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RESEARCH ARTICLE

The legacy of pine plantations on fire severity

Irene Repeto-Deudero[1,2] | **Fernando Ojeda**[1] | **Susana Gómez-González**[1,2] | **Alejandro Miranda**[2,3] | **Verónica Cruz-Alonso**[4] | **Juli G. Pausas**[5]

[1]Departamento de Biología-IVAGRO, Universidad de Cádiz, Cádiz, Spain

[2]Centro de Ciencia del Clima y la Resiliencia (CR)[2], Santiago, Chile

[3]Laboratorio de Observación de la Tierra y Ecología de Bosques, Departamento de Ciencias Forestales, Universidad de La Frontera, Temuco, Chile

[4]Departamento de Biodiversidad, Ecología y Evolución, Universidad Complutense de Madrid, Madrid, Spain

[5]CIDE-CSIC, Centro de Investigaciones sobre Desertificación, Consejo Superior de Investigaciones Científicas, Valencia, Spain

Correspondence
Irene Repeto-Deudero
Email: irene.repeto@uca.es

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Abstract

1. In Mediterranean ecosystems, afforestation efforts have created landscapes with high fuel loads and continuity that, in combination with warmer and drier conditions, may intensify fire activity. Yet, the relative contribution of afforestation to current fire severity remains little explored. We hypothesized that, under Mediterranean conditions, pine plantations can generate high-intensity fires that increase fire severity and show limited post-fire recovery compared with other vegetation types.
2. We integrated Sentinel-2 imagery with digital terrain models, vegetation maps, and national forest inventories to assess fire severity (dNBR) and one-year post-fire recovery from three large wildfires in Spain. We then used linear models to investigate the patterns of fire severity and early recovery across vegetation types. Change-point models were applied to pine plantations to evaluate whether reducing tree density beyond a specific point can limit fire severity.
3. Pine plantations exhibited significantly higher severity and lower early recovery than other vegetation types, particularly in areas with high tree densities and abundant shrub cover. Moreover, proximity to pine plantations was associated with increased fire severity in adjacent vegetation, whereas reducing tree density below a threshold of 440 trees/ha mitigated fire severity within plantation stands.
4. *Policy implications*. Our findings provide quantitative evidence that pine plantations can exacerbate fire severity under contemporary climate conditions, and that, once burned, these areas seldom recover. Effective spatial planning and management of tree plantations is therefore essential in order to promote more sustainable fire regimes in Mediterranean ecosystems. Therefore, carbon mitigation strategies should carefully consider the risk of establishing dense and continuous pine plantations when implementing future afforestation programmes.

KEYWORDS
afforestation, fire ecology, fuel management, Mediterranean landscapes, Spanish Forest Map, Spanish National Forestry Inventory, wildfires

1 | INTRODUCTION

The frequency and intensity of large and severe wildfires are increasing worldwide, in tandem with global change (Cunningham et al., 2024; Ellis et al., 2021). Fire-critical thresholds are being reached earlier and sustained for longer periods of time (Jones et al., 2022; Pausas & Keeley, 2021) and, as a result, the climatic limitations on fire activity are fading. However, in a context dominated by human-ignited fires (Balch et al., 2017), the observed changes in wildfire patterns result from the interplay between climate and land use (Ellis et al., 2021). Assuming that anthropogenic climate change is altering the relationships between climate and fuel, the recent increase in large and severe wildfires must be assessed considering the changes in the extent of continuous fuels and ignition sources (Jolly et al., 2015; Pausas & Keeley, 2021). In light of the increasing prevalence of climatic conditions favouring the spread of fire, effective management at the landscape scale emerges as a critical tool, especially in fire-prone areas (Gómez-González et al., 2023; Moreira et al., 2020; Pausas & Ribeiro, 2013).

In Mediterranean regions worldwide, fire is not only a natural phenomenon but also an essential element for ecosystem functioning (Keeley et al., 2012; Pausas & Keeley, 2021). However, the occurrence of anomalously large, severe wildfires that exceed the suppression capacity has been a relatively recent phenomenon in these regions (Fernandes et al., 2016; McWethy et al., 2018), often associated with poorly managed tree plantations (e.g. Barquín et al., 2022; Gómez-González et al., 2018). Moreover, there is a growing concern that Mediterranean forests may be losing their natural capacity to recover productivity, functions and services after severe disturbances, particularly so when the historical variability is exceeded (Miranda et al., 2023; Senf & Seidl, 2022; Xu et al., 2024). In the case of fire severity, this can potentially lead to negative ecological consequences at the landscape scale, such as soil erosion (Scott et al., 2009) and limited post-fire recovery (Grau-Andrés et al., 2024).

During the 19th and 20th centuries, Europe undertook extensive reforestation and afforestation efforts with the aim of supplying forestry products, restoring and protecting water catchments, and promoting rural employment (Pausas, Bladé, et al., 2004; Vadell et al., 2016). Since the establishment of the European Union, the expansion of tree plantations can be largely explained as a way to support the economic recovery of agricultural land by the Common Agricultural Policy (CAP; Pausas, Bladé, et al., 2004) and, more recently, as a climate mitigation strategy to reduce atmospheric CO_2 (Freer-Smith et al., 2019). Today, the European Union continues to pursue ambitious afforestation initiatives such as the *Three Billion Trees* pledge, which has faced criticism for overlooking the risk of fires associated with extensive plantations (e.g. Gómez-González et al., 2020). In Spain, over 5.6 million hectares were afforested and/or reforested in the last 150 years, predominantly with pine species (over 70% of the total forested area; see Vadell et al., 2016). Such extensive transformations of the landscape have resulted in substantial modifications to the type and structure of fuel at large spatial scales, although the potential impacts on fire severity and post-fire recovery remain to be assessed.

The spatial continuity and homogeneity of fuel generated by tree plantations facilitate fire spread, particularly if left unmanaged (Barquín et al., 2022; Lindenmayer et al., 2023). When fuels are connected horizontally (within and between stands) and vertically (between canopy and understorey), the risk of extreme wildfires increases significantly. This has recently been evidenced in Mediterranean contexts, such as the Portugal and Chile wildfires in 2017 (Gómez-González et al., 2018; McWethy et al., 2018), the Australian wildfires of 2019–2020 (Bowman et al., 2021; Lindenmayer et al., 2023), or the South African wildfires of 2021 (Giddey et al., 2021). Additionally, fires affecting tree plantations may escape and increase fire severity in adjacent areas (Bowman et al., 2020; González et al., 2023). It is therefore crucial to understand how tree density and the proximity of forest plantations shape fire severity in Mediterranean contexts, in order to prioritize management areas and prevent uncontainable wildfires.

While the impacts of *Eucalyptus* plantations on Mediterranean fire regimes have been described (e.g. Bowman et al., 2021; Fernandes et al., 2019), pine plantations have received comparatively little attention. This may be attributed to the fact that most *Pinus* species employed in forestation initiatives are native to the Mediterranean basin (e.g. *Pinus pinaster*; Vadell et al., 2016) and thus pine plantations are often mistaken for natural vegetation. Additionally, pines are often found intertwined with *Quercus* woodlands, forming mixed forests (Sheffer, 2012). Coupled with the abandonment of rural areas (see e.g. Viedma et al., 2015), the distinction between pine plantations and pine forests in the Mediterranean basin has become increasingly challenging. Consequently, most existing research does not differentiate between pine plantations and forests (Fernández-Guisuraga & Calvo, 2023; Viedma et al., 2020) or exclude tree plantations (e.g. Beltrán-Marcos et al., 2024) when assessing fire severity in Mediterranean areas. This can result in misguided policies regarding biodiversity conservation, climate change mitigation, and the provision of forest services (Liao et al., 2012). Hence, the role of pine plantations in the development of these newly emerging, high-severity wildfires has yet to be formally investigated.

Here, we aim to evaluate the effect of pine plantations on fire severity and vegetation recovery after recent wildfires in a Mediterranean system. We hypothesized that pine plantations would generate high-severity fires and low post-fire recovery compared to other vegetation types in similar conditions. We selected three large wildfires along a north–south gradient in Spain to (i) evaluate the relative contribution of pine plantations and other Mediterranean vegetation types to fire severity and early recovery at the landscape scale; (ii) evaluate how proximity to pine plantations influences fire severity of neighbouring vegetation; and (iii) identify whether there is a threshold at which tree density magnifies fire severity in pine plantations, and how silvicultural management can buffer this effect.

2 | METHODOLOGY

2.1 | Data acquisition and processing

2.1.1 | Selected wildfires

We used data from multiple open-access sources (Data Sources) to examine three large wildfires that occurred in consecutive years (2021, 2022 and 2023) distributed along a latitudinal gradient in Spain (Figure 1; Table 1). This gradient spans most bioclimatic zones in the western Mediterranean region (Bracho-Estévanez et al., 2023). All three wildfires burned extensive areas with extreme fire severity (European Forest Fire Information System; EFFIS).

These areas are characterized by a representative range of Mediterranean ecological conditions and land-use contexts, including similar vegetation structure, disturbance regimes, and land-use history (Beltrán-Marcos et al., 2024), as well as the establishment of pine plantations. *Culebra* is composed of two adjacent wildfires that occurred 32 days apart, whose perimeters have been merged and are here analysed as a single wildfire.

2.1.2 | Vegetation maps and forest inventories

The distribution of dominant vegetation and fuel structure was assessed using two complementary, open-data catalogues: the Spanish Forest Map (MFE, Spanish acronym for *Mapa Forestal de España*) and the Spanish National Forest Inventory (IFN, *Inventario Forestal Nacional*). The MFE provides basic vegetation cartography but lacks the fuel structure data available in the IFN. The MFE for the *Culebra* and *Hurdes* fires was produced in 2020 and 2018, respectively. However, the most recent MFE for the *Bermeja* fire dated from 2006. In order to ensure comparability, the *Bermeja* MFE data were updated using regional vegetation maps and fire history records (see Appendix II).

Vegetation types were then classified according to the dominant tree species in order to account for site-specific differences in species composition. In this regard, 'pine plantations' were defined as all formations of *Pinus radiata* (non-native) and/or all native *P. pinaster* and *P. sylvestris* stands located in areas with clear plantation cues. Other *P. pinaster* and *P. sylvestris* dominated formations (along with patches of the native *Pinus nigra*) with no clear plantation patterns present were classified as 'pine forests' (details below). Patches of *Quercus suber*, *Q. pyrenaica* and *Q. ilex* were classified as 'oak forest'.

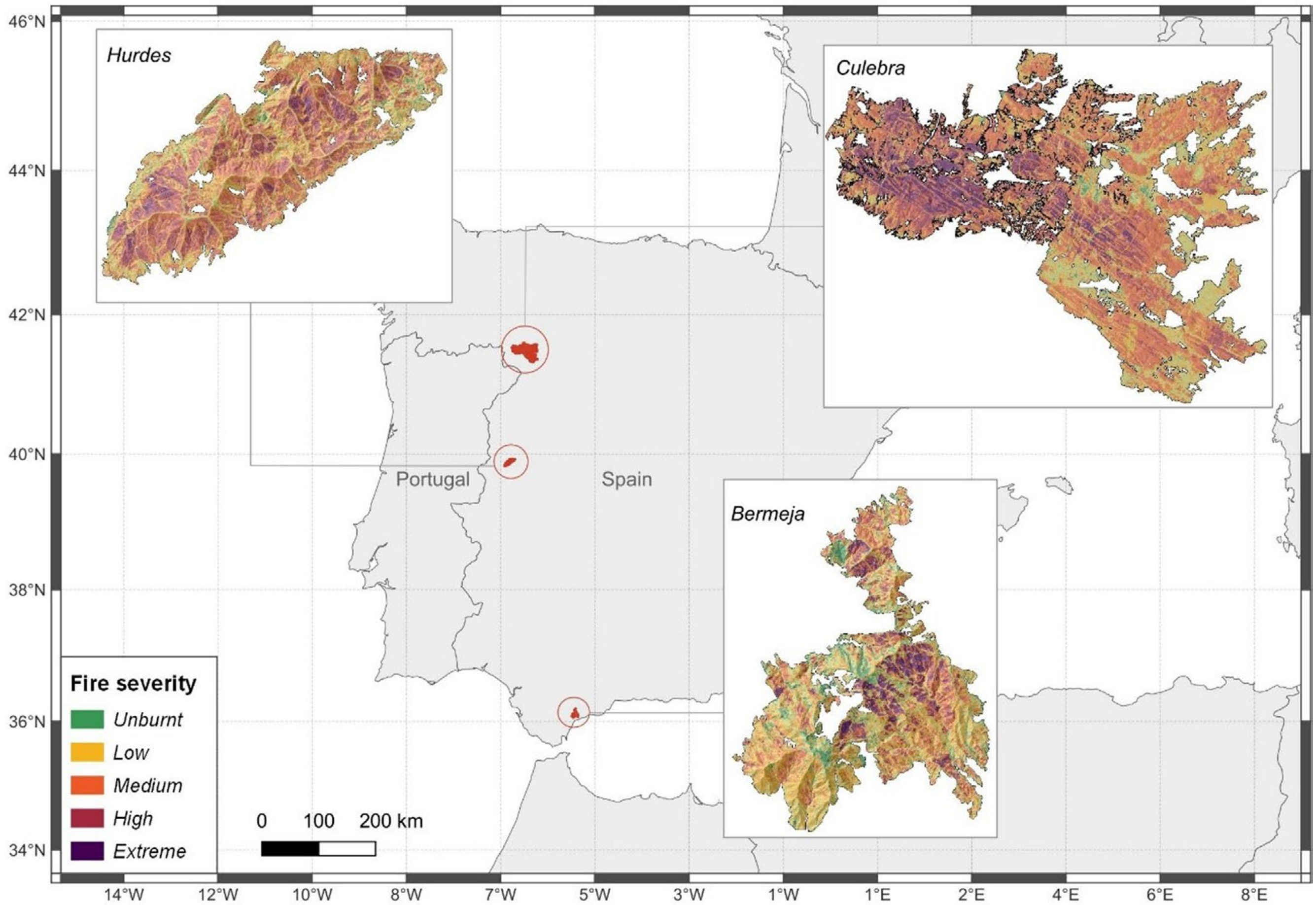

FIGURE 1 Selected wildfires in the study area. Fire severity estimated by the delta Normalized Burn Ratio (dNBR) was categorized according to EFFIS thresholds (Key & Benson, 2006). Individual, high-resolution maps are available as Supporting Information.

TABLE 1 Characteristics of the selected wildfires. Climate values were obtained from the nearest weather station (AEMET) using the *climaemet* library (Pizarro et al., 2021).

		Culebra		
Wildfire ID	***Bermeja***	**Wildfire 1**	**Wildfire 2**	***Hurdes***
Locality (Province)	Sierra Bermeja (Málaga)	Sierra de la Culebra (Zamora)	Losacio (Zamora)	Sierra de las Hurdes (Cáceres)
Burned area (ha)	9670	27,242	31,473	10,863
Start and control dates	08/09/2021, 24/09/2021	15/06/2022 24/06/2022	17/07/2022 31/08/2022	17/05/2023 22/05/2023
Mean annual precipitation (mm)	600	700	700	650
Mean annual temperature (°C)	17.5	11.5	11.5	13.5
$Temp_{10}$ (°C)[a]	23.8 (+6.3)	20.4 (+8.9)	25.2 (+13.7)	18.2 (+4.7)
RH_{10} (%)[a]	55.9	41.1	24.9	36.2
WS_{10} (m/s)[a]	2.19	2.7	1.8	2.81
Centroid coordinates	36° 30′ 42.23″ N 5° 10′ 44.58″ W	41° 54′ 57.56″ N 6° 12′ 2.12″ W	41° 52′ 24.15″ N 6° 1′ 50.06″ W	40° 17′ 6.08″ N 6° 25′ 28.07″ W

[a]Values of $Temp_{10}$ (temperature), RH_{10} (relative humidity) and WS_{10} (wind speed) correspond to the mean values for 10 days before each fire ignition date. Temperature anomaly (difference between mean annual temperature and $Temp_{10}$) is given between brackets.

Other less frequent dominant tree species such as *Arbutus unedo*, *Alnus glutinosa*, *Populus nigra*, *Fraxinus angustifolia*, *Castanea sativa* and *Abies pinsapo* were considered as 'other forests' (mostly broadleaf forests). Treeless woody vegetation was grouped as 'shrublands'. The remaining land-use types (i.e. pastures, agricultural and artificial areas) were excluded from the analysis, since concerns exist about the reliability of spectral indices in such areas (Fassnacht et al., 2021).

Pine plantations in the study areas were identified through a two-step process. First, we identified all *Pinus*-dominated patches classified as planted forests in the MFE. Second, the total extent of pine plantations was assessed through photointerpretation of historical orthoimagery. We used 0.50–1 m/pixel images from all available national flights across the affected areas to identify typical indicators of plantation features (e.g. mechanical terracing, uniform tree spacing and/or linear planting; see Figure S11). The resulting distribution of pine plantations was verified in recent pre-fire orthophotos (PNOA 2019 for *Bermeja*, and PNOA 2020 for *Culebra* and *Hurdes* wildfires) at 0.50 and 0.25 m/pixel respectively, to ensure no substantial differences. Hence, patches classified here as 'pine plantation' have been validated to the best of available sources, while the remaining pine-dominated patches are considered 'pine forest'. We acknowledge that some 'pine forest' patches may have an anthropogenic origin (old plantations without terracing, etc.), and thus, our results are conservative in relation to the hypothesis.

Forest structure and pre-fire fuel characteristics were obtained from the IFN. This inventory gathers information from thousands of plots distributed across a 1 km^2 cell grid within all Spanish forests. Each plot comprises four concentric subplots of 5, 10, 15 and 25 m radius, where the size, species identity and state (dead or alive) of trees are periodically surveyed (Alberdi et al., 2016). Using this information, basal area and tree density were calculated per species, as well as structural heterogeneity, which corresponds to the coefficient of variation of live tree heights (CVH). The shrub cover per plot was obtained by adding up the cover of all taxa measured within the 10-m radius plot in the IFN (Alberdi et al., 2016).

Forest types in the IFN plots were classified according to the dominant composition of their basal area. A plot was classified as either pine plantation (149 plots), pine forest (32) or oak forest (50) if at least 50% of the plot basal area belonged to one group of species, and as 'mixed forest' (28) otherwise, following Astigarraga et al. (2020). The IFN records the presence and absence of recent silvicultural practices at each plot (e.g. tree cutting, thinning or understorey clearing). Based on this information, we classified plots as 'managed' if any of these practices were recorded, and 'unmanaged' when the absence of recent silvicultural treatments is explicitly indicated. Most managed plots corresponded to pine plantations (34), whereas few records of management treatments were found in either oak (11), pine (3), or mixed forests plots (2).

2.1.3 | Topography

Topographic characteristics (altitude, slope, aspect and solar radiation) within the wildfire perimeters were obtained from a Digital Elevation Model (DEM) at 5 m/pixel resolution from the PNOA-LiDAR national programme. Aspect values were linearized using a cosine function, specifically cos*(degrees*pi/180). This converts angles into values from −1 (the southernmost aspect) to +1 (the northernmost aspect), since north-facing slopes are more humid and likely accumulate more fuel than in south-facing slopes in the northern hemisphere.

2.1.4 | Fire severity and early recovery mapping

Fire severity has been defined as a measure of the magnitude of fire impacts to vegetation and soils (Keeley, 2009). We selected cloud and smoke-free scenes in Sentinel-2 imagery to assess fire severity in each wildfire for three temporal windows: pre-fire, immediately post-fire, and 1-year post-fire. The anniversary dates were matched as closely as possible to original dates in order to minimize phenological differences.

The difference in the spectral response of healthy and burned vegetation is most pronounced in the near-infrared (NIR) and short-wave infrared (SWIR) regions of the electromagnetic spectrum (Key & Benson, 2006). The Normalized Burn Ratio (NBR; Figure 2, Equation 1) uses the shifts between these wavelengths to assess fire damage.

The delta NBR (dNBR; Equation 2) is a well-established indicator of fire severity at the landscape scale (Beltrán-Marcos et al., 2024), and has demonstrated strong agreement with field-based assessments in many ecosystems worldwide, including complex Mediterranean landscapes (Fassnacht et al., 2021; Llorens et al., 2020). The dNBR has also been reported as the most effective metric within the context of our study areas (García-Llamas et al., 2019).

$$\mathrm{NBR} = \frac{\mathrm{NIR} - \mathrm{SWIR}}{\mathrm{NIR} + \mathrm{SWIR}} \quad (1)$$

$$\mathrm{dNBR} = \mathrm{NBRpre} - \mathrm{NBRpost} \quad (2)$$

By measuring the relative differences in the spectral reflectance of vegetation pre- and post-fire, this index reflects the magnitude of ecological change. In this sense, low fire severity reflects minimal deviations from the pre-fire response (i.e., mostly unaffected vegetation), while high fire severity indicates strong reductions in canopy moisture and high biomass consumption (Keeley, 2009).

We defined early recovery (Equation 3) as one minus the ratio between fire severity 1 year after the fire event (dNBR 1 = pre-fire NBR − NBR at year +1) and dNBR immediately after (Equation 2).

$$\mathrm{Recovery} = 1 - \frac{\mathrm{dNBR\ 1}}{\mathrm{dNBR}} \quad (3)$$

This allows capturing the proportional decrease in spectral damage in the first year after fire, which in turn serves as an indicator of vegetation recovery. Recovery values close to 1 indicate strong recovery (i.e. spectral signal is close to pre-fire levels), whereas values near 0 indicate little to no change from the initial fire damage after 1 year. Negative recovery values would indicate further degradation, such as post-fire salvage logging. These negative values were therefore masked and removed from the final dataset in order to avoid confounding effects. Imagery processing and calculation of spectral index were conducted using Google Earth Engine (GEE).

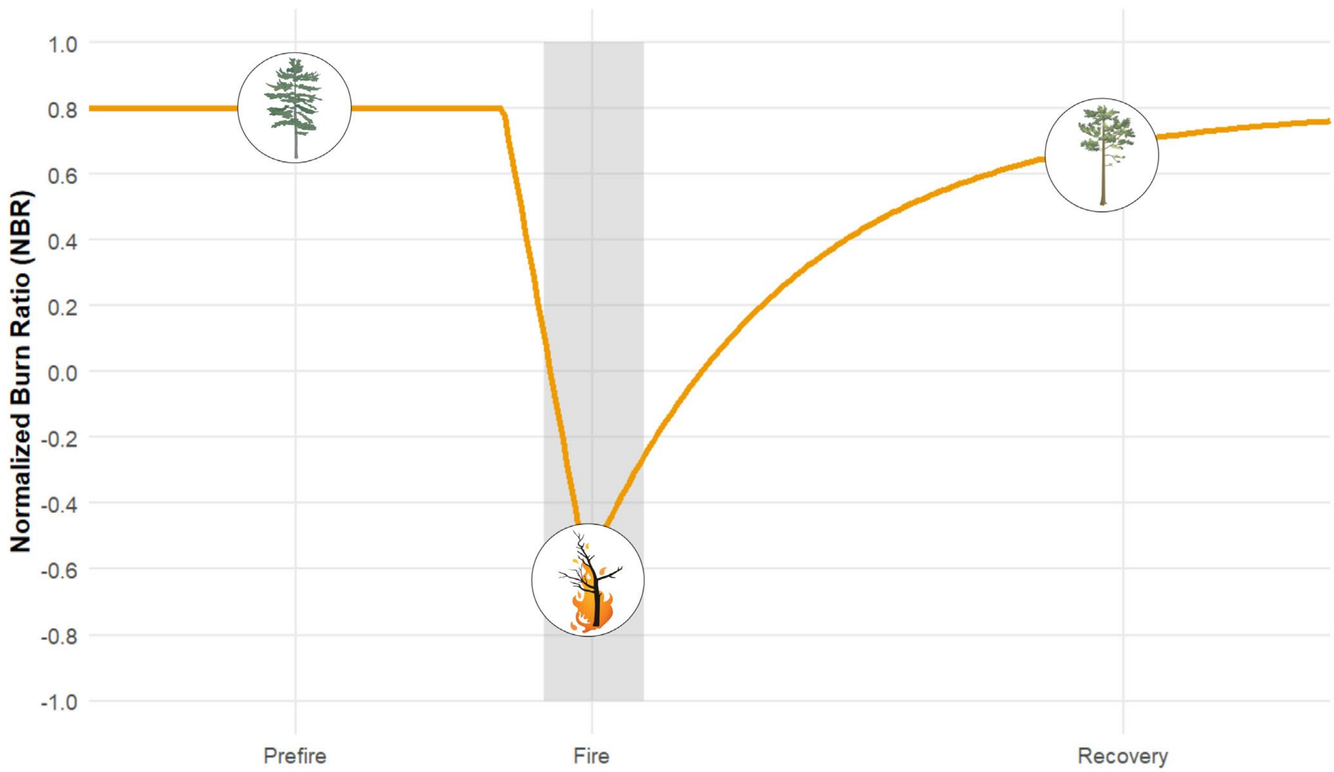

FIGURE 2 Conceptual NBR trajectory showing high pre-fire values, sharp post-fire decline, and gradual recovery towards initial values.

2.1.5 | Sample design and statistical analysis

All IFN plots within wildfire perimeters were retrieved, yielding a total of 289 plots from the third (IFN3, $n=74$ for *Bermeja*) and fourth (IFN4, $n=163$ for *Culebra* and $n=52$ for *Hurdes*) inventories. Then, plots of 50-m radius were generated by applying a circular buffer from the coordinates of each IFN plot centroid. Accordingly, random 50-m radius plots were also generated within a 1-km regular grid on the MFE dataset for each wildfire scar. This ensured the grid spacing was closely aligned with IFN plots, that at least five Sentinel-2 pixels were contained, and minimized spatial autocorrelation. This approach resulted in 756 plots across the *Bermeja* (87), *Culebra* (566) and *Hurdes* (103) perimeters.

Median values of immediate fire severity (dNBR) and early recovery (Recovery) were obtained from the two datasets (MFE and IFN), since this metric is less sensitive to extreme values (e.g. pixel errors). Negative values of fire severity (~1%) were discarded. We also obtained topographic characteristics (mean altitude, slope, aspect and solar radiation) and dominant vegetation or forest type in each plot. The distance between the centroid of each plot and the nearest pine plantation was calculated using the *st_distance* function in the *sf* R package (Pebesma, 2018).

To investigate the relative contribution of pine plantations to fire severity (dNBR) and early recovery at the landscape scale, linear models (LM) were fitted for each dataset (MFE and IFN). A stepwise forward model selection process, starting from the null model (no explanatory variables), was performed to identify the most relevant predictors of fire severity and early recovery. Correlation among predictors was assessed using Pearson correlation ($\alpha=0.05$). Uncorrelated, significant predictors were sequentially added to the model using an *F* test ($\alpha=0.05$). For MFE data, vegetation cover, wildfire ID and topographic covariables were added as fixed terms. For the IFN, the most relevant variables were forest type, wildfire ID and uncorrelated fuel structural characteristics (namely tree density, shrub cover and CVH). The absence of spatial autocorrelation in the response variables was tested using Moran's *I* index, available from the *spdep* package (Bivand, 2022). Normality of the residuals and goodness of fit were verified via the *performance* package (Lüdecke et al., 2021). Lastly, the significance of the explanatory variables of fire severity and early recovery in both databases was assessed through ANOVA tests.

To explore whether proximity to pine plantations influences fire severity in other vegetation types, a linear model (LM) was fitted to the MFE dataset (excluding pine plantations) to predict fire severity with the remaining vegetation types that were located within 2 km of a pine plantation. This model included the distance between the plot centroid and the closest pine plantation as fixed terms, while wildfire ID and average solar radiation were included as covariates.

To evaluate whether there is a threshold at which fire severity in these plantations escalates without silvicultural management, a Threshold Generalized Linear Model was fitted to pine plantations in the IFN dataset using the *chngpt* package (Fong et al., 2017). Fire severity (dNBR) was defined as the response variable, while wildfire ID and shrub cover were included as covariates. Tree density was modelled as a thresholded variable with a step change type, which assumes that its effect on fire severity changes abruptly at a given threshold. Since these models do not allow the inclusion of the same variable as both fixed and thresholded terms, and hence we cannot test for interactions between density and management; separate models were fitted for managed and unmanaged plantations. All analyses were conducted in R (version 4.3.1). Ethical approval was not required for this work.

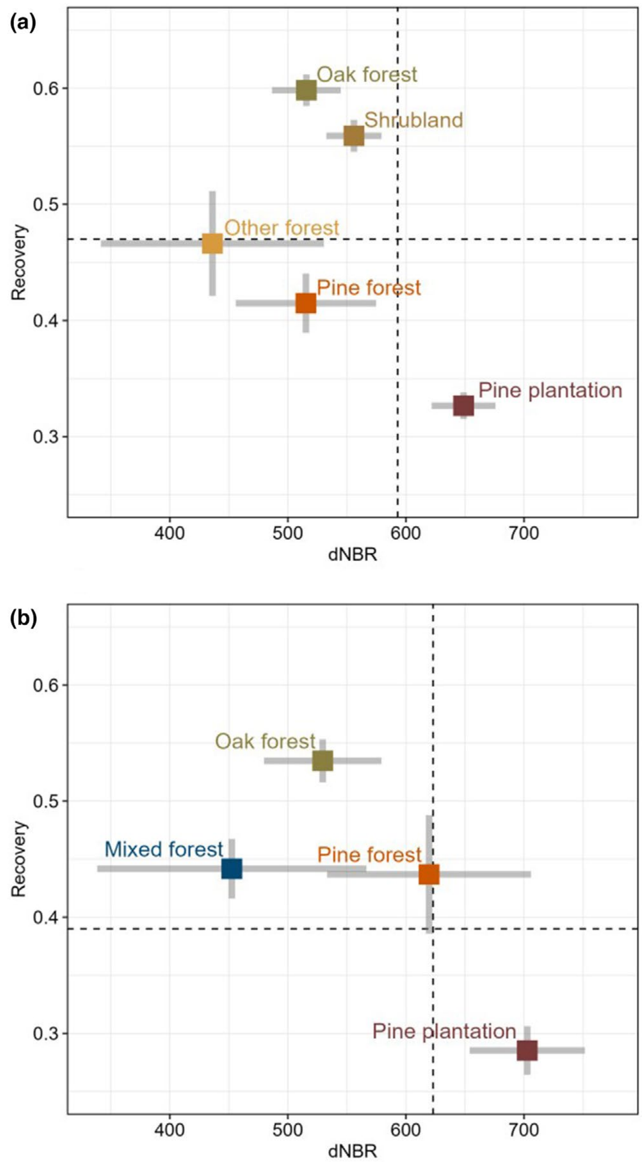

FIGURE 3 Relationship between fire severity (dNBR) and early post-fire recovery (Recovery) for the different vegetation types affected in the *Bermeja*, *Culebra* and *Hurdes* wildfires. Panel (a) corresponds to the MFE data, and panel (b) to the IFN. Dashed lines correspond to global mean values. Coloured squares represent group mean values, and grey bars indicate ±2 standard deviations.

3 | RESULTS

3.1 | Patterns of fire severity and early recovery

Between 2021 and 2023, the *Bermeja*, *Culebra* and *Hurdes* wildfires affected approximately 80,000 ha of pine plantations, pine forests, oak forests, shrublands, and mixed forests with varying degrees of fire severity and early recovery. Pine plantations burned at higher severity and showed lower early recovery values than any other vegetation type across the three wildfires investigated (Figure 3). Oak forests and shrublands generally showed moderately high levels of fire severity, with no substantial variations among them (mean dNBR values between 400 and 600; Figure 3). However, these vegetation types (particularly oak forests) exhibited a significantly stronger spectral recovery than pine plantations after 1 year, both in the MFE (Figure 3a) and IFN (Figure 3b) based models.

The most parsimonious models fitted to MFE data indicate that vegetation type played a critical role, showing a strong and significant effect on both fire severity ($F = 18.88$, $p < 0.001$; ANOVA Table S3) and early recovery ($F = 60.40$, $p < 0.001$; ANOVA Table S4). Additionally, north-facing slopes experienced significantly higher fire severity. Similarly, the model fitted to IFN data revealed that fire severity was significantly influenced by fuel structure, increasing with shrub cover and tree density (Figure 4), while showing a non-significant tendency to decrease with greater heterogeneity in tree heights. Once fuel structure was incorporated, topographical characteristics had no significant effect on either response. However, pine-dominated areas consistently showed higher fire severity than oak and mixed forests (Figure 4). The differences between vegetation types held independently of fuel structure covariates.

The *Culebra* wildfire showed higher values of fire severity (Tables S3 and S5). However, we did not find a significant effect of the two-way interaction between vegetation types and wildfire on fire severity (F (7, 602) = 1.80, Mean Sq. = 58.96, p-value = 0.08; Table S3), indicating that patterns were similar across wildfires.

3.2 | Fire severity of pine plantations: Adjacency effect and silvicultural management

Fire severity significantly decreased with increasing distance from pine plantations (ES = −0.17; Figure 5a), particularly in the *Culebra* wildfire. In other words, plots located close to pine plantations exhibited higher fire severity than those located further away, with mean fire severity being particularly high within the immediate vicinity of pine plantations. We found no significant effect of vegetation types other than pine plantations, nor of topographical characteristics.

The step change-point model identified a significant threshold effect of tree density in unmanaged pine plantations, estimated at approximately 440 trees per hectare (Figure 5b). Below this

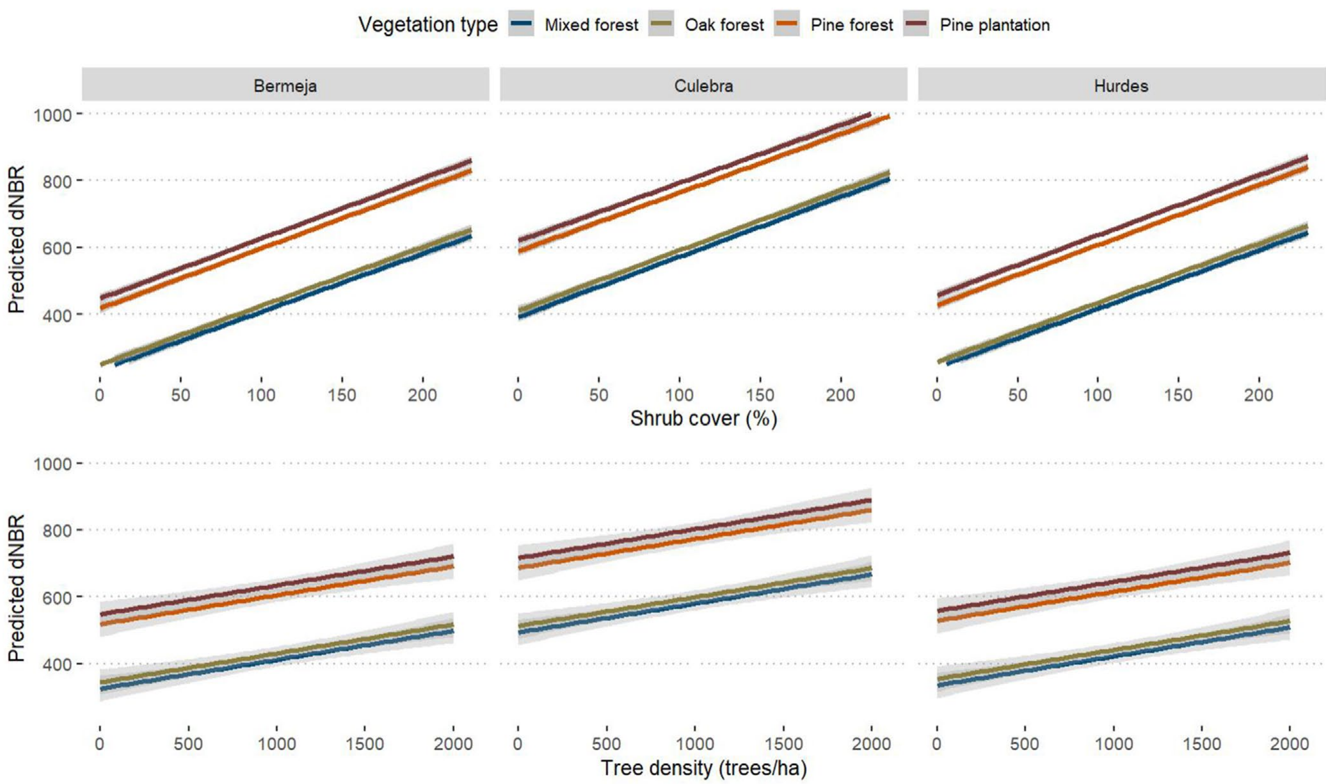

FIGURE 4 Effect of fuel structural characteristics (shrub cover and tree density) on fire severity (dNBR) of the *Bermeja*, *Culebra* and *Hurdes* wildfires. Shaded areas represent the 95% confidence interval around the fitted line. Based on the model in Table S5.

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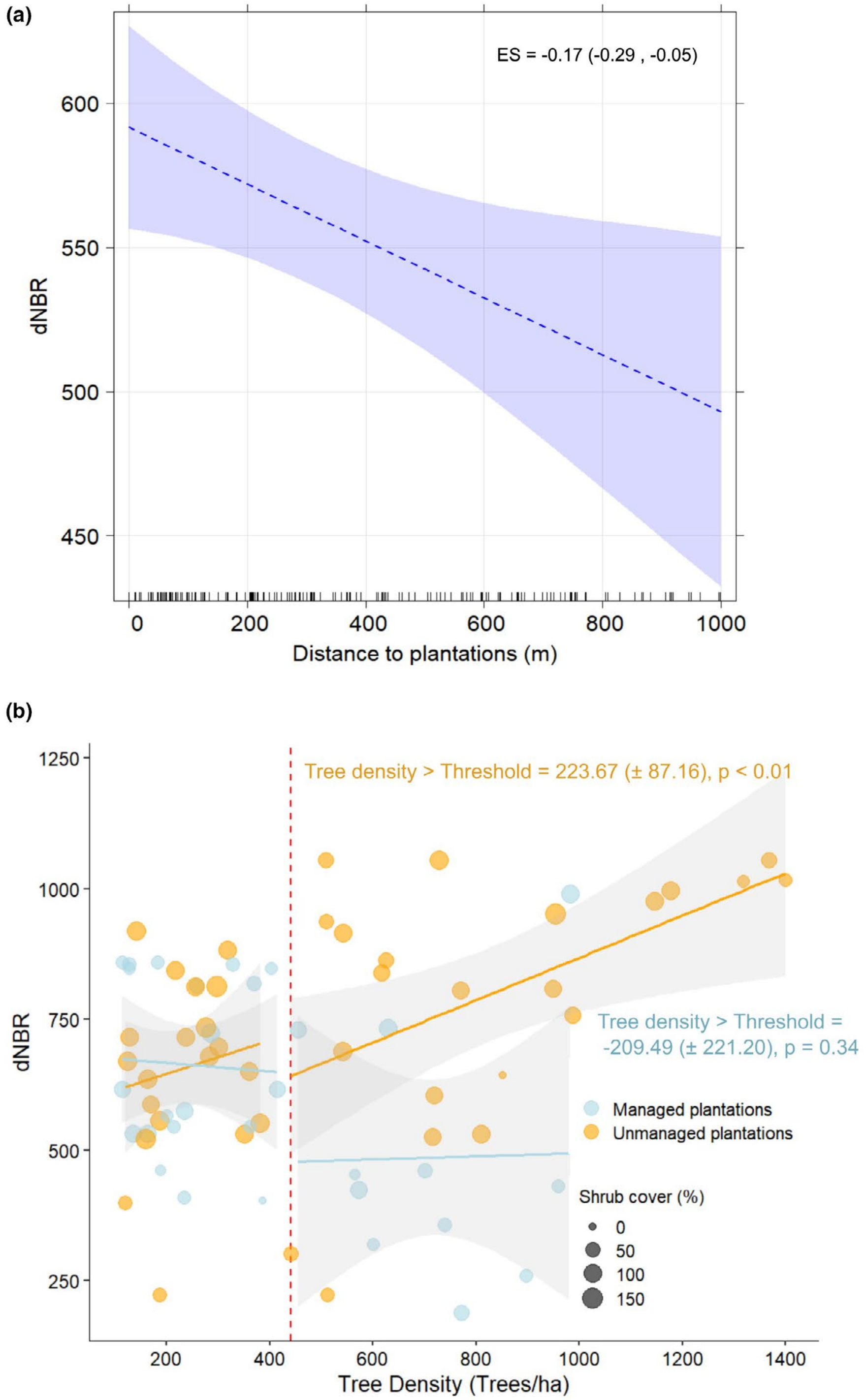

FIGURE 5 (a) Effect of distance to plantations on observed fire severity (dNBR) of other vegetation types in the first kilometre. ES value represents the effect size and 95% confidence interval. (b) Effect of tree density on fire severity (dNBR) in unmanaged (orange) and managed (blue) pine plantations. The red vertical line marks the 440 trees/ha threshold. Shaded areas represent the 95% confidence intervals.

threshold, fire severity did not differ significantly between managed and unmanaged plantations (Figure 5b). However, beyond 440 trees per hectare, the dNBR of unmanaged plantations increased sharply by 223.67 units (±87.16 SE; $p < 0.001$; Figure 5b). Although tree density ranges mostly overlap, managed plantations exhibited a lower mean tree density (411.3 trees/ha) compared to unmanaged plantations (545.6 trees/ha). We did not find a significant threshold on the fire severity of managed plantations, neither across wildfire events (Figure 5b and Table S9).

4 | DISCUSSION

Our results indicate that, in Mediterranean landscapes, pine-dominated areas generate higher fire severity and lower early recovery rates than other vegetation types, especially oak-dominated forests. We show that extreme fire severity reached within pine plantations can extend to neighbouring vegetation. Moreover, reducing tree density within plantation stands appears to effectively mitigate their fire severity. Our findings provide novel, quantitative evidence of the pivotal role that pine plantations can play in promoting severe wildfires, and underscore the importance of silvicultural management in preventing unmanageable wildfires in dense and continuous plantations.

The wildfires examined here occurred during extreme heatwaves, with temperature anomalies ranging from 4.7 to 13°C above historical averages (Table 1). These values largely exceed the natural variability range and are consistent with projections under anthropogenic climate change (Jones et al., 2022; Tejedor et al., 2024). Thus, our results do not support the assumption that all vegetation types burn similarly under extreme fire conditions. Instead, they show that differences in vegetation flammability matter.

4.1 | Fire severity and early recovery

In line with previous studies, fire severity was best explained using IFN-based models, where fuel type, structure and topography were collectively accounted for (Álvarez et al., 2024; Fernandes et al., 2010; Viedma et al., 2015). Fuel structure variables captured fine-scale characteristics that significantly influence post-fire vegetation dynamics at the landscape scale (Fernández-Alonso et al., 2013). Specifically, areas with abundant shrub understorey and dense tree overstorey supported the highest fire severity, through greater fuel load and continuity. Nonetheless, the distinct effect of pine plantations was consistent across MFE (lacking fuel structural characteristics) and IFN-based models. This agreement points to inherent ecological traits, beyond fuel structure, that strongly drive the differences in fire severity and recovery among vegetation types.

Although evidence at the landscape scale is still limited, there is broad consensus that pines have species-specific traits (e.g. high resin content, needle-type leaves and litter, retaining of dead branches; Fonda, 2001; Pausas, Bradstock, et al., 2004) that can influence flammability and may contribute to explaining the observed fire severity patterns. In the study areas, pine plantations include both exotic (*Pinus radiata*) and native species (*P. pinaster*; *P. sylvestris*), hence preventing the possibility of testing whether fire severity differs among pine species. Broader-scale assessments are needed to determine whether plantations with different pine tree species exhibit different trends in fire severity in order to optimize fuel management practices under a warming climate.

In MFE-based models, pine forests showed lower fire severity and higher spectral recovery than pine plantations, which have been scarcely documented in the Iberian context (Álvarez et al., 2024; Beltrán-Marcos et al., 2024). Although some pine forests in the study areas might be older plantations—a possibility that requires further investigation—a more open fuel distribution in pine forests could lead to more heterogeneous structures with gaps in fuel continuity, reducing fire severity and enhancing vegetation recovery (Bowman et al., 2020).

On the other hand, the low recovery observed in pine plantations after 1 year suggests that extreme fire severity may have impacted soil properties (Maia et al., 2012; Scott et al., 2009), the shrub strata and/or seed banks that critically assist the vegetation recovery process after fire (Galloway et al., 2017). In addition to the pre-fire conditions related to the plantation itself, such as the shadow effect caused by dense tree cover and increased erosion after mechanical terracing (Ojeda, 2020), the impacts of exacerbated fire severity will likely impair post-fire regeneration in these areas (Scott et al., 2009).

By contrast, oak forests showed strong spectral recovery 1 year after fire, despite initial high severity. Such recovery might be attributed to the fire-resistant strategy of most *Quercus* species, which rely on thick bark and protected bud banks for resprouting after fire (Keeley et al., 2012). This would be particularly pronounced under moderate or low severity, where these structures are less likely to be damaged. Lower fire severity may also allow a faster reestablishment of the vegetation community, protecting the topsoil layer from further erosion (Galloway et al., 2017).

Consequently, Mediterranean oak forest ecosystems may rebound earlier than pine-dominated stands after high-severity fires. This is consistent with earlier research, suggesting that broadleaf forests can contribute to alleviate fire-induced ecosystem damage and favour more resilient landscapes in the Mediterranean Basin (Álvarez et al., 2024; Moreira et al., 2009), with important implications for ecosystem management and long-term carbon storage within Mediterranean forests (Xu et al., 2024).

4.2 | Managing the landscape in and around pine plantations

We found that proximity to pine plantations can increase fire severity in adjacent vegetation, potentially compromising their persistence and recovery capacity (Beltrán-Marcos et al., 2024). This side effect can be particularly concerning when surrounding vegetation is not adapted to fire or high fire severity. For example, following the 2017 Chilean megafires, González et al. (2023) found that *Pinus radiata* showed strong post-fire recruitment in severely burnt native forest fragments nearby. Our findings corroborate existing evidence showing that impacts associated with plantations may extend beyond their boundaries (Fernandes et al., 2010; Giddey et al., 2021; Lindenmayer et al., 2023), which reinforces calls for a paradigm shift in current wildfire management strategies (Moreira et al., 2020), particularly in Mediterranean ecosystems.

Moreover, our study highlights the crucial role of silvicultural management in maintaining tree densities in pine plantations below critical levels. Notably, the identified threshold was closely aligned with earlier predictions made in fuel models for *Pinus pinaster* stands in Portugal (around 500 trees/ha; Fernandes & Rigolot, 2007). In unmanaged plantations, higher tree density significantly contributed to fuel build-up, while the lack of significance in managed plots suggests that tree biomass removal effectively lowers the risk of higher fire severity.

Fuel load reduction practices seem to effectively mitigate fire severity of pine plantations, indicating that unmanaged plantations

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are more likely to exceed a critical threshold at which fire severity escalates abruptly. This underlines the importance of integrated management strategies that address both the canopy and understorey components to ensure an effective reduction of fire hazards. In the absence of such treatments, recurrent high-severity feedbacks could trigger cascading effects, threatening the provision of key ecosystem services (Fernández-Guisuraga & Calvo, 2023). It is critical that the amount and continuity of fuel in plantations are strategically altered in order to lower the risk of extreme fire severity, particularly in dense and homogeneous stands (Gómez-González et al., 2018, 2023).

4.3 | Limitations

We acknowledge that our analysis relies exclusively on satellite-based estimations, which may not entirely capture fine-scale effects such as absolute biomass consumption or the magnitude of soil damage (Galloway et al., 2017; Scott et al., 2009). We encourage upcoming studies to validate and refine the fire severity and recovery patterns using field-based techniques. This would enhance our knowledge of the unintended effects of forest plantations in complex, fire-prone landscapes. Also, while our study did not explicitly assess within-fire weather conditions, these factors can modulate fuel moisture and fire intensity, affecting fire spread and severity (Álvarez et al., 2024; Viedma et al., 2015). Future research should consider the potential interactions between bottom-up (fuels and topography) and top-down (fire weather) modulators to better understand the drivers of high fire severity in Mediterranean ecosystems.

4.4 | Conclusions and implications for policy

The historic demand for forestry products has largely driven the expansion of pine plantations (Freer-Smith et al., 2019). However, unless substantial investments are made in fuel management treatments, the establishment of extensive plantations in Mediterranean areas can become incompatible with upcoming wildfires. As extreme events are expected to become more frequent, adaptive strategies are urgently needed in order to reduce the risk of high fire severity, especially in forest plantations.

The financial and logistical challenges associated with the management of tree plantations will only increase under a warming climate. It is therefore essential to re-evaluate past and future land-use changes that modify the type and quantity of available fuel, as well as current wildfire management policies and afforestation initiatives for climate mitigation in Mediterranean contexts.

Our results lend support to the hypothesis that dense tree plantations can significantly contribute to highly severe wildfires in Mediterranean landscapes, emphasizing the challenges posed by extensive planting initiatives in the context of global change (Stevens & Bond, 2024). Priority must be given to sustainable forestry plans (e.g. Gómez-González et al., 2022) that reconcile forest productivity with the prevention of unmanageable wildfires and the maintenance of key ecosystem services.

AUTHOR CONTRIBUTIONS

Susana Gómez-González, Juli G. Pausas and Fernando Ojeda conceived the ideas. Juli G. Pausas and Alejandro Miranda designed the methodology. Irene Repeto-Deudero and Verónica Cruz-Alonso collected the data. Irene Repeto-Deudero analysed the data and led the writing of the initial manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data available from Zenodo Digital Repository https://doi.org/10.5281/zenodo.15012068. (Repeto Deudero et al., 2025).

ORCID

Irene Repeto-Deudero https://orcid.org/0009-0008-2159-6794
Fernando Ojeda https://orcid.org/0000-0001-5480-0925
Susana Gómez-González https://orcid.org/0000-0002-3779-8701
Verónica Cruz-Alonso https://orcid.org/0000-0002-0642-036X

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1. Example of post-fire salvage logging in the perimeter of the Sierra de la Culebra.

Table S1. Total area burned by vegetation types and percentage of each class from the total.

Table S2. Graphical summary of the Spanish National Inventory (IFN) variables considered.

Figure S2. Pearson correlation between the predictor variables used to assess fire severity.

Table S3. Results from the linear model fitted using *Spanish Forest Map* (MFE) data to assess fire severity (dNBR).

Figure S3. Post-hoc comparisons among vegetation types on fire severity across wildfires.

Figure S4. Predicted values of fire severity for the affected vegetation types in each wildfire.
Table S4. Results from the linear model fitted using *Spanish Forest Map* (MFE) data to assess early recovery.
Figure S5. Post-hoc comparisons among vegetation types on early recovery across wildfires.
Figure S6. Predicted values of early recovery for the affected vegetation types in each wildfire.
Table S5. Results from the linear model fitted using Spanish National Forest Inventories (IFN) data to assess fire severity (dNBR).
Figure S7. Post-hoc comparisons among forest types on fire severity across wildfires.
Figure S8. Predicted values for fire severity of the forest types affected in each wildfire.
Table S6. Results from the linear model fitted using Spanish National Forest Inventories (IFN) data to assess early recovery.
Figure S9. Post-hoc comparisons among forest types on early recovery across wildfires.
Figure S10. Predicted values of early recovery of the forest types affected in each wildfire.
Table S7. Results from the linear model fitted using *Spanish Forest Map* (MFE) data to assess the influence of the distance to pine plantations on fire severity.
Table S8. Results from the threshold generalized linear model fitted using unmanaged pine plantations.
Table S9. Results from the threshold generalized linear model fitted using managed pine plantations.
Figure S11. Examples of anthropogenic plantation patterns found in the study areas.
Appendix I. High-resolution fire severity (dNBR) maps of the *Bermeja*, *Culebra* and *Hurdes* Wildfires.
Appendix II. Detailed workflow used to update the *Bermeja* Spanish Forest Map (MFE) data.

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