



Effects of fuel types and fire severity on atmospheric pollutant emissions in an extreme wind-driven wildfire

Albert Alvarez^{1,2} · Judit Lecina-Diaz³ ·
Miquel De Cáceres² · Jordi Vayreda² ·
Javier Retana^{1,2}

Received: 13 December 2024 / Accepted: 15 May 2025
© The Author(s) 2026

Abstract In the Mediterranean region, wind-driven crown fires are becoming more frequent, leading to increased atmospheric pollutant emissions. This study explored how the distribution of pre-fire fuels across the crown, shrub and litter layers varies among different fuel types, and how these variations were linked with fuel consumption and fire severity for each layer to quantify and compare atmospheric pollutant emissions (CO₂, CO, CH₄ and PM_{2.5}) in pine (*Pinus halepensis*) and oak (*Quercus suber*) forests. Our analysis was carried out in the Jonquera wildfire in Northeast Spain, which burned 10,264 ha. Pre-fire fuel loading among fuel types in pine and oak forests showed different vertical distributions despite, having similar fire-type patterns. Pine

forests had a higher percentage of crown and shrub fuel loading for all fuel. In contrast, oak forests had more litter than pine forest. Fuel types characterized by large trees and low densities had the lowest fire severity in both forest types. Pine forests were more resistant to the effects of surface fires than oak forests due to their taller trees, which allowed them to withstand high-intensity surface fires with less tree damage. However, these fires have resulted in higher surface fuel consumption in pine forests. Fuel types with more vertical and horizontal continuity experienced higher fire severity and fuel consumption (72–85% of high severity). Fire severity rather than species or fuel type was the primary factor influencing pollutant emissions. Emissions of CO₂ and CH₄ were higher in pine than in oak forests especially at lower severities, while at intermediate and higher severity oak forests emitted more CO and PM_{2.5}. Although remote sensing technologies are useful for fuel loading and wildfire severity assessments, field data are essential for accurately quantifying fuel consumption across fuel types and layers.

Project funding: This work has been funded by the project GREEN-RISK “Evaluation of past changes in ecosystem services and biodiversity in forests and restoration priorities under global change impacts” (PID2020-119933RB-C21/AEI/10.13039/501100011033), funded by the Spanish Ministry of Science and Innovation.

The online version is available at <https://link.springer.com/>.

Corresponding editor: Shuxuan Li

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s11676-026-01988-2>.

✉ Javier Retana
Javier.Retana@uab.es

¹ Universitat Autònoma de Barcelona,
08193 Bellaterra (Cerdanyola del Vallès), Catalonia, Spain

² CREAM, 08193 Bellaterra (Cerdanyola del Vallès), Catalonia,
Spain

³ Ecosystem Dynamics and Forest Management Group, TUM
School of Life Sciences, Technical University of Munich,
Hans-Carl-Von-Carlowitz-Platz 2, 85354 Freising, Germany

Keywords Fuel loading · Fuel consumption · Forest structure · *Pinus halepensis* · *Quercus suber* · Pyrogenic emissions

Introduction

Biomass burning by wildfires is a global phenomenon emitting significant quantities of pollutants such as atmospheric gases, aerosols and particulates into the atmosphere having a serious impact on global warming and climate (Bowman et al. 2009; Keywood et al. 2013; Knorr et al. 2016). Global average fire emissions were estimated to be 2.2 Pg a⁻¹ in the period 1997–2020 (van der Werf et al. 2017; van Bees et al. 2022). In addition, climate change directly affects wildfires

by increasing drought conditions that affect fire ignition, propagation, frequency and distribution of extremely high-intensity wildfires, and indirectly through its effects on vegetation and fuels (Ruffault et al. 2018; San-Miguel-Ayanz et al. 2022). Wildfire events release carbon mainly in the form of carbon dioxide (CO₂), carbon monoxide (CO) and methane (CH₄), constituting nearly 95% of wildfire carbon emissions. CO₂ and CH₄ have the greatest greenhouse influence, while CO is an active trace gas contributing to the secondary formation of ozone (O₃) (Pallozzi et al. 2018; Fortems-Cheiney et al. 2024). Wildfires also release particulate matter and gaseous phase pollutants with different toxicity levels depending on the location, particle size and composition, both of which can cause health issues (Naeher et al. 2007; Kocbach Bølling et al. 2009; Salana et al. 2024).

In recent years, large wildfires have repeatedly affected European countries, particularly the southern Mediterranean, which accounted for the majority of the annual burned area from 1980 to 2017 (de Rigo et al. 2017; San-Miguel-Ayanz et al. 2022) in extreme years like 2022 in Portugal, France and Spain (Rodrigues et al. 2023). At the country level, wildfire emissions have only been assessed in Portugal (Carvalho et al. 2007; Miranda et al. 2009; Rosa et al. 2011), Italy (Bacciu et al. 2015), Greece (Lazaridis et al. 2008) and Turkey (Çinar et al. 2024), usually analyzing specific fires or periods and adapting the model proposed by Seiler and Crutzen (1980). Moreover, previous studies are usually restricted to prescribed fires, which do not represent the range of severity of extreme wildfires (Urbanski 2013; Balde et al. 2023). In addition, uncertainties associated with the variables used for the estimation procedures can influence the amount of pollutants emitted by fires, which may be especially important in heterogeneous Mediterranean forests with different forest structures. The lack of data on fuel consumption and fire severity in different layers at the local scale from wildfires forces the use of general assumptions that can cause inaccuracies (Fernandes et al. 2022; Balde et al. 2023). For instance, the emission factors used in these estimates may not be accurate at the species level or for individual components (needles/leaves, branches, shrubs, litter) (Larkin et al. 2014; Poupkou et al. 2014; Pallozzi et al. 2018). In the Mediterranean area, wind-driven wildfires are common (i.e., those that follow the speed and direction of strong winds, where changes in the landscape have little influence on fire direction and behavior) (Costa et al. 2011; Ruffault et al. 2017; Duane and Brotons 2018; Lahaye et al. 2018). However, little is known about how these wildfires spread under extreme weather conditions, if fire behavior is affected by changes in forest structure and composition, and how this affects fuel consumption, fire severity and, ultimately, pollutant emissions (Fernández-García et al. 2022; Beltrán-Marcos et al. 2024). Data on fire occurrences and forest stand characteristics are still scarce in southern

Europe and are essential for understanding the impact of different fuel and cover types on fire emission estimates, especially under different burning conditions (French et al. 2011; Nunes et al. 2019; Balde et al. 2023).

Fire severity can be defined as the degree of fire-induced environmental changes in vegetation immediately after a fire (Keeley 2009; Lecina-Diaz et al. 2014). A wildfire can produce different levels of damage depending on the burned layers and their flammability (Xanthopoulos et al. 2011; Chiriaco et al. 2013). Forest structures (i.e., defined by the characteristics of the spatial distribution of biomass in forested vegetation, both vertically and horizontally) (Fernandes 2009; Kane et al. 2010; Hoff et al. 2019; Ritter et al. 2023) are usually grouped in fuel types, defined as “identifiable associations of fuel elements with distinctive species, form, size arrangement and continuity that exhibit characteristic fire behavior under defined burning conditions” (Merrill and Alexander 1987; CIFFC 2023). Although most studies recognize the importance of data on combustion factors (i.e., the proportions of fuel consumed for each severity class, Wiedinmyer et al. (2006)), there is a lack of field data for different forest and fuel types with different vertical and horizontal distributions (Jiménez et al. 2013; Lydersen et al. 2014; Balde et al. 2023; Ritter et al. 2023). Understanding how fuel consumption determines fire severity among different fuel types is also required to improve fire emission estimates (Campbell et al. 2007; De Santis et al. 2010; Surawski et al. 2016; Price et al. 2022; Balde et al. 2023).

This study combines detailed post-fire field sampling and national forest inventory data from an extreme wind-driven wildfire in a Mediterranean area to analyze and estimate atmospheric pollutant emissions in different fuel types and forest types. To this end, the specific objectives are: (1) To compare the distribution of fuel loading before the fire for different fuel types in two forest types (pine- and oak-dominated forests); (2) To evaluate the differences in the percentage of fuel consumption that determine fire severity in these different fuel and forest types; and (3) To estimate the amount of pollutant emissions (CO₂, CO, CH₄ and PM_{2.5}) across fire severities, fuel types and forest types.

Materials and methods

Study area and pre-fire vegetation

The study area was the Jonquera wildfire (Northeastern Spain, Girona province, 42°24′59″N–2°52′29″E), which occurred in the summer of 2012 and burned 10,264 ha. The region’s climate is humid sub-Mediterranean, with annual rainfall varying from 650 mm at lower altitudes to 1000 mm at higher altitudes, with the lowest precipitation occurring in July. The mean annual temperature is approximately

15–16 °C, peaking in July (23–24 °C) and dropping to its lowest values in January (8–9 °C). The study area ranges from 35 to 780 m a.s.l., with slopes between 0 and 72%. Historically important for cork production (*Quercus suber*) in the twentieth century, the area now has fewer shrublands, more forests and increased artificial surfaces due to the decline of the cork industry and population density (Badia et al. 2019; Salis et al. 2019), resulting in higher fuel load and greater landscape continuity.

Sixty percent of the area burned in 2012 had previously burned in a large wildfire in 1986 (15,000 ha), with a smaller area burning again in 2006 (420 ha). The burned area in 2012 was characterized by continuous forests of cork oak (*Quercus suber*) and Aleppo pine (*Pinus halepensis*) (hereafter, oak and pine forests/plots, respectively) (MCSC 2010). Oak forests were located in the northern part of the wildfire area (Fig. 1) on acidic soils with forest structures shaped by over 26 years without fires, resulting in high vertical and horizontal continuity (Schaffhauser et al. 2011). The understory of oak forests was characterized in the field (see “Field plot data” section) and included a high density of *Erica arborea* in combination with other species such as *Cistus monspeliensis*, *Cistus salvifolius*, *Ulex parviflorus*, *Genista scorpius*, *Quercus suber* or *Arbutus unedo*. Pine forests were located in the southern part of the burned area (Fig. 1) on sedimentary rocks with diverse forest structures ranging from medium (1300–3000 ind. ha⁻¹) to low density (< 1300 ind. ha⁻¹). The pine forest understory was dominated by *Quercus coccifera*, *Pistacea lentiscus* and *Rosmarinus officinalis*, with lower proportions of *Phyllirea latifolia*, *Quercus ilex*, *Rhamnus alaternus* and *Viburnum tinus*. In both forest types, the main herbaceous species was *Brachypodium retusum*. First, we studied in each area the species composition using plots from the National Forest Inventory before the fire inside the perimeter of the fire. Afterwards, during the field surveys, we could confirm data from those plots inside the fire studying the remains in green and scorch areas and learning to identify the species when they were almost consumed, also considering if it could be visible like resprouters, or not visible like seeders species.

Fire description and weather conditions during the fire

The Jonquera wildfire started at 12.54 pm on July 22 2012, and was controlled by 7.48 am on July 27. The wildfire started and burned mainly as a wind-driven wildfire characterized by continuous, strong northwest-oriented winds (known locally as “Tramuntana”), with speeds of 10–50 km h⁻¹ and gusts up to 70 km h⁻¹. It was a large wildfire at the wildland urban interface (WUI), affecting thousands of people and property. Fire behavior was extreme, including crown fires and prolific spotting activity, with intermediate distances of 200–400 m and a maximum of

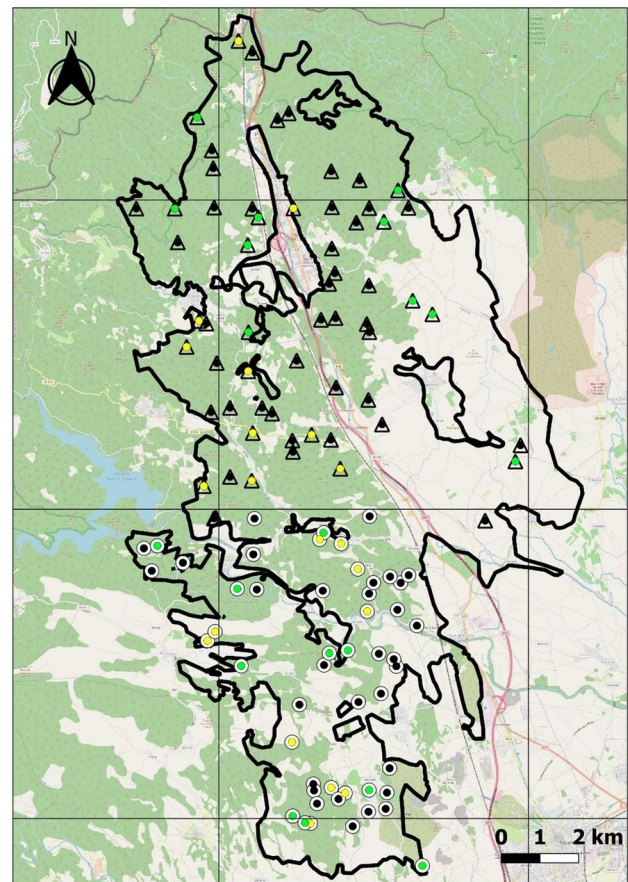


Fig. 1 Field plots distributed in the fire area (black perimeter). The triangles represent *Quercus suber* plots, and the circles *Pinus halepensis* plots. The inner dot colors indicate the fire severity: low (green), moderate (yellow) and high (black). The base map was sourced from © OpenStreetMap contributors 2024. Distributed under the Open Data Commons Open Database License (ODbL) v1.0. The fire perimeter was provided by The Generalitat de Catalunya firefighters corps

1 km. The fire had high fire spread rates of 1.8 km h⁻¹ and explosive runs reaching 7.7 km h⁻¹ (DARP 2012), making it an extreme wildfire event (EWE) (Tedim et al. 2018). On the second day, as the fire transitioned to a lower-intensity topographic wildfire, its behavior moderated. When the fire started, the temperature was 24 °C (“el Pertús”) and relative humidity was 24% due to the dry northern winds (DARP 2012). Relative humidity fluctuated between 19 and 36% during the main spread period and only recovered in the second afternoon (> 60%) (data from the meteorological stations of Espolla and Cabanes, at 13 and 20 km from the fire’s origin). The Standardized Precipitation-Evapotranspiration Index (SPEI) (Vicente-Serrano et al. 2010) indicated a moderate short-term drought with only mild medium and long-term drought. Fuel moisture was typical for the station and area, with values of 69% for *Rosmarinus officinalis* and 66% for *Cistus monspeliensis*, while *P. halepensis* moisture

content was 105%, slightly below the seasonal average (station of Port de la Selva, located 30 km from the burned area for the 18th July 2012) (Gabriel et al. 2021).

Field plot data, fire severity and fuel consumption

One and a half months after the fire, with the assistance of two scheduled helicopter flights by the fire department, we obtained videos and hundreds of photos that allowed us to understand the fire's overall spread and capture detailed information on the fire severity distribution. Field sampling began two months after the wildfire and lasted approximately six months. We selected plots that represented a range of fire severities (understood as the immediate effect of fire from the forest's biotic components) (Keeley 2009) for *Pinus halepensis* and *Quercus suber* forests. The number of plots was proportional to each forest type and fire severity within the study area, with a minimum of ten plots per fire severity class and forest type. We obtained data from 109 circular plots, each 20 m in diameter, 59 oak forests and 50 pine forests (Fig. 1). At the tree level, fire severity was measured using an adapted version of the Jain and Graham (2007) method using the proportion of residual crown severity (green, scorched, charred) (Alvarez et al. 2013) (Appendix A-fire severity description). For each tree, we also identified the species and measured the diameter at breast height (DBH) and total height. At the plot level, following Alvarez et al. (2013), we classified the plots into three fire severity classes based on the percentage of trees in each severity class: green plots ($\geq 50\%$ green trees), scorched plots ($\geq 50\%$ scorched trees and $\leq 25\%$ green trees), and charred plots ($\sim 100\%$ charred trees).

We determined the fire behavior and fuel consumption in each plot. We first assessed the fire type (surface fire, passive crown fire or active crown fire) in each plot (see Appendix A, fire type descriptions). For trees, we visually estimated the proportion of leaves and fine branches (< 6 mm) remaining on each tree after the fire and obtained the percentage consumed in each tree. We calculated the combustion factor for leaves and branches at the plot level by averaging the values for all trees in the plot. For shrubs, we estimated two consumption categories at the plot level: the percentage of fine fuel consumed (i.e., leaves and fine branches < 0.6 cm) and the percentage of coarse fuel (> 0.6 cm) consumed.

Finally, we visually assessed the percentage of litter consumed in the entire plot, minimizing inherent uncertainties of the method with previous training (see Section “Uncertainties and limitations in emission estimates”).

Fuel type classification

We classified each plot into one of the 20 forest structures in Alvarez et al. (2012a), based on the number of layers, the percentage of tree types (small, medium and large) and tree density. Subsequently, we grouped these forest structures into four fuel types according to their forest structure characteristics and fire types (Table S1). Fuel type 1 (FT1) is characterized by open forest structures with variable proportions of large trees and very low tree density (< 500 ind. ha^{-1}). These forests have very low horizontal tree continuity, which is associated with surface fires or passive crown fires. Fuel type 2 (FT2), characterized by large trees and low tree density, can mainly spread surface fires because of its lack of vertical continuity, but if the fire reaches the plot as a crown fire, FT2 can also spread crown fires. Fuel type 3 (FT3) has a forest structure similar to that of FT2 but with a higher tree density, a lower percentage of large trees and a second tree layer, which leads to a higher risk of crown fire compared to FT2. Finally, fuel type 4 (FT4) consists of forest structures with less than 60% large trees, high vertical continuity and high tree density (> 1300 ind. ha^{-1}) with high horizontal continuity. This fuel type is most likely to generate crown fires (Fig. 2). This fuel type classification has been successfully applied in studies related to potential fire behavior (Alvarez et al. 2012a), assessing changes in fire type risk using inventory plots and ecosystem services loss (Alvarez et al. 2012b; Lecina-Díaz et al. 2021). See Appendix B for further details on this methodology.

Area burned and pre-fire fuel loading

The burned area was determined from the fire perimeter, which was measured by firefighter crews in the field immediately after the wildfire using GPS. During fieldwork, we established plots of two sizes, ranging from 316 to 716 m^2 , depending on the heterogeneity of the area and density. To estimate fine fuel loading before the fire in these plots, we distinguished three components: trees,

Fig. 2 Schematic view of the four fuel types (FT1 to FT4) used in this study (based on Fig. 2 of Alvarez et al. 2012b)



shrubs and litter (Fig. S2). For trees, we calculated the fine fuel loading (leaves and fine branches < 0.6 cm) using allometric equations provided in the “medfate” R package (De Cáceres et al. 2015), using data from the Ecological Forestry Inventory of Catalunya (IEFC) (Gracia 2001). For shrublands, we first selected the closest National Forest Inventory (NFI3) plot for each field plot, ensuring that 70% of the dominant area was either *Q. suber* or *P. halepensis* and that the same dominant shrub species found in the field were present. A total of 83 NFI3 oak plots and 51 pine plots were located within the fire perimeter and county region (Alt Empordà). We applied the same procedure to calculate the total and fine shrub fuel loading. Afterwards, we developed a multiple linear regression model (MLR) to create allometric equations linking the total shrub fuel loading estimated with ‘medfate’ with the shrub cover and dominant species, for oak and pine forests separately (Table S2). Using simple linear regression models, we derived fine shrub fuel loading from the total fuel loading for each forest type (Table S3). We applied these allometric equations to our field plots, using shrub cover and dominant species data obtained in the field to compute the total and fine shrub fuel loading for each field plot. To assess litter fuel loading, we created allometric equations using previous data from the IEFC (Gracia 2001), as litter fuel loading was not available in the NFI3 data (Vayreda et al. 2016). We selected the IEFC plots closest to the wildfire with the same dominant tree and shrub species found in the field. This included 91 oak and 190 pine plots located within or near the fire perimeter. We then applied a stepwise backward method to obtain the best MLR model to estimate litter fuel loading in the IEFC plots (Table S4). Finally, we applied these equations to our field data to obtain litter fuel loading in our field plots (Fig. S2). The procedure followed in the multiple linear regression (MLR) analyses to compute shrub and litter fuel loading variables of the two species is described in Appendix D of the Supplementary Material.

Emission factors

We considered CO₂ and CH₄ (two gases that contribute to the greenhouse effect), CO (an active trace gas that aids in the secondary formation of O₃) and PM_{2.5} (which are indicators of health hazards). The emission factors, which represent the mass of a compound emitted per mass of dry fuel consumed (Urbanski 2014), were taken from the literature: Miranda (2004, 2005) for trees and shrub layers and Pallozzi et al. (2018) for trees (Table S5). We applied the formula to each plot to obtain the total emissions of CO₂, CO, CH₄ and PM_{2.5}.

Estimation of pyrogenic emissions

To assess pyrogenic emissions, we applied the Seiler and Crutzen (1980) method (Appendix C) following Eq. 1:

$$EM = A \times B \times C \times D \quad (1)$$

where, EM is the total emissions (Mg ha⁻¹), A is the burned area (ha), B is the fuel loading before the fire (Mg ha⁻¹), C is the combustion factor (%) and D is the emission factor (g kg⁻¹).

Statistical analysis

To determine the differences in fuel loading at the field plot level among the fuel types for oak and pine before the fire, we performed two analyses. First, we compared the fuel loading between species and fuel types using a two-factor ANOVA, with fuel type and species as factors. Then, to study the vertical distribution of fuel loading, we calculated the percentage of fuel loading per layer (i.e., crown, shrub and litter) for each fuel type and species, and carried out a three-factor ANOVA to analyze differences in fuel loading among layers and fuel types for each species (log-transforming fuel loading to achieve normality). Subsequently, we determined the percentage of plots with different fire severity classes for each fuel and forest type using field data. To analyze differences in emissions of CO₂, CO, CH₄ and PM_{2.5} across forest types, fire severities and fuel types, we first analyzed the emissions in green, scorched and charred plots of the two forest types with two-way ANOVAs, independent of the fuel types. We then assessed for each forest type the effect of fire severity (green, scorched and charred) and fuel type (FT2 and FT3 for pines, and FT3 and FT4 for oaks) on atmospheric pollutant emissions. For all ANOVA analyses, we represented the residuals of the models with the predicted values, and examining the graphical representation of the residuals against the expected values allowed us to assess a series of assumptions made about the quality of the model fit: normality, linearity and homoscedasticity. In all cases, whenever ANOVA tests resulted in significant differences, we used a post-hoc Tukey test to identify the levels that were different ($P < 0.05$).

Results

Differences in pre-fire fuel loading among fuel types in pines and oaks

We found significant differences in fuel loading before the fire among fuel types (ANOVA test, $F = 11.94$ $P < 0.001$) but not between species ($F = 3.14$, $P = 0.08$). The interaction between the fuel type and species was also significant

($F=4.28$, $P=0.007$). The total fuel loading was higher in oaks than in pines in FT1, with the lowest tree density. Fuel loading was higher in pine than in oak plots in FT2, whereas fuel loading was similar among species in FT3 and FT4 (Fig. 3).

Fuel loading was distributed differently in different layers of the oak and pine plots (Fig. 4). The percentage of fuel loading in the crown was higher in pine than in oak plots (Fig. 4). In oak plots, there was more fuel loading in the litter than in the shrub layer, whereas in pine plots, the distribution was the opposite (with more fuel loading in the shrub layer). Moreover, pine plots had a higher percentage of fuel loading in the crown than oak plots. In pine plots

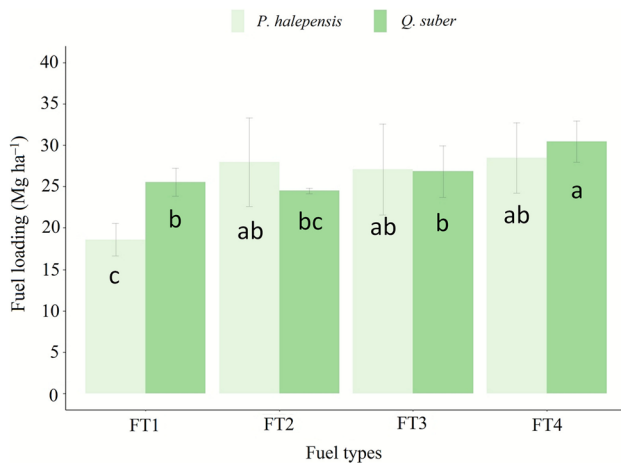


Fig. 3 Mean ($\pm$ standard deviation) values of fuel loading (Mg ha^{-1}) for different fuel types (FT) in the pine and oak plots. For the crown, we only counted fine fuel loading (<6 mm). Descriptions of the four fuel types are given in the Methods section. Different letters indicate significant differences according to Tukey's post hoc test

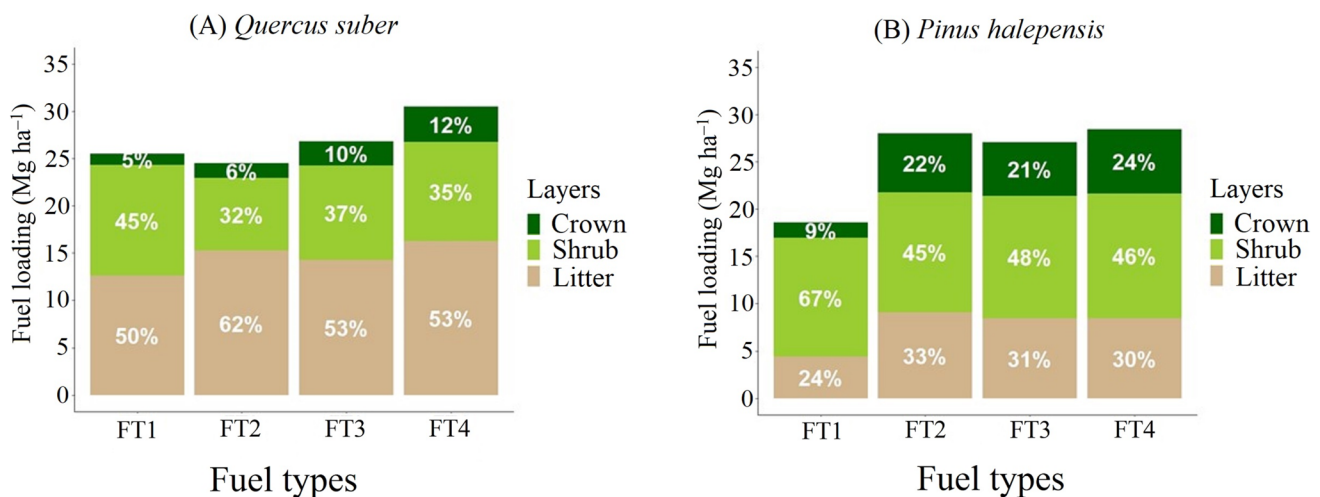


Fig. 4 Distribution of fuel loading (Mg ha^{-1}) before the fire in the crown (only fine fuel loading <6 mm), shrub and litter layers and the four fuel types for **a** oak plots and **b** pine plots. Numbers in the bars indicate the percentages of fuel loading in each layer for each fuel type

we also found differences between FT1 (the fuel type with the lowest tree density) and the other fuel types. FT1 had a higher percentage of shrub fuel loading and much less fuel in the crowns compared to the other fuel types (Fig. 4). When analyzing fuel loading distribution among the 3 layers (crown, shrub and litter) across fuel types and species, we found that the interaction of all three factors was not significant (three-way ANOVA, $F=1.47$ $P=0.18$). However, we found significant differences in fuel loading between species ($F=8.53$ $P=0.004$), layers ($F=509.83$ $P<0.001$) and fuel types ($F=49.35$ $P<0.001$). Paired interactions between the factors were also significant (ANOVA test, $P<0.001$ for all three cases; Fig. S3). Concerning the interaction between species and layer, oak plots had higher litter fuel loading, while pine plots had more crown fuel loading and, to a lesser extent, shrub fuel loading. The interaction of species and fuel type showed similar fuel loading, with pines having slightly lower fuel loading in FT1 compared to the other three fuel types. Finally, the significant interaction between fuel type and layers indicated that FT2 and FT4 had the highest crown fuel loading, FT1 had the lowest crown fuel loading, while shrub fuel loading was similar across all fuel types.

Fire severity and combustion factors

The percentage of plots with green, scorched and charred severities varied across fuel types and species. Crown fires were predominant in oak forests (50–89%, Fig. 5a). In oak plots, FT3 (characterized by large trees with a second tree layer and higher density than FT2) had the highest proportion of green plots (26%). In contrast, FT1 (with the lowest tree density) and FT4 (with higher vertical and horizontal continuity but fewer large trees) had the highest percentage

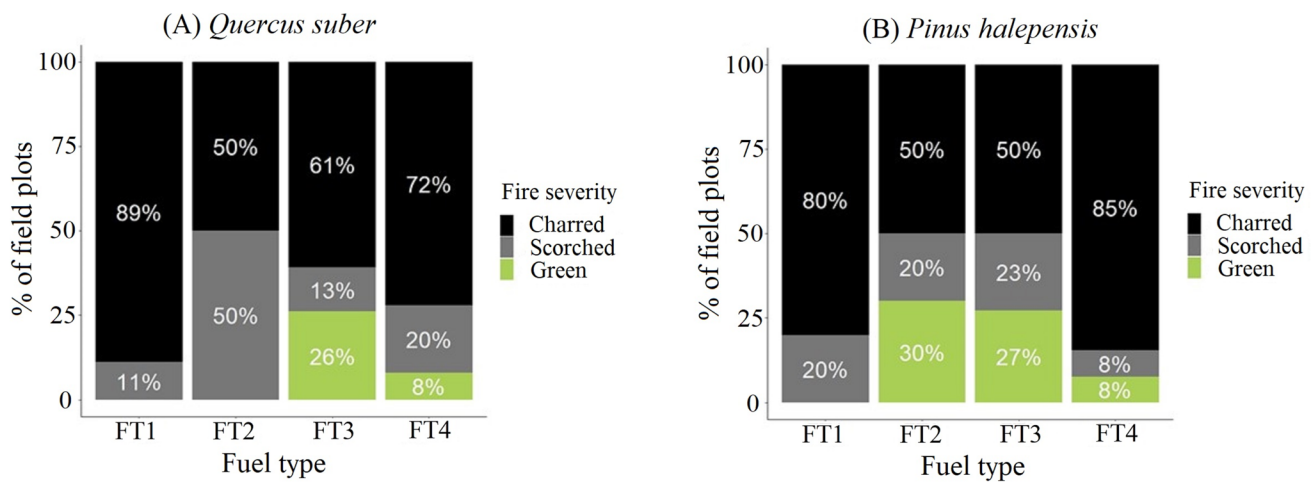


Fig. 5 Percentage of plots with different fire severities (i.e., green, scorched, charred) and different fuel types (FT1 to FT4) in (A) oak (*Quercus suber*) and (B) pine (*Pinus halepensis*) forests. See distribution of fire severity in Fig. 1

of charred plots (89% and 72%, respectively, Fig. 5a). In pine forests, FT2 (characterized by large trees and low tree density) had the highest proportion of green plots (30%, Fig. 5b). FT2 and FT3 had the lowest percentage of charred plots (50%), while FT1 and FT4 had the highest proportion of charred plots (80% and 85%, respectively, Fig. 5b).

Combustion factors from the field varied among species and fuel types depending on fire severity (Table 1). In green plots, FT2 and FT3 of pines had the highest consumption values for litter and shrub layers and, to a lesser extent, for crown fuels. In contrast, FT3 and FT4 of oaks (the only fuel types available in the field) had lower consumption factors

Table 1 Mean of combustion factors (% of fuel consumption) with the coefficient of variation shown in parentheses, for the different layers (crown leaves, crown fine branches, fine shrub fuel, coarse shrub

fuel and litter), species (pines and oaks), fuel type (FT1 to FT4) and severity classes (green, scorched and charred)

Forest	Fuel type	Severity	n	Crown leaves	Crown fine branches	Fine shrub fuel	Coarse shrub fuel	Litter
Oak	FT1	Scorched	1	57	21	70	33	90
		Charred	8	99 (1%)	76 (25%)	99 (1%)	94 (5%)	100 (0%)
	FT2	Scorched	1	30	8	68	30	95
		Charred	1	90	22	100	90	100
	FT3	Green	6	6 (100%)	2 (150%)	42 (52%)	19 (89%)	46 (67%)
		Scorched	3	29 (52%)	11 (64%)	93 (11%)	83 (29%)	94 (9%)
		Charred	14	100 (1%)	82 (20%)	100 (1%)	95 (5%)	100 (0%)
	FT4	Green	2	7 (57%)	2 (50%)	42 (10%)	20 (65%)	38 (79%)
		Scorched	5	15 (47%)	4 (50%)	82 (15%)	67 (45%)	93 (4%)
		Charred	18	100 (0%)	90 (13%)	100 (0%)	84 (18%)	99 (2%)
Pine	FT1	Scorched	1	68	24	100	95	98
		Charred	4	99 (2%)	69 (45%)	100 (1%)	96 (4%)	100 (0%)
	FT2	Green	3	7 (86%)	3 (67%)	89 (17%)	77 (34%)	77 (13%)
		Scorched	2	20 (15%)	11 (9%)	100 (0%)	95 (0%)	99 (1%)
		Charred	5	96 (8%)	67 (25%)	100 (0%)	98 (0%)	100 (1%)
	FT3	Green	6	15 (67%)	6 (67%)	69 (22%)	48 (23%)	62 (19%)
		Scorched	5	38 (34%)	22 (50%)	97 (7%)	82 (21%)	85 (22%)
		Charred	11	97 (4%)	75 (19%)	100 (1%)	91 (5%)	98 (2%)
	FT4	Green	1	23	10	35	10	57
		Scorched	1	25	13	95	70	90
		Charred	11	98 (6%)	82 (16%)	99 (1%)	90 (11%)	99 (2%)

than those of the corresponding pine fuel types (Table 1). In scorched plots, the two species had similar percentages of litter consumption, with higher values in FT1 and FT2 plots. However, in the shrub layer, fine and coarse fuel consumption was higher in pine than in oak plots. Among oak fuel types, FT3 and FT4 had higher consumption factors, whereas they were similar in pine plots. Crown consumption factors were highest in FT1 for both species. In charred plots, litter and fine shrub fuels were completely consumed, while coarse shrub fuel consumption ranged from 84 to 98% across species and fuel types (Table 1). The percentage of crown leaves consumed in charred plots was nearly 100% in all cases, while a lower proportion of fine branches was consumed.

Atmospheric pollutant emissions

The total emissions of all pollutants varied with fire severity, and CO₂ and CH₄ emissions varied with species (Table 2). In particular CO₂, CO, CH₄ and PM_{2.5} emissions were the highest in charred plots, followed by scorched plots and green

plots, while CO₂ and CH₄ emissions were higher in pine than in oak plots (Fig. 6).

Fire severity showed significant differences for all pollutants in the two forest types (Table 3), while there was no significant difference between fuel types, and the interaction of the two factors was only marginally significant in the case of CH₄ for oak plots.

Discussion

Pre-fire fuel loading across different forest and fuel types

Total pre-fire fuel loading is a critical factor to accurately estimating wildfire emissions, but comparisons are challenging due to differing criteria for measuring fuel loading in crowns, shrubs and litter (Abdollahi and Yebra 2023). The use of constant average fuel loading and consumption values in areas with inherent variability, such as Mediterranean forests, or the lack of fuel components such as litter,

Table 2 Two-way ANOVA values of the effects of fire severity, species and their interaction on the emissions of CO₂, CO, CH₄ and PM_{2.5}

Variable	Fire severity	Species	Fire severity*Species
CO ₂	$F=97.55, P<0.001$	$F=7.27, P=0.01$	$F=1.41, P=0.24$
CO	$F=60.02, P<0.001$	$F=0.11, P=0.74$	$F=2.47, P=0.09$
CH ₄	$F=158.03, P<0.001$	$F=60.29, P<0.001$	$F=0.72, P=0.49$
PM _{2.5}	$F=84.81, P<0.001$	$F=0.11, P=0.91$	$F=2.78, P=0.07$

Fig. 6 Mean ($\pm$ standard deviation) values of (A) CO₂, (B) CO, (C) CH₄, and (D) PM_{2.5} emissions (Mg ha⁻¹) in green, scorched and charred plots of pines (light green) and oaks (green)

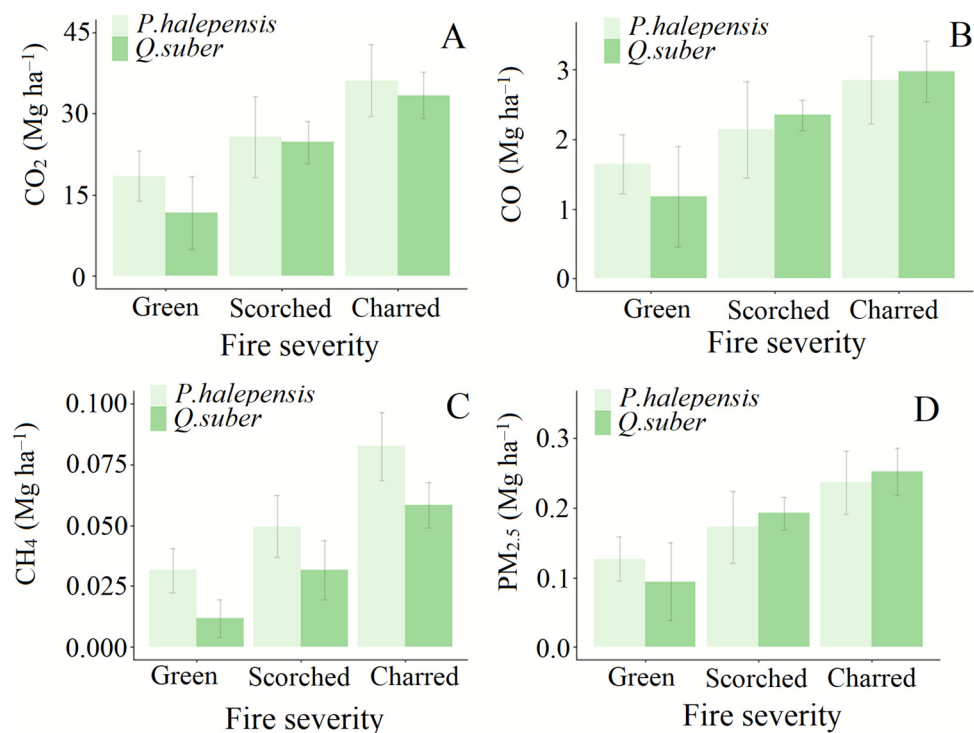


Table 3 Two-way ANOVA values of the effects of severity (green, scorched and charred), fuel type (FT2 and FT3 for pines, FT 3 and FT4 for oaks) and their interaction on the emissions of CO₂, CO, CH₄ and PM_{2.5} of the two forest types considered separately

Emissions (Mg ha ⁻¹)	Forest	Fuel type	Severity	Fuel type * Severity	Green (mean ± SD)	Scorched (mean ± SD)	Charred (mean ± SD)
CO ₂	Pine	$F=1.90$ $P=0.17$	$F=27.60$ $P<0.001$	$F=0.01$ $P=0.99$	18.9 ± 4.8^B	25.1 ± 7.9^B	38.6 ± 6^A
	Oak	$F=0.01$ $P=0.92$	$F=66.70$ $P<0.001$	$F=2.15$ $P=0.13$	11.7 ± 6.7^C	25.7 ± 3.7^B	34.3 ± 4.3^A
CO	Pine	$F=2.04$ $P=0.16$	$F=18.27$ $P=0.001$	$F=0.15$ $P=0.90$	1.66 ± 0.45^B	2.07 ± 0.66^B	3.09 ± 0.58^A
	Oak	$F=0.61$ $P=0.44$	$F=50.61$ $P<0.001$	$F=2.23$ $P=0.11$	1.18 ± 0.72^C	2.40 ± 0.21^B	3.07 ± 0.44^A
CH ₄	Pine	$F=1.59$ $P=0.21$	$F=60.78$ $P<0.001$	$F=0.10$ $P=0.90$	0.033 ± 0.008^C	0.049 ± 0.014^B	0.087 ± 0.01^A
	Oak	$F=0.54$ $P=0.46$	$F=95.54$ $P<0.001$	$F=4.40$ $P=0.02$	0.012 ± 0.007^C	0.034 ± 0.012^B	0.060 ± 0.008^A
PM _{2.5}	Pine	$F=1.89$ $P=0.17$	$F=24.09$ $P<0.001$	$F=0.01$ $P=0.98$	0.13 ± 0.03^B	0.17 ± 0.05^B	0.26 ± 0.04^A
	Oak	$F=0.21$ $P=0.64$	$F=60.13$ $P<0.001$	$F=2.21$ $P=0.12$	0.09 ± 0.06^C	0.2 ± 0.02^B	0.26 ± 0.03^A

Significant differences among fire severities for each pollutant and species according to the Tukey post hoc test are indicated with different letters

can lead to substantial over- or underestimation of wildfire emissions (Fernandes and Loureiro 2013; Lydersen et al. 2014; Lasslop and Kloster 2015). For example, including or excluding litter, variations in its definition, or differences in collection methods can significantly alter total fuel loading estimates (Ganteaume et al. 2014; Morgan Varner et al. 2015). Taking these facts into consideration, we found small differences in total fuel loading, but consistent with previous analysis showing similar values (i.e., 10–30 Mg ha⁻¹ for *Q. suber* and conifer forests) and distribution among layers. Our results are in line with other studies that found that oak forests had more litter, while pine forests exhibited higher shrub and crown fuel loading (Silva et al. 2006; Rosa et al. 2011). Fuel loading in oak stands varied with the time since the last fire. In areas without fire for over 25 years, shrubs were dominated by *E. arborea* or *A. unedo*, while areas with frequent fires were dominated by *Cistus* or *Genista* species (Curt et al. 2011; Schaffhauser et al. 2011, 2015). In the Jonquera wildfire, 61% of plots were dominated by *E. arborea* or *A. unedo* with an average fuel loading of 12 Mg ha⁻¹, while there were only 8% of plots with *Cistus/Genista* with lower fuel loading (8.3 Mg ha⁻¹), similar to the 6.26 Mg ha⁻¹ reported in southern Spain (Pasalodos-Tato et al. 2015). This distinction has been noted by other authors (Casals et al. 2023) and explains differences in fire behavior, with surface fires occurring in *Cistus*-dominated stands and crown fires in *Erica*-dominated stands (Schaffhauser et al. 2011).

In pine stands, the shrub layer contained the highest percentage of fuel loading, followed by litter and crown (Fig. 4). The FT1 of pine stands (characterized by higher open areas and low tree density) showed the lowest total fuel loading due to low litter accumulation, while the other fuel types had similar total fuel loading (Fig. 4). The low values of FT1 can be attributed to the open nature of these areas, and with characteristics associated with low site quality (i.e., low values of water availability and high percentage of stoniness, Alvarez et al. 2012b) that limit potential high tree densities. In

other Mediterranean regions, high total fuel loading has been reported for pure maquis shrublands (i.e., 25–53 Mg ha⁻¹ in Greece, Dimitrakopoulos 2002). In our study, shrubland fuel loading ranged from 4 to 21 Mg ha⁻¹, similar to the understory of other *P. halepensis* forests (Dimitrakopoulos et al. 2007). The dominant shrub species in pine plots were *Q. coccifera* and *P. lentiscus* (58% of pine plots), corresponding to formations with an average of 18 Mg ha⁻¹, followed by *R. officinalis* with 16 Mg ha⁻¹. These results are in line with previous research in southern Spain, which reported high variability in shrub fuel loading, and even exceeded those measured in Catalonia (Pasalodos-Tato et al. 2015; Casals et al. 2023). Litter fuel loading in *P. halepensis* was also consistent with other studies in Greece (Dimitrakopoulos et al. 2007) and France (Ganteaume et al. 2009). Fine crown fuel loading in pine stands was highly variable, and higher values were previously found in Greece were associated with forests with greater basal area, ranging from 33 to 69 m² ha⁻¹ (Mitsopoulos and Dimitrakopoulos 2007). Moreover, crown fuel loading assessments often include leaves and branches up to 0.6 cm (Jiménez et al. 2013), whereas in our study only branches below 0.6 cm were consumed. This methodological difference likely explains why our crown fuel loading estimates were lower than those from other studies that included larger branch sizes (Jiménez et al. 2013; Balde et al. 2023).

Fire severity and combustion factors

In the Jonquera wildfire, fire severity was similar between oak and pine stands, with a dominance of charred plots with 69% and 62% respectively and a small proportion of scorch and green plots with 31% in oak and 38% in pine. Previous studies found that wind speed affects fire behavior and fire severity depending on the forest type. In broadleaf forests, wind-driven fire behavior is shaped by litter type, thickness and topography, while in coniferous forests, particularly pine stands, wind plays a dominant role in driving surface fire

spread and increasing the likelihood of crown fires (Aghajani et al. 2014; Cruz et al. 2025). However, our results suggest that while wind is a key factor in the two forest types, its influence is more pronounced under extreme conditions, overriding landscape or species-specific factors (Aghajani et al. 2014; Keeley and Syphard 2019; Prichard et al. 2020b). In contrast, fire severity varied among fuel types for both species due to differences in stand structure, but in a lower way than other past wind-driven wildfires in the same area, most likely due to differences in fuel and meteorological conditions. FT2, characterized by large trees and low tree density often linked to surface fires, was rare in oak plots but common in pine stands, where it had the highest percentage of green plots (30%) (Fig. 5). This aligns with findings from the Ventalló wildfire (Alvarez et al. 2012a), where FT2 also exhibited the lowest severity. However, in the Jonquera fire, FT2 pine stands spread more crown fires, most likely due to differences in fuel and meteorological conditions. Stands with high basal area, larger tree size, and less variability in tree diameter characteristics associated with FT2 have been linked to higher post-fire survival and lower damage in Northeastern Spain in models using forest inventories and fire data (Gonzalez et al. 2007). Similar trends have been observed between large trees with low densities and live post-fire structures in *Pinus nigra* and *Pinus pinaster* forests with recurrent low-severity fires in Spain (Fulé et al. 2008; Viedma et al. 2020), in Portugal (Fernandes et al. 2015) or in fire-tolerant trees in Oregon, USA (Hessburg et al. 2016). FT3, with higher tree density than FT2 and a second tree layer, showed variable severity. In oaks, FT3 burned with the greatest range in severity linked to the highest shrub and litter fuel loading variability. This fuel type also had the highest proportion of green plots partly due to the lower fuel loading without *E. arborea* and possibly past fuel management identified during the fieldwork. In contrast, FT4, with higher vertical and horizontal continuity, had very high percentages of charred plots caused by active crown fires in both species as previously reported by other authors, who modelled the effect of forest structure on the potential for crown fires and large tree consumption (Ritter et al. 2023). High intensity fires in forests with these characteristics is a phenomenon widely described in pine forests, but also oak forests in areas of southern France and north-eastern Catalonia (Schaffhauser et al. 2011; Alvarez et al. 2012a; Sánchez-Pinillos et al. 2021). This is in accordance with previous research in which, independently of the species, relative stand damage increases when there is a large variation in tree size, lower basal area and mean diameter (Gonzalez et al. 2007; Jain and Graham 2007). Finally, FT1, with the lowest tree density and open forest structures, had similar high fire severity in both species. Its sparse canopy, combined with higher understory density and increased wind exposure, increased the availability of surface fuels and favored

intense surface fires and passive crown fires (Alvarez et al. 2012a; Sánchez-Pinillos et al. 2021). This pattern of severe fires in open, understory-dominated stands has been widely documented in Mediterranean forests (Oliveras et al. 2009; Viedma et al. 2020).

Combustion factors are influenced by variables such as fuel moisture content, fire behavior, spread direction and fire phase (Urbanski 2013; Surawski et al. 2016; Fernandes et al. 2022). In the Mediterranean area, there is limited data on combustion factors across forest or fire types (topographic, convective or wind-driven fires) (Costa et al. 2011). In our study, combustion factors of all layers (Table 1) were consistent with wind-driven fires (Costa et al. 2011), and showed the expected correlation between higher fire severity and increased fuel consumption (Campbell et al. 2007; Molina et al. 2019; Price et al. 2022). In charred plots, litter and fine/coarse shrub fuels were nearly completely consumed, aligning with experimental surface burnings in *Pinus pinaster* in Portugal (59–100% for litter, 24–95% for understory fuels; Fernandes and Loureiro 2013), but higher than the 70% litter consumption observed in low-severity fires in Oregon (Campbell et al. 2007). Direct comparisons, however, are difficult due to differing fuel definitions and data availability. Crown leaves were completely consumed in charred plots, typical of high-intensity crown fires (Mitsopoulos and Dimitrakopoulos 2014), but 10–33% of fine crown branches remained unconsumed. Wind-driven fires, which spread faster than convective or topographic fires, can result in partial branch combustion even when reaching the canopy if there is an appropriate combination of windspeed and fine fuel moisture content, as seen in the Sobradelo wildfire in Spain (Jiménez et al. 2013). In scorched plots but especially in green plots (in this case only comparing FT3 and FT4, the only available for the two species), consumption was generally higher in pine than oak stands. Green plots showed sensitivity to the relationship between percentage of large trees, tree height and surface fire intensity. Studies in California found higher burn efficiency in conifers than in hardwood at higher severities (De Santis et al. 2010), while in Spain, oak species showed higher efficiency at lower severities (Balde et al. 2023). However, some of these studies did not differentiate biomass into fractions or different components for burn efficiency calculations as we tried to assess in the field work. Improved understanding of species-specific and size-specific consumption patterns across fire severities is essential to refine emission estimates and reduce uncertainties (Balde et al. 2023). FT2 pine plots were the most fire-resistant, retaining green trees despite burning in high-intensity surface fires. This resistance was attributed to their taller trees and lower tree density, which allowed for higher surface fuel consumption without severe crown damage. In FT3 and FT4 oak stands, lower litter consumption in green plots reflected the inability to burn more

surface fuel without transitioning to a higher severity class. In contrast, pine plots consumed more surface fuels in green severities without causing greater tree damage, as their taller trees were less affected. Similarly, Busby et al. (2023) found that overstory canopy height was the only influential factor in reducing fire severity under strong winds.

Atmospheric pollutant emissions

Quantifying and comparing wildfire emissions is challenging due to substantial variation in both quantity and chemical composition among fires (Sommers et al. 2014; Filonchik et al. 2025). In our study, fire severity was the main factor that determined emissions for all pollutants, consistent with established relationships between fire severity, burning efficiency and emission levels (De Santis et al. 2010; Balde et al. 2023). CO₂ emissions were slightly higher in pine than in oak forests, with the Jonquera fire with charred plots emitting 34–39 Mg ha⁻¹ of CO₂ and 12–19 Mg ha⁻¹ in green plots for both oak and pine (Table 3). While these values are considerably lower than those reported from other large fires in Australia (Surawski et al. 2016) or Canada (Filonchik et al. 2025), they are in line with studies in similar regions and climates (Bacciu et al. 2009), with some variations depending on the forest type and severity. However, these studies did not account for litter or included crown branches larger than 0.6 cm, which were not consumed in our study. Despite advances in methods for estimating fuel loading and crown fuel consumption (Fernández-Guisuraga and Fernandes 2023; Labenski et al. 2023; Wiedinmyer et al. 2023; Conrad et al. 2024), wildfire consumption and emissions in Mediterranean areas are often underestimated due to the exclusion of litter and shrub layers (Domingo et al. 2017; Balde et al. 2023).

Epidemiological literature has focused on health effects of PM_{2.5} concentrations (Chowdhury et al. 2024), but estimating CO emissions is gaining importance because elevated CO levels during and after wildfires pose risks to all breathing organisms for up to a month post-fire (Griffin et al. 2023; Yilmaz et al. 2023). Here, we only assessed immediate CO and PM_{2.5} emissions, which were lower (5.3 Mg ha⁻¹ and 0.5 Mg ha⁻¹, respectively) than those reported for the Andilla fire but similar to emissions from the maquis forest fire in Cortes de Pallás (Chaves Naharro 2015). In contrast, Bacciu et al. (2009) documented CO emissions of 0.3–7 Mg ha⁻¹ and PM_{2.5} emissions of 0.04–0.08 Mg ha⁻¹ in a Mediterranean maquis fire. CH₄ emissions, less studied in Mediterranean wildfires due to limited emission factor data compared to other countries such as the USA (Prichard et al. 2020a; Urbanski et al. 2022), were higher in pines than in oaks in our study (Tables 2 and 3). CH₄ emissions in pine stands also exhibited greater variability across fire severities compared to other gases, likely due to the higher resin

content in pines, which is carbon-rich and conducive to CH₄ production. The CH₄ values observed here exceeded those reported for the Andilla and Cortes de Pallás fires (0.037 and 0.022 Mg ha⁻¹) and were markedly higher than the 0.019–0.036 Mg ha⁻¹ range reported by Bacciu et al. (2009). Moreover, the higher consumption rates in pines at low fire severities may offset the higher litter fuel loading observed in oak forests. On the other hand, the shrub species typical of oak forests have greater water retention capacity and greater fire resistance than those in pine stands, likely resulting in less combustion and lower CH₄ emissions.

Limitations and uncertainties

Important limitations need to be considered when interpreting our findings. First, the differences in fuel distribution observed between pine- and oak-dominated forests in this study could have been better understood by considering ecological factors such as litter chemistry, soil properties and topography, all of which have been shown to influence fire behavior and fuel consumption (Alves et al. 2011; Ma et al. 2022; Akin et al. 2023). Litter chemistry plays a significant role, as forests with higher concentrations of volatile compounds in their litter, such as pine forests with their resin-rich needles, tend to exhibit higher fuel loads and more intense fires compared to oak forests, where tannins and lignin in the litter can dampen combustibility (Ficken and Wright 2017; Liang et al. 2022). Additionally, soil properties, particularly water retention capacity, influence fuel consumption rates, with soils that have high clay content tending to retain moisture, reducing combustion efficiency, whereas drier, sandy soils allow for greater fuel consumption (Ma et al. 2022). Furthermore, topography, including aspects like slope and elevation, affects fuel distribution by concentrating fuel on steeper slopes and enhancing its susceptibility to ignition, as shown in previous studies (Lecina-Diaz et al. 2014; Conedera et al. 2024). Taken together, litter chemistry, soil properties and topography could provide a deeper insight into the mechanisms behind fuel distribution and combustion, which in turn could influence the emissions of gases like CO₂, CO, CH₄ and particulate matter.

Important sources of uncertainty also arise in emission estimates. According to the Corinair90 inventory, the contribution of forest fires to the total emissions is small, but uncertainties are very large (San-Miguel-Ayán and Steinbrecher 2019). Emissions from wildfires require accurate and complex estimations, often based on the Seiler and Crutzen (1980) equation. The components of this equation are derived in various ways, introducing significant uncertainties at each component (Larkin et al. 2014; Lydersen et al. 2014). Among these, pre-fire fuel loading is a primary source of uncertainty, with variations in fuel characteristics contributing up to 83% of the uncertainty in emission

estimates (Ottmar et al. 2008; Fernandes et al. 2022). While measuring fuel loading directly before a wildfire is ideal, it is rarely feasible. To mitigate this uncertainty, we avoided reliance on average fuel values that fail to capture the variability of wildland fuels (Kennedy et al. 2020). Instead, we used species-specific allometric equations derived from Spanish National Forest Inventory (NFI) data for canopies and shrubs, supplemented by the Ecological Forestry Inventory of Catalunya (IEFC) for litter assessment (Gracia 2001; Vayreda et al. 2016). Although more precise shrubland equations exist in Spain, their application post-fire is challenging, as variables such as shrub height or diameter may no longer be measurable (Pasalodos-Tato et al. 2015; Casals et al. 2016, 2023). Litter is one of the most uncertain components in wildfire emission calculations (Fernandes and Loureiro 2013; Kennedy et al. 2020). If litter content is not measured before the fire, underestimations of fuel loading and emissions are inevitable, particularly in extreme wildfires. Litter consumption can range from near 0 to 100%, making its inclusion crucial because only subsurface carbon, litter, duff and mineral soil carbon, account for 65% of total fire-wide carbon emissions (Campbell et al. 2007; Vilen and Fernandes 2011). Prioritizing litter measurements is essential for reducing overall uncertainty, especially in the Mediterranean region, where studies of litter fuel loading for wildfire emissions are scarce (Fernandes and Loureiro 2013; Garcia-Hurtado et al. 2013; Kennedy et al. 2020).

Combustion factors for each layer are another important source of uncertainty in wildfire emissions, with over 30% (Ottmar et al. 2008; Fernandes et al. 2022; Prichard et al. 2022) that could be higher depending on the position of the fire (head, flank and back), fuel type, arrangement and fuel moisture (Surawski et al. 2016). In the Mediterranean area, there are limited field-based examples, and no standardized methods are widely adopted for estimating fuel consumption after a fire, particularly when applying the Seiler and Crutzen (1980) method (De Santis et al. 2010; Jiménez et al. 2013; Balde et al. 2023). Few attempts have been made to estimate fuel consumption in active crown fires (Stocks 1987b, 1989; Call and Albin 1997), and some used visual assessment of consumption percentage, but knowing pre-fire fuel loading (Stocks 1987a, b). Because of this, our results should be interpreted cautiously as there is still a subjective element despite efforts to minimize bias. To improve accuracy, we visited unburned areas within and bordering the fire perimeter to identify potential fuels and species before and during the fieldwork. During plot measurements, we spent considerable time recording shrub diameters and fine tree branches. All plots were measured by two people to reduce observer bias, and each percentage estimate was averaged between both to minimize perceptual errors. Additionally, we took numerous photos from various angles at each plot to document trees, shrubs and litter. Afterwards, we

reviewed and adjusted the percentages to ensure consistency across plots, aiming for a more balanced interpretation. These efforts helped mitigate biases and allowed us to accurately reflect the differences observed between plots. Our results offer a logical and valuable contribution, particularly in understanding the fraction of canopy consumed during these fires. Grouping fuel consumption into four types based on vertical and horizontal continuity, and comparing them across species, aligns with recent studies on how tree continuity influences crown fire potential and fuel consumption (Ritter et al. 2023).

Finally, emission factors are another significant source of uncertainty in emission estimations, contributing to about 16% of the total error (Ottmar et al. 2008; Bacciu et al. 2015; Fernandes et al. 2022). This uncertainty could be higher, as emission factors are influenced by many factors including fuel moisture, fire type, fuel type and combustion phase. The latter can lead to substantially different emissions when using average values, causing variations in pollutants that are highly dependent on them (e.g. CO, CH₄ or PM) (Prichard et al. 2020a). The main challenge is that most available data on emission factors is from USA forests (Urbanski 2013; Prichard et al. 2020a; Urbanski et al. 2022), which may not be applicable to European wildfires due to differences in species composition and combustion characteristics, such as the flaming and smoldering phases (Fernandes et al. 2022). In our study, we did not differentiate between flaming or smoldering combustion phases nor did we consider fuel moisture, which could further affect the accuracy of emission estimates. Efforts have been made in Europe to refine emission factors for species like pine and oak, but further research is needed, particularly for southern European conditions. In this study, we used litter emission factors from another species of the genus, *Q. pubescens* (Pallozzi et al. 2018) for *Q. suber*, as it was the only available data. It is crucial to continue research and field measurements in Mediterranean regions to improve emission factor accuracy (Fernandes et al. 2022). Despite these limitations, this study contributes valuable knowledge, especially in establishing relationships between pre-fire fuel loading, fire severity, fuel consumption and emissions in heterogeneous Mediterranean forests.

Conclusion

With the growing risk of wildfires in Europe, especially in Mediterranean ecosystems, the loss of forest ecosystem services and increased pollutant emissions have become more pronounced (Zheng et al. 2021; Shakoor et al. 2023; Lecina-Diaz et al. 2024). While remote sensing tools are valuable for estimating emission components (Labenski et al. 2023), field data remains critical for validating and calibrating

models, and for quantifying fuel consumption and emissions across fuel types and layers (Jiménez et al. 2013). Our study reveals that in Mediterranean pine and oak forests, fuel types with greater vertical and horizontal continuity had higher fire severity and fuel consumption. At high severities, oak forests emitted more CO and PM_{2.5} while pine emitted more CO₂ and CH₄. While fire severity was the primary factor influencing pollutant emissions, pine forests emitted higher levels of pollutants than oak forests at lower severities, highlighting the importance of understanding pre-fire fuel distribution for accurate emission quantification. Forest management should aim to create structures with predominantly taller trees, resembling the highly fire-resistant fuel types (FT2 and FT3) in both pine and oak forests. Efforts should also focus on reducing understory shrub density to reduce surface fire intensity, thereby decreasing fire severity and emissions, as demonstrated in our study. Special attention should be given to oak forests with understories dominated by *Erica* and pine stands with understories composed of *Pistacia lentiscus*, *Quercus coccifera* or *Rosmarinus officinalis*. By implementing targeted management strategies to achieve these specific vegetation structures, we can significantly reduce emissions and minimize the environmental impact of wildfires in these ecosystems.

Acknowledgements We thank The Generalitat de Catalunya fire-fighters corps for providing the helicopter flights as well as the use of the Llança fire station during field work and the GRAF unit for some reports and information gathered from visits to the fire. Thanks to Roberto Molowny for statistical consultations, Monserrat Monge for graphic illustrations and Paul Abbott for English language corrections.

Funding Open Access Funding provided by Universitat Autònoma de Barcelona.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

References

- Abdollahi A, Yebra M (2023) Forest fuel type classification: review of remote sensing techniques, constraints and future trends. *J Environ Manage* 342:118315. <https://doi.org/10.1016/j.jenvm.2023.118315>
- Aghajani H, Fallah A, Fazlollah Emadian S (2014) Modelling and analyzing the surface fire behaviour in Hyrcanian forest of Iran. *J for Sci* 60(9):353–362. <https://doi.org/10.17221/97/2013-jfs>
- Akin ID, Akinleye TO, Robichaud PR (2023) Changes in soil properties over time after a wildfire and implications to slope stability. *J Geotech Geoenviron Eng* 149(7):04023045. <https://doi.org/10.1061/jggef.k.gteng-11348>
- Alves CA, Vicente A, Monteiro C, Gonçalves C, Evtyugina M, Pio C (2011) Emission of trace gases and organic components in smoke particles from a wildfire in a mixed-evergreen forest in Portugal. *Sci Total Environ* 409(8):1466–1475. <https://doi.org/10.1016/j.scitotenv.2010.12.025>
- Alvarez A, Gracia M, Retana J (2012a) Fuel types and crown fire potential in *Pinus halepensis* forests. *Eur J for Res* 131(2):463–474. <https://doi.org/10.1007/s10342-011-0520-6>
- Alvarez A, Gracia M, Vayreda J, Retana J (2012b) Patterns of fuel types and crown fire potential in *Pinus halepensis* forests in the Western Mediterranean Basin. *For Ecol Manag* 270:282–290. <https://doi.org/10.1016/j.foreco.2011.01.039>
- Alvarez A, Gracia M, Castellnou M, Retana J (2013) Variables that influence changes in fire severity and their relationship with changes between surface and crown fires in a wind-driven wildfire. *For Sci* 59(2):139–150. <https://doi.org/10.5849/forsci.10-140>
- Bacciu V, Salis M, Spano D, Arca B, Pellizzaro G, Duce P (2009) Assessment of smoke emission and carbon estimates from Mediterranean Maquis fire events. In: Eighth symposium on fire and forest meteorology. American Meteorological Society, Kalispell. https://ams.confex.com/ams/8Fire/techprogram/paper_156372.htm
- Bacciu V, Spano D, Salis M (2015) Emissions from forest fires: methods of estimation and national results. In: Valentini R, Miglietta F (eds) *The greenhouse gas balance of Italy: an insight on managed and natural terrestrial ecosystems*. Springer, Berlin, Heidelberg, pp 87–102
- Badia A, Pallares-Barbera M, Valdeperas N, Gisbert M (2019) Wildfires in the wildland-urban interface in Catalonia: vulnerability analysis based on land use and land cover change. *Sci Total Environ* 673:184–196. <https://doi.org/10.1016/j.scitotenv.2019.04.012>
- Balde B, Vega-Garcia C, Gelabert PJ, Ameztegui A, Rodrigues M (2023) The relationship between fire severity and burning efficiency for estimating wildfire emissions in Mediterranean forests. *J for Res* 34(5):1195–1206. <https://doi.org/10.1007/s11676-023-01599-1>
- Beltrán-Marcos D, Suárez-Seoane S, Fernández-Guisuraga JM, Azevedo JC, Calvo L (2024) Fire regime attributes shape pre-fire vegetation characteristics controlling extreme fire behavior under different bioregions in Spain. *Fire Ecol* 20(1):47. <https://doi.org/10.1186/s42408-024-00276-w>
- Bowman DMJS, Balch JK, Artaxo P, Bond WJ, Carlson JM, Cochrane MA, D'Antonio CM, DeFries RS, Doyle JC, Harrison SP, Johnston FH, Keeley JE, Krawchuk MA, Kull CA, Marston JB, Moritz MA, Prentice IC, Roos CI, Scott AC, Swetnam TW, van der Werf GR, Pyne SJ (2009) Fire in the Earth system. *Science* 324(5926):481–484. <https://doi.org/10.1126/science.1163886>
- Busby SU, Klock AM, Fried JS (2023) Inventory analysis of fire effects wrought by wind-driven megafires in relation to weather and pre-fire forest structure in the western Cascades. *Fire Ecol* 19(1):58. <https://doi.org/10.1186/s42408-023-00219-x>
- Call PT, Albini FA (1997) Aerial and surface fuel consumption in crown fires. *Int J Wildland Fire* 7(3):259–264. <https://doi.org/10.1071/wf9970259>
- Campbell J, Donato D, Azuma D, Law B (2007) Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J Geophys Res Biogeosciences* 112(G4):2007JG000451. <https://doi.org/10.1029/2007JG000451>
- Carvalho A, Monteiro A, Flannigan M et al (2007) Forest fire emissions under climate change: impacts on air quality. In: Seventh symposium on fire and forest meteorology. American Meteorological

- Society, The Turrets. https://ams.confex.com/ams/7firenortheast/techprogram/paper_126854.htm
- Casals P, Valor T, Besalú A, Molina-Terrén D (2016) Understory fuel load and structure eight to nine years after prescribed burning in Mediterranean pine forests. *For Ecol Manag* 362:156–168. <https://doi.org/10.1016/j.foreco.2015.11.050>
- Casals P, Gabriel E, De Cáceres M, Ríos AI, Castro X (2023) Composition and structure of Mediterranean shrublands for fuel characterization. *Ann for Sci* 80(1):23. <https://doi.org/10.1186/s13595-023-01190-y>
- Chaves Naharro J (2015) Forest fires and climate change. Estimation of emissions from the fires in Andilla and Cortes de Pallás. Master's Thesis. Universitat Politècnica de València. <https://riunet.upv.es/handle/10251/51839>
- Chiriaco MV, Perugini L, Cimini D, D'Amato E, Valentini R, Bovio G, Corona P, Barbati A (2013) Comparison of approaches for reporting forest fire-related biomass loss and greenhouse gas emissions in southern Europe. *Int J Wildland Fire* 22(6):730–738. <https://doi.org/10.1071/wf12011>
- Chowdhury S, Hänninen R, Sofiev M, Aunan K (2024) Fires as a source of annual ambient PM_{2.5} exposure and chronic health impacts in Europe. *Sci Total Environ* 922:171314. <https://doi.org/10.1016/j.scitotenv.2024.171314>
- CIFFC (2023) Canadian Wildland Fire Management Glossary. Can Interag For Fire Cent Winnipeg, MB, Canada. https://www.cifcc.ca/sites/default/files/2023-04/CWFM_glossary_v2023-04-24-EN.pdf
- Conedera M, Feusi J, Pezzatti GB, Krebs P (2024) Linking the future likelihood of large fires to occur on mountain slopes with fuel connectivity and topography. *Nat Hazards* 120(5):4657–4673. <https://doi.org/10.1007/s11069-023-06395-y>
- Conrad ET, Jolly WM, Brown TP, Hillman SC (2024) Branching out: species-specific canopy architecture limits live crown fuel consumption in Intermountain West USA conifers. *Fire Ecol* 20(1):28. <https://doi.org/10.1186/s42408-024-00261-3>
- Costa P, Castellnou M, Larrañaga A, Miralles M, Kraus D (2011) Prevention of Large Wildfires using the Fire Types Concept. Departament de Interior. Generalitat de Catalunya., Cerdanyola del Vallès, Barcelona, Spain. https://interior.gencat.cat/ca/el_departament/publicacions/proteccio_civil/la_prevenccio_dels_grans_incendis_forestals_adaptada_a_l_incendi_tipus/index.html
- CREAF (2009) Land Cover Map of Catalonia (MCSC). <http://www.creaf.uab.es/mcsc/descriptiu.htm>
- Curt T, Schaffhauser A, Borgniet L, Dumas C, Estève R, Ganteaume A, Jappiot M, Martin W, N'Diaye A, Poilvet B (2011) Litter flammability in oak woodlands and shrublands of southeastern France. *For Ecol Manag* 261(12):2214–2222. <https://doi.org/10.1016/j.foreco.2010.12.002>
- Cruz MG, Hoffman CM, Fernandes PM (2025) Global synthesis of quantification of fire behaviour characteristics in forests and shrublands: recent progress. *Curr for Rep* 11(1):8. <https://doi.org/10.1007/s40725-024-00241-5>
- Çinar T, Taşpınar F, Aydın A (2024) Analysis and estimation of gaseous air pollutant emissions emitted into the atmosphere during Manavgat and Milas wildfire episodes using remote sensing data and ground measurements. *Air Qual Atmos Health* 17(3):559–579. <https://doi.org/10.1007/s11869-023-01463-5>
- DARP (2012) Report on the forest fire of July 22, 2012 in La Jonquera (Alt Empordà), Generalitat de Catalunya, Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural, Girona, Spain. https://agricultura.gencat.cat/web/contenut/06-medi-natural/bosc/gestio-forestal/obres/restauracio-forestal/restauracio-hidrologica/fitxers-binarijs/jonquera_informe_incendi.pdf
- De Cáceres M, Martínez-Vilalta J, Coll L, Llorens P, Casals P, Poyatos R, Pausas JG, Brotons L (2015) Coupling a water balance model with forest inventory data to predict drought stress: the role of forest structural changes vs. climate changes. *Agric for Meteorol* 213:77–90. <https://doi.org/10.1016/j.agrformet.2015.06.012>
- de Rigo D, Libertà G, Houston Durrant T, Artés T, San Miguel Ayanz J (2017) Forest fire danger extremes in Europe under climate change: variability and uncertainty. *Publ Off Eur Union. JRC108974*. <https://doi.org/10.2760/13180>
- De Santis A, Asner GP, Vaughan PJ, Knapp DE (2010) Mapping burn severity and burning efficiency in California using simulation models and Landsat imagery. *Remote Sens Environ* 114(7):1535–1545. <https://doi.org/10.1016/j.rse.2010.02.008>
- Dimitrakopoulos AP (2002) Mediterranean fuel models and potential fire behaviour in Greece. *Int J Wildland Fire* 11(2):127–130. <https://doi.org/10.1071/wf02018>
- Dimitrakopoulos AP, Mitsopoulos ID, Raptis DI (2007) Nomographs for predicting crown fire initiation in Aleppo pine (*Pinus halepensis* Mill.) forests. *Eur J Forest Res* 126(4):555–561. <https://doi.org/10.1007/s10342-007-0176-4>
- Domingo D, Lamelas-Gracia MT, Montealegre-Gracia AL, de la Riva-Fernández J (2017) Comparison of regression models to estimate biomass losses and CO₂ emissions using low-density airborne laser scanning data in a burnt Aleppo pine forest. *Eur J Remote Sens* 50(1):384–396. <https://doi.org/10.1080/22797254.2017.1336067>
- Duane A, Brotons L (2018) Synoptic weather conditions and changing fire regimes in a Mediterranean environment. *Agric for Meteorol* 253:190–202. <https://doi.org/10.1016/j.agrformet.2018.02.014>
- Fernandes PM (2009) Combining forest structure data and fuel modelling to classify fire hazard in Portugal. *Ann for Sci* 66:415. <https://doi.org/10.1051/forest/2009013>
- Fernandes PM, Fernandes MM, Loureiro C (2015) Post-fire live residuals of maritime pine plantations in Portugal: structure, burn severity, and fire recurrence. *For Ecol Manag* 347:170–179. <https://doi.org/10.1016/j.foreco.2015.03.023>
- Fernandes AP, Lopes D, Sorte S, Monteiro A, Gama C, Reis J, Menezes I, Osswald T, Borrego C, Almeida M, Ribeiro LM, Viegas DX, Miranda AI (2022) Smoke emissions from the extreme wildfire events in central Portugal in October 2017. *Int J Wildland Fire* 31(11):989–1001. <https://doi.org/10.1071/wf21097>
- Fernandes PM, Loureiro C (2013) Fine fuels consumption and CO₂ emissions from surface fire experiments in maritime pine stands in northern Portugal. *For Ecol Manag* 291:344–356. <https://doi.org/10.1016/j.foreco.2012.11.037>
- Fernández-García V, Beltrán-Marcos D, Fernández-Guisuraga JM, Marcos E, Calvo L (2022) Predicting potential wildfire severity across Southern Europe with global data sources. *Sci Total Environ* 829:154729. <https://doi.org/10.1016/j.scitotenv.2022.154729>
- Fernández-Guisuraga JM, Fernandes PM, Fernández-Guisuraga JM, Fernandes PM (2023) Using pre-fire high point cloud density LiDAR data to predict fire severity in central Portugal. *Remote Sens* 15(3):768. <https://doi.org/10.3390/rs15030768>
- Ficken CD, Wright JP (2017) Effects of fire frequency on litter decomposition as mediated by changes to litter chemistry and soil environmental conditions. *PLoS ONE* 12(10):e0186292. <https://doi.org/10.1371/journal.pone.0186292>
- Filonchik M, Peterson MP, Zhang LM, Zhang LF, He Y (2025) Estimating air pollutant emissions from the 2024 wildfires in Canada and the impact on air quality. *Gondwana Res* 140:194–204. <https://doi.org/10.1016/j.gr.2024.12.012>
- Fortems-Cheiney A, Broquet G, Potier E, Plauchu R, Berchet A, Pison I, van der Denier Gon H, Dellaert S (2024) CO anthropogenic emissions in Europe from 2011 to 2021: insights from Measurement of Pollution in the Troposphere (MOPITT) satellite data. *Atmos Chem Phys* 24(8):4635–4649. <https://doi.org/10.5194/acp-24-4635-2024>

- French NHF, de Groot WJ, Jenkins LK, Rogers BM, Alvarado E, Amiro B, de Jong B, Goetz S, Hoy E, Hyer E, Keane R, Law BE, McKenzie D, McNulty SG, Ottmar R, Pérez-Salicrup DR, Randerson J, Robertson KM, Turetsky M (2011) Model comparisons for estimating carbon emissions from North American wildland fire. *J Geophys Res* 116:G00K05. <https://doi.org/10.1029/2010jg001469>
- Fulé PZ, Ribas M, Gutiérrez E, Vallejo R, Kaye MW (2008) Forest structure and fire history in an old *Pinus nigra* forest, eastern Spain. *For Ecol Manag* 255(3–4):1234–1242. <https://doi.org/10.1016/j.foreco.2007.10.046>
- Gabriel E, Delgado-Dávila R, De Cáceres M, Casals P, Tudela A, Castro X (2021) Live fuel moisture content time series in Catalonia since 1998. *Ann for Sci* 78(2):44. <https://doi.org/10.1007/s13595-021-01057-0>
- Ganteaume A, Jappiot M, Curt T, Lampin C, Borgniet L (2014) Flammability of litter sampled according to two different methods: comparison of results in laboratory experiments. *Int J Wildland Fire* 23(8):1061–1075. <https://doi.org/10.1071/wf13045>
- Ganteaume A, Lampin-Maillet C, Guijarro M, Hernando C, Jappiot M, Fonturbel T, Pérez-Gorostiaga P, Vega JA (2009) Spot fires: fuel bed flammability and capability of firebrands to ignite fuel beds. *Int J Wildland Fire* 18(8):951–969. <https://doi.org/10.1071/wf07111>
- García-Hurtado E, Pey J, Baeza MJ, Carrara A, Llovet J, Querol X, Alastuey A, Vallejo VR (2013) Carbon emissions in Mediterranean shrubland wildfires: an experimental approach. *Atmos Environ* 69:86–93. <https://doi.org/10.1016/j.atmosenv.2012.11.063>
- González JR, Trasobares A, Palahí M, Pukkala T (2007) Predicting stand damage and tree survival in burned forests in Catalonia (North-East Spain). *Ann for Sci* 64(7):733–742. <https://doi.org/10.1051/forest:2007053>
- Gracia C (2001) (Ed) *Inventari Ecològic i Forestal de Catalunya, Regio forestal III*, CREA, Bellaterra, Spain
- Griffin D, Chen J, Anderson K, Makar P, McLinden CA, Dammers E, Fogal A (2023) Towards an improved understanding of wildfire CO emissions: a satellite remote-sensing perspective. *Egusphere* 2023:1–37. <https://doi.org/10.5194/egusphere-2023-649>
- Hessburg PF, Spies TA, Perry DA, Skinner CN, Taylor AH, Brown PM, Stephens SL, Larson AJ, Churchill DJ, Povak NA, Singleton PH, McComb B, Zielinski WJ, Collins BM, Salter RB, Keane JJ, Franklin JF, Riegel G (2016) Tamm review: management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. *For Ecol Manag* 366:221–250. <https://doi.org/10.1016/j.foreco.2016.01.034>
- Hoff V, Rowell E, Teske C, Queen L, Wallace T, Hoff V, Rowell E, Teske C, Queen L, Wallace T (2019) Assessing the relationship between forest structure and fire severity on the north rim of the Grand Canyon. *Fire* 2(1):10. <https://doi.org/10.3390/fire2010010>
- Jain TB, Graham RT (2007) The relation between tree burn severity and forest structure in the rocky mountains. *Gen Tech Rep PSW-GTR-203USDA For Serv Rocky Mt Res Stn*. https://www.fs.usda.gov/psw/publications/documents/psw_gtr203/psw_gtr203_017jain.pdf
- Jiménez E, Vega JA, Ruiz-González AD, Guijarro M, Alvarez-González JG, Madrigal J, Cuiñas P, Hernando C, Fernández-Alonso JM (2013) Carbon emissions and vertical pattern of canopy fuel consumption in three *Pinus pinaster* Ait. active crown fires in Galicia (NW Spain). *Ecol Eng* 54:202–209. <https://doi.org/10.1016/j.ecoleng.2013.01.039>
- Kane VR, McGaughey RJ, Bakker JD, Gersonde RF, Lutz JA, Franklin JF (2010) Comparisons between field- and LiDAR-based measures of stand structural complexity. *Can J for Res* 40(4):761–773. <https://doi.org/10.1139/x10-024>
- Keeley JE (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *Int J Wildland Fire* 18(1):116–126. <https://doi.org/10.1071/wf07049>
- Keeley JE, Syphard AD (2019) Twenty-first century California, USA, wildfires: fuel-dominated vs. wind-dominated fires. *Fire Ecol* 15(1):24. <https://doi.org/10.1186/s42408-019-0041-0>
- Kennedy MC, Prichard SJ, McKenzie D, French NHF (2020) Quantifying how sources of uncertainty in combustible biomass propagate to prediction of wildland fire emissions. *Int J Wildland Fire* 29(9):793–806. <https://doi.org/10.1071/wf19160>
- Keywood M, Kanakidou M, Stohl A, Dentener F, Grassi G, Meyer CP, Torseth K, Edwards D, Thompson AM, Lohmann U, Burrows J (2013) Fire in the air: biomass burning impacts in a changing climate. *Crit Rev Environ Sci Technol* 43(1):40–83. <https://doi.org/10.1080/10643389.2011.604248>
- Knorr W, Jiang L, Arneth A (2016) Climate, CO₂ and human population impacts on global wildfire emissions. *Biogeosciences* 13(1):267–282. <https://doi.org/10.5194/bg-13-267-2016>
- Kocbach Bølling A, Pagels J, Yttri KE, Barregard L, Sallsten G, Schwarze PE, Boman C (2009) Health effects of residential wood smoke particles: the importance of combustion conditions and physicochemical particle properties. *Part Fibre Toxicol* 6:29. <https://doi.org/10.1186/1743-8977-6-29>
- Labenski P, Ewald M, Schmidlein S, Heinsch FA, Fassnacht FE (2023) Quantifying surface fuels for fire modelling in temperate forests using airborne lidar and Sentinel-2: potential and limitations. *Remote Sens Environ* 295:113711. <https://doi.org/10.1016/j.rse.2023.113711>
- Lahaye S, Curt T, Fréjaville T, Sharples J, Paradis L, Hély C (2018) What are the drivers of dangerous fires in Mediterranean France. *Int J Wildland Fire* 27(3):155–163. <https://doi.org/10.1071/wf17087>
- Larkin NK, Raffuse SM, Strand TM (2014) Wildland fire emissions, carbon, and climate: U.S. emissions inventories. *For Ecol Manag* 317:61–69. <https://doi.org/10.1016/j.foreco.2013.09.012>
- Lasslop G, Kloster S (2015) Impact of fuel variability on wildfire emission estimates. *Atmos Environ* 121:93–102. <https://doi.org/10.1016/j.atmosenv.2015.05.040>
- Lazaridis M, Latos M, Aleksandropoulou V, Hov Ø, Papayannis A, Tørseth K (2008) Contribution of forest fire emissions to atmospheric pollution in Greece. *Air Qual Atmos Health* 1(3):143–158. <https://doi.org/10.1007/s11869-008-0020-0>
- Lecina-Díaz J, Alvarez A, Retana J (2014) Extreme fire severity patterns in topographic, convective and wind-driven historical wildfires of Mediterranean pine forests. *PLoS ONE* 9(1):e85127. <https://doi.org/10.1371/journal.pone.0085127>
- Lecina-Díaz J, Martínez-Vilalta J, Alvarez A, Vayreda J, Retana J (2021) Assessing the risk of losing forest ecosystem services due to wildfires. *Ecosystems* 24(7):1687–1701. <https://doi.org/10.1007/s10021-021-00611-1>
- Lecina-Díaz J, Senf C, Grünig M, Seidl R (2024) Ecosystem services at risk from disturbance in Europe's forests. *Glob Change Biol* 30(3):e17242. <https://doi.org/10.1111/gcb.17242>
- Liang YT, Stamatis C, Fortner EC, Wernis RA, Van Rooy P, Majluf F, Yacovitch TI, Daube C, Herndon SC, Kreisberg NM, Barsanti KC, Goldstein AH (2022) Emissions of organic compounds from western US wildfires and their near-fire transformations. *Atmos Chem Phys* 22(15):9877–9893. <https://doi.org/10.5194/acp-22-9877-2022>
- Lydersen JM, Collins BM, Ewell CM, Reiner AL, Fites JA, Dow CB, Gonzalez P, Saah DS, Battles JJ (2014) Using field data to assess model predictions of surface and ground fuel consumption by wildfire in coniferous forests of California. *J Geophys Res Biogeosci* 119(3):223–235. <https://doi.org/10.1002/2013JG002475>
- Ma YF, Yang S, Zhu ZP, Wang GY, Tigabu M, Guo YX, Zheng WX, Guo FT (2022) Emissions of gaseous pollutants released by

- forest fire in relation to litter fuel moisture content. *Atmos Environ* 284:119215. <https://doi.org/10.1016/j.atmosenv.2022.119215>
- MCSC (2010) <https://www.mcsc.creaf.cat/>
- Merrill DF, Alexander ME (1987) Glossary of forest fire management terms. Can Comm For Fire Manag Natl Res Counc Canada Ottawa
- Miranda AI (2004) An integrated numerical system to estimate air quality effects of forest fires. *Int J Wildl Fire* 13(2):217–226. <https://doi.org/10.1071/WF02047>
- Miranda AI, Ferreira J, Valente J, Santos P, Amorim JH, Borrego C (2005) Smoke measurements during Gestosa-2002 experimental field fires. *Int J Wildland Fire* 14(1):107–116. <https://doi.org/10.1071/wf04069>
- Miranda AI, Borrego C, Martins H, Martins V, Amorim JH, Valente J, Carvalho A (2009) Forest fire emissions and air pollution in southern Europe. In: *Earth observation of wildland fires in Mediterranean ecosystems*. Springer, pp 171–187. https://doi.org/10.1007/978-3-642-01754-4_12
- Mitsopoulos ID, Dimitrakopoulos AP (2007) Canopy fuel characteristics and potential crown fire behavior in Aleppo pine (*Pinus halepensis* Mill.) forests. *Ann for Sci* 64(3):287–299. <https://doi.org/10.1051/forest:2007006>
- Mitsopoulos ID, Dimitrakopoulos AP (2014) Estimation of canopy fuel characteristics of Aleppo pine (*Pinus halepensis* Mill.) forests in Greece based on common stand parameters. *Eur J Forest Res* 133(1):73–79. <https://doi.org/10.1007/s10342-013-0740-z>
- Molina JR, Herrera MA, Rodríguez y Silva F (2019) Wildfire-induced reduction in the carbon storage of Mediterranean ecosystems: an application to brush and forest fires impacts assessment. *Environ Impact Assess Rev* 76:88–97. <https://doi.org/10.1016/j.eiar.2019.02.008>
- Naeher LP, Brauer M, Lipsett M, Zelikoff JT, Simpson CD, Koenig JQ, Smith KR (2007) Woodsmoke health effects: a review. *Inhal Toxicol* 19(1):67–106. <https://doi.org/10.1080/08958370600985875>
- Nunes L, Álvarez-González J, Alberdi I, Silva V, Rocha M, Rego FC (2019) Analysis of the occurrence of wildfires in the Iberian Peninsula based on harmonised data from national forest inventories. *Ann for Sci* 76(1):27. <https://doi.org/10.1007/s13595-019-0811-5>
- Oliveras I, Gracia M, Moré G, Retana J (2009) Factors influencing the pattern of fire severities in a large wildfire under extreme meteorological conditions in the Mediterranean basin. *Int J Wildland Fire* 18(7):755–764. <https://doi.org/10.1071/wf08070>
- Ottmar RD, Miranda AI, Sandberg DV (2008) Chapter 3 characterizing sources of emissions from wildland fires. In: *Wildland fires and air pollution*. Elsevier, pp 61–78. [https://doi.org/10.1016/s1474-8177\(08\)00003-x](https://doi.org/10.1016/s1474-8177(08)00003-x)
- Pallozzi E, Lusini I, Cherubini L, Hajiaghayeva RA, Ciccioli P, Calafapietra C (2018) Differences between a deciduous and a conifer tree species in gaseous and particulate emissions from biomass burning. *Environ Pollut* 234:457–467. <https://doi.org/10.1016/j.envpol.2017.11.080>
- Pasalodos-Tato M, Ruiz-Peinado R, del Río M, Montero G (2015) Shrub biomass accumulation and growth rate models to quantify carbon stocks and fluxes for the Mediterranean region. *Eur J Forest Res* 134(3):537–553. <https://doi.org/10.1007/s10342-015-0870-6>
- Poupkou A, Markakis K, Liora N, Giannaros TM, Zanis P, Im U, Daskalakis N, Myriokefalitakis S, Kaiser JW, Melas D, Kanakidou M, Karacostas T, Zerefos C (2014) A modeling study of the impact of the 2007 Greek forest fires on the gaseous pollutant levels in the Eastern Mediterranean. *Atmos Res* 149:1–17. <https://doi.org/10.1016/j.atmosres.2014.05.015>
- Price OH, Nolan RH, Samson SA (2022) Fuel consumption rates in resprouting eucalypt forest during hazard reduction burns, cultural burns and wildfires. *For Ecol Manag* 505:119894. <https://doi.org/10.1016/j.foreco.2021.119894>
- Prichard SJ, O'Neill SM, Eagle P, Andreu AG, Drye B, Dubowy J, Urbanski S, Strand TM (2020a) Wildland fire emission factors in North America: synthesis of existing data, measurement needs and management applications. *Int J Wildland Fire* 29(2):132–147. <https://doi.org/10.1071/wf19066>
- Prichard SJ, Povak NA, Kennedy MC, Peterson DW (2020b) Fuel treatment effectiveness in the context of landform, vegetation, and large, wind-driven wildfires. *Ecol Appl* 30(5):e02104. <https://doi.org/10.1002/eap.2104>
- Prichard SJ, Rowell EM, Hudak AT, Keane RE, Loudermilk EL, Lutes DC, Ottmar RD, Chappell LM, Hall JA, Hornsby BS (2022) Fuels and consumption. In: *Wildland fire smoke in the United States*. Springer, pp 11–49. https://doi.org/10.1007/978-3-030-87045-4_2
- Ritter SM, Hoffman CM, Battaglia MA, Linn R, Mell WE (2023) Vertical and horizontal crown fuel continuity influences group-scale ignition and fuel consumption. *Fire* 6(8):321. <https://doi.org/10.3390/fire6080321>
- Rodrigues M, Cunill Camprubí À, Balaguer-Romano R, Coco Megía CJ, Castañares F, Ruffault J, Fernandes PM, de Resco Dios V (2023) Drivers and implications of the extreme 2022 wildfire season in Southwest Europe. *Sci Total Environ* 859:160320. <https://doi.org/10.1016/j.scitotenv.2022.160320>
- Rosa IMD, Pereira JMC, Tarantola S (2011) Atmospheric emissions from vegetation fires in Portugal (1990–2008): estimates, uncertainty analysis, and sensitivity analysis. *Atmos Chem Phys* 11(6):2625–2640. <https://doi.org/10.5194/acp-11-2625-2011>
- Ruffault J, Moron V, Trigo RM, Curt T (2017) Daily synoptic conditions associated with large fire occurrence in Mediterranean France: evidence for a wind-driven fire regime. *Int J Climatol* 37(1):524–533. <https://doi.org/10.1002/joc.4680>
- Ruffault J, Curt T, Martin-StPaul NK, Moron V, Trigo RM (2018) Extreme wildfire events are linked to global-change-type droughts in the northern Mediterranean. *Nat Hazards Earth Syst Sci* 18(3):847–856. <https://doi.org/10.5194/nhess-18-847-2018>
- Salana S, Yu HR, Dai ZY, Ganesh Subramanian PS, Puthussery JV, Wang YX, Singh A, Pope FD, Leiva GMA, Rastogi N, Tripathi SN, Weber RJ, Verma V (2024) Inter-continental variability in the relationship of oxidative potential and cytotoxicity with PM_{2.5} mass. *Nat Commun* 15(1):5263. <https://doi.org/10.1038/s41467-024-49649-4>
- Salis M, Arca B, Alcasena-Urdiroz F, Massaiu A, Bacciu V, Bosseur F, Caramelle P, Dettori S, de Fernans Oliveira AS, Molina-Terren D, Pellizzaro G, Santoni PA, Spano D, Vega-García C, Duce P (2019) Analyzing the recent dynamics of wildland fires in *Quercus suber* L. woodlands in Sardinia (Italy), Corsica (France) and Catalonia (Spain). *Eur J Forest Res* 138(3):415–431. <https://doi.org/10.1007/s10342-019-01179-1>
- San-Miguel-Ayaz J, Steinbrecher R (2019) EMEP/EEA air pollutant emission inventory guidebook 2019: Forest and other vegetation fires, European Environment Agency, Tech. <https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/11-natural-sources/11-b-forest-fires/view>
- San-Miguel-Ayaz J, Durrant T, Boca R, Maianti P, Liberta G, Jacome Felix Oom D, Branco A, De Rigo D, Suarez-Moreno M, Ferrari D, Roglia E, Scionti N, Broglia M, Onida M, Tistan A, Löffler P (2022) Forest Fires in Europe, Middle East and North Africa 2022
- Sánchez-Pinillos M, De Cáceres M, Casals P, Alvarez A, Beltrán M, Pausas JG, Vayreda J, Coll L (2021) Spatial and temporal variations of overstory and understory fuels in Mediterranean landscapes. *For Ecol Manage* 490:119094. <https://doi.org/10.1016/j.foreco.2021.119094>
- Schaffhauser A, Curt T, Tatoni T (2011) Fire-vegetation interplay in a mosaic structure of *Quercus suber* woodlands and Mediterranean

- maquis under recurrent fires. For Ecol Manage 262(5):730–738. <https://doi.org/10.1016/j.foreco.2011.05.003>
- Schaffhauser A, Pimont F, Curt T, Cassagne N, Dupuy JL, Tatoni T (2015) Effects of fire recurrence on fire behaviour in cork oak woodlands (*Quercus suber* L.) and Mediterranean shrublands over the last fifty years. CR Biol 338(12):812–824. <https://doi.org/10.1016/j.crvi.2015.10.001>
- Seiler W, Crutzen PJ (1980) Estimates of gross and net fluxes of carbon between the biosphere and the atmosphere from biomass burning. Clim Change 2(3):207–247. <https://doi.org/10.1007/BF00137988>
- Shakoor A, Farooq TH, Arif MS, Shahzad SM (2023) Unprecedented wildfires in Canada and transboundary effects of carbon monoxide pollution. Nat Hazards J Int Soc Prev Mitig Nat Hazards 118(3):2711–2713
- Silva TP, Pereira JMC, Paúl JCP, Santos MTN, Vasconcelo MJP (2006) Estimated atmospheric emissions originated from rural fires in Portugal. Silva Lusit 14(2):239–263
- Sommers WT, Loehman RA, Hardy CC (2014) Wildland fire emissions, carbon, and climate: science overview and knowledge needs. For Ecol Manage 317:1–8. <https://doi.org/10.1016/j.foreco.2013.12.014>
- Stocks BJ (1987a) Fire potential in the spruce budworm-damaged forests of Ontario. For Chron 63(1):8–14. <https://doi.org/10.5558/tfc63008-1>
- Stocks BJ (1987b) Fire behavior in immature jack pine. Can J for Res 17(1):80–86. <https://doi.org/10.1139/x87-014>
- Stocks BJ (1989) Fire behavior in mature jack pine. Can J for Res 19(6):783–790. <https://doi.org/10.1139/x89-119>
- Surawski NC, Sullivan AL, Roxburgh SH, Polglase PJ (2016) Estimates of greenhouse gas and black carbon emissions from a major Australian wildfire with high spatiotemporal resolution. J Geophys Res Atmos 121(16):9892–9907. <https://doi.org/10.1002/2016JD025087>
- Tedim F, Leone V, Amraoui M, Bouillon C, Coughlan M, Delogu G, Fernandes P, Ferreira C, McCaffrey S, McGee T, Parente J, Paton D, Pereira M, Ribeiro L, Viegas D, Xanthopoulos G (2018) Defining extreme wildfire events: difficulties, challenges, and impacts. Fire 1(1):9. <https://doi.org/10.3390/fire1010009>
- Urbanski SP (2013) Combustion efficiency and emission factors for wildfire-season fires in mixed conifer forests of the northern Rocky Mountains, US. Atmos Chem Phys 13(14):7241–7262. <https://doi.org/10.5194/acp-13-7241-2013>
- Urbanski S (2014) Wildland fire emissions, carbon, and climate: emission factors. For Ecol Manage 317:51–60. <https://doi.org/10.1016/j.foreco.2013.05.045>
- Urbanski SP, Long RW, Halliday H, Lincoln EN, Habel A, Landis MS (2022) Fuel layer specific pollutant emission factors for fire prone forest ecosystems of the western U.S. and Canada. Atmos Environ X 16:1–17. <https://doi.org/10.1016/j.aeaoa.2022.100188>
- Varner JM, Kane JM, Kreye JK, Engber E (2015) The flammability of forest and woodland litter: a synthesis. Curr Rep 1(2):91–99. <https://doi.org/10.1007/s40725-015-0012-x>
- van der Werf GR, Randerson JT, Giglio L, van Leeuwen TT, Chen Y, Rogers BM, Mu MQ, van Marle MJE, Morton DC, Collatz GJ, Yokelson RJ, Kasibhatla PS (2017) Global fire emissions estimates during 1997–2016. Earth Syst Sci Data 9(2):697–720. <https://doi.org/10.5194/essd-9-697-2017>
- van Wees D, van der Werf GR, Randerson JT, Rogers BM, Chen Y, Veraverbeke S, Giglio L, Morton DC (2022) Global biomass burning fuel consumption and emissions at 500 m spatial resolution based on the Global Fire Emissions Database (GFED). Geosci Model Dev 15(22):8411–8437. <https://doi.org/10.5194/gmd-15-8411-2022>
- Vayreda J, Martínez-Vilalta J, Vilà-Cabrera A (2016) The ecological forest inventory of Catalonia: a tool for functional ecology. Ecosistemas 25(3):70–79. <https://doi.org/10.7818/ecos.2016.25-3.08>
- Vicente-Serrano SM, Beguería S, López-Moreno JI (2010) A multiscalar drought index sensitive to global warming: the standardized precipitation evapotranspiration index. J Clim 23(7):1696–1718. <https://doi.org/10.1175/2009JCLI2909.1>
- Viedma O, Chico F, Fernández JJ, Madrigal C, Safford HD, Moreno JM (2020) Disentangling the role of prefire vegetation vs. burning conditions on fire severity in a large forest fire in SE Spain. Remote Sens Environ 247:111891. <https://doi.org/10.1016/j.rse.2020.111891>
- Vilén T, Fernandes PM (2011) Forest fires in Mediterranean countries: CO₂ emissions and mitigation possibilities through prescribed burning. Environ Manage 48(3):558–567. <https://doi.org/10.1007/s00267-011-9681-9>
- Wiedinmyer C, Quayle B, Geron C, Belote A, McKenzie D, Zhang XY, O'Neill S, Wynne KK (2006) Estimating emissions from fires in North America for air quality modeling. Atmos Environ 40(19):3419–3432. <https://doi.org/10.1016/j.atmosenv.2006.02.010>
- Wiedinmyer C, Kimura Y, McDonald-Buller EC, Emmons LK, Buchholz RR, Tang WF, Seto K, Joseph MB, Barsanti KC, Carlton AG, Yokelson R (2023) The fire inventory from NCAR version 2.5: an updated global fire emissions model for climate and chemistry applications. Geosci Model Dev 16(13):3873–3891. <https://doi.org/10.5194/gmd-16-3873-2023>
- Xanthopoulos G, Calfapietra C, Fernandes P (2011) Fire hazard and flammability of European forest types. In: Post-fire management and restoration of southern European forests. Springer, pp 79–92. https://doi.org/10.1007/978-94-007-2208-8_4
- Yilmaz OS, Acar U, Sanli FB, Gulgen F, Ates AM (2023) Mapping burn severity and monitoring CO content in Türkiye's 2021 wildfires, using Sentinel-2 and Sentinel-5P satellite data on the GEE platform. Earth Sci Inform 16(1):221–240. <https://doi.org/10.1007/s12145-023-00933-9>
- Zheng B, Ciais P, Chevallier F, Chuvieco E, Chen Y, Yang H (2021) Increasing forest fire emissions despite the decline in global burned area. Sci Adv 7(39):eabh2646. <https://doi.org/10.1126/sciadv.abh2646>

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.