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Beyond the forest-centric bias: rethinking extensive pine plantations in mediterranean landscapes

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ABSTRACT

This contribution examines the significant ecological consequences of extensive pine afforestation in the western Iberian Peninsula since the mid-20th century. While initially established for economic purposes and watershed protection, these plantations—primarily of *Pinus pinaster*—have fundamentally altered the natural landscape and introduced unforeseen environmental risks. We highlight a forest-centric management bias that has systematically undervalued open ecosystems, such as the Mediterranean heathland, often mischaracterizing them as degraded stages of forest. Evidence demonstrates that replacing open habitats with dense pine stands has increased soil erosion, reduced water availability, and diminished biodiversity, including endemic flora and pollinator fauna. Furthermore, the high flammability and landscape-scale continuity of pine plantations, combined with the spontaneous post-fire expansion of *P. pinaster* cultivars, have facilitated the emergence of "megafires". While natural *P. pinaster* forests remain of high conservation value, contemporary plantations require separate management strategies. We advocate for a broader shift in environmental education to recognize the intrinsic value of open ecosystems and a move toward landscape restoration that favours resilient Mediterranean mosaics over homogenous tree stands.

1. Introduction

The intensity and extension of the wildfires that occurred on the Iberian Peninsula during the summer of 2025, with over 650,000 ha affected across Spain (ca. 390,000 ha) and Portugal (ca. 280,000 ha) (Sánchez-Hernández et al., 2025; Beltrán-Marcos et al., 2026), reinforce the public perception of fire as a destroyer of wildlife and soil. However, while the harm and loss caused to people is undeniable and painful, this negative perception of fire's impact on the natural landscape and biodiversity contrasts with scientific evidence (e.g., Bond and Keeley, 2005; Keeley and Pausas, 2022). Many Mediterranean ecosystems are *pyrophilic*; that is, they are not only resilient to fire but are actually vulnerable to its absence or suppression (Keeley et al., 2012). Their biodiversity and function have been closely associated with the recurring presence of fire throughout their evolutionary history (Pausas et al., 2025). Their resilience is determined by specific adaptations for the persistence and regeneration of species following a fire (Ojeda, 2001; Rundel et al., 2018; He et al., 2019; Pausas, 2024). This relationship with fire is not limited to plant species; the faunal biodiversity of pyrophilic ecosystems can also be favored by the occurrence of fire (Tingley et al., 2016; Kral et al., 2017; Álvarez-Ruíz et al., 2021; Moritz et al., 2023). Nonetheless, the association between biodiversity and fire depends not only on its occurrence but on the seasonality, frequency, severity, and extent of the wildfires (Keeley et al., 2011; Kelly and Brotons, 2017; Jones and Tingley, 2022; Fernando and McCarthy, 2025). The alteration of these parameters by human activity can generate adverse ecological impacts even in pyrophilic ecosystems (Keeley and Pausas, 2019).

The magnitude of wildfires on the Iberian Peninsula over the last 30 years is certainly alarming. Many are referred to as "megafires" due to their size, intensity, and suppression difficulty (Linley et al., 2022; Costa-Saura et al., 2025). Media outlets point to human action, whether intentional or accidental, as the primary cause of wildfire ignition, and to weather conditions associated with climate change to explain their

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severity and extent. Indeed, a strong association between increasing occurrence of heatwaves and megafires has been evidenced (Costa-Saura et al., 2025). However, social forums rarely discuss *what* is burning; that is, how vegetation type (fuel), its continuity and flammability are key to explaining and predicting the severity and extent of wildfires (Pausas and Keeley, 2021). Reference is frequently made only to woodlands or vegetation in general, overlooking the fact that not all woody formations are equally flammable (Saura-Mas et al., 2010) and that anthropogenic landscape modifications largely determine the occurrence of these extreme events (Pausas et al., 2017; Gómez-González et al., 2018, 2022; Cappelluti et al., 2024). In this context, we re-examine the recent history of pine plantations in the western Iberian Peninsula to expose how these homogenized landscapes may act as a primary catalyst for biodiversity loss and the intensification of the regional wildfire regime. We focus on the western Iberian Peninsula, particularly in the northwestern quarter, where the large-scale replacement of heterogeneous mosaics with *Pinus pinaster* stands has created a vast, uninterrupted fuel bed responsible for the recent history of extreme fire events (e.g., Fernandes et al., 2010; Nunes et al., 2019).

2. Pine plantations and their unforeseen ecological risks

Since the mid-20th century, the landscape of the Iberian Peninsula has been strikingly modified by extensive afforestation with fast-growing tree species, primarily pine and eucalyptus plantations (Pemán et al., 2017), predominantly in monospecific stands. These plantations were established largely on low-productivity soils to secure economic benefits from timber, resin and cellulose production, as well as to provide watershed protection against soil erosion and reservoir siltation (Iriarte, 2017). The legacy of this period is most evident in the northwestern Iberian Peninsula, which maintains the greatest extent and intensity of these afforested areas (Fig. 1). These areas roughly coincide with more acidic (Fig. 2) and, consequently, nutrient-poor soils.

At the landscape scale, the most acidic and infertile soils of western Iberia occur on the high slopes of mountain ranges, where the natural landscape is characterized by the presence of heathlands, either treeless or with open cover of Pyrenean oak (*Quercus pyrenaica*) and/or cork oak (*Quercus suber*) (Gil-López et al., 2018; Fig. 2). Pine tree species, such as *Pinus pinaster* Ait. (maritime pine) and *Pinus sylvestris* L. (Scots pine), would have been naturally found scattered across high ridges of the mountains, where soil is scarce and water retention capacity is low (Font i Quer, 1930; Pallarés et al., 2001; Robledo-Arnuncio et al., 2005; Ramírez-Valiente and Robledo-Arnuncio, 2014). While *P. pinaster* formations have been listed as a habitat of European community interest (subtype 42.81 within habitat type 9540: Mediterranean pine forests; Ruiz Benito et al., 2009), it is widely recognized that the majority of the current stands of this pine species are actually the legacy of extensive afforestation efforts (Ruiz-Benito et al., 2009; Ramírez-Valiente and Robledo-Arnuncio, 2014). Indeed, seeking to increase productivity, large portions of the heathlands on infertile soils were afforested, primarily with *P. pinaster* (Rico, 1995, 2008a,b; Aguiar et al., 2007; Simpson and Ojeda, 2010; Vadell et al., 2016). This species was prioritized in the western Iberian Peninsula for its native status, its high adaptability to infertile, acidic soils (Scott 1962; Vadell et al., 2016), and, crucially, its significant economic profitability through resin and timber production (Pemán et al., 2017; Soliño et al., 2018). However, these afforestation efforts initially faced fierce opposition from local communities, who saw their traditional grazing and farming land-use rights curtailed without receiving a share of the benefits generated by the plantations (Radich and Monteiro-Alves, 2000; Rico, 2000; Iriarte, 2017; Picos-Martín, 2017).

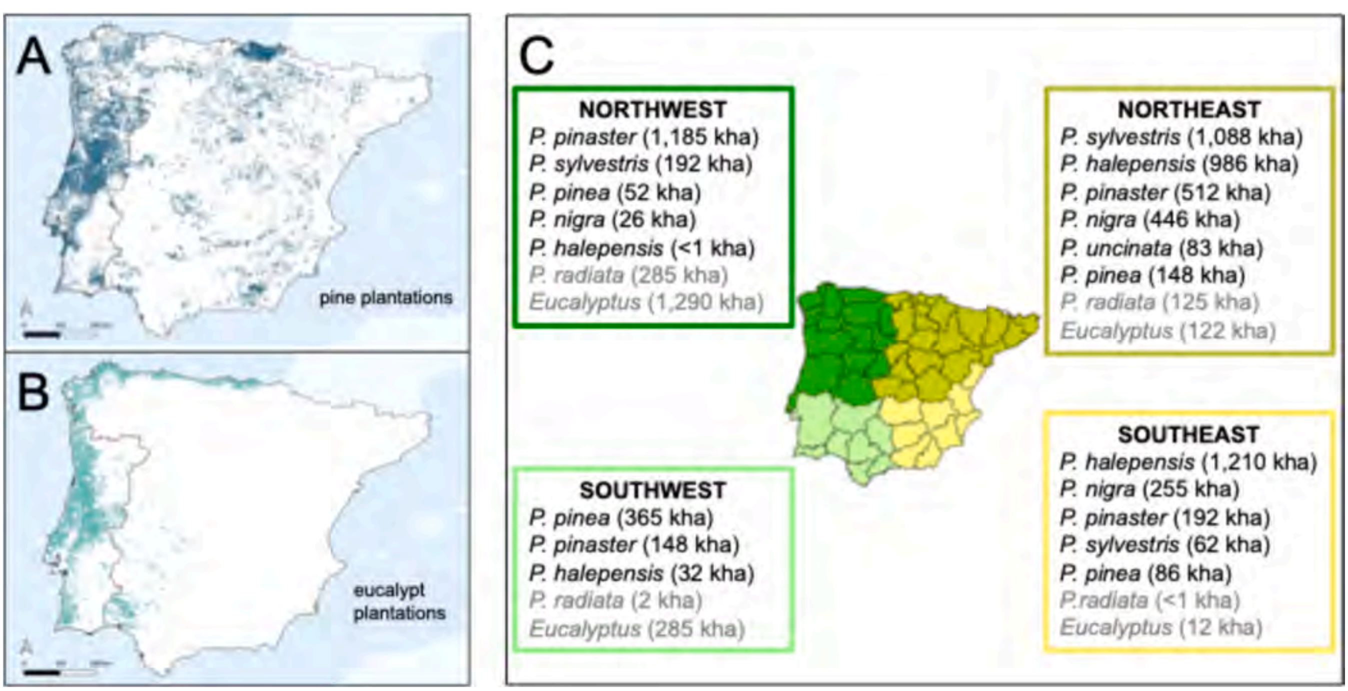

Fig. 1. Geographic extension of (A) pine plantations and (B) eucalypt plantations in the Iberian peninsula (obtained from the *Mapa Forestal de España* (MFE) for 1956, 1985, and 2020 in Spain, and the *Carta de Uso e Ocupação do Solo* (COS) for 1995 and 2018 in Portugal). (C) Estimated total extension (in kilohectares; 1 kha= 10^3 ha) of the different pine tree species formations and eucalypt plantations (considering all *Eucalyptus* species together). Native species appear in black font and exotic ones in grey. For each *Pinus* species, stands were defined where the target species accounts for >50% of the basal area or canopy cover. For the native *Pinus* species, figures encompass autochthonous relict forests, naturalized stands, and both protective and commercial plantations. Data for the Spanish provinces are derived from the third and fourth National Forest Inventories (IFN3: 1997–2007; IFN4: 2008–present), provided by *MITECO* (*Ministerio para la Transición Ecológica y el Reto Demográfico*). The most recent available provincial data was prioritized, with the IFN4 serving as the primary source for the Northwest and Northeast sectors. Data for the Portuguese NUTS II territorial divisions are derived from the 6th National Forest Inventory (IFN6: 2013–2015), provided by the *Instituto da Conservação da Natureza e das Florestas* (ICNF).

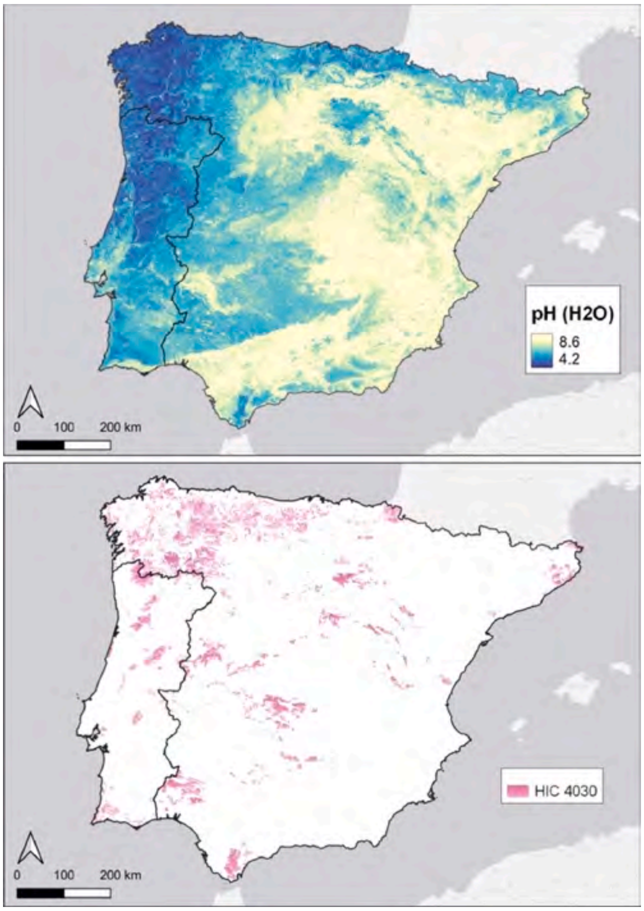

Fig. 2. Top: soil pH map of the Iberian Peninsula (European Commission, 2010). Bottom: distribution of the Habitat of Community Interest (HCI) 4030 *European dry heaths*, created from the *Cartografía de Uso e Ocupação do Solo* (COS for 1995 and 2018; Portugal) and the *Ministerio para la Transición Ecológica y el Reto Demográfico* (MITECO; Spain).

Pine and eucalyptus plantations are characterized by high flammability (Barquín et al., 2022; Lindermayer et al., 2023). Among these, *P. pinaster* plantations are particularly noteworthy, as they constitute one of the largest afforested and reforested areas in the region (Vadell et al., 2016; Moreno et al., 2021) and are the dominant type in the northwestern Iberian Peninsula (Fig. 1C). Their structural homogeneity and continuity effectively facilitate the spread of large wildfires (Fernandes and Rigolot 2007; Pasalodos-Tato et al., 2010; Moreno et al., 2021), a hazard further aggravated by the increasing frequency of climate-driven heatwaves (Costa-Saura et al., 2025) and the neglect of silvicultural management due to declining profitability (Pasalodos-Tato et al., 2010; Guijarro et al., 2017). These monospecific stands typically burn with significantly higher severity than other Mediterranean vegetation types while exhibiting comparatively lower post-fire recovery rates (Fig. 3), particularly in high-density stands lacking recent management (Repeto-Deudero et al., 2025). Furthermore, proximity to plantation borders increases the likelihood of high-severity fire in adjacent vegetation (Repeto-Deudero et al., 2025), thereby promoting fire propagation across the landscape. While other vegetation types, particularly shrublands and heathlands, often exceed these stands in total burned area during megafires (Table 1; Sánchez-Hernández et al., 2025), pine plantations act as drivers of fire spread and constitute the most vulnerable components of the landscape due to the combination of high burning severity and poor post-fire regeneration (Fig. 3).

On the other hand, the protective and restorative roles of pine plantations regarding soil and landscape (Vadell et al., 2016; González-Doncel and Vicente, 2017; Torres et al., 2021) are debatable and contingent upon the compared landscapes. For instance, the afforestation of ruderal landscapes following agricultural abandonment can certainly contribute to soil protection and improvement (Martín-Peinado et al., 2016; Mongil-Manso et al., 2022). However, by focusing exclusively and systematically on trees (Pemán et al., 2017), the role of shrublands as critical agents of soil retention and generation has been neglected, even despite early warnings (e.g., Ceballos, 1945). Paradoxically, one of the consequences of replacing shrublands with tree plantations in Mediterranean landscapes is an increase in soil erosion. Specifically, *P. pinaster* plantations are often less effective at soil protection than shrublands, or even grasslands, largely due to the aggressive land-preparation techniques used for afforestation (Ternan et al., 1996; Nunes et al., 2011).

In traditional forestry practices, particularly during the mid-20th century, rockrose and heather scrubs (i.e., heathlands) were dismissed as unproductive obstacles to afforestation (Rico, 2008a, 2016). Their

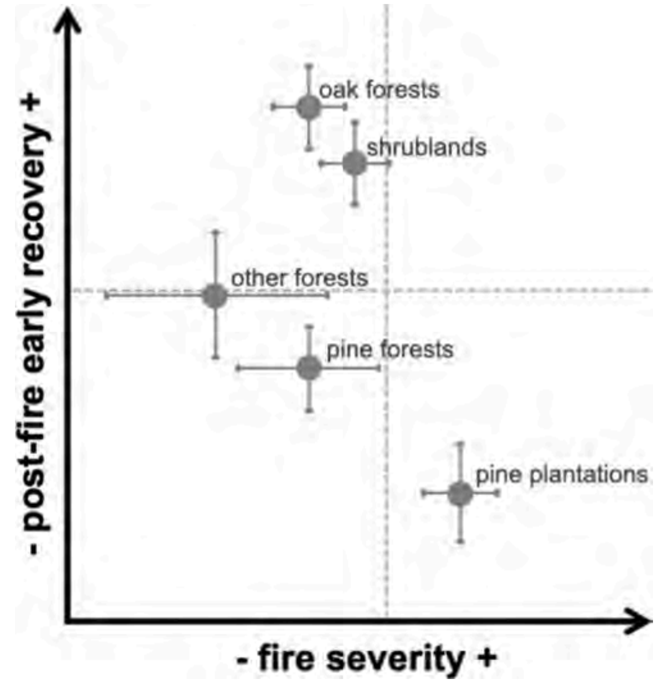

Fig. 3. Average early post-fire recovery (one-year post-fire) and fire severity metrics for the various vegetation types in the *Bermeja, Culebra* and *Hurdes* wildfires (see Table 1). Dashed lines represent global mean values, and error bars represent two standard deviations from the mean. Modified from Repeto-Deudero et al., (2025).

Table 1
Total burned area (in ha) of different vegetation types (with percentage of the total in parenthesis) in three megafires that occurred in western Spain (data taken from Repeto-Deudero et al., 2025).

Vegetation type	*Bermeja* fire (year 2021; *c.* 10,000 ha)	*Culebra* fire (year 2022; *c.* 60,000 ha)	*Hurdes* fire (year 2023; *c.* 11, 000 ha)
pine plantations	2062.7 (22%)	12,804.8 (22%)	8689.3 (80%)
pine forests*	5489.9 (58%)	0	48.3 (< 1%)
oak forests	270.3 (3%)	13,907.9 (24%)	453.1 (4%)
other forests	201.6 (2%)	501.5 (1%)	838.1 (8%)
shrubs and heaths	1080.9 (11%)	16,169.5 (28%)	287.7 (3%)
grasslands	197.3 (2%)	2002.5 (4%)	27.0 (< 1%)
Agricultural and urban	172.0 (2%)	11,705.8 (21%)	520.4 (5%)

* All pine-dominated patches lacking validation as plantations according to the criteria in Repeto-Deudero et al. (2025).

pyrophytic nature and high flammability (Ojeda, 2001) were also viewed negatively, as they exacerbated crown fires within tree plantations (Simpson and Ojeda 2010; Guijarro et al., 2017). Consequently, scrublands and heathlands were systematically cleared, often using heavy machinery (Rico, 2008a; Vadell et al., 2017), despite the detrimental effects on soil retention (Nunes et al., 2011; Martins et al., 2013). This increased soil fragility under forest plantations is aggravated after a fire (Martins et al., 2013), as the elimination of the tree canopy by fire leaves the soil exposed to erosion by water and wind. In contrast, burnt heathlands prove to be resistant to soil erosion (Jordán et al., 2010; Hancock and Legg, 2012), likely due to the importance of resprouting plants (Fernández and Vega, 2014) and their characteristic network of roots, rhizomes, and lignotubers that hold the soil in place. Therefore, it is difficult to reconcile that scrublands and heathlands were cleared to establish forest plantations with the aim of protecting soils against erosion. Arguably, economic motivations for forest plantations might have prevailed over protective ones, at least during the second half of the 20th century (Rico, 2008b; Iriarte, 2017).

3. The forest-centric paradigm: impacts on ecosystem services and biodiversity

In recent years, there has been a renewed emphasis on the environmental benefits and ecosystem services provided by forest plantations, such as landscape restoration, atmospheric carbon sequestration, the alleviation of water stress, and even the preservation of biodiversity (Torres et al., 2021; Navarro-Cerrillo et al., 2024; Tudor et al., 2025). Regarding landscape restoration, there is a dominant trend driven by a positive perception of wooded landscapes and a negative view of open, often treeless environments (González-Doncel and Vicente, 2017; Pausas and Bond, 2019; Ojeda, 2020). The view of scrubland as a stage of forest degradation —a core tenet of sigmatist phytosociology (e.g., Peinado-Lorca and Rivas-Martínez, 1987)— permeated nature conservation and landscape management in Spain and Portugal during the final decades of the 20th century (Simón-Navarrete, 1993; Madrigal, 1994; Mendes, 1999). Consequently, until relatively recently, the management of natural areas was carried out from a perspective that was more silvicultural than ecological, with trees as the primary focus (González-Doncel and Vicente, 2017). Scrublands and grasslands were therefore considered degraded habitats destined to be transformed into forests through clearing and afforestation with pines (Vadell et al., 2017), particularly the Mediterranean-type heathland or *herriza* of the western Iberian Peninsula (Molinero et al., 2008; Ojeda, 2020).

On the other hand, the assumed role of pine plantations in atmospheric carbon sequestration and, consequently, in climate change mitigation (Serrada-Hierro, 2017; Navarrete-Poyatos et al., 2019; Tudor et al., 2025), has also been called into question. Unlike *Quercus* forests, heathlands and scrublands, the majority of carbon retention in pine stands is located in the above-ground biomass. This makes the carbon pool highly vulnerable to fire, particularly within the framework of an increasingly warm climate. Consequently, its medium-to-long-term retention capacity is limited, as shown by van den Bor et al. (2024) when comparing carbon stocks between monospecific plantations of *Pinus halepensis* and native *Quercus* forests in central Spain. This limitation has also been unequivocally demonstrated in the pine afforestation of Andean grasslands (Farley et al., 2004) and Scottish heathlands (Friggens et al., 2020). In both cases, afforested patches significantly decreased the soil's ability to retain both carbon and water compared to adjacent non-forested areas.

Regarding water, the role of pine plantations in protecting watersheds and mitigating water stress (e.g., Simón-Navarrete, 1993) has similarly been questioned when natural habitats such as heathlands, scrublands, or grasslands are afforested. Studies in other regions globally illustrate how the establishment of pine plantations in non-wooded habitats drastically reduces soil water availability (Scott et al., 1998; Farley et al., 2005; Stock et al., 2012; Friggens et al., 2020). A study conducted in the Los Alcornocales Natural Park (southern Spain) showed that the afforestation of Mediterranean heathland patches with pines significantly reduces the primary productivity and radial growth of downslope cork oak forests (Repeto-Deudero et al., 2024), likely due to a decrease in soil water availability.

Therefore, a forest-centric view of natural landscape management (e.g., Peinado-Lorca and Rivas-Martínez, 1987; Pemán et al., 2017) not only underestimates the intrinsic beauty and ecological functionality of treeless habitats, but also alters that functionality and threatens their biodiversity (Bremer and Farley, 2010; Gómez-González et al., 2020; Ojeda, 2020). For instance, the afforestation of heathlands with pines drastically decreases the biodiversity of woody flora, both in terms of the number of endemic species and functional diversity (Andrés and Ojeda, 2002). It has also been shown to reduce the richness of pollinating insects and simplify plant-pollinator interaction networks, with these impacts being directly related to the degree of the pine canopy cover (Pérez-Gómez et al., 2024, 2025). In fact, scrublands are the essence of biodiversity in Mediterranean ecosystems (Guarino et al., 2020). Consequently, it seems contradictory to attribute a biodiversity preservation function to pine plantations when their establishment involves the replacement and degradation of inherently diverse, open habitats (e.g., Ojeda, 2020).

4. Pine plantations: facilitators or threats to Mediterranean woodlands?

Another aspect of landscape restoration frequently attributed to pine plantations is their ability to facilitate the transition to Mediterranean oak forests (Sheffer, 2012; Navarro-González et al., 2013; Villar-Salvador, 2016; Navarro-Cerrillo et al., 2024). However, it should be noted that the conversion of a pine stand into a holm, cork, or Pyrenean oak forest is an overly protracted process that requires specialized, diligent, and long-term silvicultural intervention (Gómez-Aparicio et al., 2009; Villar-Salvador, 2016; Martín-Alcón et al., 2017). Conversely, the opposite process —the invasion of natural habitats by pines— often occurs spontaneously and relatively quickly. For example, *P. pinaster*, widely used in commercial plantations globally (Scott, 1962), behaves as an invasive species, spreading spontaneously into natural habitats in South Africa, Chile, Australia, and New Zealand (Richardson and Higgins, 1998). In the western Mediterranean basin, where it is the most widely used species in reforestation, it has also been reported to invade habitats of high ecological value, such as cork oak forests (Fernandes et al., 2016; Benamirouche et al., 2024). Generally, in Mediterranean ecosystems, the colonization of oak formations by pines is more frequent than the reverse (Sheffer, 2012).

The spontaneous expansion of *P. pinaster* occurs primarily after a wildfire (Gil-Sánchez et al., 2008; Fernandes et al., 2016; Fig. 4), facilitated by the serotinous nature of its cones (Hernández-Serrano et al., 2013). In turn, the volume and flammability of this species' biomass, as well as its landscape-scale extent, promote the incidence and severity of large wildfires (Fernandes and Rigolot, 2007; Viedma et al., 2015; Repeto-Deudero et al., 2025). Therefore, the high frequency of megafires —disproportionate in terms of intensity, severity, and extent— is often linked to the presence of vast areas of homogeneous *P. pinaster* plantations and naturalized stands (Fernandes and Rigolot, 2007; Gómez-González et al., 2018; Ojeda, 2020; Repeto-Deudero et al., 2025). This association between large wildfires and extensive pine plantations is not exclusive to the Iberian Peninsula; it also occurs in other regions of the world, such as France (Vallet et al., 2023), Chile (Gómez-González et al., 2018), and South Africa (Kraaij et al., 2018). The catastrophic wildfires in central Chile in early 2026 serve as a stark, recent example of this phenomenon (https://globaljusticeecology.org/chile-engulfed-in-plantation-climate-fueled-mega-fires-yet-again/).

Technically, *P. pinaster* cannot be considered an invasive species within the landscapes of the Iberian Peninsula, as it is a native taxon (Amaral-Franco, 1986; Salvador et al., 2000). Nevertheless, the

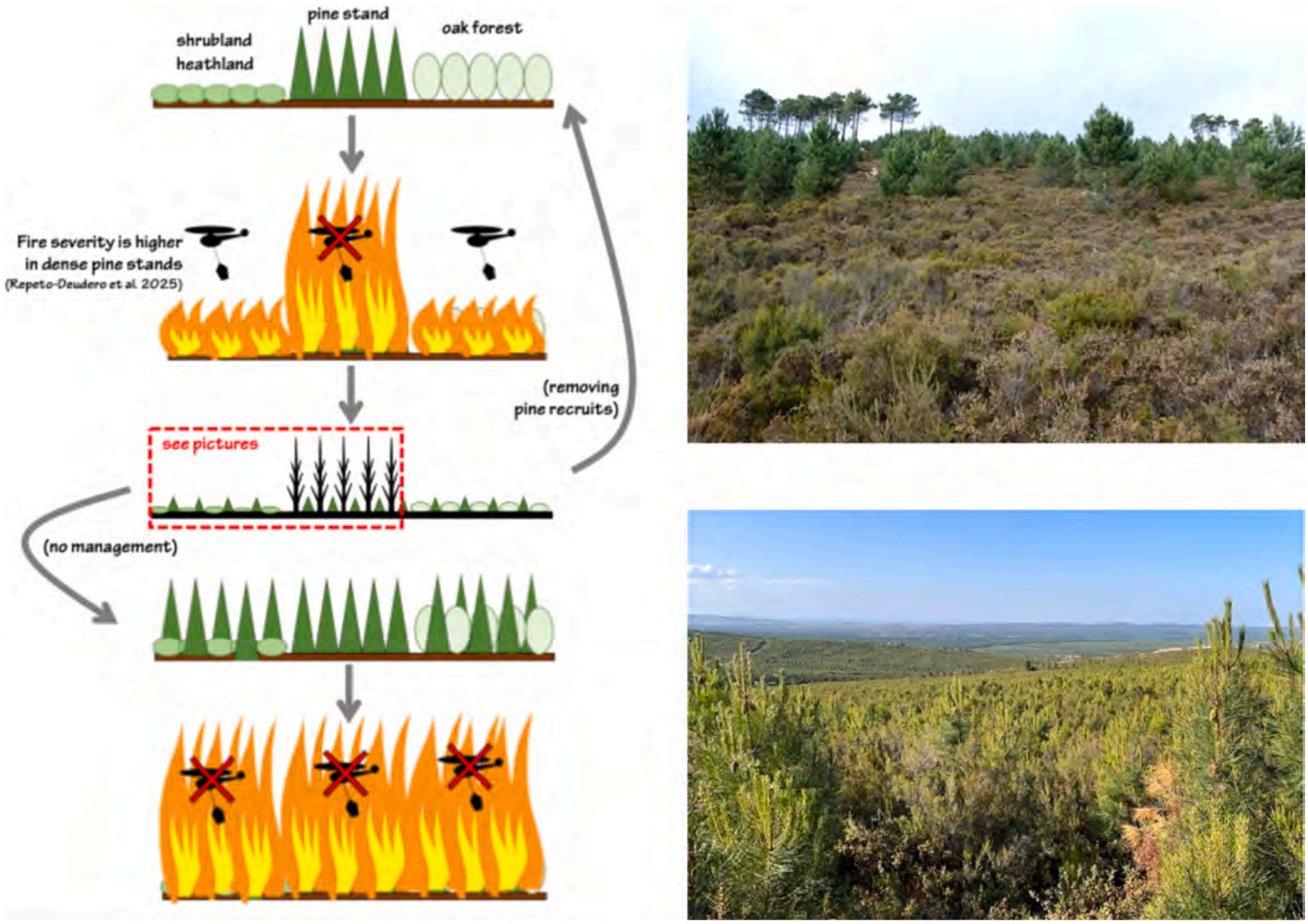

Fig. 4. Schematic representation of the fire-driven spontaneous expansion of *Pinus pinaster* into adjacent vegetation (shrublands and oak forests). Under no management, successful post-fire pine recruitment results in a continuous, high-fuel landscape where fire severity is significantly higher (Repeto-Deudero et al., 2025), potentially leading to megafires that exceed current suppression capacity (represented by the crossed-out helicopter). Conversely, active management through the removal of young pine saplings (see text) prevents this flammable continuity, maintaining or restoring landscapes toward more diverse and resilient Mediterranean mosaics. *Top picture*: expansion of *P. pinaster* into a Mediterranean heathland (herriza) patch 18 years after a 1990 fire in El Marrufo (Los Alcornocales Natural Park, southern Spain; January 2008). *Bottom picture*: spontaneous recruitment of *P. pinaster* in heathland within the Sierra del Teleno (Luyego, northwestern Spain) 13 years after a major 12,000-hectare wildfire in 2012 (June 2025).

spontaneous expansion of native species can also alter ecosystem functionality and threaten biodiversity (e.g., Shackelford et al., 2015; Nackley et al., 2017, 2018). In this regard, it is essential to recognize that while *P. pinaster* is native, the stock used in afforestation and reforestation programs is often the product of extensive artificial selection or domestication (Scott, 1962; Butcher and Hopkins, 1993; Alía and Moro, 1996). Such domestication may explain the species' high expansion potential into other habitats (e.g., Fernandes et al., 2016; Selvi et al., 2016) —encroaching even upon its own natural populations (Ramírez-Valiente and Robledo-Arnuncio, 2014)— as well as its significant invasive potential on a global scale (Scott, 1962; Richardson and Rejmánek, 2011).

5. Conclusions and proposals for the future

This contribution does not seek to disparage *P. pinas*ter as a species. On the contrary, its natural forest formations are of high conservation value, recognized as Habitat Type 9540 under the Habitats Directive (92/43/EEC). A critical distinction must be made, however, between these natural populations and the majority of contemporary stands, which are the legacy of intensive artificial selection and historical afforestation (Scott, 1962; Aguiar et al., 2007; Ruiz-Benito et al., 2009). These plantations require separate categorization and management, especially as the spontaneous expansion of *P. pinaster* cultivars threatens the ecological and genetic integrity of natural autochthonous populations (Ramírez-Valiente and Robledo-Arnuncio, 2014).

Many pine plantations that were sustainable under the silvicultural practices and climatic conditions of the previous century have become ecologically untenable due to management abandonment and 21st-century anthropogenic climate change. These areas have transitioned into extensive, homogenous stands that not only reduce landscape beta-diversity but also accumulate high loads of flammable biomass, fueling large-scale, high-severity wildfires that compromise post-fire resilience (Levine et al., 2025; Repeto-Deudero et al., 2025; Fig. 5). In the interest of wildfire prevention and the recovery of landscape heterogeneity, recently burned areas offer a strategic opportunity to restore landscapes toward more diverse and resilient Mediterranean mosaics, better equipped to withstand the impacts of a changing climate (Leverkus et al., 2019).

First, it is essential to acknowledge the intrinsic ecological value of shrublands and heathlands, which serve as vital biodiversity reservoirs (Guarino et al., 2020; Ojeda, 2020; Olmeda et al., 2020). The afforestation of these open ecosystems —often justified under the guise of climate mitigation, erosion control, or biodiversity enhancement— is largely unsupported by scientific evidence and should be avoided (e.g., Gómez-González et al., 2020; Ojeda, 2020; Pérez-Gómez et al., 2024;

Fig. 5. *Pinus pinaster* stand in Sierra de la Culebra (Zamora, NW Spain) in April 2023, ten months after the 2022 megafire. The contrast between the barren understorey and the regrowth along the road verge (foreground) illustrates the impact of fire severity. High-severity conditions within the stand likely resulted in lethal soil temperatures, destroying subterranean regenerative organs and depleting the seedbank (Maia et al., 2012).

van der Bor et al., 2024). Regarding existing pine plantations, their spontaneous post-fire self-perpetuation and expansion into adjacent natural areas must be curtailed through active management (Fig. 4). This should include targeted early interventions, such as the manual removal of seedlings or the use of hand-held brush cutters by specialized forestry crews. Conversely, naturally occurring, autochthonous pine populations must be strictly preserved and protected to safeguard their unique genetic and ecological legacies (Robledo-Arnuncio et al., 2005; Ramírez-Valiente and Robledo-Arnuncio, 2014).

Plantations currently under productive exploitation should be managed through ecologically informed silviculture, integrating nature-based solutions that favour biodiversity and limit fire spread (Gómez-González et al., 2022; Pérez-Gómez et al., 2025). On the other hand, a broader shift in environmental education is required to emphasize the intrinsic ecological value of open ecosystems such as shrublands, heathlands and grasslands, and to raise awareness about the role of fire as a natural driver of biodiversity. Such a shift in perception is vital to ensure that public support aligns with appropriate management actions. The pervasive narrative that 'fires are extinguished in winter by cleaning the forest' should be applied to tree plantations through adequate silvicultural management. Except for strategic sites for fire protection (e.g., in the wildland-urban interface), this 'cleaning' approach must not be extended to natural open habitats, specifically shrublands and heathlands, where fire behaviour is more manageable and where the biomass often characterized as 'flammable debris' is an essential component of the ecosystem's structural and biological diversity (Ojeda, 2020).

CRediT authorship contribution statement

Fernando Ojeda: Writing – review & editing, Writing – original draft, Conceptualization. **Irene Repeto-Deudero:** Writing – review & editing. **Álvaro Pérez-Gómez:** Writing – review & editing. **Susana Gómez-González:** Writing – review & editing. **Juli G. Pausas:** Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

No data was used for the research described in the article.

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