2022). Galicia's scheme operates with internally managed funds and handles the organizational and logistical aspects, such as the human and material resources, including supplies for the treatments, ensuring its availability (Fernández et al. 2019).

Another necessary change is to separate the funding for emergency stabilization treatments from the funding for rehabilitation and recovery treatments. Current emergency stabilization reports in Portugal assess not only short-term emergency treatments, but also treatments such as fuel management, invasive species control or sowing/plantation, which are usually considered at the rehabilitation or at the recovery stage (medium and long-term stages) (Vallejo and Moreira 2010; Vega et al. 2013a). The inclusion of all these treatments as eligible expenses in the calls increases the complexity of the funding process, which is consequently prolonged in time, leading to a very delayed execution. This consequently leads to underperformance of the interventions, and conditions future restoration projects by ineffectively consuming resources (Higgs 1997; DGRF 2015). The creation of a separate funding specifically oriented for emergency stabilization would certainly increase the effectiveness of the whole process and facilitate the execution of emergency restoration actions within the required time interval after fire. An example of the proposed approach is the already mentioned United States BAER public program, which concurrently develops emergency stabilization and rehabilitation plans for the burned areas, but separates the funding process (US-BLM 2007). The BAER program is activated for all fires larger than 200 ha (and smaller fires if critical values are involved) and implements emergency stabilization treatments in the first year with a ''cost-risk analysis'', with eventual monitoring and follow-up treatments for up to 3 years (Meyer et al. 2021).

Public assessment reports and calls should also be revised for a careful re-evaluation of eligible treatments, based on technical information, monitoring data if available, and on total budget presented in the submitted projects. In fact, the large difference observed between projects total budget and eligible expenses suggests that beneficiaries aim to execute treatments that are not eligible by funding calls. As such, to improve the funding process, it would be relevant to analyse which other treatments are being considered by beneficiaries but are not eligible.

Additionally, in most cases, there is no clear link between the location of treatments identified in the reports and the location of those treatments in the projects submitted. Inclusion of the spatial location of required treatments in the reports would allow beneficiaries to optimize treatments, minimize costs and improve the efficiency of project execution. Finally, the absence of systematic monitoring is another factor that limits the effectiveness of emergency stabilization in Portugal since it does not allow assessing the impacts of treatments (DGRF 2015; Ribeiro et al. 2020), neither to assess if a follow-up or re-treatment is necessary (Nilsson et al. 2016; USDA Forest Service 2020).

2.4.3. Future perspectives

Postfire restoration is an extremely important tool for impact mitigation and ecosystem recovery in Europe, where wildfires present increasing socioeconomic and ecological impacts, particularly in southern European countries with Mediterranean-climate and vegetation (Rego et al. 2018). However, the rural territory in southern Europe (including Portugal) is characterized by rural poverty, aging population and rural abandonment (Mancini et al. 2018; Oliveira and Zêzere 2020), which may hinder the use of available funds that demand an initial effort of investment from the landowner (Valente et al. 2015), as shown by our results (general underspending of the money available within calls). Accordingly, the uptake of forestry measures (including forest restoration) supported by the European Common Agricultural Policy (CAP) in 2014-2020 was low and decreased considerably along the programming period due to insufficient attractiveness of the premium and the lack of capacity building support through advisory services, among other reasons (European Commission 2021c).

Postfire restoration funding policies need also to be tailored for the local stakeholders (landowners and managers), to motivate and engage them in the decision processes, likely leading to greater social acceptance of restoration (Ribeiro et al., 2015). For example, misperceptions about fire impacts can result in misperceptions about ecosystem recovery and in negative opinions on restoration and management actions (Kooistra et al. 2018). Furthermore, there has been a tendency in Europe in recent years to shift from the classic landowner mainly concerned with financial return to a multi-objective, environmentalist or indifferent/passive owner, which was also observed in

Portugal (Valente et al. 2015). These new emerging stakeholders, with different perceptions, values and attitudes towards land management, need to be considered in the future funding policy and process (Fabra-Crespo et al. 2012).

The recent EU Forest Strategy for 2030 (European Commission 2021c) aims to address targeted financial incentives to strengthen the engagement, motivation and dedication of forest landowners and managers for forest protection and restoration, particularly in regions with a higher vulnerability to climate change and where rural areas have suffered from the loss of income, livelihoods and even lives (due to wildfires for example), as is the case of Portugal (European Commission 2021c). Following European policies, recently launched national plans and programs in Portugal led to changes in the postfire restoration planning process, by strengthening the cooperation between stakeholders, and by reorganizing/clarifying the chain of processes and responsibility of each entity involved at postfire emergency stabilization, as a response to very large fires in previous years (Conselho de Ministros 2020a; Conselho de Ministros 2020b). The new planning process also envisages the need to motivate landowners to invest and manage their properties, including postfire restoration, for example through aggregated landscape units for joint management of small landholdings (Conselho de Ministros 2020c). The creation of an emergency fund specifically for postfire recovery was also proposed, although without detailed information on its operationalization.

This study provides a better understanding of how the funding process limits the efficacy of postfire emergency restoration in Portugal, contributing also to improve the general understanding of how to optimize the cost-effectiveness of environmental and forest policy measures funded by the European Union (Pacini et al. 2015). Postfire ecosystem restoration needs to be supported by funding that is aligned with policies and tailored for the local specificities, allowing for a technically sound and fast execution of interventions and overcoming landowner's investment limitations.

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2.7. Supporting Information

2.7.1. Tables

Table S1 - Temporal evolution of timing for execution, selection criteria and support levels

<table>
<tr><th>Program</th><th colspan="2">PRODER (2009-2013)</th><th colspan="3">PDR2020 (2014-2018)</th></tr>
<tr><th>Year</th><td>2009 -2010</td><td>2012 - 2013</td><td>2015</td><td>2016 - 2017</td><td>2018</td></tr>
<tr><th rowspan="2">Timing for execution</th><td colspan="2">Since fire occurrence.</td><td rowspan="2">After project submission.</td><td colspan="2">Since fire occurrence, as long as not fully executed before project submission.</td></tr>
<tr><td>Until a maximum period of 12 months after the fire.</td><td>Until December 31st, 2014.</td><td>-</td><td>Up to 4 months after project contracting in large scale areas in the case of abiotic treatments.</td></tr>
<tr><th>Selection criteria</th><td colspan="2">Ordered by decreasing order of importance: (a) within classified areas or under the forest regime; and (b) forestry intervention zones or common lands, followed by managing entities of grouped areas, local and central administration bodies, and finally, remaining beneficiaries. Tiebreaker by project treatment area.</td><td colspan="3">Ordered by decreasing order of importance: (a) burned area (if > than 750 ha); (b) within forest intervention zones or common lands; (c) within natural protected area or area susceptible to desertification. Tiebreaker by percentage of forest stands affected and extent of protected area.</td></tr>
<tr><th>Support levels</th><td colspan="2">Supported at 100%, with maximum limits per type of beneficiary, initially ranging from 75 000 € to 1 million €. Posteriorly unified to 1 million € per beneficiary (08/2010) and later to 2.5 million € (06/2011).</td><td colspan="3">Variable according to the type of expense and beneficiary. Ranging from 100% for public entities, forestry intervention zones or common lands, in large scale areas and without equipment acquisition, to 50% for equipment acquisition and for specific types of beneficiaries.</td></tr>
</table>

Table S2 - Time intervals between different stages of the funding process (Median, interquartile range and extreme values).

Time intervals in days. N corresponds to the number of observations. x̃ used as symbol for median.

	2009-2013					2014-2018					2009-2018				
Period	N	x̃	IQR	Min	Max	N	x̃	IQR	Min	Max	N	x̃	IQR	Min	Max
Fire extinction and approved starting date	111	222	232	0	788	374	78	44	8	330	485	88	100	0	788
Fire extinction and predicted date for project conclusion	113	893	290	322	1649	374	1656	547	473	2418	487	1614	653	322	2418
Fire extinction and 1st payment request	113	491	272	226	1038	211	713	385	153	1577	324	602	396	153	1577
Fire extinction and 1st payment transfer	113	545	246	249	1080	162	696	395	225	1570	275	635	357	225	1570
1st payment request and transfer	113	65	37	18	286	162	67	67	6	405	275	67	49	6	405
Fire extinction and date of approval	108	208	63	142	450	339	205	116	111	618	447	205	93	111	618
End of the call and date of approval	108	57	33	2	340	340	120	58	40	349	447	110	76	2	349
Date of approval and 1st payment request	107	323	223	90	860	161	471	391	49	1276	268	408	340	49	1276
Approved starting date and date of approval	107	-17	237	-552	234	339	119	59	-102	349	446	106	84	-552	349
Fire extinction and opening of the call	32	99	50	27	307	89	37	45	18	203	121	60	61	18	307
Fire extinction and closure of the call	32	143	39	102	339	89	84	41	55	269	121	108	68	55	339

2.7.2. Figures

Figure S1 - Average number of treatment categories for postfire emergency stabilization in emergency stabilization reports in 2009-2018.

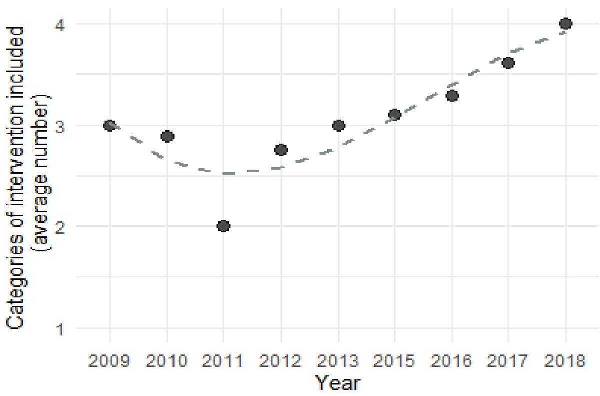
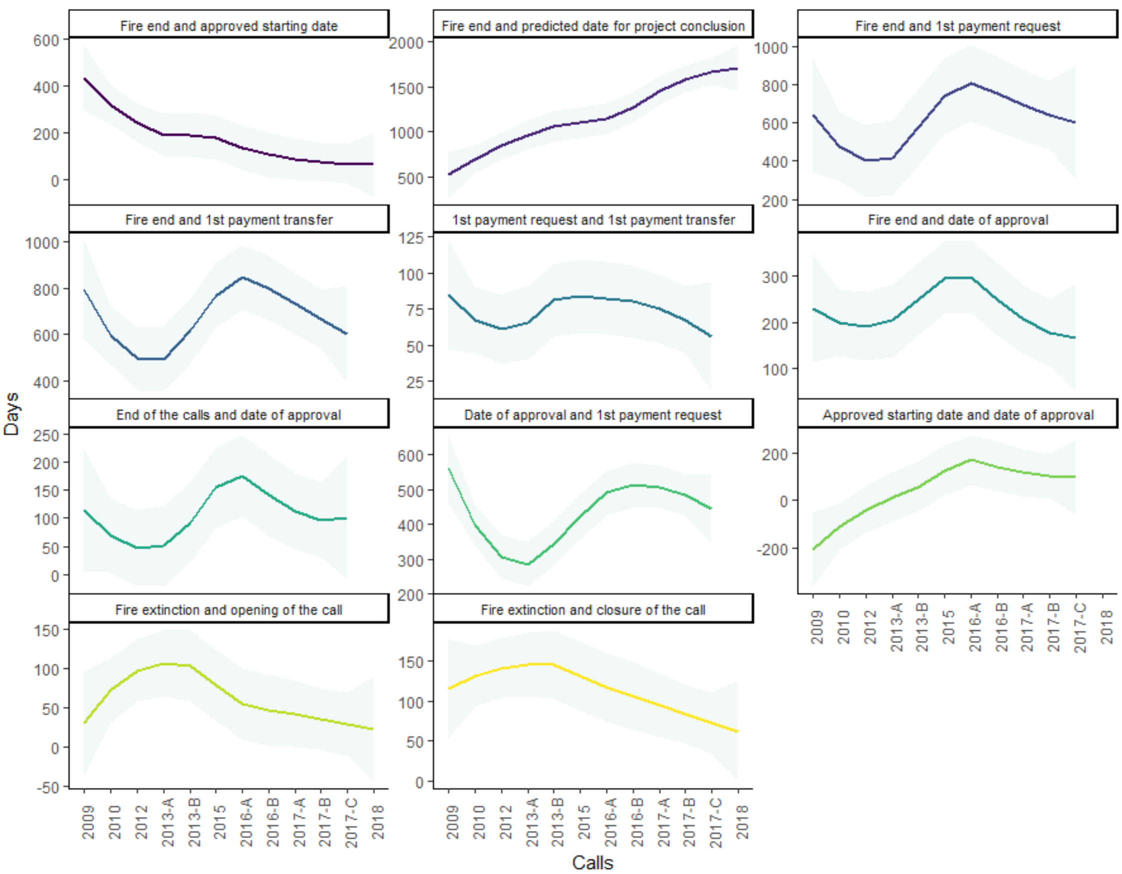


Figure S2 - Time intervals per call (tendency lines based on median values).



Chapter 3 - A remote sensing assessment of oak forest recovery after postfire restoration

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Abstract

Mediterranean Europe is witnessing an increase in extreme wildfires, which has led to increasing socioeconomic and ecological impacts. Postfire restoration emerges as an important tool for impact mitigation and ecosystem recovery, but the ecological effects of such interventions remain poorly understood.

We used remote sensing to assess the impacts of postfire restoration on the recovery of deciduous oak forests in Portugal, based on 3013 sampling points in areas with and without postfire interventions. We quantified vegetation indexes as proxies of oak forest recovery for a period of 4 years after the fire, for fire events that occurred in 2016 and 2017. We used a Generalized Additive Mixed Model (GAMM) to analyse temporal changes in NDVI as a function of postfire restoration, fire characteristics, topography and postfire drought events. The GAMM showed that postfire restoration had a significant positive effect on NDVI recovery over time, although this effect was small. Severe drought and fire recurrence up to a maximum of 6 fires had a negative effect on NDVI recovery. On the opposite, severe wetness, and low or high burn severities had a positive effect on recovery.

Our study emphasizes the importance of monitoring postfire restoration effects on forest recovery and identifies deciduous oak stands as potentially valuable for implementing postfire emergency stabilization measures, supporting regeneration capability and enhancing ecosystem resilience. This becomes even more relevant under forecasted scenarios of increased wildfire frequency and severity interacting with other climate-driven disturbances, which will further impact the capacity of forest recovery.

Keywords

Postfire restoration; Remote sensing; Vegetation recovery; Wildfires; Vegetation indices; Deciduous oaks

3.1. Background

In the last decades, global changes have led to a change in wildfire regimes (e.g., frequency and severity) across the world, with an increase of extreme wildfire events (Rogers et al. 2020; Pausas and Keeley 2021; Harvey and Enright 2022). In Mediterranean Europe in particular, and despite the decreasing trend in annual burned area since the 1980s, there has been an increment in extreme wildfire events, which has led to growing socioeconomic and ecological impacts (Tedim et al. 2015; Doerr and Santín 2016; Turco et al. 2016; Fernandez-Anez et al. 2021). For example, changes in wildfire regimes have impacted Mediterranean forests in different ways, by converting forests into shrublands in a self-reinforcing feedback loop, by creating more connected and homogeneous vegetation patches with increased wildfire risk, and by forcing forest management to adapt to the new wildfire regime (e.g., selection of species with higher fire resilience or managed with short-rotation silviculture), among other impacts (Moreira et al. 2011; Vallejo et al. 2012c; Mateus and Fernandes 2014; Navarro and Pereira 2015; Tessler et al. 2016; Tepley et al. 2018; Francos and Lemus Canovas 2021).

Wildfires affect plant survival, regeneration and growth (Bright et al. 2019; Pausas and Keeley 2019; Castro Rego et al. 2021a) and therefore modify the structure, composition, and dynamics of ecosystems. Thus, fire impacts can limit or delay forest recovery (McLauchlan et al. 2020). Fire characteristics that may influence postfire forest recovery include fire return interval, time since last fire and burn severity. Longer fire return intervals (time between successive fires) tend to favour the establishment and growth of woody plants such as shrubs and trees, while more frequent fires (recurring at shorter intervals) can compromise the regenerative capacity of woody plants by preventing the establishment and growth of seedlings and saplings. A longer period of time since the last fire influences the composition and structure of plant communities, with early successional species being gradually replaced by mid-successional and late-successional species that are more competitive and shade-tolerant. A higher burn severity will likely lead to a more limited/lower postfire regeneration of vegetation, when compared to lower burn severity, because fires with higher severity will cause a higher removal of vegetation, a stronger decrease in soil fertility and structure, and a lower seed availability and germination (Moreira et al. 2012a; Tepley et al. 2018; McLauchlan et al. 2020; Castro Rego et al. 2021a).

The rate of postfire forest recovery is also influenced by other factors such as postfire climatic conditions and topography. Postfire weather conditions can be determinant for vegetation response (Pausas 1999a; Torres et al. 2018a; McLauchlan et al. 2020), especially precipitation and temperature (Díaz-Delgado et al. 2002; Torres et al. 2018a), facilitating or hindering their regrowth and recovery from the damage caused by the fire (Millington et al. 2009; Parra and Moreno 2017). Topography has the potential to alter microclimatic conditions (Correia et al. 2015), thus affecting tree growth and its resilience to disturbances (Roder et al. 2008; Viana-Soto et al. 2017). Additionally, site topography (e.g., slope) directly affects fire spread, which will indirectly affect fire impacts (burn severity) (Hawbaker et al. 2013; Fernandes et al. 2016b). The trajectories of forest recovery after a wildfire may also be conditioned by the regeneration capacity of the dominant tree species. Resprouting individuals can persist after fires, resulting in long-lived specimens and more stable populations (Pausas 1999a; Lamont et al. 2011). Additionally, resprouter species regenerate vegetatively after fire and can therefore accomplish faster postfire ground cover, in comparison to seeders (Pausas 1999a; Pausas 1999b; Maia 2014).

Lastly, forest recovery following a wildfire may be further conditioned by postfire restoration interventions (Moreira et al. 2012a), which are commonly grouped into three stages: (i) emergency stabilization (applied in the short-term after fire extinction), (ii) the intermediate stage (up to 2 years after the fire), and (iii) the recovery stage (long-term) (Moreira et al. 2014). Postfire emergency stabilization is important for efficiently stabilizing the area and preserving the soil, particularly when the risk of soil erosion is high and plant regeneration is slow (Vallejo et al. 2012c; Vega et al. 2013b). Emergency stabilization measures should be implemented shortly after the wildfire and, if possible, before the first autumn rains, in order to prevent further degradation, mitigate risks to people and assets, and minimize the significant loss of ashes and soil that typically occurs within the initial four months after the fire (Vallejo et al. 2012c; Ferreira et al. 2015; Girona-García et al. 2021). However, emergency stabilization may also cause negative impacts; for example, mulching may cause vegetation suppression by creating a physical barrier that inhibits the establishment of native vegetation (Fernández and Vega 2016; Bontrager et al. 2019).

In recent decades, there has been a transformation of European Union policies towards the restoration of forest ecosystems affected by wildfires (Faivre et al. 2018). However, monitoring of vegetation recovery after postfire restoration is seldom done in Europe (Ribeiro et al. 2020) and there is a large knowledge gap on the ecological effects of postfire restoration (Robichaud et al. 2009), with very few studies available (e.g. (Bontrager et al. 2019). Postfire forest recovery can be assessed with remote sensing techniques and indices (Bastos et al. 2011; Gitas et al. 2012; Viana-Soto et al. 2017; Hislop et al. 2018; Gouveia et al. 2018; Torres et al. 2018a; Meneses 2021; Pérez-Cabello et al. 2021), including assessments under different types of restoration (e.g., Chen et al. 2014), or related with the implementation of postfire restoration treatments (Vo and Kinoshita 2020; Carrari et al. 2022). Remote sensing provides biophysical measurements in a time-efficient and less costly manner, in comparison to fieldwork (Szpakowski and Jensen 2019; Pérez-Cabello et al. 2021). In addition, the recent developments in remote sensing are leading to a transition from bitemporal to continuous approaches, allowing analysis of spectral trajectories across regular time series

(Woodcock et al. 2020; Pérez-Cabello et al. 2021). By supplying frequent medium to high resolution (<100 m) observations, it is possible to monitor short-term postfire dynamics (Szpakowski and Jensen 2019; Pérez-Cabello et al. 2021).

Portugal is the southern European country most affected by wildfires, with large burned areas resulting from a small number of fire events, with extraordinary negative environmental, social, and economic impacts (Pereira et al. 2006; Mateus and Fernandes 2014; Tonini et al. 2017). In Portugal, wildfires occur predominantly in the northern and central regions (Pereira et al. 2006; Tonini et al. 2017), once dominated by native deciduous oak forests (reaching a maximum between ca. 9000 BP and ca. 5000 BP), but today replaced mostly by eucalypt and pine forests Deciduous oak forests in Portugal have decreased gradually over time, mostly due to anthropogenic disturbances (Fletcher et al. 2007; Reboredo and Pais 2014). In the present, only some original traces of those forests remain, across small fragmented patches (Silva 2007; ICNF 2019a). Oak species are resprouters, with the ability to regenerate after major disturbances such as wildfires, even at young ages (Alves et al. 2018; Marañón et al. 2020). Such postfire regeneration capacity, in association with wetter environments and subsequent fire resistant fuels commonly found in mature oak forests (Dimitrakopoulos and Papaioannou 2001; Fernandes 2009), turn these forests into fire-resilient landscapes (Díaz-Delgado et al. 2002; Botequim et al. 2017), able to recover forest canopy in about four years after fire (Calvo et al 2002 for *Quercus pyrenaica*).

In Portugal, postfire restoration has been increasingly integrated into national policies, especially since the beginning of the twenty-first century, being implemented mainly through public funding subsidized by EU funds (Ribeiro et al. 2018). Furthermore, since 2007, emergency stabilization measures have been incorporated into these funding programs (Pinho 2017; Ribeiro et al. 2018). However, postfire restoration in Portugal is often ineffective (Lopes et al. 2023) and the lack of systematic monitoring constrains treatments optimization (Observatório Técnico Independente 2019c; Ribeiro et al. 2020). In particular, little is known about postfire restoration effects on the regeneration of Mediterranean deciduous oaks (Espelta et al. 2012). In this study, we analysed the effects of postfire emergency stabilization on oak forest recovery for a period of 4 years after fire, for fire events that occurred in 2016 and 2017, by comparing recovery between areas with and without postfire emergency stabilization interventions. We asked the questions: i) Did postfire emergency stabilization have a positive impact on the recovery of oak forests? and ii) How did fire characteristics, topography and postfire drought events affected oak forest recovery? Assessing the effects of postfire restoration on forest recovery will contribute to improving knowledge on the effectiveness of implemented practices and optimize postfire forest management, particularly under increased wildfire severity predicted for the Mediterranean region (Dupuy et al. 2020; Rogers et al. 2020; Lucas-Borja et al. 2021).

3.2. Methods

3.2.1. Study area

Mainland Portugal (latitude 37º-42ºN, longitude 6º-10ºW) is positioned in the extreme southwest of continental Europe, on the Iberian Peninsula. The Portuguese climate is Mediterranean, with a north-south gradient (Mora and Vieira 2020). There are considerable physiographic differences between northern and southern regions, with the northern half having a more rugged topography, a denser river network, and the majority of forests and semi-natural areas (Tonini et al. 2017). In 2015, Portugal was predominantly covered by forests (36%), shrublands and pastures (31%), and agricultural land (23%) (ICNF 2019a). The main forest species in the country are blue gum eucalypt (*Eucalyptus globulus*), cork oak (*Quercus suber*) and maritime pine (*Pinus pinaster*), with 26.2%, 22.3% and 22.1% of total forest area, respectively. Deciduous oaks account for only 2.5% of total forest area (ICNF 2019a).

3.2.2. Selection of spatial sampling units

We collected all the projects with postfire emergency stabilization interventions after the fires of 2016 and 2017, which were approved and funded by the Portuguese Rural Development Program. The expenses eligible by this funding program include interventions such as mulching, erosion barriers, shredding of organic/forest residues, and also sowing/planting (normally considered as intermediate-stage interventions) (Ministério da Agricultura e do Mar 2015). Each project may include any of these interventions implemented at one or several parcels, and each parcel is identified by a unique number. Project data and parcel identification numbers were provided by the Financing Institute for Agriculture and Fisheries (IFAP) and parcel spatial location was obtained from IFAP online services, which show the parcel coordinates based on its identification (ICNF 2021).

Spatial data on oak land cover was extracted from the 2015 National Land Use and Land Cover (LULC) Map (hereafter COS2015, Cartografia de Ocupação do Solo 2015, Direção-Geral do Território 2022). We selected “oak forests” land cover class from COS2015, which includes three main species, namely, Pyrenean oak *(Quercus* *pyrenaica* (Willd.)), Portuguese oak (*Quercus faginea* (Lam.)), and Pedunculate oak (*Quercus robur* (L.)), since map categories do not distinguish the deciduous oak species. This class is dominated by deciduous oaks, including pure stands or mixed stands (with other broadleaves).

Burned areas between 2016 and 2017 were collected from the national database of burned areas, which contains this information in vector format since 1975 (ICNF 2021).

We intersected the three spatial layers (in vector format), namely, (i) polygons (parcels) with postfire restoration projects, (ii) polygons dominated by deciduous oak forests and (iii) burned areas, in order to obtain the final polygons for analysis, which corresponded to areas dominated by oak forest that burned in 2016-2017 where postfire restoration was subsequently implemented. Polygons with an area lower than 1000 m^2 were excluded from further analysis.

We created a buffer area of 500 meters around each final polygon to produce control polygons without postfire restoration interventions. Control polygons were also dominated by deciduous oak forests that burned in 2016-2017, and with at least 1000 m^2. Contiguous buffers were aggregated to constitute unique spatial units (with a single identification number), in a total of 60 final units.

Lastly, we created random points within the final polygons (differentiated between intervention and control points, if located within polygons with postfire restoration or within control polygons, respectively), with a minimum distance of 100 m between points. In total, 3013 random points were originated, of which 43.6% (*n* = 1314) were located within polygons with postfire restoration projects, and 56.4% (n = 1699) within control polygons (Figure 3.1). Spatial analysis was performed in ArcGIS Pro (Esri Inc. 2022). Random points were used as the sampling units for data collection and analysis.

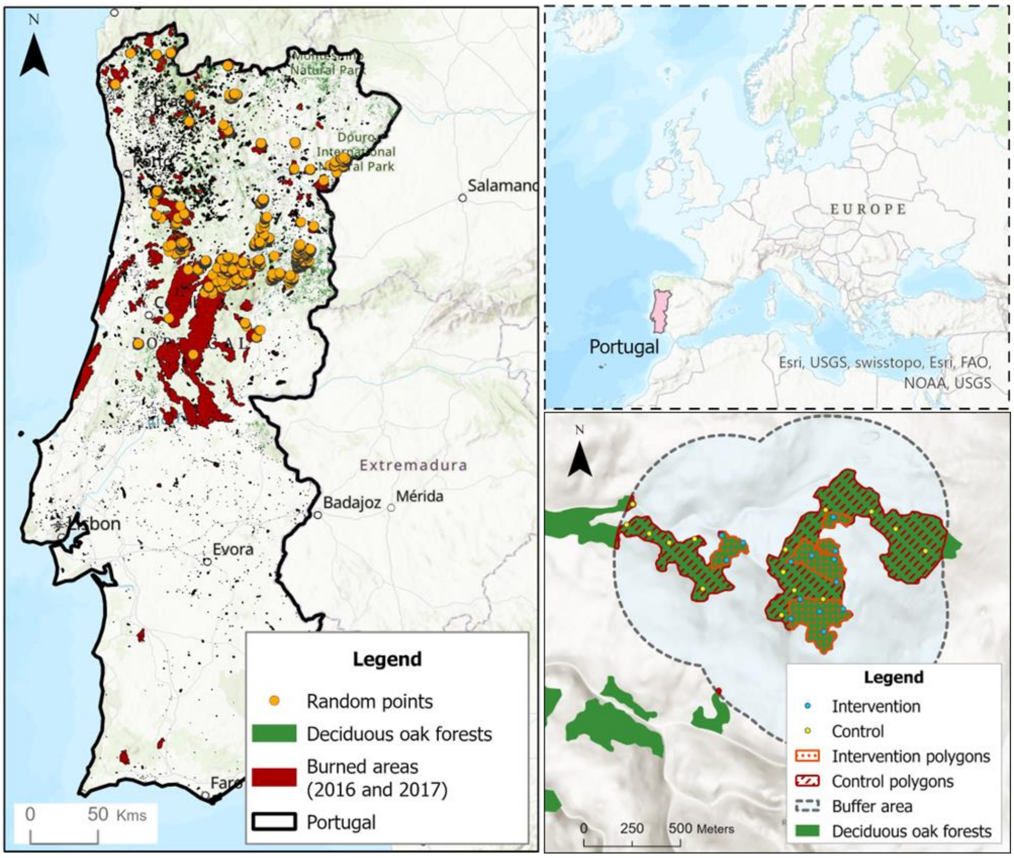

Figure 3.1 - Spatial location of the random points analysed, burned areas between 2016 and 2017, and deciduous oak forests (Left). Spatial location of Portugal in Europe (Upper right). Representation of random points (intervention and control), polygons with postfire restoration projects and corresponding control polygons, buffer area of 500 m and deciduous oak forest land cover (Lower right).

3.2.3. Data collection

For each sampling point, we collected remote sensing data using the Google Earth Engine (GEE) platform (Gorelick et al. 2017) and used GIS to obtain complementary spatial data on variables related with fire characteristics and topography (Table 3.1). We used data from Sentinel-2 and QA (Quality Assurance) band to mask clouds (clouds and cirrus) in images, thus ensuring clear conditions.

We quantified two vegetation spectral indices: the Normalized Difference Vegetation Index (NDVI) and the Modified Soil Adjusted Vegetation Index 2 (MSAVI2). NDVI is widely used to analyse the state of recovery of burnt vegetation (Paci et al. 2017; Torres et al. 2018a; Pérez-Cabello et al. 2021), and is the most robust vegetation index for vegetation recovery assessments (Veraverbeke et al. 2012; Szpakowski and Jensen 2019), showing a strong relationship with aboveground biomass for a wide range of ecosystems (Gitas et al. 2012). Since all vegetation indices have advantages and disadvantages, it is common to work simultaneously with more than one index

(Wegmann et al. 2016). Therefore, we selected MSAVI2 since it minimizes the effect of bare soil of the Soil Adjusted Vegetation Index (SAVI), which is in turn used to rectify the effect of soil brightness in areas where vegetative cover is low (Qi et al. 1994). Both vegetation indices range from -1 to 1, with negative values indicating clouds and water, positive values near zero indicating bare soil, and higher positive values indicating denser vegetation (the closer to 1, the denser the vegetation). Postfire vegetation indices (NDVI and MSAVI2) were used as proxies of oak forest recovery and were calculated as monthly median values for the 48 months after the fire, for each random point. Monthly medians were computed to diminish the leverage effect of extreme values.

We also quantified the differenced Normalized Burn Ratio (dNBR) for each random point. The index dNBR is important to discriminate different degrees of severity on surfaces affected by fire (Keeley 2009) and is commonly used for burn severity assessment (Szpakowski and Jensen 2019). To obtain the dNBR, we calculated the difference between the Normalized Burn Index (NBR) for prefire and postfire (NBR prefire - NBR postfire) and used the time interval of 21 days before the fire start date and 21 days after the fire end date. To ensure accurate assessment of burn severity, we calculated dNBR offset to account for temporal and seasonal differences between pre-fire and post-fire satellite images. For that, we used a total of 1996 random points located outside the fire perimeter (> 1000 m) and within the same vegetation type (deciduous oaks), obtaining a mean dNBR value of -16.9, which is within the dNBR offset acceptable range (-50 to 50) (Picotte et al. 2020). Burn severity levels employed correspond to those established by the United States Geological Survey (USGS) (Key and Benson 2006). Due to the possible existence of unburned islands inside the burned areas (Tedim and et. al. 2018), 11.2% of sampling points (n = 338) presented dNBR values that corresponded to the levels of "Unburned" and "Regrowth" (dNBR < 100, Key & Benson (2006)). Therefore, those sampling points were excluded from subsequent analysis, with a total of 2675 points remaining.

We collected additional variables for each random point, namely: (i) fire recurrence (fires occurred between 1975 and the analysed fire event, 2016 or 2017) and time (number of years) since the last fire (before 2016 or 2017), both collected from the national database of burned areas, which starts in 1975 (ICNF 2021); (ii) postfire monthly values of the Palmer Drought Severity Index (PDSI), which uses temperature and precipitation records to estimate relative dryness (values between -1 and -4 correspond to different drought levels) from TerraClimate (Abatzoglou et al. 2018); and (iii) topographic information (aspect, elevation, and slope) from the NASA DEM Digital Elevation Model (NASA JPL 2020).

Data collected to characterize the sampling points are shown in Table 3.1.

Table 3.1 - Data collected, respective time periods, resolution, and data sources. *Depending on the fire year (2016 or 2017).

Group	Variable	Period*	Temporal resolution	Spatial resolution	Data source
Fire variables	Fire recurrence Time since the last fire	1975-2015 1975-2016	Minimum detectable area variable over time. Between 1975-1983 = 35ha; 1984-2004 = 5 ha; since 2005 it is possible to detect smaller fires		National database of burned areas (ICNF 2021)
	Burn severity (dNBR)	2016-2020 2017-2021	Monthly	10 m	Sentinel-2 (Copernicus Sentinel 2022)
Postfire vegetation indices	NDVI MSAVI2				
Postfire drought	PDSI		Monthly	4638 m	TerraClimate (Abatzoglou et al. 2018)
Topography	Aspect Elevation Slope	-	-	30 m	NASA SRTM Digital Elevation (NASA JPL 2020)

Spatial analysis was performed in ArcGIS Pro (Esri Inc. 2022).

3.2.4. Data analysis

To evaluate the impact of postfire restoration projects, fire variables, topography and postfire drought events on oak forest recovery, we modelled changes in time (48 months after fire) of two vegetation indices, NDVI and MSAVI2, as a function of burn severity, time since the last fire, fire recurrence, PDSI, aspect, elevation, and slope (see Table 3.1**Error! Reference source not found.**), using a Generalized Additive Mixed Model (GAMM) from the R package *mgcv* (Wood 2022). The effect of postfire restoration was added to the model as a factorial variable called "Intervention" which takes the value 1 for restored areas and 0 for control areas. We also added the variable *Time* that indicates how much time has passed since the fire. Since NDVI/MSAVI2 values present temporal correlation, we added a correlation structure of order 1 (corAR1) to the model's residuals. We used the identification of each buffer (*buffer* id) as a random effect to account for discrete spatial heterogeneity.

We fitted GAMMs, one for each vegetation index (MSAVI2 and NDVI). As preliminary analysis, we evaluated the pairwise relationship using pair plots, in order to assess the association between each predictor variables and each vegetation index. However, we observed that MSAVI2 was poorly related to the predictors, and only NDVI provided a valid model, corroborated through model checking tools. Hence, MSAVI2 model was discarded from further analyses. The final predictor variables selected for the NDVI model were postfire drought (PDSI), burn severity, fire recurrence, Time (that accounts for temporal correlation), and the factorial variable Intervention. To improve model fitting we transformed the variable PDSI by log(PDSI+6). We added 6 to avoid negative values. Statistical analyses were carried out with R (4.0.3) statistical environment (R Core Team 2020). Data were handled with the support of *Dplyr* and *Tidyr* packages (Wickham 2020; Wickham et al. 2020). Except when specified, statistical analyses were performed with base R functions.

R scripts with data analysis are shown as Supporting Information (Annex I).

3.3. Results

3.3.1. Characterization of sampling points: fire characteristics, postfire drought and topography

The different levels of burn severity showed a similar distribution between sampling points with and without (control) postfire interventions, with the moderate-high burn severity level being the most frequent (33.64%), followed by high (26.13%), and moderate-low levels (24.07%) (Table 3.2). The points with values equivalent to "Unburned" and "Regrowth" (dNBR < 100) were removed from the model analysis, as already mentioned in the methods section (and were not included in Table 3.2).

Table 3.2 - Contingency table of burn severity and intervention

Burn severity levels	Regrowth	Unburned	Low	Moderate-low	Moderate-high	High	Total
dNBR values	-250 ≥ & < -100	-100 ≥ & < 100	100 ≥ & < 270	270 ≥ & < 440	440 ≥ & < 660	≥ 660	Total
Points with postfire Intervention	7 (0.2%)	140 (4.6%)	223 (7.4%)	287 (9.5%)	371 (12.3%)	286 (9.5%)	1314 (43.6%)
Control Points	2 (0.1%)	189 (6.3%)	209 (6.9%)	357 (11.8%)	529 (17.6%)	413 (13.7%)	1699 (56.4%)
Total	9 (0.3%)	432 (10.9%)	432 (14.3%)	644 (21.4%)	900 (29.9%)	699 (23.2%)	3013 (100%)

Table 3.3 shows the descriptive statistics of the predictor variables used in the initial model. The altitude of random points locations ranged from 22 m to about 1300 m and the majority of the sampling points was exposed to ESE (East-South-East, between 90º and 135º) or WNW (West-North-West, between 270º and 315º) aspects, with highly variable slopes. Fire recurrence ranged from 0 to a maximum of 10 fire events in 1975-2016/2017, from which 24% of sampling points (650) never burned, 44% (1182 points) burned once or twice, and 32% (843 points) burned 3 or more times. Time since last fire was also highly variable, ranging from 1 year until a maximum of 42 years (for points that never burned), in accordance with the variability of fire recurrence. Burn severity, measured by dNBR, ranged from 100 (defined as minimum limit) to 1171.

Table 3.3 - Descriptive statistics of predictor variables. Aspect (º) was excluded from the table due to its 0-360 scale. Refer to the text for further details. [a] The values are limited by the temporal resolution, which only allows a maximum value of 42 years (1975-2017).

Group	Variable	Min	Mean	Max	SD
Fire characteristics	Fire recurrence	0	2	10	1.9
	Time since last fire [a] (years)	1	15	42	9.3
	Burn severity (dNBR)	101	503	1171	218
Postfire drought	PDSI	-5.5	-1.8	3.4	2
Topography	Elevation (m)	22	652	1326	197
	Slope (º)	0	10	47	8

Postfire drought (PDSI monthly values) showed similar seasonal patterns across sampling points over the two analysed periods (2016-2020 or 2017-2021, depending on the fire event) (Figure 3.2). For the majority of sampling points, severe to extreme drought levels (PDSI< -3) were registered in the first 3 to 5 months after fire, followed by mild to moderate drought levels (-1<PDSI<-3). Normal to moderate wet periods (3<PDSI<0) were mostly concentrated between the 5^{th} and the 15^{th} months after the start of both 2016 and 2017-fires. In contrast, the sampling points located in fires that started in September 2017 (n = 14) showed an unexpected and opposed pattern; these points belong to two spatially close buffers (1.7 Km - distance), forming a cluster situated on a south-east facing hillside near a village, which may be subjected to specific microclimatic conditions.

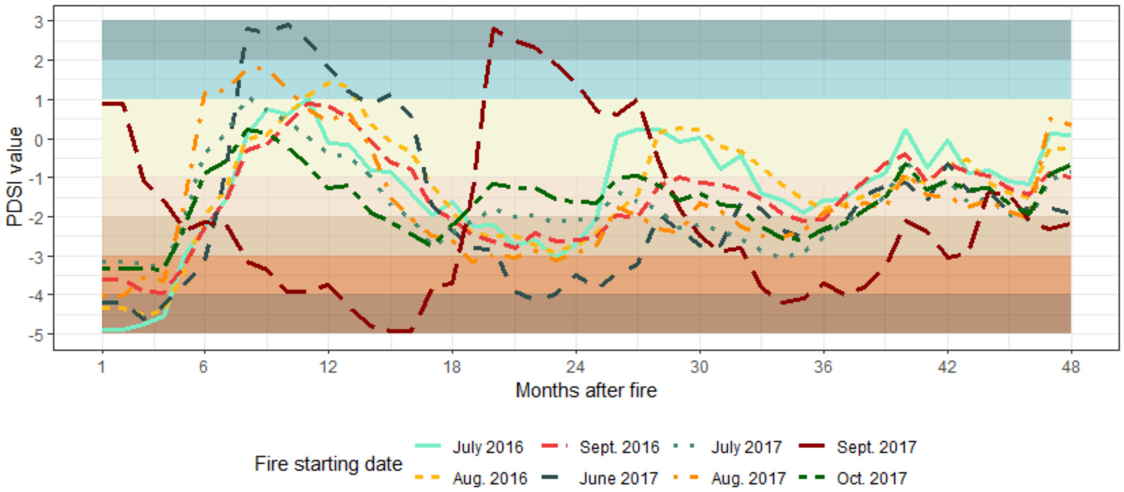

Figure 3.2 - Mean monthly PDSI values over the 48 months after fire, grouped by fire starting date. Sampling points with the same fire-starting month are grouped with the same line colour, and years are differentiated by lighter (2016) or darker (2017) colours. Colour areas in the background indicate PDSI categories, as seen in (Pires 2003).

3.3.2. Effects of postfire restoration, fire characteristics and postfire drought events on oak forests recovery

Overall, the model explained 26.4% of the variance of NDVI, with model checking revealing a well-fitted model (see SM – Annex I). Results show that the effect of postfire restoration intervention was weakly positive but statistically significant (β = 0.011, p < 0.001). This result means that the NDVI in restored areas tended to be 1% higher than in control areas.

The smooth terms of covariables were all significant (p < 0.001). Partial effects plots (Figure 3.3) show the effect of each variable on NDVI recovery rate (model response), namely: i) a negative effect of severe drought events and a positive effect of severe wetness; ii) a positive effect of low burn severities (dNBR < 270), a null effect of moderate-low (270 > dNBR > 440) and moderate-high burn severities (440 > dNBR > 660), and an increasingly positive effect with higher burn severities (dNBR > 660); iii) an increasingly negative effect of fire recurrence (one or more fires), which appears to achieve its maximum with a probability of 5 or 6 fires. Above 6 fires, there is a decrease of the negative effect, associated with wider confidence intervals. Lastly, the variable time shows a well-adjusted undulating trend (small confidence intervals) associated with yearly seasons, but with a clear recovery over time as shown by the maximum annual values.

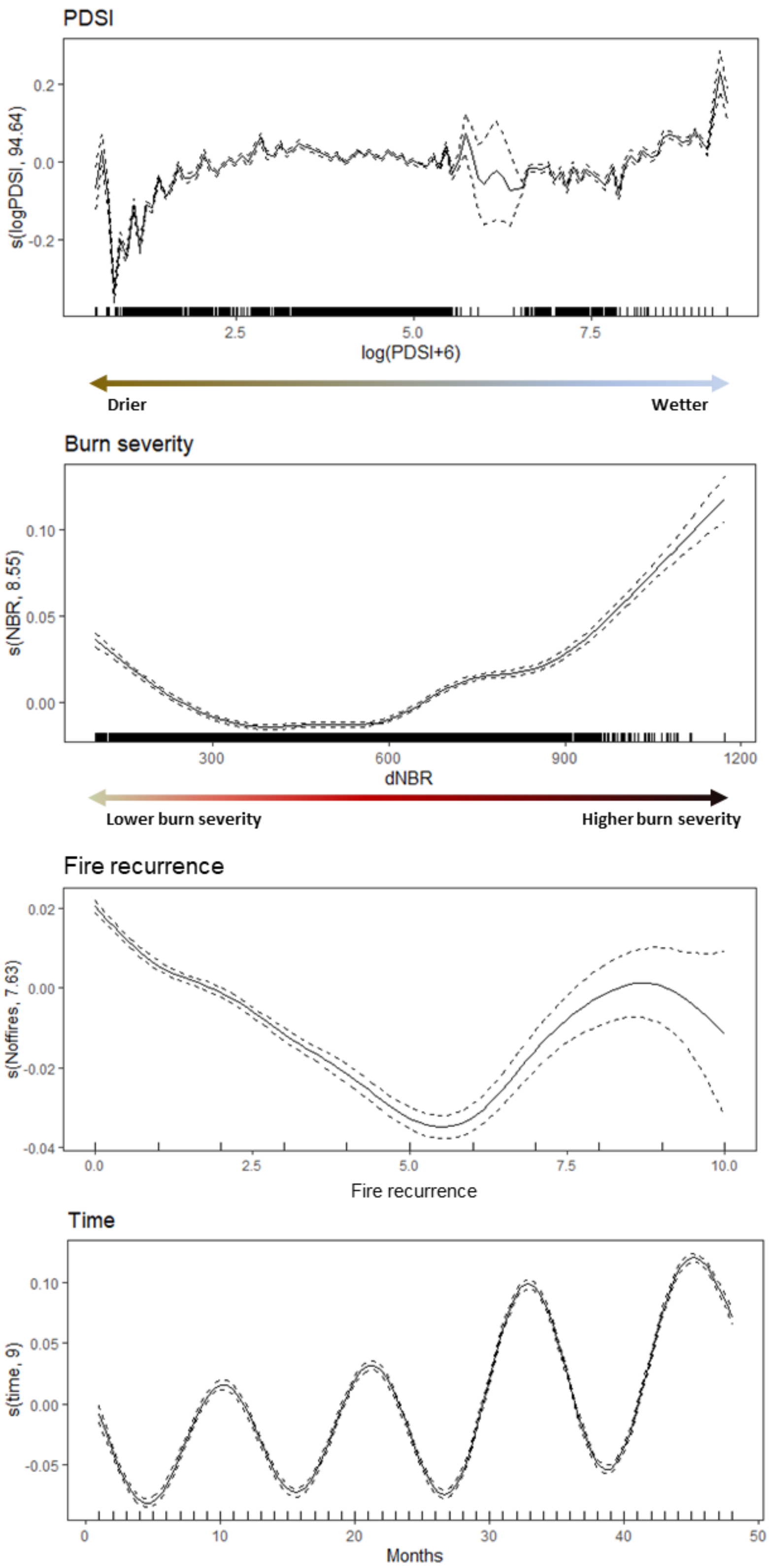

Figure 3.3 - GAMM partial effects plots of smooth variables on NDVI recovery rate. X-axis shows predictor variable values. The y-axis represents the effect of each variable. Tick marks on the x-axis are observed data points. Dotted lines indicate the 95% confidence intervals.

3.4. Discussion

3.4.1. Effects of postfire restoration interventions on oak forest recovery

We found a tendency for an increase in NDVI over time, which indicates a generalized postfire oak forest recovery in the study areas. NDVI maximum annual values increased during the spring meteorological season, which corresponds to the period between the start of the Mediterranean growing season (March) and the maximum of photosynthetic activity (June) (Piedallu et al. 2019). We also observed a shift in annual maximum NDVI values amid the second and the third year. Such trends agree with other studies that show a gradual recovery of oak forest communities, dominated by the recovery of herbaceous species during the first two years, with the recovery of woody species (shrubs and trees) becoming stronger from the third year onwards (Calvo et al. 2002; Calvo et al. 2003; Capitanio and Carcaillet 2008).

Postfire restoration interventions had a significant positive impact on oak forest recovery, however with a small model effect. The small effect observed may be the result of the execution of interventions outside the optimal timeframe. Delayed postfire emergency stabilization is common in public-funded restoration projects in Portugal and can limit vegetation recovery instead of facilitating it, as the substantial loss of ashes and soil has already occurred, leading to reduced benefits (Lopes et al. 2023).

The marginal effect of postfire emergency stabilization on oak recovery (NDVI) can also be attributed to the natural capacity of oaks to regenerate following a fire (Catry et al. 2013b). As a result, the distinction between project areas and the control group was not substantial. This knowledge becomes crucial in guiding decision-making processes and prioritizing specific areas for implementation of postfire stabilization measures. By giving preference to these identified areas, the effectiveness of stabilization efforts can be maximized while minimizing potential adverse effects on the natural oak regeneration process, as well as reducing overall restoration costs.

3.4.2. Effects of postfire drought events and fire characteristics on oak forest recovery

The drought index PDSI was the variable with the higher effect in the postfire recovery of NDVI. During the analysed period, PDSI monthly values were mostly negative (reflecting mild to extreme drought levels), with the most severe drought levels observed in the first months after the fire. In fact, the hydrological year 2016-2017 was the 9th driest year since 1931, and in 2017-2018 most of mainland Portugal was under severe to extreme drought. In addition, this drought event was distinct from former drought years since drought severity was aggravated in the autumn of 2017 (IPMA 2020). Our results showed that postfire recovery of NDVI was negatively affected by severe drought levels and positively affected by increased wetness, in accordance with available literature. Indeed, the initial stages of the oak life cycle are especially vulnerable to the reduction of precipitation (Montagnoli et al. 2016; Marañón et al. 2020), and severe droughts will likely affect postfire regeneration capacity of oak forests (Acácio et al. 2017; Marañón et al. 2020). A recent study also showed that precipitation deficits were associated with changes from deciduous oak forests to other land cover types in Portugal (Acácio et al. 2017). In general, post-fire climatic conditions have been pointed out as one of the most important predictive factors for postfire vegetation recovery (Pausas et al. 1999; Torres et al. 2018a; Nolan et al. 2021).

Regarding burn severity, our results showed that NDVI responded positively to both low and high burn severities, while intermediate severities showed a reduced effect on NDVI recovery rate. Although both extreme categories appear to have a similar outcome, the NDVI response can be justified by different reasons. On one hand, low burn-severity fires will generally provide beneficial consequences, since trees will not be top-killed, surface litter will be only partially consumed, and the soil organic layer will remain largely intact (Keeley 2009; Fernandes et al. 2010). This will result in availability of soil nutrients and faster habitat rejuvenation (Castro Rego et al. 2021a), as shown in this study by the higher NDVI recovery rate.

On the other hand, higher burn severity levels will cause higher impacts on the soil and vegetation when compared to lower severity fires (Viana-Soto et al. 2017; Castro Rego et al. 2021a), and will consequently lead to oak forest degradation (e.g. changes in structure, loss of habitat quality or increased susceptibility to pests and diseases) and difficulties in oak natural regeneration (Tarrega and Luis-Calabuig 1987; Álvarez et al. 2009; Clarke et al. 2013; Acácio et al. 2017; Monteiro-Henriques and Fernandes 2018a). High-severity fires (often crown fires) commonly originate near complete mortality of the above-ground vegetation, including a significant amount of post-fire stem mortality in deciduous oak trees (Catry et al. 2010b; Catry et al. 2013b), and total consumption of the forest floor (Keeley 2009; Tepley et al. 2018). High-severity fires also lead to changes in ecosystem recovery dynamics, decreasing the dispersal, viability and success rate of tree seeds (Marzano et al. 2012). Therefore, high burn severities likely imply more time for the recovery of the tree canopy (Keeley 2009; Vega et al. 2013a; Tepley et al. 2018). As such, NDVI increase under high burn severity is partially explained by the fast recovery of the understorey vegetation that dominates the burned area (Castro Rego et al. 2021a) and also by the extensive tree resprouting response stimulated by the greater canopy damage (top kill) (Moreira et al. 2009a; Frelich et al. 2015). Such differences in the type of vegetation that recovers cannot be detected by NDVI measurements, which show vegetation greenness and do not guarantee that the same type of pre-fire vegetation is being regenerated (Gouveia et al. 2010; Meneses 2021). In agreement with our results, postfire NDVI values in pine forests mixed with oaks and other woody species under moderate/high burn severities increased quickly over time, achieving those of the lightly burned areas two years after the fire (Lee and Chow 2015).

Lastly, fire recurrence showed a small negative effect on NDVI recovery rate, up to a maximum frequency of six fires. Higher fire frequency may lead to a diminishing capacity of oaks to resprout (Frelich et al. 2015), and its effect seems to be increasingly negative on sapling density, with Pedunculate oak and Pyrenean oak saplings being especially sensitive (Monteiro-Henriques and Fernandes 2018a). Hence, recurrent fires may alter the stand structure and lead to a decrease in oak dominance in favour of other tree species, with the magnitude of this effect depending on the fire recurrence (Burton et al. 2010; Frelich et al. 2015). For example, repeated burning can transform an uneven-aged oak stand into an even-aged stand (Knapp et al. 2017). However, and as our results indicate, if fire frequency is above a certain threshold, it will lead to lower fuel loading and consequent lower burn severity, which will cause lower impacts in the oak forest (Burton et al. 2010; Steel et al. 2015; Knapp et al. 2017).

3.4.3. Future research needs

Spectral indices are increasingly used in fire ecology (Szpakowski and Jensen 2019) and the majority of the authors considers NDVI as an extremely valuable tool, applicable without additional field validation (Gitas et al. 2012). Spectral indices also show relatively high accuracies of estimation. For example, the index dNBR showed a global accuracy of 81 % to estimate burn severity with Sentinel-2 satellite imagery, in comparison with in situ measurements (Sobrino et al. 2019). Nevertheless, remote sensing data are approximations of reality (Szpakowski and Jensen 2019). There is a wide range of possible indices available, each with its own strengths and weaknesses which will have an impact on the results (Fernández-Guisuraga et al. 2018; Torres et al. 2018b; Szpakowski and Jensen 2019). In addition, remote sensing data are generally limited by atmospheric effects, saturation, and sensor characteristics, and as such, a careful interpretation of results is recommended (Huang et al. 2021). Nevertheless, newer techniques are approaching maturity (e.g. hyperspectral imagery or Unmanned Aerial

Vehicles to obtain ultra-high resolution imagery), which will enable more accurate collection of data (Fernández-Guisuraga et al. 2018; Veraverbeke et al. 2018; van Gerrevink and Veraverbeke 2021), and hence better models.

Future changes in fire regimes are expected worldwide as a result of climate change, in particular for southern Europe (e.g., higher frequency and severity) (Dupuy et al. 2020). Although resprouting species such as oaks are likely the most resilient to changing fire regimes, postfire oak recruitment and forest recovery may be hindered by overlapping climate-driven stressors such as pest outbreaks, droughts and heatwaves, which may significantly limit the capacity of vegetation to recover in the future (Nolan et al. 2021). It is therefore necessary to continue studying the impacts of postfire restoration interventions on forest recovery and integrate the possible ecological consequences of restoration into the decision-making process (Robichaud et al. 2009), and accept that, for example, non-interventions can be the most adequate postfire management strategy for the Mediterranean broadleaved forests (Carrari et al. 2022). Ultimately, a better understanding of postfire vegetation dynamics, with further research on the direct and indirect impacts of the various types of postfire restoration measures, will lead to better decisions regarding postfire forest management.

3.5. Declarations

Availability of data and material

The dataset generated in the present study is available and can be found here: https://doi.org/10.6084/m9.figshare.22015970

Code availability

Information can be found in Supplementary Material - Annex I “R code for reproducing the analysis”.

Competing interests

The authors declare that they have no competing interests.

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Authors' contributions

LFL, VA conceived and designed the analysis; LFL performed data collection; data analysis and interpretation were performed by LFL in collaboration with FSD, PMF, VA; all authors participated in writing/edition of the manuscript.

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3.8. Supporting Information

ANNEX I

A remote-sensing assessment of oak forest recovery after postfire restoration

Supplementary Material - R code for reproducing the analysis
European Journal of Forest Research

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Contents

Chapter 4 - Postfire oak afforestation in Portugal reveals trade-offs in the supply of ecosystem services

Lopes LF, Santos E, Nunes L, Fernandes PM, Acácio V (submitted) Postfire oak afforestation in Portugal reveals trade-offs in the supply of ecosystem services. Ecosystem Services.

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Abstract

Forests have long been recognized as important sources of ecosystem services (ES), particularly in the Mediterranean Basin. In this region, oak forests are an important component of the landscape and provide numerous ES, including fire mitigation. However, the increasing incidence of catastrophic fires may lead to the degradation of oak forests decreasing their supply of ES. In this context, postfire forest restoration can assist in the recovery of ecosystem functions and associated ES, but the performance of postfire restoration is seldom evaluated and monitored. In this study, we aim to understand how postfire restoration with oak afforestation affected the supply of ES and to use the observed ES supply to evaluate restoration effectiveness. We selected 15 postfire afforestation projects with Pyrenean oak in Portugal, subsidized by public funding in 1994-2006. For each afforestation project, we established a paired control area affected by the same fire event but without oak afforestation. We quantified and compared ecosystem services in afforested and control areas, including site conditions, stand characteristics, forest biometry, understory vegetation, floristic richness and diversity, natural regeneration, and litter and soil characteristics. Results show that postfire oak afforestation with understory management improved wood provision, carbon storage in the tree layer and fire protection services, while likely contributing to higher oak natural regeneration. Conversely, postfire oak afforestation led to a decrease in understory and soil organic carbon and had no apparent effect on floristic richness and diversity. Our study reveals that the implementation of postfire oak afforestation in oak forests may have a positive, neutral or negative impact on ES, highlighting the existence of ES trade-offs due to management choices. Understanding the impact of postfire forest restoration on the supply of ES will contribute to advancing restoration efforts and policies, particularly in the face of increasing wildfire severity in Mediterranean Europe.

Keywords

Environmental benefits; Afforestation; Postfire recovery; *Quercus pyrenaica*; Mediterranean region; Soil-plant system

Highlights

- The impacts of postfire oak afforestation on the supply of ecosystem services ranged from positive to neutral or negative, depending on the service.
- Ecosystem services differed between afforested and non-afforested areas, revealing trade-offs among services.
- Postfire passive restoration of oak forests may be preferable to active restoration with oak afforestation.

4.1. Introduction

Forests have long been recognized as important sources of ecosystem services (ES), such as carbon sequestration, provision of biodiversity and soil protection, among many others (Myers 1997; Millennium Ecosystem Assessment 2005). In the Mediterranean Basin in particular, one of the world biodiversity hotspots (Myers et al. 2000), forests are essential for sustaining the livelihoods and well-being of communities (Palahi et al. 2008), providing important externalities and supporting services (e.g. watershed protection and soil conservation) that maintain vital ecosystem functions and processes (Croitoru 2007; Nocentini et al. 2022).

Oak forests and woodlands (dominated by the genus *Quercus*) are an important component of the Mediterranean landscape and offer a multifaceted array of ES, some of which are distinct from what is commonly observed in other forest ecosystems (Marañón et al. 2020; Stavi et al. 2022). Oak forests usually present high biodiversity at both local and regional scales, are resilient in nutrient-poor soils, and have high economic, cultural and aesthetic values (Alves et al. 2018; Marañón et al. 2020). In addition, mature deciduous oak forests often have understories with the ability to retain moisture, which promotes fire-resistant environments (Fernandes 2009; Botequim et al. 2017) and oak species are able to regenerate and resprout after wildfires, even at a young age (Calvo et al. 2003; Catry et al. 2013b). As a result, oak forests are commonly acknowledged as fire-resilient landscapes, contributing to fire mitigation, which can itself be considered as an ecosystem service (Brockerhoff et al. 2017; Parthum et al. 2017).

In recent decades, fire risk mitigation provided by forests has become an important service to buffer the increased severity and/or frequency of wildfires worldwide, as a result from changes in fire regimes (Rogers et al. 2020; Pausas and Keeley 2021). In particular, Mediterranean Europe is a highly fire-prone region where the frequency of large and high-impact fires has been increasing, especially in countries such as Portugal, Greece, and Spain (Dupuy et al. 2020). Increased wildfire severity, with more extreme wildfires, can decrease the supply of ES from forests, such as water quality, erosion control and climate regulation (Thom and Seidl 2016; Harvey and Enright 2022; Roces-Díaz et al. 2022). Regarding oak forests, excessively frequent or severe fires can impede natural oak regeneration, decrease oak ability to resprout after fire (Mauri and Pons 2019) and result in the encroachment of fire-adapted shrublands (Tarrega and Luis-Calabuig 1987; Acácio et al. 2017; Monteiro-Henriques and Fernandes 2018b), leading to oak forest degradation. Such negative impacts may affect the provision of many ES from oak forests (Mauri and Pons 2019; Nocentini et al. 2022), further enhanced by climate change (Morán-Ordóñez et al. 2020; Morán-Ordóñez et al. 2021). In this context, postfire management and restoration of burned forests can assist the recovery of ecosystem functions and promote the provision of ES (Espelta et al. 2012; Leverkus 2017; Lucas-Borja et al. 2021).

Restoration techniques such as tree planting and seeding have been frequently employed after fires since the late nineteenth century, as a result of strong political pressure and to meet societal expectations (Merino et al. 2019). However, the performance of tree planting is seldom evaluated with monitoring activities (Bautista et al. 2010b; Gann et al. 2019) and many restoration programs lack a follow-up assessment (Mansourian et al. 2017; Lindenmayer 2020). Moreover, postfire soil conditions and soil recovery, which inevitably affect the success of plant development, are rarely evaluated (Holl 2020). In recent years, ES have been increasingly used as indicators to monitor and assess the progress and success of restoration programs (Jackson et al. 2022; Liu et al. 2023). Hence, the supply of ES, beyond a restoration goal, is also a tool to measure the performance of implemented practices (Jackson et al. 2022).

In this study, we aim to understand: (1) how the supply of ES was affected by postfire restoration with oak afforestation and (2) to use the observed ES supply to evaluate restoration effectiveness. We use postfire afforestation projects with the Pyrenean oak (*Quercus pyrenaica* Willd.) implemented in 1994-2006 in the centre and north of Portugal and quantify ecosystem services in afforested and non-afforested (control) areas. Postfire restoration in Portugal has been supported by European funding since 1986 when the country entered European Community (Pinho 2017) and has gradually gained political attention (Ribeiro et al. 2018). However, assessment and monitoring of postfire restoration in Portugal is seldom performed but has been pointed out as extremely necessary (DGRF 2015). This study will contribute to improving knowledge on the impacts of postfire forest management and restoration on ES, thereby helping to optimize decision-making.

4.2. Material and methods

4.2.1. Study area and target species

Mainland Portugal (latitude 37º-42ºN, longitude 6º-10ºW) is positioned in the extreme southwest of continental Europe on the Iberian Peninsula and experiences a Mediterranean climate characterized by a north-south gradient (Mora and Vieira 2020). The native forest in Portugal was primarily composed of deciduous oaks, which have gradually declined over the centuries due to human activities, leaving behind fragmented remnants (Figueiral and Bettencourt 2004; Reboredo and Pais 2014; ICNF 2019a). Today, deciduous oak forests in Portugal (*Q. pyrenaica*, *Quercus faginea* Lam. and *Quercus robur* L.) cover only 2.5% of the country's forested land (ICNF 2019), with *Q. pyrenaica* occupying the largest area (Carvalho 2007). The species is found in the western Atlantic-Mediterranean region, including Portugal, Spain, France, and Morocco, with Spain and Portugal accounting for approximately 95% of its natural distribution, rendering the species nearly endemic to the Iberian Peninsula (Rushforth 1999; Nieto Quintano et al. 2016). It is a species that flourishes in mesophytic conditions (moderate moisture conditions) and is adapted to a large range of soil types but prefers soils developed on siliceous bedrock (Vila-Viçosa et al. 2022). It is adaptable to sub-humid to hyper-humid climatic conditions, displaying marcescent behaviour in thermophilic areas and tolerating summer drought (Vila-Viçosa et al. 2022). In Portugal, woodlands of *Q. pyrenaica* are protected under the Natura2000 European network, but many are in an intermediate stage of ecological succession, with high density of young trees and high shrub coverage (ICNB 2000).

4.2.2. Selection of afforestation projects and control areas

We collected data from afforestation projects subsidized by public funding in 1994-2006 in Portugal from the Financing Institute for Agriculture and Fisheries (IFAP), including funding program, project starting date and spatial location. Available information was collected from two European funding programmes for the afforestation of marginal agricultural areas, namely Regulation (CEE) 2080/92 (1994-1999) and Rural Development Program - RURIS (2000-2006). Projects were selected for analysis based on three criteria: 1) use of *Q. pyrenaica* as afforestation species; 2) experiencing at least one fire in the five years preceding the year of project implementation, with no further fires after that; and 3) be classified as deciduous oak land cover in 2018 (the most recent year with public Land Use/Land Cover data for Portugal; note that national Land Use/Land Cover cartography groups deciduous oaks into the same class without distinguishing the oak species) (DGT 2018). This last criterion was used to exclude projects without deciduous oaks in the present, which could not be compared with the remaining projects due to absence of the target species (*Q. pyrenaica*). In total, we identified 92 forest projects that met these selection criteria, distributed across two regions with similar bioclimatic characteristics (northeast and central-east Portugal). Projects implemented in the same year and spatially contiguous, were considered as a single project. Hence, a final set of 43 afforestation projects remained for further analysis.

After an initial field evaluation of the 43 afforestation projects, 24 were excluded due to absence of the target species and 4 were excluded due to inaccessibility (fences or other barriers), with a final sample of 15 projects for analysis (Figure 4.1). For each of these, we selected a spatially close (< 500 m) control area affected by the same fire event but without evidence of oak afforestation or other management practices.

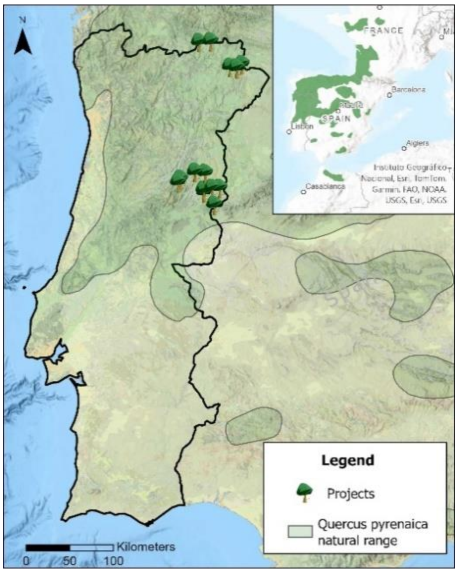


Figure 4.1 - Project location in Portugal (blue pinpoints, N = 15 projects + 15 control areas). Quercus pyrenaica natural range from European Atlas of Forest Tree Species (green color, Caudullo et al. 2017).

4.2.3. Data Collection

We used circular sampling plots of 500 m^2 to collect field data on each selected project and control areas. Collected data was divided into eight categories, namely: site conditions, stand characteristics, forest biometry, understory vegetation, floristic richness and diversity, natural regeneration, litter characteristics, and soil characteristics (Table 4.1). Within each plot, sampling area varied with the type of data collected (see Table 4.1 and Figure 4.2 for further details). Fieldwork was carried out in summer 2021, with the exception of floristic richness and natural regeneration data, which was collected during spring 2022. Methodology used to collect data on site conditions, stand characteristics and forest biometry was adapted from the Portuguese National Forest Inventory field manual (AFN 2009). We used the line transect method (Canfield, 1941) to collect data on understory vegetation, which was the baseline for litter and soil sampling. Soil samples were analysed in the laboratory to obtain soil characteristics (Table 4.1), including: percentage of fine and coarse fractions (Dry sieving), pH in H_2O (1:2.5 *m*:*V*), Corg (Springer and Klee 1954), total N (Kjedahl method), extractable P (Olsen method), K in available fraction (DTPA method; (Lindsay and Norvell 1978), and cation exchange capacity (CEC). CEC was the sum of exchangeable non-acid cations (Ca, Mg, Na, K) determined by Madeira et al. (2003) and exchange acidity (Logan et al. 1985). Floristic richness was based on the number of species present and floristic diversity was quantified with the Shannon diversity index (*H*) that encompasses both species richness and evenness (Magurran 1988).

Additionally, we quantified the Fire Return Interval (average time between wildfires from 1975 until the start of project) using the national burned area mapping (ICNF 2021b).

Table 4.1 - Description of data collected.

Category	Sampling area	Variables	Description
Site conditions	main sampling plot of 500 m^2 (r = 12.62 m)	Topography	Altitude (m), aspect (compass direction as north, south, east, west) and slope (º)
Stand characteristics	main sampling plot of 500 m^2 (r = 12.62 m)	Stand Composition	Pure or mixed
		Management of understory vegetation	Brush cutter, disc harrow, other or none
		Silvicultural treatments	Visual evidence of silvicultural treatments such as pruning and thinning
Forest biometry	main sampling plot of 500 m^2 (r = 12.62 m) Smaller trees: 5 subplots, each with 10 m^2 (r = 1.78 m)	Tree density, tree height and diameter at breast height (DBH)	Density of all adult trees (DBH ≥ 7.5 cm, including *Q. pyrenaica* and other species, number of trees per unit plot area); Height (m); Diameter at breast height (cm); Density of smaller trees (DBH < 7.5 cm and height ≥ 50 cm)
Understory vegetation	3 linear transects, each one with 10 m	Understory height and cover	Average height and cover percentage of each species belonging to the herbaceous, shrubs, ferns or tree natural regeneration (seedlings < 50 cm) groups (within the transect)
Floristic richness and diversity	20 m^2 rectangular plot (10 m x 2 m)	Species richness and diversity	Total number of species and proportion of area occupied by each species, belonging to the herbaceous, shrubs, ferns or tree seedlings (< 50 cm) groups, within a rectangular plot around one of the linear transects
Oak natural regeneration	5 subplots, each with 10 m^2 (r = 1.78 m)	*Q. pyrenaica* natural regeneration	Number of *Q. pyrenaica* seedlings (height < 50 cm)
Litter	3 random points, each with 0.0625 m^2 (0.25 m x 0.25 m)	Litter characteristics	Depth (cm) and weight (t/ha) of the forest floor
Soil	A composite sample (from 2 subsamples) of the superficial layer (0-5 cm of depth) from each of 3 linear transects	Soil characteristics	Percentage of fine and coarse fractions; pH (H_2O); organic Carbon Content (Corg); total Nitrogen; extractable Phosphorus; exchangeable acidity; and exchangeable non-acid cations (Calcium; Magnesium; Sodium; Potassium)

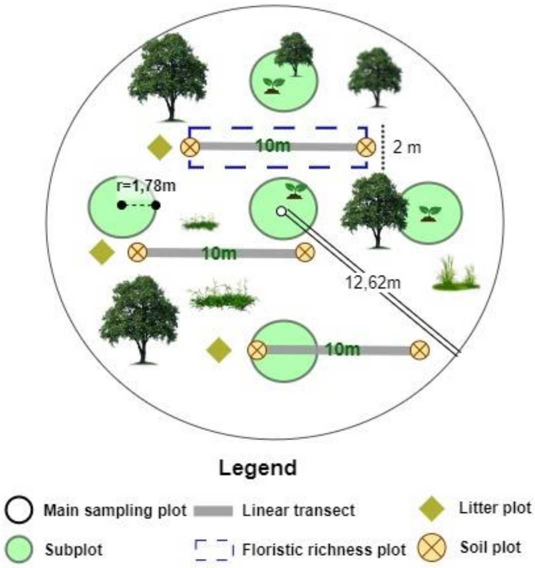


Figure 4.2 - Plot layout.

4.2.4. Data Analysis

Data collected in the field and obtained in the laboratory analysis was used to quantify additional variables, which were used as proxies of ecosystem services (according to the Common International Classification of Ecosystem Services, CICES; Haines-Young and Potschin 2018), namely: (i) forest stand volume and *Q. pyrenaica* stand volume; (ii) biomass and carbon at forest stand level; (iii) understory biomass and carbon; (iv) litter biomass and carbon content; (v) fuel loading; (vi) soil quality index (SQI) and (vii) C/N ratio (Table 4.1).

We used volume estimation equations based on tree DBH and height to calculate total forest stand volume (including all trees present in the plot with DBH > 7.5 cm) and *Q. pyrenaica* stand volume. We used volume estimation equations for oaks (Carvalho 2000), maritime pine (*Pinus pinaster* Aiton, Tomé et al. 2007), and chestnut trees (*Castanea sativa* Mill., Patrício 2006). In the absence of specific volume estimation equations for some of the tree's species observed in the plots (e.g. *Betula* spp. or *Fraxinus* spp.), we applied the previous equations. Two other volume equations were used for smaller trees (DBH < 7.5 cm), independently of species (ICNF 2019b). Similarly, we estimated biomass from DBH and tree height (using biomass estimation equations) to calculate stand biomass, separately for tree species and parameterized for trunk, branches, leaves, needles, and roots (Montero et al. 2005 and Mendes et al. 2013 for oaks and other broadleaves; Tomé et al. 2007 and Montero et al. 2005 for maritime pine; Patrício 2006 and Montero et al. 2005 for chestnut). Biometric equations (volume estimation equations for mature trees and young trees, and biomass estimation equations) are described in Supplementary Material.

To obtain understory biomass per species and total understory biomass per plot (sum of biomass for all species), we used understory height and cover to calculate phytovolume for each species (Botequim et al. 2015) and multiplied it by the species-specific bulk density (Nunes et al. 2022). A conversion factor of 0.81 was used for the seedlings present in the understory (*Acer pseudoplatanus* L., *Pinus pinaster*, *Q. pyrenaica*, and *Quercus suber* L.) (Botequim et al. 2015). Tree and understory biomass were converted into carbon content with a conversion rate of 50% (Houghton et al. 2009).

We used litter depth and weight to calculate litter bulk density and quantify litter biomass (Ottmar and Andreu 2007). Carbon content in litter samples was determined by Loss-On-Ignition Method. Fuel loading was quantified as the sum of the understory vegetation biomass and the litter biomass (Cowman and Russell 2021).

SQIs are highly valuable for evaluating the condition and overall soil health (Sillero-Medina et al. 2020). As such, we calculated the SQI per plot with a principal component analysis (PCA) with standardized soil characteristics. We chose principal components that accounted for a minimum of 5% of the total data variance, aiming to accumulate 90% variance. The Soil Quality Index (SQI) was then derived by multiplying the score of each principal component obtained from PCA by its corresponding explained variance. Subsequently, the calculated values were summed. (Andrés-Abellán et al. 2019; Abdel-Fattah et al. 2021). The C/N ratio is a proxy for the quality of organic matter entering forest soils (Ostrowska and Porębska 2015).

Table 4.2 shows the proxy variables used to estimate ES, divided into categories and classes (according to CICES, Haines-Young and Potschin 2018).

Table 4.2 - Proxy variables used to quantify ecosystem services.

Ecosystem service category	Ecosystem service class	CICES code	Proxy variable
Provisioning	*Materials for direct use or processing*	1.1.1.2	– Forest stand volume (m^3/ha) – *Q. pyrenaica* stand volume (m3/ha)
Regulation and Maintenance	*Regulation of chemical composition of atmosphere and oceans*	2.2.6.1	– Tree carbon (t/ha) – Understory carbon (t/ha) – Litter carbon (t/ha) – Total carbon (t/ha)

Fire protection	2.2.1.5	– Fuel loading (t/ha)
Maintaining nursery populations and habitats	2.2.2.3	– Floristic species richness (number of species/plot) – Floristic species diversity (Shannon index - *H*)
Seed dispersal	2.2.2.2	– *Q. pyrenaica* natural regeneration (number/ha)
Weathering processes and their effect on soil quality	2.2.4.1	– Soil Quality Index (SQI)
Decomposition and fixing processes and their effect on soil quality	2.2.4.2	– Organic Carbon Content (Corg) – C/N ratio

To assess how postfire oak afforestation affected the supply of ecosystem services, we compared the quantification of proxy variables between plots with afforestation projects and corresponding control plots (without afforestation), using the non-parametric paired Wilcoxon signed-rank test (the variables did not meet normality assumptions, significance level was < 0.05). We did not perform separate analyses for pure and mixed stands because the Kruskal-Wallis test did not reveal significant differences in proxy variables between these two groups.

Statistical analyses were carried out with R (4.0.3) (R Core Team 2020). Statistical tests were performed with base R functions. Descriptive analyses were formulated with the support of Dplyr, Tidyr, ggplot2 and Factoextra packages (Kassambara and Mundt 2016; Wickham 2016; Wickham et al. 2020; Wickham 2020).

4.3. Results

4.3.1. Characterization of project and control plots

Afforestation projects were implemented over a period of 14 years, from 1996 (during the first funding period) until 2009 (exceeding the initially scheduled duration of the second funding programme). At the time of the assessment (2021/2022), the majority of projects (66.7%) was between 12 and 17 years old, while the remaining projects (33.3%) were between 21 and 25 years old (since their implementation). Fire return interval at the plots showed a mean value of 17.5 years (SD = 7.8). Most plots with afforestation projects (86.7%) showed evidence of active management of the understory. Among these, 40% included soil mobilization, with disc harrowing as the most employed technique. Clear evidence of recent silvicultural treatments, such as pruning and thinning, was observed in 46.6% of the sampled plots with afforestation projects. Approximately 46.6% of afforested plots (7) were pure oak stands, while the remaining afforested plots (8) were mixed stands, with varying species composition. The identified tree species of mixed stands comprised *Q. pyrenaica*, *Acer pseudoplatanus, Betula pubescens* Ehrh.*, Castanea sativa, Fraxinus angustifolia* Vahl*, Pinus pinaster, Q. rotundifolia* L.*, Q. rubra* L.*,* and species of *Salix*.

Descriptive statistics of data collected in the field is shown in Table 4.3. Topographic conditions (altitude, slope and aspect) were similar between project and control plots. Project plots showed higher adult tree density (472 trees/ha), in comparison to control plots (173 trees/ha), with a similar trend regarding the density of *Q. pyrenaica* adult trees (305 and 156 trees/ha respectively). Nonetheless, the difference in tree density between afforested and control plots was lower for smaller Q. *pyrenaica* trees (DBH < 7.5 cm; 117 and 100 trees/ha for project and control plots, respectively), although showing similar DBH (11.6 cm and 11.2 cm).

The understory vegetation of project plots was dominated by herbaceous species (41.9%) or bare ground (29.2%), while the majority of control plots was covered by shrub species (70.9%) (mainly by C*ytisus* sp. and *Genista* sp.). As such, average height of understory vegetation in project plots (47.4 cm) was considerably lower than in control plots (205 cm). Litter layer was lighter but slightly thicker in afforested plots (mainly composed of leaves from trees) than in control plots (mainly composed of woody biomass from the shrub understory).

Sampled soils were slightly acidic (pH around 5) and were classified as cambisols (53%) or leptosols (47%) (APA 2014; IUSS Working Group WRB 2015). The average percentage of fine fraction (< 2 mm) was between 67% and 69%. Soils from project plots showed higher content of available K, and control plots showed higher values of Corg, N, extractable P and CEC. Observed CEC values are considered low for both afforested and control plots (<10 cmolc/kg; de Varennes 2003).

Table 4.3 - Descriptive statistic of collected variables.

Categories	Variables		Plots with oak afforestation projects		Plots without oak afforestation (control)	
			Mean	***SD***	***Mean***	***SD***
Site conditions	Altitude (m)		878	171	880	169
	Slope (m)		5.7	4.7	5.7	3.8
Forest biometry	Density (number of trees per ha)	Total tree density	472	264	173	184
		Q. pyrenaica	305	242	156	182
		Smaller trees	140	92.9	108	113
		Smaller *Q. pyrenaica* trees	117	105	100	103
	Q. pyrenaica DBH		11.6	4.5	11.2	3.9
Understory vegetation	Average height (cm)		47.4	20.7	205	138
	Shrub cover (%)		19.8	19.6	70.9	29.3
	Herbaceous cover (%)		41.9	21.1	20.5	29.9
	Bare ground cover (%)		29.2	27	0.3	0.06
	Floristic species richness		13.4	4.4	12.8	3.8
	Floristic species diversity		2.1	0.5	1.9	0.6
Oak natural regeneration	Number of *Q. pyrenaica* seedlings per ha		947	1110	573	650
Litter	Litter depth (cm)		2.9	1.6	2.6	0.9
	Litter weight (t/ha)		3.8	3.4	5	6
Soil	Fine fraction (% of <2mm)		69.9	12.3	67.3	10.7
	pH (H20)		5.1	0.5	4.7	0.7
	Corg (g/kg)		36.2	26.9	61.4	34.2
	CEC ($cmol_c$/kg)		4.6	2.4	5.5	1.5
	Total N (g/kg)		2.4	1.8	3.7	2.0
	Extractable P (mg/kg)		20.3	19.6	25.1	21.5
	Available K (mg/kg)		117.0	54.8	94.6	49.5

4.3.2. Comparison of ecosystem services between project and control plots

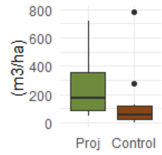
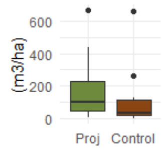
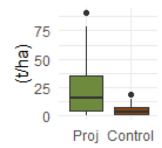
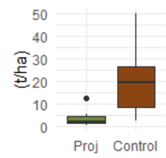
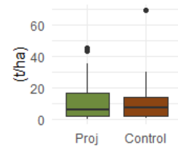
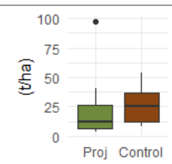
Ecosystem services (proxy variables) quantification and comparison between plots with oak afforestation projects and control plots (without afforestation) is shown in Table 4.4 (boxplots and test results).

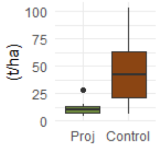
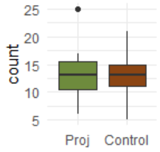
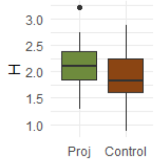
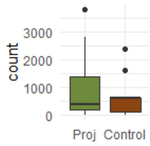
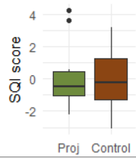
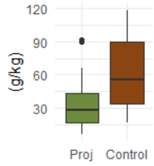
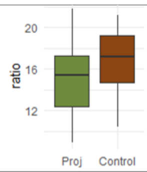
Regarding provisioning ecosystem services, plots with afforestation projects showed significantly higher wood volume (both total and *Q. pyrenaica* volume) than control plots.

Concerning regulation and maintenance services, total carbon content in the ecosystem was slightly higher in control plots but not significantly different from project plots. Nevertheless, when looking at carbon stored in the different forest layers, carbon in the tree layer was significantly higher in project plots, but conversely, carbon in the understory layer was significantly higher in control plots. We found no significant differences in litter carbon stock between project and control plots. There was a significantly higher fire protection service (lower fuel loading) in afforested plots than in control plots.

Furthermore, we found no significant differences between afforested and control plots regarding proxy variables used to estimate the maintenance of nursery populations and habitats (floristic species richness and diversity) and seed dispersal (*Q. pyrenaica* natural regeneration). Lastly, among soil-related services, only organic Carbon content in soils showed a significant difference between paired plots (being higher in control plots). Soil quality index presented a larger interquartile range in control plots, while C/N ratio was similar between paired plots.

Table 4.4 - Descriptive statistics (boxplot) and Wilcoxon signed-rank test results for variables used as proxies for ecosystem services quantification. * statistically significant ($p < 0.05$).

Ecosystem service category	Ecosystem servise class	Variable	Boxplot	Wilcoxon signed-rank
Provisioning	*Materials for direct use or processing*	Forest stand volume		Test result: 98 p-value: 0.005 *
		Q. pyrenaica stand volume		Test result: 97 p-value: 0.038 *
Regulation and Maintenance	*Regulation of chemical composition of atmosphere and oceans*	Tree carbon		Test result: 99 p-value: 0.004 *
		Understory carbon		Test result: 0 p-value: < 0.001 *
		Litter carbon		Test result: 52 p-value: 0.679
		Total carbon		Test result: 35 p-value: 0.169

	Fire protection	Fuel loading		Test result: 1 p-value: < 0.001 *
	Maintaining nursery populations and habitats	Floristic species richness		Test result: 64.5 p-value: 0.468
		Floristic species diversity		Test result: 80 p-value: 0.277
	Seed dispersal	*Oak (Q. pyrenaica)* natural regeneration		Test result: 61 p-value: 0.289
	Weathering processes and their effect on soil quality	Soil Quality Index (SQI)		Test result: 51 p-value: 0.639
	Decomposition and fixing processes and their effect on soil quality	Soil Organic Carbon (Corg)		Test result: 8 p-value: 0.002 *
		C/N ratio		Test result: 40 p-value: 0.277

4.4. Discussion and conclusions

4.4.1. Effects of postfire oak afforestation on wood provision, carbon storage and fire protection

The plantation of oak trees after wildfires clearly led to a higher wood provision and consequent carbon storage in the tree layer of afforested areas. Carbon content quantified in the tree layer falls ($\overline{x}$ = 24.74 t/ha) within the range reported for deciduous oak forests in Portugal (e.g., the 6th Portuguese National Forest Inventory reports 30.51 t/ha (ICNF 2016)). However, carbon in the understory layer was significantly lower in afforested areas where there is regular shrub cutting and superficial soil mobilization. Hence, afforested areas also presented lower fuel loading, which resulted in a higher supply of fire protection service. Conversely, in areas without postfire oak afforestation (control plots), there was higher carbon storage in the understory layer, which balanced the carbon content in the tree layer of afforested areas (resulting in similar values of total carbon between afforested and control plots). However, the presence of a shrubby understory (fuel load) in control plots decreased its fire protection service.

Our results therefore show that postfire restoration with oak afforestation was successful regarding wood provision due to a higher tree density, and regarding fire protection due to lower fuel loading and associated fire risk (Botequim et al. 2015; Enes et al. 2020). In Mediterranean Europe, the value of wood provision from *Q. pyrenaica* forests has diminished in recent decades, largely due to its predominant use as coppice forest for firewood or charcoal (co-existing with grazing areas). Nevertheless, in response to changing demands, some coppice forests are being converted into high forests, with benefits in terms of wood valorization. This conversion allows for the production of multiple uses, namely timber and firewood (from thinning and pruning) (Núñez et al. 2012; Nieto Quintano et al. 2016; Carvalho 2023).

Without oak afforestation, the recovery of oak forests after wildfires is largely influenced by factors such as fire disturbance history, prefire forest composition and structure, shrub layer development at the onset of fire, and fire severity (Calvo et al. 1999; Proença et al. 2010; Cutino et al. 2018). As already stated, ecosystems dominated by *Q. pyrenaica* are adapted to frequent fires due to the vigorous resprouting capacity of this species and many coexisting plants (Luis-Calabuig et al. 2006; Álvarez et al. 2009). Nevertheless, severe fires can halt postfire oak recovery (Vega et al. 2005; Proença et al. 2010; Catry et al. 2013b; Cutino et al. 2018) and high fire frequency can lead to resprouting exhaustion syndrome, which occurs when species reach the limit of their resprouting capacity (Nolan et al. 2021). In our study plots, the fire return interval was 17.5 years on average, which represents a fire regime with high frequency that may halt postfire oak recovery. If the fire return interval decreases even more, it may cause a landscape transition from oak forests to shrublands or grasslands (Mateus and Fernandes 2014).

4.4.2. Effects of postfire oak afforestation on the maintenance of populations, habitats and seed dispersal

We found no significant differences in floristic species richness and diversity of the understory layer (proxies of maintenance of populations and habitats) between areas with and without oak afforestation, despite their differences in understory management and composition (recall that afforested plots were dominated by herbaceous species, while control plots by shrub species). Nevertheless, areas without afforestation (control plots) showed on average lower floristic diversity than afforested areas. Our results agree with a study reporting similar species richness between *Q. pyrenaica* shrublands and woodlands with an open understory, which did not burn for at least three years, but with lower species diversity in oak shrublands (Tárrega et al. 2006). Likewise, another study showed no significant differences in woody species richness and diversity among *Q. pyrenaica* forests at different successional stages due to high spatial heterogeneity at all stages (Álvarez et al. 2009).

Oak forests in afforested plots in our study can be considered in the initial stage of the succession (0-20 years old, Álvarez et al. 2009) since planted oaks were 12 to 25 years old and the study plots did not burn for at least 12 years. At this initial stage, plant communities in oak forests are expected to be more homogeneous, despite high spatial heterogeneity (Álvarez et al. 2009), which might explain the similarity in floristic richness across plots. In addition, lower diversity (proportion of the different species) in control plots may be the result of the dominance of fewer species (e.g., *Cytisus* sp. and *Genista* sp.) within a more stable (unmanaged) and homogeneous community in the initial stage of the succession. Species richness in our plots is below the range observed for *Q. pyrenaica* forests in the initial stage of the succession (Álvarez et al. 2009) but is higher than the richness observed in other forest types such as pine forests (López-Marcos et al. 2020). Our results indicate that oak forests promote high biodiversity independently of postfire afforestation, likely because oak forests are fire-resilient landscapes, able to quickly recover after fires, as previously discussed.

We also found no significant differences in *Q. pyrenaica* natural regeneration (proxy of seed dispersal) between areas with and without postfire oak afforestation, although afforested areas presented higher number of oak seedlings on average (about the double of what was found in control plots), likely as a result of higher oak density. Afforested areas also presented a much higher variability of oak natural regeneration among plots, which likely resulted from the implementation of distinct management practices across afforested plots, with different impacts on natural regeneration. For example, soil scarification may enhance acorn establishment (Parrott et al. 2013), while understory clearing may cause negative effects on oak natural regeneration (Castro et al. 2006; Monteiro-Henriques and Fernandes 2018b). Nevertheless, the number of oak seedlings recorded in our study plots was lower than the range observed in other *Q. pyrenaica* forests (ranging from ≈ 1900 seedlings per ha Plieninger et al., 2010, to more than 30 000 seedlings/ha Camisón et al., 2015), probably because of the young age of oak trees (below 20 years old).

In our study, the quantification of oak natural regeneration was used as a proxy of seed dispersal, although we did not identify if oak seedlings resulted from seeds or from asexual reproduction. Biotic seed dispersal service in postfire restoration in Portugal facilitates the genetic exchange among areas (Valbuena-Carabaña et al. 2008) and was recently evaluated in about 23 million euros per year (Benedicto-Royuela et al. 2023).

4.4.3. Effects of postfire oak afforestation on litter and soil characteristics

We found no differences in carbon stored in litter between afforested and non-afforested areas. Nevertheless, litter composition showed visual differences between areas. In afforested areas, litter was primarily composed of *Q. pyrenaica* foliage (from higher tree density), while in non-afforested areas, litter was dominated by woody elements resulting from the higher shrub canopy. Litter composition and its C:N ratio will impact the rate of decomposition, which will consequently influence nutrients cycling and soil fertility (Santa Regina 2000; Bravo-Oviedo et al. 2017). In our study plots, *Q. pyrenaica* foliage will generally undergo fast decomposition and consequent nutrients release (Santa Regina et al. 1997; Gallardo et al. 1998). However, afforested areas presented a significantly lower content of soil organic carbon (Corg) (36 g/kg on average) than areas without oak afforestation (61.4 g/kg on average), contrarily to what was expected due the greater presence of oak leaf litter in afforested areas. Plot management was likely the factor responsible for the lower Corg content found at afforested plots. Management practices such as clearing of understory vegetation, pruning and thinning can impact Corg levels by changing the quantity and quality of organic biomass inputs to the soil (Rodeghiero et al. 2011; Tonon et al. 2011). In addition, the disturbance caused by soil mobilization activities, particularly in the superficial layer where Corg is mostly accumulated, may increase organic matter decomposition (Magdoff and Weil 2004). Yet, other studies found a large variation of soil Corg across various *Q. pyrenaica* sites in Spain (32 to 97 g/kg, Castro and Fernandez-Nuñez 2014; 13 to 71 g/kg , Turrión et al. 2009).

We found no significant differences in C/N and Soil Quality Index (SQI) between afforested and non-afforested areas, likely due to the slow soil recovery after afforestation, causing difficulties in discerning differences in soil variables between areas with natural succession (control plots) and areas with afforestation (Nadal-Romero et al.

2016; Sillero-Medina et al. 2020). Nevertheless, C/N content was slightly higher in non-afforested areas, likely due to the higher values of Corg here found. The C/N values quantified at our soils lie within the intervals reported for *Q. pyrenaica* on soils developed on granite rocks (Santa Regina et al. 1997). To comprehensively understand the observed patterns and its relation with the rate of litter decay, a more detailed analysis would be necessary, considering the intricate interconnections of litter quality, microbial activity, environmental conditions, and organic matter stabilization processes (Berg and Laskowski 2006). Further studies aimed at understanding how management options influence these soil dynamics are crucial for enhancing strategies in sustainable forest management. Forest soils constitute approximately 45% of the carbon stock in the forest ecosystem and play a pivotal role in regulating climate processes (Pan et al. 2011).

4.4.4. Implications for decision-making and management

Our study shows that implementation of postfire oak afforestation in burned oak forests may have a positive, neutral or negative impact on ES, depending on the service under analysis. It highlights the existence of ES trade-offs due to management choices. In sum, oak afforestation with understory management improved wood provision, carbon storage in the tree layer and fire protection services, and likely contributed to higher oak natural regeneration. On the other hand, it caused a decrease in understory and soil organic carbon and no apparent effect on floristic richness and diversity. As climate change poses a threat to the provision of ES by Mediterranean forests, a thorough understanding of how forest management practices may impact these services and potential trade-offs between ES is essential (Morán-Ordóñez et al. 2021). Nevertheless, most studies on the management of Mediterranean forests towards the supply of ES focus on three or fewer ecosystem services (Nocentini et al. 2022).

Our study also highlights that postfire restoration requires an holistic approach and a comprehensive analysis of the resources available, of vegetation responses to fire, and of potential outcomes from passive restoration, in order to help decision-making (Holl and Aide 2011; Vallejo et al. 2012a). In the case of oak forests, postfire passive restoration may be preferable to active restoration with afforestation, since oak forests are able to quickly recover after wildfires and hence provide many ES (Alessandro et al. 2012). Furthermore, postfire responses of oak forests (e.g., tree mortality and resprouting) may be predicted based on the knowledge of stand characteristics, fire history and fire characteristics (e.g., severity) for a specific location and fire event (Vega et al. 2005; Catry et al. 2013b; Nunes et al. 2019), which will help decision-making.

Postfire passive restoration represents a more cost-effective approach (Chazdon 2008; Wilson et al. 2011), particularly in forest ecosystems dominated by fire resilient species such as oaks (Vallejo et al. 2012a; Merino et al. 2019). Additionally, passive restoration techniques such as the assistance of postfire tree natural regeneration not only contribute to a more natural stand composition and structure, but also lead to an accelerated growth in tree height and basal diameter, and enhanced survival rates (Moreira et al. 2009b; Vallejo et al. 2012a). On the opposite, postfire active restoration with afforestation implies expensive operations (e.g. plant acquisition, site preparation or human labour) and may lead to high mortality rates, especially in Mediterranean climate regions (Vallejo et al. 2012a).

A better monetary evaluation of ES would help improve the assessment of trade-offs and facilitate postfire management and decision-making. However, accounting for many benefits beyond market-priced services is extremely difficult (de Groot et al. 2012; Costanza et al. 2017; Enríquez-de-Salamanca 2023). In this regard, the New EU Forest Strategy for 2030 (European Commission 2021b) sets the objective of introducing payment schemes for forest owners and managers to provide ecosystem services. Furthermore, discussions on public and private voluntary payment schemes for forest ecosystem services have unfolded at the European level, with some national programs already established (European Commission 2023a), including Portugal (European Commission 2021d). As the challenge persists in converting valuations into payments, it is noteworthy that Europeans are prepared, for example, to contribute 30 billion euros annually for biodiversity preservation (La Notte et al. 2021). Understanding the impact of postfire forest restoration on the supply of ES and associated trade-offs will contribute to advance restoration efforts and policies, particularly under increasing wildfire severity, in line with the objectives recently established by the EU Nature Restoration Law (European Commission 2023b) and the United Nations Decade on Ecosystem Restoration (FAO et al. 2023).

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4.7. Supplementary Material

Biometry equations

Volume estimation equations for mature trees		
Oaks (*Quercus* spp.) and other broadleaves	$V = \frac{\beta 0}{1000}\,(d^2\,h)^{\beta 1}$, where β0 = 0.08011; and β1 = 0.922	(Carvalho 2000)
Maritime pine (*Pinus pinaster*)	$V = \beta 0 \left(\frac{d}{100}\right)^{\beta 1} h^{\beta 2}$, where β0 = 0.7520; β1 = 2.0706; β2 = 0.8031,	(Tomé et al. 2007)
Chestnut (*Castanea sativa*)	$V = \beta 0\, d^2\, h$, where β0 = 0.00003299	(Patrício 2006)
Volume estimation equations for smaller trees		
If DBH < 5 cm	$V = NC\ x\ (0.025)^2\ x\ Hc\ x\ 0{,}6$ where Nc = number of trees and Hc = mean height (m)	(ICNF 2019b)
If 50 mm < DBH < 5 cm	$V = NC\ x\ (0.0625)^2\ x\ Hc\ x\ 0{,}6$ where Nc = number of trees and Hc = mean height (m)	
In all the equations, d refers to the DBH (cm) and to h refers to tree height (m). All the results are in m³.		

Biomass estimation equations		
Oaks and other broadleaves		
Trunk	$Biomass = \beta 0\, c^{\beta 1}\, h^{\beta 2}$ where for **trunk:** β0 = 0.00245; β1 = 1.7191; and β2 = 1.4353	(Mendes et al. 2013)
Branches	for **branches:** β0 = 0.00025; β1 = 2.6695; and β2 = 0.56471	
Leaves	for **leaves:** β0 = 0.00399; β1 = 1.88754; and β2 = 0	
Roots	$Br = e^{\frac{\beta 2^2}{2}}\, e^{\beta 0}\, d^{\beta 1}$, Where: β0 = -2.4543; β1 = 2.13346; and β2 = 0.242145	(Montero et al. 2005)
Maritime pine		
Trunk	$Bt = \beta 0\, d^{\beta 1}\, h^{\beta 2}$, Where: β0 = 0.0146; β1 = 1.94687; and β2 = 1.106577	(Tomé et al. 2007)
Branches	$B = \beta 0\, d^{\beta 1} \left(\frac{h}{d}\right)^{\beta 2}$, Where for **branches:** β0 = 0.00308; β1 = 2.757606; and β2 = -0.39381	
Needles	for **needles:** β0 = 0.0998; β1 = 1.392518; and β2 = -0.71962	
Roots	$B == \beta 0\, d^{\beta 1}$, Where: β0 = 0.4522; and β1 = 1.1294	(Montero et al. 2005)
Chestnut		
Trunk	$Bt = \beta 0\, d^{\beta 1}\, h^{\beta 2}$, Where: β0 = 0.0244; β1 = 1.76603; and β2 = 1.16402	(Patrício 2006)
Branches	$Bb = \beta 0\, d^{\beta 1}$, Where: β0 = 0.06574; and β1 = 1.84096	
Leaves	$Bl = \beta 0\, d^2\, h$, Where: β0 = 0.0044	
Roots	$Bb = \beta 0\, d^{\beta 1}$, Where: β0 = 0.018973; and β1 = 2.83892	(Montero et al. 2005)
In all the equations, c refers to tree circumference (cm) measured at breast height, d refers to the DBH (cm) and h refers to tree height (m). All biomasses are in kg.		

Chapter 5 - Synthesis

5.1. Brief recapitulation

The main objective of this thesis was to understand the effectiveness of various public environmental measures employed in postfire restoration in Portugal over recent decades, with a focus on deciduous oak forests. As such, several research questions were analysed throughout the different chapters, using diverse approaches and methodologies, at varied spatial scales.

Chapter 2 aimed to understand how the public funding process affected the efficiency of postfire emergency stabilization in Portugal. Based on data from public reports, calls, and projects for postfire emergency stabilization subsidized by public funding, I found three important barriers for an effective emergency stabilization in burned areas, namely: (i) the financing model, (ii) an oversimplified approach to eligible interventions, and (iii) bureaucratic delays in decision-making and execution, which often exceeded the optimal timeframe for the implementation of interventions.

Chapter 3 aimed to understand how the recovery of deciduous oak forests was affected by postfire emergency stabilization interventions, as well as by fire characteristics, topography, and postfire drought events. The recovery of oak forests was analysed for a period of four years after fire using remote sensing techniques, specifically, using NDVI as a proxy of oak recovery. I found that postfire emergency stabilization had a marginal positive effect on oak recovery. Possible reasons for the small effect observed included: the resilience of deciduous oak forests after fire; the unspecified types of interventions analysed (each with different levels of effectiveness that may have reduced the overall effect); and the implementation of interventions outside the optimal timeframe as identified in Chapter 2. In addition, postfire drought events (based on the drought severity index PDSI) were identified as one of the most important predictors of oak forest recovery (which decreased with more drought events).

Chapter 4 aimed to understand how postfire restoration with Pyrenean oak afforestation affected the supply of ecosystem services and to use the observed supply to evaluate restoration effectiveness. Results showed that postfire oak afforestation had varying impacts on ecosystem services, ranging from positive to neutral or negative, depending on the service analysed. These outcomes underscore the importance of understanding the trade-offs between ecosystem services and the influence of management options on their supply. This chapter demonstrates the importance of monitoring postfire interventions to enhance the understanding of their impacts and to facilitate better postfire management and decision-making.

The main outcomes of the thesis are synthesized in Figure 4.3, summarizing key findings and insights.

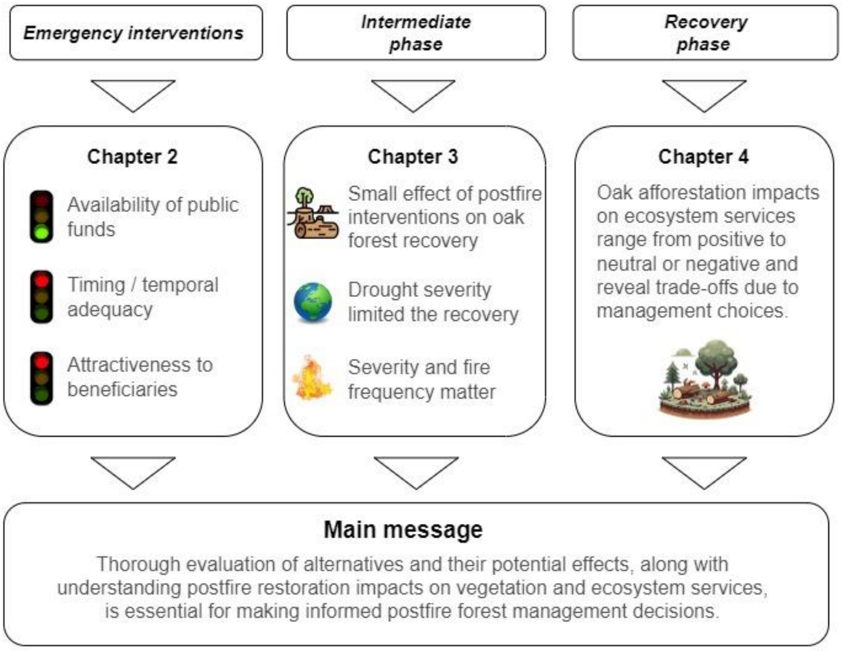

Figure 4.3 - Summary of key findings and insights

5.2. Exploring Connections: Implications and Applications

The findings of this thesis allow a significant and innovative understanding of the effectiveness of postfire restoration in Portuguese deciduous oak forests, offering valuable insights for both the academic and policy-making communities.

Overall, the findings of this thesis show that it is crucial to conduct a broad assessment of postfire restoration options to ensure the achievement of restoration objectives and the preservation of oak forests. It is imperative to thoroughly evaluate possible alternatives and their potential effects, while also considering the effective implementation of the chosen solutions. Careful consideration of available options is key to make informed decisions. In this effort to contribute to existing knowledge, I analysed the different phases of postfire restoration in Portuguese deciduous oak forests and identified potential areas for improvement.

Throughout the process of public funding, I identified three main limitations for postfire emergency stabilization in Portugal (Chapter 2), which likely decreased its effectiveness (adequacy to accomplish a purpose by producing the intended or expected result) and efficiency (achieving the intended results with the least resources consumption) and should therefore be considered by the responsible public entities. Indeed, the recent legislative changes in response to the severe 2017-fire season in Portugal, addressed some of the limitations identified in this thesis. For example, the National Plan for Integrated Rural Fire Management (Conselho de Ministros 2020a) introduced new methodologies, including a faster decision-making and execution within the government budget rules, which can already be observed in the most recent emergency stabilization reports (2019-2023). These reports comprehensively outline the emergency stabilization measures to be executed, including the definition of the priority areas and measures for intervention, along with the proposed intervention model to be implemented (e.g. funding opportunities to enable timely intervention). Reports also engage the stakeholders responsible for the management of the burned area, whilst expediting the activation of program funding agreements, thereby enhancing the efficacy of ground-level interventions. These new changes have improved the efficiency of the most recent emergency stabilization actions (see for example ICNF 2023a; ICNF 2023b).

I also found a positive but reduced effect of postfire emergency stabilization on oak forest recovery, which decreased with postfire drought events and high antecedent fire frequency (Chapter 3, see also Blanco-Rodríguez et al. 2023). Furthermore, I found trade-offs in the supply of ecosystem services between the implementation of postfire oak afforestation and the absence of management (without postfire afforestation) (Chapter 4), indicating that postfire interventions should be used strategically. Postfire restoration can be expensive, may have unintended consequences (Fernández and Vega 2016; Bontrager et al. 2019), and may not always be accepted by society (Petratou et al. 2023). Thus, in addition to a cost-effectiveness analysis, which addresses the direct objective of a specific treatment, e.g. soil erosion mitigation (Girona-García et al. 2023), it is necessary to examine the repercussions of postfire management choices on vegetation (e.g. patterns of vegetation succession and changes in composition or biodiversity) and soil properties (e.g. chemical, physical-chemical and microbiological soil properties) (e.g. Donato et al. (2015); Gómez-Sánchez et al. (2019); Lucas-Borja et al. (2021). The ecological consequences of postfire treatments must be evaluated and integrated within the decision-making process of where, when, and how treatments are implemented (Robichaud et al. 2009; Mavsar et al. 2012a).

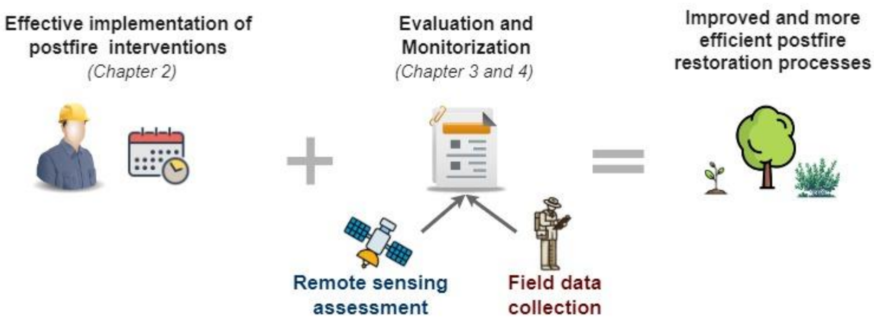

Figure 4.4 – Integrating thesis main outputs to enhance postfire restoration.

From the key insights gained, improving postfire restoration processes in Portugal (and applicable to other regions) calls for a comprehensive and integrated approach (Figure 4.4). It involves both the effective implementation of postfire interventions (particularly emergency stabilization measures, where time is critical and can constrain future options) and the establishment of robust evaluation and monitoring schemes.

Regarding the evaluation and monitorization component, Chapter 3 concentrated on remote sensing-based assessments, while Chapter 4 focused on field data collection to quantify ecosystem services. However, these approaches are not mutually exclusive. Instead, their integration should be a focal point in the development and execution of evaluation and monitoring frameworks.

In this context, although advancements in remote sensing techniques have greatly enhanced the ability to quantify ecosystem services, allowing the development of datasets specifically designed for ES estimations (Andrew et al. 2014; Jullian et al. 2021), fieldwork remains crucial for complementing these approaches. The validation and calibration of remotely sensed observations through field data are vital for maintaining accuracy and reliability, a process that is frequently constrained by the lack of sufficient ground truth data (Cord et al. 2017; Le Clec'h et al. 2018; Almeida et al. 2024).

These schemes are essential for understanding ecosystem responses and assessing the effectiveness of the postfire interventions undertaken, ultimately leading to more efficient and tailored postfire restoration processes. Yet, in Southern Europe, the limited involvement of local communities in postfire recovery efforts has often reduced these processes to replanting activities. This has resulted in political pressure to prioritize rapid reforestation or afforestation of burned areas, sometimes at the expense of more strategic, long-term restoration approaches (Moreira et al. 2012b). Despite Portugal's historical inclination towards tree planting rather than assisted restoration (e.g. by natural regeneration), the latter presents economic and ecological advantages worthy of consideration (Catry et al. 2010a). Paradoxically, funding allocation is typically more accessible for programs dedicated to tree planting than for initiatives aimed at promoting and overseeing the natural regeneration of forests (Mansourian et al. 2005; Varela et al. 2020).

To tackle the problem of expensive restoration work, which often fails to fully capture the benefits of forest ecosystems in the market, ecosystem services payments have emerged as a viable solution (Bautista et al. 2010a; Suding 2011a). Nonetheless, translating economic valuation into a tangible revenue is still a significant obstacle. The underlying issue is that numerous forest benefits are externalities, which are not comprehensively factored into economic considerations (Enríquez-de-Salamanca 2023). As Europe seeks solutions to this issue, Portugal has been used as a case study. The country has initiated the implementation of innovative environmental policies, which are being tested through pilot projects that utilise payment for ecosystem services schemes. These policies aim to prevent fires and protect biodiversity by providing financial incentives for ecosystem services (European Commission 2021d; Martins Pacheco 2022; European Commission 2023a).

Europe's forest area has increased by 17 million hectares between 1990 and 2015, with more than half being planted forest (EEA 2015). The European Union has pledged to plant an additional 3 billion trees by 2030 (European Commission 2021e) as part of the European Green Deal. The EU Forest Strategy for 2030 (European Commission 2021c) recognizes the central and multifunctional role of forests in achieving a sustainable and climate-neutral economy. However, despite this positive trend, the future of deciduous oak forests may be in jeopardy due to negative impacts from climate change, land use changes, declining oak health caused by pollution, biotic agents, and other factors (Gómez-Aparicio et al. 2008; Acácio et al. 2017; Marañón et al. 2020). These impacts may further decrease the ecosystem services provided by oak forests (Marañón et al. 2020; Morán-Ordóñez et al. 2021).

In order to be successful, postfire restoration programs need to adapt to the effects of climate change, by considering present and future conditions, especially as extreme climate events and severe wildfires are expected to continue increasing (Jolly et al. 2015; Rego et al. 2021). Furthermore, restoration programs must consider the local ecological context and incorporate monitoring actions and adaptive management throughout and beyond the lifetime of the project (FAO et al. 2023). Therefore, a deeper understanding of how postfire restoration affects vegetation dynamics and ecosystem services is essential for making informed decisions regarding postfire forest management.

5.3. Research limitations and a look to the future

The methodology adopted for this dissertation reflects my commitment to the Parsimonious Principle. I aimed to select the simplest methodologies that adequately addressed my research objectives, prioritizing clarity and efficiency. This approach was chosen to minimize unnecessary complexity, decrease the risk of errors, and enhance the overall effectiveness of my research processes. Consequently, it was necessary to proceed to adjustments throughout this thesis, with the methodologies presented in chapters 2, 3, and 4 being often preceded by different approaches, as I continually sought to refine and optimize the research methods.

Regarding the analysis of how the public funding process affected the efficiency of postfire emergency stabilization in Portugal, I could not overcome data limitations, despite my best efforts. Thus, it was impossible to respond to two research questions, namely *i) was there a difference between the physical execution of emergency stabilization interventions and the interventions initially predicted by the reports?* and *ii) the interventions were implemented according to optimal technical guidelines?* In essence, obtaining wider access to data on emergency stabilization initiatives, including specific interventions and their spatial distribution, would have been immensely valuable. This information, already gathered by the governmental agencies during planning and funding processes, could help address the current lack of detailed insight. Furthermore, as these projects have been in place at least since 2009, it would enable a comprehensive, long-term analysis. Moreover, compiling pre-fire stand characteristics and forest management practices via direct contact with land managers would provide additional layers of pertinent data.

Secondly, to evaluate the recovery of deciduous oak forests following postfire emergency stabilization interventions from a remote sensing perspective, various methodologies have been tested. These approaches included bitemporal change detection, which involves comparing pre-fire and postfire vegetation spectral indices values to assess recovery status after four years, as well as spectral trajectory analysis to evaluate recovery rate as an ongoing process. The determination of the recovery rate using the tendency line or trend (slope), indicated a small yet significant difference in the recovery rate of areas with and without postfire emergency intervention. To go further, I used a Generalized Additive Model (GAM), which later evolved to a Generalized Additive Mixed Model (GAMM) to incorporate random effects to account for the correlation structure in the data, with which I was able to continue detecting an effect of postfire emergency stabilization interventions on deciduous oak forest recovery. Additionally, I was able to identify a subset of variables (i.e. fire characteristics, topography, and postfire drought events) that exhibited a relationship with the oak recovery process.

Given the impossibility of tracking in-situ vegetation recovery after past interventions, remote sensing has been confirmed to be a robust tool. While remote sensing technologies have been present for many years, recently I assisted to a "democratization" in terms of access to satellite products which transformed the paradigm of earth system monitorization, allowing low-cost, detailed and accurate time-series (Szpakowski and Jensen 2019; Woodcock et al. 2020; Hird et al. 2021; Pérez-Cabello et al. 2021). As such, similarly to the limitations in Chapter 2, the main limitation in Chapter 3 was the lack of base data related to the emergency stabilization projects, which conditioned the analysis, either in terms of detail (e.g. categories per type of intervention) or in robustness (e.g. increased sample size or longer time series).

Regarding the assessment of the impact of postfire oak afforestation on the provision of ecosystem services, the primary limitation was also linked to data constraints. This is evident in the limited number of projects that met the defined criteria. The scarcity of available projects can be attributed to inadequate data management practices related with data preservation and protection over the years. Specifically, information pertaining to afforestation projects that were funded before the Regulation (CEE) 2080/92 program (that is, prior to 1994), has either been lost or destroyed. Even for this program and for the Rural Development Program – RURIS (between 2000 and 2006), the available information is restricted to the tree species, year of execution and plantation location. However, comprehensive data including projected tree density, previous land occupation, and terrain preparation methods employed for the plantation would produce more meaningful insights into the outcomes achieved. In the case of more recent projects, the General Data Protection Regulation (GDPR) has hindered funding agencies from providing detailed information, and my contacts with forestry associations during the development of the thesis were ineffective. Lastly, undertaking fieldwork can be both physically and financially demanding. However, with the emergence of new techniques such as Lidar technology and photogrammetry, in-situ work can be minimized by remotely generating 3-D models of the forest stands, mapping surface fuels and other vegetation structural properties, and even vegetation recovery monitoring (Szpakowski and Jensen 2019).

Given the increasing prevalence of open data policies, as exemplified by Portugal's adoption of the Open Data Directive 1024/2019 (Assembleia da República 2021; Carsaniga et al. 2023), I am optimistic about the future. I anticipate that the availability of more comprehensive information will enable a more thorough analysis of the effects of postfire restoration on forest recovery and ecosystem services supplied by forests.

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Appendix A – Paper proofs

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RESEARCH ARTICLE

Public funding constrains effective postfire emergency restoration in Portugal

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RESEARCH

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