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RESEARCH ARTICLE

Long-term comparison shows protected and non-protected forests differ in harvesting, but not in wildfires or drought-driven dieback

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Abstract

1. While disturbances are essential for biodiversity, their escalation driven by climate change may threaten forest ecosystems. Contrasting approaches to adapt forests to disturbances—intensifying management versus encouraging natural succession towards more mature ecosystems—have sparked a debate about whether protection influences forests' vulnerability to disturbance. This question, however, has barely been investigated.
2. Natura 2000 network is the backbone of biodiversity protection in Europe. We compared the long-term incidence of harvesting, wildfires and drought-driven forest dieback inside and outside Natura 2000 areas in Catalonia (NE Spain) by combining remote sensing-derived maps of harvesting and wildfires (1985–2023), an exhaustive ground survey on forest dieback (2012–2023) and forest characteristics extracted from 3400 permanent plots inventoried in 1990, 2000 and 2015.
3. From 1985 to 2023, remote sensing-identified wildfires and harvesting affected 20% of the total forest area, with 60% attributed to harvesting and 40% to wildfires, highlighting the strong influence of wildfires on Mediterranean landscapes. From 2012 to 2023, the forest area affected by drought-driven dieback (11%) matched the sum of the area of wildfires and harvesting for the same period or that of wildfires for 40 years, which suggests an increasing impact of drought-driven dieback.
4. Harvesting occurrence and intensity were significantly higher outside Natura 2000 sites, whereas protection did not influence wildfires or dieback, triggered by environmental and forest characteristics, that is, bioclimatic region, topography or leaf habit. Ultimately, a higher harvesting intensity did not prevent forests from experiencing drought-driven dieback later.
5. *Synthesis and applications*. Lower forest harvesting in Natura 2000 sites may align with socio-economic barriers often claimed by local communities, but protection

does not influence vulnerability to other disturbances. In a general scenario of reduced forest harvesting in the region, we argue that differences in harvesting due to protection are statistically significant but ecologically irrelevant in influencing wildfires or drought-driven dieback. Moreover, beyond protection status, the lack of effects of the current harvesting intensities in halting drought-driven dieback suggests they may be insufficient for supporting forests' adaptation to climate change. Additionally, other measures (e.g. promoting more drought-tolerant tree species and genotypes) should also be considered.

KEYWORDS
climate change, European Forest Disturbance Atlas, forest disturbances, habitats directive, Natura 2000

1 | INTRODUCTION

Disturbances are discrete and stochastic events that modify the ecosystem structure and functioning, changing resource availability or the physical environment (Turner, 2010). Disturbances are broadly characterized according to the causing agent (biotic vs. abiotic), intensity, seasonality, frequency, return interval or size extent. This set of characteristics for a particular forest type over a long period of time and at a determined spatial scale constitutes the so-called disturbance regime. Eventually, disturbances and their interplay with other drivers of demographic processes (e.g. competition for resources) are essential in determining species composition, the structure and functioning of forests, the provision of ecosystem services and the maintenance of biodiversity (Thom & Seidl, 2016). Notwithstanding this key role, modifications in the disturbance regime (e.g. increases in the frequency, intensity or the size of the events) may threaten forests' continuity, eventually resulting in their collapse (Lloret & Batllori, 2021).

Climate change is escalating the occurrence of natural disturbances (Seidl et al., 2017), and numerous studies have highlighted the increasing severity of extreme events in recent decades (Patacca et al., 2023). Tree species composition and forest structure are among the most crucial resilience predictors for a wide array of forest characteristics and ecosystem services (Lloret et al., 2024). Therefore, by modifying these attributes at the stand level and the spatial arrangement of forest areas at the landscape scale, forest management can reduce the occurrence and intensity of disturbance events (Messier et al., 2019). For example, the spread and intensity of wildfires can be diminished by favouring the presence of less flammable species, reducing the fuel load and controlling the development of a fuel ladder connecting the understory and the canopy layer (Palmero-Iniesta et al., 2017). In a similar way, resistance to drought episodes can be potentially ameliorated by promoting tree species richness, especially by selecting for specific species' arrays (Grossiord, 2020) as well as by reducing competition among trees through density reduction (Moreau et al., 2022). Regarding pests, a higher diversification of the tree layer and heterogeneous stand structures can reduce their impact and enhance the survival of the affected trees (Blanco-Rodríguez & Espelta, 2022). Nonetheless, aside from active management, similar changes in forest composition and structure may also passively occur, triggered through natural succession (Paluots et al., 2024). Therefore, mature or old-growth forests are often considered more resilient to disturbances than managed ones. Their greater functional diversity, structural complexity and varied microclimatic conditions help buffer against extreme environmental changes (Frey et al., 2016). Moreover, these forests retain a larger amount of abiotic and biotic 'legacies' such as nutrients or reproductive structures (seed- or bud-banks), essential to facilitate post-disturbance recovery (Johnstone et al., 2016; Lloret et al., 2012).

The existence of different and somewhat opposite itineraries to foster forest resilience (i.e. increasing forest management vs. promoting natural succession) has often resulted in a lively debate on whether forests in protected areas may be more or less prone to experiencing natural disturbance events in comparison to more intensively managed ones (De Koning et al., 2014). Global-level studies have shown that protected areas generally experience lower rates of anthropogenic disturbance; although, particularly for some regions, illegal activities may still pose significant threats (Wade et al., 2020; see also Geldmann et al., 2019). On the other hand, protected areas also tend to exhibit more natural regimes for other disturbances, such as wildfires (Mansuy et al., 2019). Yet, we still lack comprehensive analyses of the role of protection in the co-occurrence of anthropogenic and natural disturbances, and their potential interaction.

Europe is one of the territories with a higher percentage of protected land (17.5%; Evans, 2012) and the backbone of this protection system was the creation of the Natura 2000 network under the Habitats (92/43/EEC) and Birds Directives (79/409/EEC; 2009/147/EU). The approach taken in the implementation of Natura 2000 followed an integrative strategy aimed at balancing nature conservation with maintaining human activities (e.g. agricultural and forestry practices) to ensure inhabitants' well-being (Jones et al., 2015). However, there are several studies pinpointing the negative perception of this protection system by stakeholders who consider protection to hamper rural development and diminish local well-being (Rodríguez-Rodríguez & López, 2020). The main constraints argued include restrictions on management activities (e.g. reduced logging intensity, pesticide bans), excessive bureaucracy and a highly conservative mindset among protected area staff (Rodríguez-Rodríguez et al., 2021). Ultimately, it has been argued that protection leads to reduced forest management (de Dios et al., 2025), resulting in forest structures and landscape patterns that are more prone to suffering disturbances (see for wildfires Rodrigues et al., 2023; Kirkland et al., 2023; but Kirkland et al., 2024).

Despite the importance of the protection versus non-protection debate, no long-term study, to our knowledge, has examined whether forests inside and outside Natura 2000 sites differ in the occurrence of natural (wildfires, drought-induced dieback) and anthropogenic (forest harvesting) disturbances. Furthermore, the influence of potential differences on harvesting extent on the occurrence of these natural disturbances also remains unexplored. Thus, the main aim of this study was to investigate the occurrence of natural disturbances and the extent of forest harvesting inside and outside Natura 2000 areas in Catalonia (NE Spain). To address this question, we integrated multiple data sources: a long series of maps of natural and anthropogenic disturbances (1985–2023) derived from the European Forest Disturbance Atlas (EFDA; Viana-Soto & Senf, 2025), detailed forest characteristics obtained from ca. 3400 plots surveyed across three consecutive forest inventories (1990, 2000 and 2015) and an exhaustive ground survey documenting drought-induced forest dieback episodes (2012–2023).

2 | MATERIALS AND METHODS

2.1 | Study area

Catalonia (NE of continental Spain) is a region covering 32,114 km^2, limited to the north by the Pyrenees range and to the east by the Mediterranean Sea (Figure 1a). It presents a rough topography, a high altitudinal range and an intense coastal-inland temperature

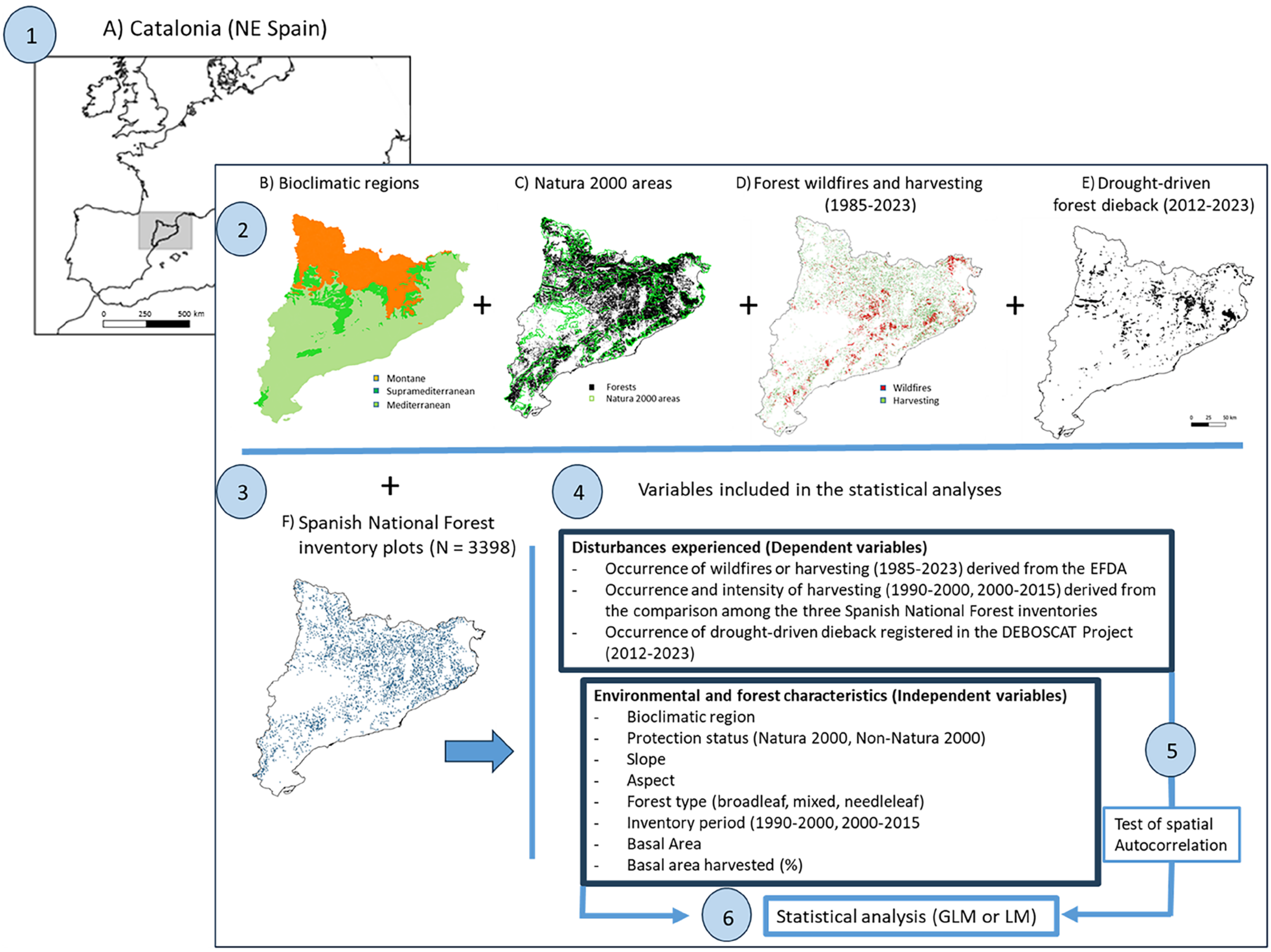

FIGURE 1 Schematic diagram of the methodological approach used: (1) The study was conducted in Catalonia (NE Spain); (2) different layers of environmental, forest protection status and disturbance events were compiled; (3) layers were intersected with the Spanish National Forest Inventory (SNFI) network of permanent plots (N=3398); and (4) values of the aforementioned variables were assigned to each SNFI plot and information on forest characteristics from the inventories conducted in 1990, 2000 and 2015 was obtained. These variables were then used as either independent or dependent variables in the subsequent analyses. (4) Spatial autocorrelation in the dependent variables was checked and corrected. (5) Finally, we applied general or generalized linear models to test the influence of protection status and other environmental and forest variables on the occurrence of disturbance events.

gradient, defining three well-contrasted bioclimatic zones (Figure 1b). The region encompasses several compelling characteristics for the purposes of this research as it is one of the most forested regions in continental Europe (ca. 43% of the total area), with nearly 40% of forest surface included in Natura 2000 sites and hosting a wide diversity of tree species (Table S1) and forest structures (Selwyn et al., 2024).

2.2 | Data sources

To investigate the role of forest protection on the occurrence of wildfires, harvesting and drought-driven forest dieback, we combined different data sources.

2.2.1 | Map of Natura 2000 sites

We used the Map of Natura 2000 areas only considering terrestrial sites where forests are present (https://mediambient.gencat.cat/). According to this map and the Spanish forest map, 41.7% of the total forest surface in the region is included in protected areas (Figure 1c), with almost 11.6% with a strict protection regime. Except for these areas of strict protection, Natura 2000 does not impose a rigorous preservation model but promotes a sustainable and compatible use of forest resources integrating conservation and forestry practices. Therefore, logging, hunting and other traditional uses are allowed, provided they do not negatively affect conservation objectives (European Commission, DG Environment, 2015). Yet, as for forestry practices, the introduction of non-native tree species is rarely admitted while best practices such as retaining biodiversity structures (e.g., snags, veteran trees), and adaptive management are encouraged (European Commission, DG Environment, 2015).

2.2.2 | Map of occurrence of forest disturbances and disturbance agents (1985–2023)

This map (Figure 1d) was obtained from the EFDA, a monitoring system based on Landsat images for mapping annual forest disturbances across continental Europe developed by Viana-Soto and Senf (2025). The atlas contains annual maps of the occurrence of forest disturbances in Europe from 1985 to 2023 and the underlying agent; either wildfire, wind/bark beetle events or harvesting (Senf & Seidl, 2021a, 2021b; Viana-Soto & Senf, 2025). Detailed information on the technical procedures to develop these maps can be found in the above-mentioned publications and, therefore, only a summary is provided here. In this atlas, a disturbance is defined as a relatively sudden change in the canopy cover of a forest. Disturbance events were identified by detecting spectral changes between successive Landsat images (30 × 30 m resolution) for the period 1985–2023 based on the distinctive shifts in the spectral signal during a disturbance event compared to the stability of undisturbed forests. Disturbed and undisturbed pixels were classified by means of a random forest model calibrated and validated using a reference dataset of manually interpreted samples from satellite data and high-resolution imagery available in Google Earth (c.a. 20,000 thousand pixels). Once disturbance events were identified, the likeliest causal agent was assigned by means of a random forest model calibrated with presence points for natural disturbances (fire, wind/bark beetle events) and pseudo-absence points representing forest harvesting (Senf & Seidl, 2021b). As wind and bark beetle effects are difficult to disentangle owing to their very similar spatial patterns and conjunct occurrence (Seidl & Rammer, 2017), these two agents are grouped in the EFDA. Opportunely, this is not a major concern for our study region as these two disturbance types are extraordinarily rare (see Section 3). Once the disturbance maps were obtained, different post-processing steps were performed to avoid spatial and temporal artefacts (Viana-Soto & Senf, 2025). Despite this, there may be potential errors in attributing the exact disturbance year due to temporal inaccuracies or gaps in the satellite data, resulting in a mean absolute error of 1.9 years. However, this imprecision does not hinder a consistent analysis of temporal trends over the extended period covered in this study (39 years).

The nature of disturbance maps derived from satellite imagery included in the EFDA may also have limitations regarding the detection of low-severity disturbances, such as drought-driven forest dieback or low-intensity harvesting (Senf & Seidl, 2021b). Conversely, this problem does not occur with forest fires, as in our study region they overwhelmingly tend to be canopy-spreading fires resulting in stand-replacing events with a distinctive spectral signal. Therefore, for the study of drought-driven dieback and harvesting impact of lower intensity, we used alternative data sources.

2.2.3 | Map of drought driven forest dieback episodes (2012–2023)

This map (Figure 1e) was obtained from the DEBOSCAT project (https://laboratoriforestal.creaf.cat/deboscat_app/) that monitors forest dieback episodes, mostly related to drought events, since 2012. Trained observers from the Rural Agents Corp perform an exhaustive yearly survey covering the whole forested area of Catalonia in September, immediately after the driest season of the year. A forest patch is considered to suffer dieback when it has a size ≥3 ha and hosts at least one dominant tree species with ≥5% canopy mortality or ≥50% canopy discoloration (browning) or leaf loss (see examples in Figure S1).

2.2.4 | Spanish National Forest Inventory (SNFI; 1990, 2000 and 2015)

To investigate the potential role of forest protection on harvesting of lower intensity than that detected by satellite imagery, and to later determine the influence of the intensity of harvesting in preventing drought-driven forest dieback, we used the data obtained in the SNFI (https://www.miteco.gob.es/). We used data only from the sub-set of plots consecutively sampled in the second (SNFI2), third (SNFI3) and fourth and last inventory (SNFI4), which in Catalonia took place in 1990, 2000 and 2015, respectively ($N=3398$; Figure 1f). To evaluate the occurrence and the intensity of harvesting in these plots, we calculated the amount of basal area removed between consecutive inventories (SFNI2-SFNI3, SFNI3-SFNI4), accounting for trees recorded as cut. From this comparison, we determined the occurrence of harvesting (no = 0, yes = 1) and the intensity of this practice (% of basal area removed). In addition, to account for the influence of forest characteristics on the successive analyses about the role of forest protection on disturbances, we obtained the following characteristics per plot: basal area ($m^2\,ha^{-1}$), forest leaf habit (broadleaf, mixed or needleleaf forest), slope (°) and aspect (N, S, E, W). With this selection, we ensured a balanced representation of typical environmental variables (bioclimatic zone, slope and aspect), forest functional types (leaf habit) and forest structure (basal area) that may influence the occurrence of disturbances, while avoiding over-parameterization of the statistical models by excluding other highly correlated variables (e.g. bioclimatic zone with altitude, basal area with tree density).

See Figure 1 for a synthesis of the procedure on combining all the above-mentioned data sources and the set of variables used in the statistical analyses.

2.3 | Data analysis

Interannual trends in forest area affected by wildfires or harvesting, as derived from the EFDA from 1985 to 2023, were analysed using the Mann-Kendall test for the entire study region and separately according to protection status (Natura 2000 vs. non-Natura 2000 forests).

To analyse the effect of protection status, together with other environmental and forest characteristics, on the occurrence of wildfires, harvesting and drought-driven forest dieback, the layer of the SFNI plots was intersected with the EFDA and the DEBOSCAT maps. Then, aside from forest characteristics and information on forest harvesting per plot derived from the comparison among different inventories, each SNFI plot was assigned a condition regarding the occurrence of a disturbance (no = 0, yes = 1) and the causal agent (wildfire, windstorm/bark beetle, harvesting) obtained from the EFDA maps (1985–2023), and the occurrence (no = 0, yes = 1) of a drought-driven dieback episode recorded in the DEBOSCAT (2012–2023) project. Then, we used this plot dataset ($N=3398$ plots) to apply generalized (GLM) or general (LM) linear models depending, respectively, on the categorical or continuous nature of the following response variables analysed:

- Occurrence (yes, no) of a wildfire, harvesting event or unspecific disturbance event (either wildfire or harvest) between 1985 and 2023 recorded in the EFDA.
- Occurrence of forest harvesting (yes, no) and harvesting intensity (% of basal area removed) in the SNFI plots by comparing the three inventories conducted in 1990 (SNFI2), 2000 (SNFI3) and 2015 (SNFI4). To explore potential temporal differences, the occurrence and intensity of harvesting were calculated for each period between inventories: 1990–2000 and 2000–2015.
- Occurrence of a drought-driven dieback episode (yes, no) from 2012 to 2023 in the SNFI plots according to the DEBOSCAT survey.

Prior to running the statistical tests, we checked for the existence of spatial autocorrelation in the dependent variables as well as the maximum distance in this effect. That critical distance (hereafter cut-off value) represented our choice for a spatial filter, which once applied to our dataset balanced the need to avoid spatial autocorrelation among sampling points (SNFI plots) and the requirement of having the larger possible dataset for the analyses (see Table S2 for the algorithm and the script implemented). Spatial autocorrelation was observed for the occurrence of unspecific disturbances (sum of harvesting and wildfires in the EFDA), wildfires and forest dieback episodes (Figure S2). Thus, cut-off values were used to ensure that this artefact did not affect the analysis of these variables. For that purpose, first, the random spatial filter was applied to the corresponding dataset retaining only those plots beyond the cut-off distance. Second, a GLM with a binomial model distribution was applied to the filtered data. This procedure was repeated 100 times and mean values and confidence intervals for the coefficients were calculated. A factor was considered to have a significant effect when the confidence interval of the estimate parameter excluded the value of 0. As no autocorrelation effect was observed for forest harvesting, neither assessed from the EFDA nor from the comparison of inventories, only standard GLMs or LMs were carried out.

In all analyses, aside from the effect of forest protection, we included other potentially explanatory co-variables accounting for the effects of environmental (bioclimatic region, aspect, slope) and forest (leaf habit, basal area) characteristics. In addition, for the models on harvesting extent derived from the SNFI data, we also included as a fixed factor the period analysed: 1990–2000 (SNFI2–SNFI3) or 2000–2015 (SNFI3–SNFI4) and plot as a random factor to account for the repeated measures nature of this analysis. In these analyses we used basal area of the initial inventory for each period analysed to evaluate how forest structure could influence the occurrence and intensity of harvesting. For the analysis of drought-driven forest dieback, we followed the same procedure but also included the percentage of basal area previously harvested as an explanatory variable to explore whether reduced tree competition after harvesting might help prevent future drought-driven forest dieback. To this end we included the percentage of basal area removed between the two latest SNFI inventories (SNFI3–SNFI4: 2000–2015) and therefore considered only forest dieback events which occurred between 2016 and 2023; the period when more episodes were detected (see Results). In all analyses, we included the second-order interaction of the main effect (protection status) with the rest of covariables and retained those interactions with a significant effect.

3 | RESULTS

Disturbances detected by satellite imagery affected 20.1% of the total forest surface from 1985 to 2023. From the overall disturbed area, less than 1‰ corresponded to windstorm/bark beetle events, 41.4% to wildfires and 58.6% to harvesting. The temporal trend (1985–2023) in the occurrence of disturbances shows a moderate but significant increasing relevance of harvesting in comparison to wildfires ($\tau = 0.26$, $p < 0.05$; Figure S3A). Yet, this pattern was mostly due to the decrease in the amount of burnt surface rather than by a real increase of the harvested surface (Figure S3B). As for forest protection, the overall percentage of forest surface disturbed in Natura 2000 areas (26.8%) was lower than in non-protected areas (32.6%). This was both due to the slightly lower proportion of forest surface affected by wildfires, but mostly by the reduced area harvested in Natura 2000 sites (11.9% and 14.9%, respectively) compared to non-protected areas (13.0% and 19.6%, respectively). The proportion of forest disturbed in protected versus non-protected areas did not show a temporal trend ($\tau = 0.11$, $p > 0.05$; Figure S4). Regarding the separate comparisons of wildfires and harvesting inside and outside protected areas, they paralleled the reported overall pattern of a moderate, but significant, increase in the prevalence of harvesting over wildfires with time ($\tau = 0.23$, $p < 0.05$ and $\tau = 0.22$, $p < 0.05$, respectively; Figure S5); in both cases due to a substantial overall decrease in burned surface (Figure S6). For the shorter period when drought-driven forest dieback episodes were monitored (2012–2023) they affected 10.6% of the total forest surface. Dieback episodes showed a high temporal variability (Figure 2), but with a progressive and outstanding increase in recent years (2021, 2022 and 2023), representing nearly two-third of the total surface affected. Interestingly, for the period when information for both drought-driven forest dieback episodes and remote sensing detected disturbances is available (2012–2023), the affected surface was very similar (119,988 ha for dieback and 122,178 ha for the sum of wildfires and harvesting).

The results of the GLMs indicated a lower occurrence of unspecific disturbances detected by satellite imagery inside Natura 2000 areas, while regarding environmental variables, montane forests and those in sites with higher slope were less prone to be disturbed (Table S3; Figure 3a). When wildfires and harvesting were considered separately, they exhibited different patterns. For wildfires, no significant influence of protection status was observed, and regarding forest and environmental characteristics, only a lower incidence in the montane region was observed (Table S4; Figure 3b). The model for harvesting occurrence revealed that it was significantly lower in Natura 2000 forests, and irrespective of protection, it increased with basal area and was lower in steeper and north-facing sites (Table S5; Figure 3c).

The assessment of harvesting extent through the comparison of consecutive forest inventories revealed a trend similar to that recorded by satellite imagery regarding forest protection status. However, due to its ability to detect less intense interventions, the former method produced higher overall figures of harvesting in both forest conditions. The occurrence of harvesting was lower in protected areas than in non-protected areas (19.2% vs. 26.1%, respectively), while regarding environmental and forest characteristics it was lower in forests in the montane region or with higher slope, whereas it was positively influenced by basal area and more likely in mixed forests (Table S6A; Figure 4a). The significance of the interaction between protection status and basal area revealed that the increase in harvesting with increasing basal area was higher in non-protected sites (Figure 5). Concerning temporal patterns, harvesting occurrence was slightly lower during the second period analysed: 24.9% of forest plots from 1990 to 2000 and 21.1% from 2000 to 2015, particularly

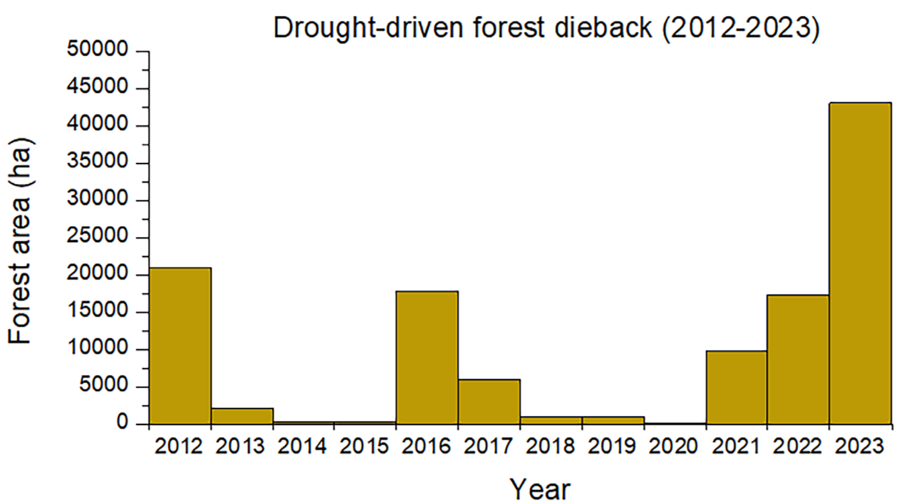

FIGURE 2 Total surface of drought-driven forest dieback episodes yearly registered in Catalonia by the DEBOSCAT project from 2012 to 2023.

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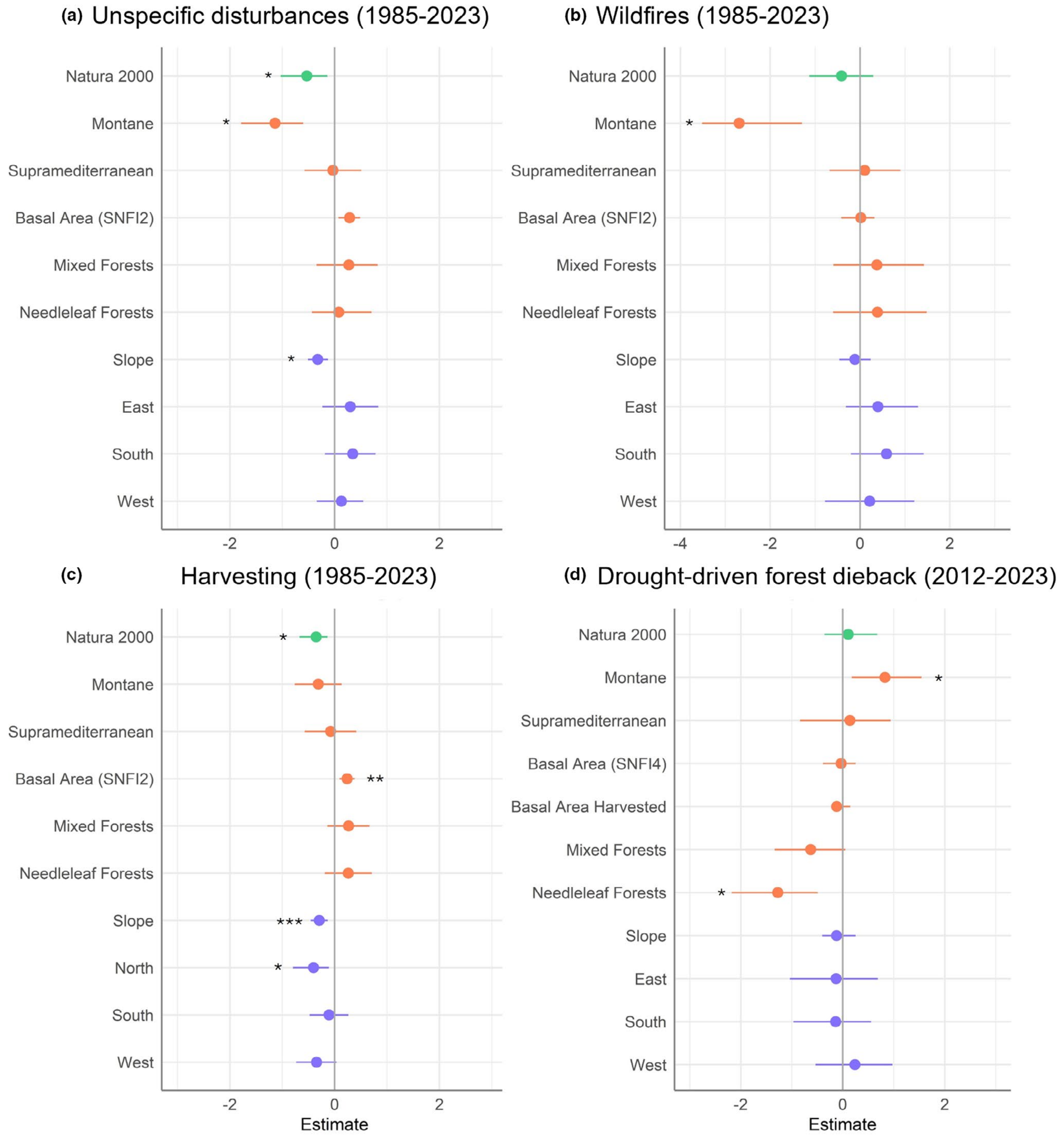

FIGURE 3 Parameter estimates for the generalized linear models of the effects of forest protection (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2 ha^{-1}$), % basal area removed (for forest dieback), forest leaf habit (broadleaf, needleleaf, mixed), slope and aspect (north, east, south, west) on the occurrence of the following: (a) unspecific disturbances (wildfires or harvesting; 1985–2023), (b) wildfires (1985–2023), (c) harvesting (1985–2023) and (d) drought-driven forest dieback (2012–2023). In panels (a), (b) and (d), asterisks indicate the significance of effects with the confidence interval of the parameter estimate excluding 0, obtained from a generalized linear model (GLM) repeated 100 times, applied to the filtered data (excluding autocorrelation); in panel (c), asterisks indicate significant effects ('***' <0.001, '**' <0.01, '*' <0.05) for a GLM on the unfiltered dataset due to the lack of autocorrelation.

in protected areas (interaction between protection status and period in Table S6A and Figure S7). Similar to harvesting occurrence, harvesting intensity (percentage of basal area removed) was also lower in protected than in non-protected areas (27.6% ± 0.7% vs. 32.2% ± 1.0%, respectively) and it was lower in needleleaf forests and under steeper slopes, irrespective of protection status

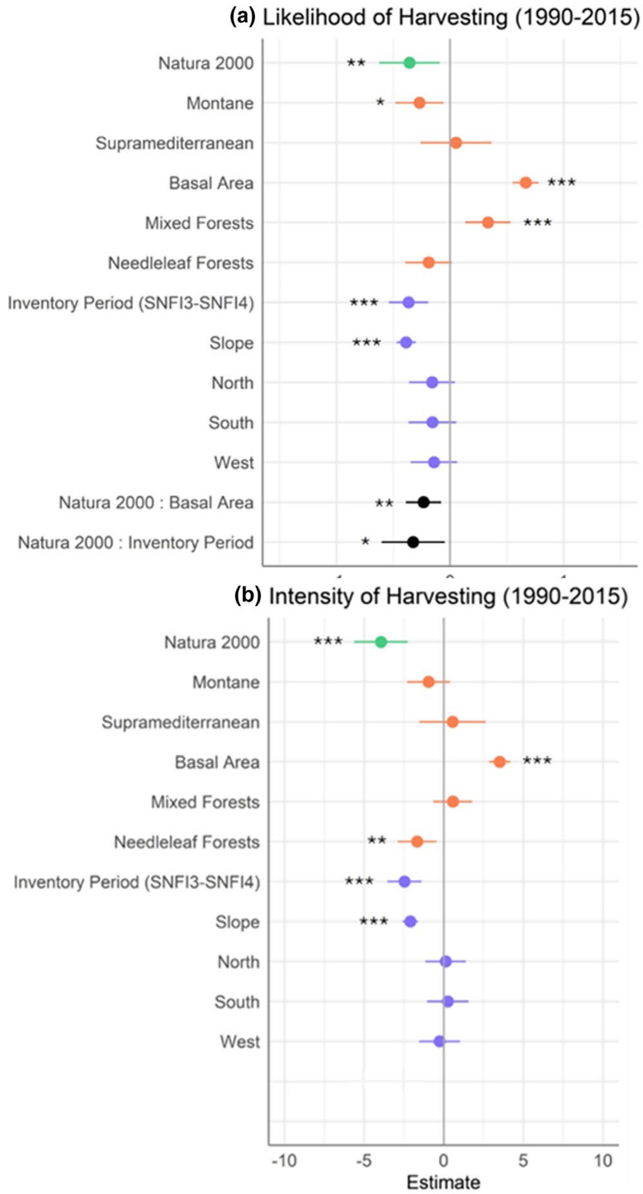

FIGURE 4 Parameter estimates for the general linear models of the effects of forest protection (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2 ha^{-1}$), forest leaf habit (broadleaf, needleleaf, mixed), inventory period (SNFI2–SNFI3, SNFI3–SNFI4), slope and aspect (north, east, south, west) on: (a) the occurrence of harvesting (0, 1) and (b) intensity of harvesting (% of basal area removed). Asterisks denote significant effects: '***' <0.001, '**' <0.01, '*' <0.05.

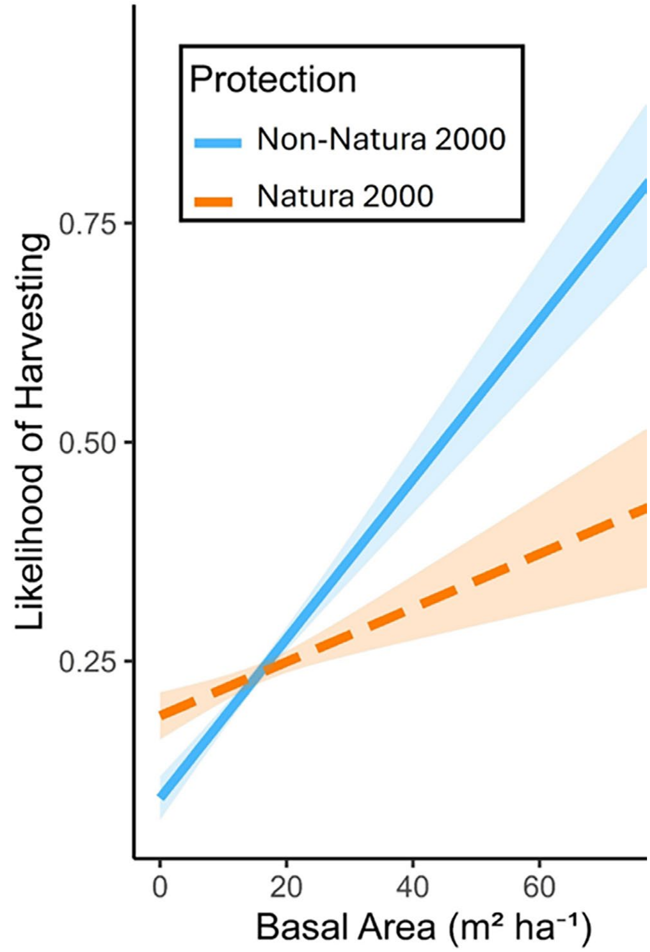

FIGURE 5 Results of the interaction between the effect of forest protection (Natura 2000, Non-Natura 2000) and forest basal area ($m^2 ha^{-1}$) on the likelihood of harvesting (0, 1) according to the data from the comparison of Spanish National Forest Inventories.

(Table S6B). Also, harvesting intensity was slightly but significantly lower during the second period analysed (Figure 4b): 31.8% ± 0.7% of basal area removed from 1990 to 2000 and 28.7% ± 0.8% from 2000 to 2015.

Conversely to the proportion of forest surface affected by wildfires and harvesting, which was higher in non-protected areas, the proportion of forest area affected by die-off events was similar inside and outside Natura 2000 sites (11.4% vs. 9.9%, respectively). This was supported by the GLM analysis which revealed no influence of forest protection in the occurrence of drought-driven dieback (Table S7; Figure 3d). As for environmental and forest characteristics, dieback episodes were more common in montane areas and less abundant in needleleaf forests, with no influence of forest basal area or the intensity of previous harvesting in the plot (Table S7; Figure 3d).

4 | DISCUSSION

From 1985 to 2023, the proportion of disturbed forests by harvesting or wildfires in Catalonia (20.1%) was noticeably similar to the figure reported for entire Europe during the same period (20.9% in Viana-Soto and Senf (2025)). Yet, dramatic differences emerge when considering the nature of the agents. In Europe, 14% of the disturbed forest area corresponded to natural disturbances (7% wildfires and 7% windstorms/bark-beetle attacks; Senf & Seidl, 2021b); whereas in our Mediterranean area, wildfires accounted for an overwhelming proportion (41.1%). These results highlight that disturbance agents are operating very differently across Europe,

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with a predominant role of natural disturbances towards southern regions and of harvesting towards northern latitudes (Seidl & Senf, 2024; Senf & Seidl, 2021a). This pattern points to the urgent need for tailoring regional level policies across Europe to address local differences in the ecology and socio-economic impact of forest disturbances (Cantarello et al., 2024). Interestingly, we observed a progressive reduction in the pre-eminence of wildfires in the last decades, as has been reported in general for south-western Europe (Bountzouklis et al., 2022) in contrast with the increasing trend in temperate and boreal Europe (Grünig et al., 2023). Even considering the high year-to-year stochasticity that governs wildfire occurrence, we should account for the likely positive effects of certain social and governance innovations in wildfire prevention and suppression that were implemented in the study region after the dramatic wildfire events of the 1990s (Fernández-Blanco et al., 2022). Conversely, the almost flat trend observed in forest harvesting suggests that further efforts are needed to promote forest management to boost bioeconomy. Besides the incidence of wildfires and forest harvesting, drought-driven forest dieback episodes are gaining momentum in recent years, as has been consistently observed across Europe (Senf et al., 2020). While such events undoubtedly occurred before systematic monitoring began, our results show they affected an area comparable to that of wildfires and harvesting combined over the same period. This highlights the alarming escalation of drought dieback and the urgent need for adaptive measures to address this disturbance (Ripple et al., 2024), which remains less central to forest policies compared to wildfire prevention.

The extent of harvesting and its intensity was significantly influenced by forest characteristics and topography. In line with sustainable forestry principles, more capitalized forests—characterized by higher basal area—were more intensively managed, whereas forests on steeper slopes, where mechanization is limited and soil erosion risk is higher, were less managed (Puettmann et al., 2009). Aside from these drivers, both satellite images and the comparison of the forest inventories indicated a lower likelihood and intensity of harvesting in Natura 2000 areas compared to non-protected sites. Refraining from harvesting in Natura 2000 areas could be a deliberate decision, but it may also indicate the existence of unintended bureaucratic and regulatory barriers that constrain forest management initiatives, as sometimes reported in protected sites (Schneider et al., 2020). Interestingly, the fact that differences between non-protected and protected sites were greater for the occurrence of harvesting than for its intensity (36% and 16% higher in non-Natura 2000 sites, respectively) suggests that the initial decision of whether to harvest plays a larger role in protected areas, whereas once harvesting is decided, its intensity is much more similar to that in non-protected sites. Clearly, if the maintenance of local communities' well-being is considered a cornerstone of the Natura 2000 network, potential barriers halting forestry activities should be addressed (Paletto et al., 2019).

We observed some overall effects of the climatic bioregion on the occurrence of wildfires (lower in montane areas); yet the occurrence of these disturbances was unrelated to the protection status of forests. Forests in montane areas in Catalonia are certainly less prone to wildfires owing to their more favourable environmental characteristics (lower temperatures and higher precipitation), but they may also benefit from being in more remote areas with lower population density and, therefore, fewer human-driven fire risks. As for the role of protection, our findings contrast with studies that reported a higher (Kirkland et al., 2023; Rodrigues et al., 2023) or a lower (Kirkland et al., 2024) incidence of wildfires in protected sites. This disagreement may stem from differences in the length of the time series or in the type of landscape considered, for a highly variable phenomenon, such as wildfires. For example, while our analysis spans over 39 years, the study by Rodrigues et al. (2023) covers just 1 year. In turn, Kirkland et al. (2023) analysed a smaller and very different ecological system (i.e. wildfires initiated in croplands spreading into protected areas richer in wildfire-sensitive vegetation). In any case, all studies assessing wildfire incidence relative to forest protection status highlight the importance of the potential differences in forest structure and forest cover between protected and non-protected sites as the primary cause (de Dios et al., 2025). Within this framework, even considering that our protected and non-protected forests significantly differed in harvesting extent, we hypothesize these differences were not large enough to drastically modify forest structure and therefore the likelihood of the occurrence of wildfires. For example, in the two most fire-prone bioclimatic regions (Mediterranean and Supra-Mediterranean), the difference in average basal area between protected ($19.3 \pm 0.36\,m^2\,ha^{-1}$) and non-protected forests ($16.8 \pm 0.28\,m^2\,ha^{-1}$) is only 16%, probably too small to drastically change wildfire risk related to forest structure (Palmero-Iniesta et al., 2017). Additionally, regarding the role of forest connectivity in wildfire spread, we must stress that forests across our study region exhibit a high and similar level of connectivity, regardless of their protection status (Selwyn et al., 2024).

In contrast to the lower incidence of wildfires, drought-driven forest dieback episodes were more abundant in montane forests; however, as with wildfires, we did not observe significant differences related to forest protection. The higher susceptibility of montane forests to drought may be explained by the fact that they host several tree species growing near their southwestern distribution limit in Europe (*Abies alba*, *Fagus sylvatica* and *Pinus sylvestris* in Table S1), which are particularly sensitive to drought (Leuschner, 2020; Martínez-Vilalta & Piñol, 2002). Regarding leaf habit, although our results indicate that forest dieback was lower in needleleaf forests than in broadleaved ones, the long-term impact of drought is likely to be higher in the former, as conifers are ultimately more prone to collapse whereas most broadleaf species are able to resprout and reconstruct their canopy after drought-driven defoliation (Hartmann et al., 2022). Once again protection status did not influence the extent of drought-driven forest dieback. The reduction of tree density by harvesting has been extensively suggested as a mechanism to reduce competition and increase the resistance of trees to drought (Willig et al., 2025). Yet, this effect can be extremely contextual, and it can largely depend, among others, on thinning intensity (Moreau et al., 2022). Thus, similar to wildfires, we hypothesize that although protected and non-protected forests significantly differ in the intensity of harvesting (a 16% higher mean basal area removal outside Natura 2000 areas), this difference may be ecologically irrelevant when facing extreme drought events, given the generally overstocked condition of forests in the region (Roces-Díaz et al., 2021). This rationale is also supported by evidence from previous local studies that reported limited and transient effects of traditional thinning intensities to cope with drought (Cotillas et al., 2009) or no effects of stand basal area on the extent of drought-driven dieback in *P. sylvestris* forests (Chowdhury et al., 2024).

Protected areas—typically static in space—are facing growing challenges as climate change conditions may force species to migrate or it intensifies the occurrence of disturbance events (Alagador et al., 2014). In this framework, a debate exists on whether redesigning protected networks or adding new protection areas may be more cost-effective (Hannah et al., 2007) as well as, particularly for forests, whether increasing active management may contribute to decreasing the vulnerability to disturbances (de Dios et al., 2025; De Koning et al., 2014). Ultimately, our results report a lower harvesting extent in Natura 2000 than in non-Natura 2000 forests, but they do not support a direct effect of protection on the occurrence of wildfires or drought-driven forest dieback. We argue that, although the reported differences in harvesting associated with protection are statistically significant, they are likely ecologically irrelevant in influencing the occurrence of other disturbances, at least when compared to the demonstrated effects of other factors investigated (i.e. bioclimatic region, topography, species identity), as well as others not covered in this study (e.g. ignition sources for wildfires, soil water-holding capacity for drought).

After a long period of forest management abandonment in Mediterranean-type forests, it has been suggested the overall need to increase forest harvesting to promote climate change adaptation (Roces-Díaz et al., 2021). Yet, our results indicate that traditional thinning intensities applied up to date may not suffice to meet this objective. Therefore, a key question is whether efforts to halt climate change-driven disturbances should focus on expanding the managed area or increasing management intensity in specific locations. In the latter case, determining the extent to which intensity could be increased will be a challenge, especially for Natura 2000 areas, given the need to protect biodiversity, highly menaced even in these protected sites (EEA Report, 2020). Moreover, this debate should also consider the EU Biodiversity Strategy requirement that 10% of each country's territory be strictly protected, a target for which Natura 2000 sites likely represent the main potential. This implies that the scope for intensifying management within these areas may be even more limited than commonly assumed, although the consequences for disturbance seem to be probably negligible. Outside strictly protected areas, silvicultural approaches such as close-to-nature forest management (Larsen et al., 2022) or smart silviculture for more effective large-scale wildfire risk reduction (Fernandes, 2013) should be promoted and combined. In addition, in both protected and non-protected sites, additional measures may be required to increase forest resilience, such as promoting tree species diversity, particularly the presence of autochthonous drought-tolerant species (Selwyn et al., 2024).

AUTHOR CONTRIBUTIONS

Conceptualization: Josep Maria Espelta; data provision: Josep Maria Espelta, Alba Viana-Soto, Miquel de Caceres and Mireia Banqué; formal analysis: Josep Maria Espelta and Roberto Molowny-Horas; writing—original draft: Josep Maria Espelta; writing—review and editing: Josep Maria Espelta, Alba Viana-Soto, Miquel de Caceres, Miriam Selwyn, Mireia Banqué, Lluís Brotons, Francisco Lloret, Jordi Martínez-Vilalta, Miriam Piqué and Cornelius Senf.

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CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

DATA AVAILABILITY STATEMENT

Data are available from the Dryad Digital Repository: https://doi.org/10.5061/dryad.b2rbnzsv7 (Espelta et al., 2026).

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SUPPORTING INFORMATION

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

Table S1. Identity of tree species recorded in the National Forest Inventory plots in Catalonia across different bioclimatic regions (Mediterranean, Supra-Mediterranean and montane) and according to protection status (Non-Natura 2000 vs. Natura 2000 areas).

Table S2. Method used for the analysis of spatial autocorrelation and determination of cut-off values (A) and example of R-script for the analysis of spatial autocorrelation, selection of cut-off value and GLM analysis for the occurrence of wildfires in protected and non-protected areas (B).

Table S3. Results of the generalized linear model for the effects of forest protection status (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2\,ha^{-1}$), forest leaf habit (broadleaf, needle leaf, mixed), slope and aspect (north, east, south, west) on the occurrence of unspecific disturbances (wildfires or harvesting) detected by remote sensing-derived images from the European Forest Disturbance Atlas from 1985 to 2023.

Table S4. Results of the generalized linear model for the effects of forest protection status (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2\,ha^{-1}$), forest leaf habit (broadleaf, needle leaf, mixed), slope and aspect (north, east, south, west) on the occurrence of wildfires detected by remote sensing-derived images from the European Forest Disturbance Atlas from 1985 to 2023.

Table S5. Results of the generalized linear model for the effects of forest protection status (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2\,ha^{-1}$), forest leaf habit (broadleaf, needle leaf, mixed), slope and aspect (north, east, south, west) on the occurrence of harvesting detected by remote sensing-derived images from the European Forest Disturbance Atlas from 1985 to 2023.

Table S6. Results of the general linear model for the effects of forest protection status (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2\,ha^{-1}$), period inventoried (1990–2000, 2000–2015) forest leaf habit (broadleaf, needle leaf, mixed), slope and aspect (north, east, south, west) on (A) harvesting occurrence and (B) harvesting intensity (% basal area removed) obtained by the comparison of the three national forest inventories conducted in the study area (SNFI2, SNFI3, SNFI4) in 1990, 2000 and 2015.

Table S7. Results of the generalized linear model for the effects of forest protection status (Natura 2000, Non-Natura 2000), bioclimatic region (Mediterranean, Supra-Mediterranean, montane), basal area ($m^2\,ha^{-1}$), basal area harvested, forest leaf habit (broadleaf, needle leaf, mixed), slope and aspect (north, east, south, west) on the likelihood of the occurrence of drought-driven forest dieback from 2016 to 2023.

Figure S1. Images of different drought-driven forest dieback episodes recorded in the DEBOSCAT project.

Figure S2. Mean join-count p-values as a function of cut-off distance for: (A) unspecific disturbances (wildfires + harvesting), (B) wildfires, (C) harvesting and (D) drought-driven forest dieback.

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Figure S3. Yearly proportion of burned (red) and harvested (green) surface (A) and total disturbed surface (B) from 1985 to 2023, according to remote sensing-derived images-from the European Forest Disturbance Atlas.

Figure S4. Yearly proportion (A) and total surface (B) of disturbed surface in Natura 2000 (green) and Non-Natura 2000 (orange) sites from 1985 to 2023, according to the analysis of remote sensing-derived images from the European Forest Disturbance Atlas.

Figure S5. Yearly proportion of disturbed surface by wildfires (red) or harvesting (green) in Natura 2000 (A) and non-Natura 2000 (B) forests from 1985 to 2023, according to remote sensing-derived images from the European Forest Disturbance Atlas.

Figure S6. Yearly total disturbed surface by wildfires (red) or harvesting (green) in Natura 2000 (A) and Non-Natura 2000 (B) forests from 1985 to 2023, according to remote sensing-derived images from the European Forest Disturbance Atlas.

Figure S7. Results of the interaction between the effect of forest protection status (Natura 2000, Non-Natura 2000) and the inventory period (SNF2–SNFI3, SNFI3–SNFI4) on the occurrence of harvesting (0, 1) according to the data from the comparison of the three successive Spanish National Forest Inventories (SNFI2, SNFI3, SNFI4).

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