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Fine-tuning the BFOLDS Fire Regime Module to support the assessment of fire-related functions and services in a changing Mediterranean mountain landscape

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ABSTRACT

Fire simulation models are useful to advance fire research and improve landscape management. However, a better understanding of these tools is crucial to increase their reliability and expansion into research fields where their application remains limited (e.g., ecosystem services). We evaluated several components of the BFOLDS Fire Regime Module and then tested its ability to simulate fire regime attributes in a Mediterranean mountainous landscape. Based on model outputs, we assessed the landscape fire regulation capacity over time and its implications for supporting the climate regulation ecosystem service. We found that input data quality and the adjustment of fuel and fire behaviour parameters are crucial to accurately emulating key fire regime attributes. Besides, the high predictive capacity shown by BFOLDS-FRM allows to reliably inform the planning and sustainable management of fire-prone mountainous areas of the Mediterranean. Moreover, we identified and discussed modelling limitations and made recommendations to improve future model applications.

1. Introduction

Modelling of landscape disturbances, such as fire, has expanded over the last decades in forest ecology studies (Albrich et al., 2020; Seidl et al., 2011) because of the increasing ability of models to capture complex phenomena and of the advances in computational capacity to run models that adequately simulate those phenomena (Perera et al., 2015). The growing interest in landscape scale processes for management and planning (Turner and Gardner, 2015) and the potential application of modelling tools to support decision-making (Seidl et al., 2011) further increased the relevance of landscape studies using a simulation modelling approach. In addition, real world field experiments over large areas are costly in time and money, and often unfeasible (Turner and Gardner, 2015).

Simulation models are mathematical simplifications of an ecological system (Perera et al., 2015) whose implementation as a computer program seeks either to solve complex relationships or to describe and understand patterns of behaviour of a target system (Durán, 2021). Their ability to test a wide range of conditions makes simulation models suitable to study complex (socio-) ecological phenomena such as wildfires and their impacts on real landscapes across space and time (He, 2008). Landscape and ecological modelling has been applied to the development of tools (Synes et al., 2016) that can be used to simulate fire disturbance processes on a broad range of spatial and temporal scales, and levels of complexity (Dai et al., 2015; Keane et al., 2004). Simulation modelling tools addressing fire can be typified into three major categories (Herawati et al., 2015): (i) landscape fire behaviour models, e.g. FlamMap (Finney, 2006); (ii) integrated fire-vegetation

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models, e.g. FSim (Finney et al., 2011); and (iii) dynamic global vegetation, e.g. LPJ-DGVM (Sitch et al., 2003) and landscape models, e.g. LANDIS-II (Scheller et al., 2007).

The application of a simulation approach to model landscape-level disturbances is a useful way to predict and forecast wildfire events, estimating their occurrence and behaviour to support fire management, planning and fighting operations (Pacheco et al., 2015). It also allows to explore and simulate 'what-if' scenarios to assess long-term patterns, e.g. fire regimes (Keane et al., 2004) or the ecological impacts of fires (Pais et al., 2020). Notwithstanding the usefulness of these tools in supporting fire-related research, limitations in their application may arise, often derived from incomplete understanding and knowledge about the modelled processes (Perera et al., 2015). Besides, modellers have to handle difficulties related to limitations in the quality of data resources (e.g., spatial, temporal or thematic resolution, spatial or temporal extent) needed to calibrate and validate models (Perera et al., 2015), uncertainties about model parameters or the accuracy of model predictions (Alexander and Cruz, 2013), or regional model extrapolation (Seidl et al., 2011).

In this regard, comprehensive model evaluation methods are available (Cruz et al., 2003b; Jorgensen and Fath, 2011) to better understand model behaviour and provide information to identify and adjust key inputs, increasing accuracy and reliability, which is crucial when models are used to support fire management and decision-making (Riley and Thompson, 2016). Some examples of model evaluation can be found in the scientific literature addressing fire modelling, focused on model sensitivity (Cary et al., 2006; Hummel et al., 2013; Sturtevant et al., 2009), uncertainty analysis (Benali et al., 2016; Pinto et al., 2016), or scenario analysis (Perera and Cui, 2010; Perera et al., 2009). Such approaches allow for a comprehensive assessment of these modelling tools, which may partially overcome difficulties in their application (Perera et al., 2015).

Despite the widespread use of fire simulation tools in multiple fire-related fields, their application to characterize fire-related ecosystem functions, services and disservices remains limited (Baskent, 2020; Sil et al., 2019b). Still, the identification of ecosystem attributes that promote their fire regulation capacity, i.e., the ability of ecosystems and landscapes to regulate spatiotemporal attributes of fire regimes through the control of factors affecting fire behaviour resulting from the interaction between ecosystem processes (fire) and biophysical structures (vegetation types and spatial patterns) (Depietri and Orenstein, 2019; Haines-Young and Potschin, 2018; Pettorelli et al., 2018; Sil et al., 2019a, 2019b), has been fostering the application of fire modelling tools to assess fire-related services at the landscape scale, such as the fire protection ecosystem service (Sil et al., 2019b), or the impact of fire on the climate regulation ecosystem service (Pais et al., 2020).

In this study, we aimed to evaluate and apply the Boreal Forest Landscape Dynamics Simulator – Fire Regime Module (BFOLDS-FRM) (Ouellette et al., 2020; Perera et al., 2014) in a dynamic mountain landscape under Mediterranean type of climate in Europe. BFOLDS-FRM is a spatially explicit process-based model that simulates fire growth, implemented in the LANDIS-II ecosystem modelling framework (Scheller et al., 2007). BFOLDS-FRM was designed primarily to simulate fire regimes in boreal forests and has been used to study fire regimes and to support forest management and policymaking (Perera and Cui, 2010; Rempel et al., 2007). However, its application and evaluation in other biogeographic and landscape settings is scarce.

This study builds on experience from a previous application of BFOLDS-FRM (Sil et al., 2019b) and seeks to better understand BFOLDS-FRM behaviour and its ability to simulate fire regimes in a fire-prone Mediterranean mountain landscape, and to support the assessment of its fire regulation capacity (FRC) and the potential provision of the climate regulation ecosystem service (CRES) through the carbon stocks balance. We addressed the following research questions: (i) how sensitive are the model outputs to changes in input data quality attributes and user assumptions?; (ii) do model parameters need adjustments to emulate real-world conditions in the study area?; (iii) does BFOLDS-FRM accurately predict observed fire patterns in the study area?; (iv) to what extent do BFOLDS-FRM outcomes support the assessment of FRC and its impact on CRES in the study area?; (v) what are the potential strengths and limitations of the model and how can its application be improved? This study contributes to identify relevant aspects of the BFOLDS-FRM model for broader applications, particularly in the context of fire regimes in mountain regions under a Mediterranean type of climate, as well as to support the application of the fire regulation concept in the management of fire-prone landscapes.

2. Methods

2.1. General approach

We evaluated the Boreal Forest Landscape Dynamics Simulator – Fire Regime Module (BFOLDS-FRM) in terms of its behaviour and capabilities to explore and predict fire regime attributes in the context of fire-prone mountain landscapes in the Mediterranean region. We run BFOLDS-FRM as a stand-alone module, using the Sabor River upper basin (NE Portugal) as the focal area to carry out model sensitivity analysis, to support calibration and validation. Besides, we applied BFOLDS-FRM according to an ecosystem services-based conceptual framework to assess fire regulation capacity (FRC) and the supply of the climate regulation ecosystem service (CRES) in the study area (Fig. 1).

A framework was developed to evaluate BFOLDS-FRM based on the available standardized ecological modelling guidelines for developing and evaluating simulation models (Cruz et al., 2003b; Jorgensen and Fath, 2011; Swannack et al., 2012; Waveren et al., 1999). Four main steps were taken in the process 1) baseline simulations to define model reference behaviour for subsequent model evaluation steps; 2) assessment of model outputs sensitivity to changes in inputs and parameters to better understand their relative effects on model outcomes; 3) adjustment of parameters to improve the correspondence between model behaviour and expected fire patterns (calibration); and 4) evaluation of its predictive accuracy when applied to a set of observed data (validation). We then used BFOLDS-FRM to assess shifts in fire-related ecosystem functions, i.e., FRC, and its impact on services, i.e., CRES, in the focal area. We used the simulated fire regime attributes 'burned area' and 'fire intensity' to characterize FRC, and combined fire modelling with carbon storage modelling to assess CRES supply in the study area.

2.2. Context of the application

The European Mediterranean basin accounts for most of the annual burned area (94%) and fire ignitions (74%) in the EU (San-Miguel-Ayanz et al., 2019). Fire in the Mediterranean basin has historically been human-driven due to intensive land use and continuous use of fire in rural activities (e.g. for pasture improvement), which shaped landscapes, ecosystems, and ultimately fire regimes (Keeley et al., 2012). The fire regimes (or pyromes) associated with Mediterranean vegetation can be typified as high-intensity large fires and low-intensity small fires with short fire seasons (Archibald et al., 2013). The high fire hazard of Mediterranean regions is partially a consequence of favourable conditions for vegetation growth during the rainy season followed by warm dry summers (Keeley et al., 2012), which is particularly relevant to the fire regime in Mediterranean mountains (Fréjaville and Curt, 2015). In addition, socio-economic factors particularly relevant in mountainous areas (Lasanta et al., 2017), and contemporary land management and fire-suppression policies across the Mediterranean (Rigolot et al., 2009) further increased landscape homogenization, and fuel accumulation and connectivity (Moreira et al., 2011).

The study was conducted in the Sabor River upper basin, a fire-prone mountain area located in northeast Portugal, at the southwestern end of the Cantabrian mountain range (Körner et al., 2016) (Fig. 2). The area is

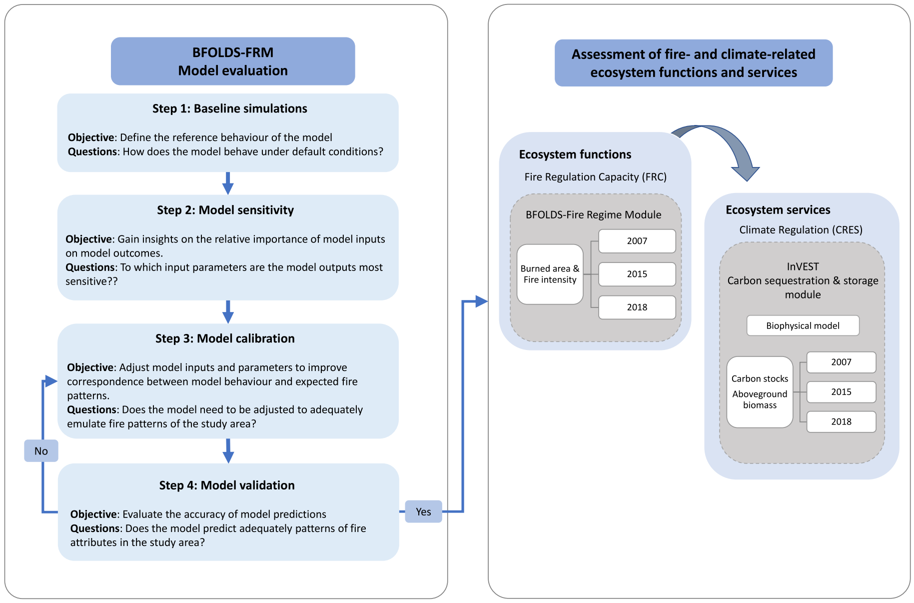


Fig. 1. Diagram of the modelling workflow applied to test BFOLDS-FRM and to assess ecological functions and services related to fire and climate.

approximately 30650 ha and elevation ranges from 484 to 1487 m (Fig. 2). Climate is Mediterranean (Beck et al., 2018) with average annual precipitation ranging from 806 to 1262 mm, and average annual temperature ranging from 8.5 to 12.8 °C (Sil et al., 2017). Seminatural habitats (*Cistus* spp., *Cytisus* spp. and *Erica* spp.) dominate the landscape, although the areas of native forests (*Quercus pyrenaica* and *Q. rotundifolia*), forest stands of maritime pine (*Pinus pinaster*), and agroforestry systems (*Castanea sativa*) have been increasing over the last decades (Sil et al., 2016). On the other hand, demographic and socioeconomic factors contributed to farmland abandonment and the conversion of these areas over time, which modified landscape composition and configuration and favoured more hazardous fuels and fuel continuity (Azevedo et al., 2011), which potentially decreased the fire regulating capacity over time (Sil et al., 2019b).

2.3. BFOLDS-FRM model description

Boreal Forest Landscape Dynamics Simulator – Fire Regime Module (BFOLDS-FRM) simulates fire mechanistically, i.e., computing fire on the landscape at hourly time steps, from fire weather, fuel type, and slope inputs, providing spatially explicit information about the location of the ignition, the pixels burned in each simulated fire event, and the intensity at which each pixel burned. BFOLDS-FRM uses the CFFWIS - Canadian Forest Fire Weather Index System (Van Wagner and Pickett, 1985) to describe fuel moisture conditions and provide weather-related inputs to fire simulation. The weather parameters (wind speed and wind direction) and the CFFWIS components computed from consecutive sequences of daily weather conditions measured at noon (temperature, relative humidity, and wind speed) and the 24-h cumulative rainfall, are supplied to BFOLDS-FRM to interpolate weather data across the simulated area. The CFFWIS indexes are numeric ratings for the moisture content of litter and other dead fine fuels (FFMC), the moisture content of loosely compacted organic layers of moderate depth (DMC), the amount of fuel available for combustion (BUI). Each land cover category is assigned to one of the 16 fuel types described in the Canadian Forest Fire Behaviour Prediction (FBP) system (Forestry Canada Fire Danger Group, 1992) to predict fire behaviour in BFOLDS-FRM. Standard fuel types in BFOLDS-FRM are partially user-modifiable, allowing incorporation into the simulations of vegetation type features for a given area or region.

Fire ignition information is supplied daily and its placement on the landscape is spatially explicit, but whether it spreads or not is determined by the combination of daily fire weather, fuel type at the location point, and a user-defined DMC threshold. Fire ignition location is seeded randomly, either weighted or unweighted. The weighted mode allows the user to define the exact location of the ignition to start a fire event or to supply a surface that spatially weights the probability of ignition on the landscape, while the unweighted ignition seeding is spatially random. Fire extinguishment is caused by absence of burnable cells adjoining the fire perimeter, weather conditions below the DMC threshold for fire spread, or when the fire season ends, meaning that no fire management activities are implemented and used to control the process.

2.4. Model evaluation

2.4.1. Baseline simulations

Baseline simulations comprised the application of BFOLDS-FRM to the Sabor River upper basin taking 2007 as the reference year (Appendix A and Appendix B). Overall, model parameterization included the reclassification of land use and land cover (LULC) spatial data for 2007 (DGT, 2019) as burnable (i.e., natural grasslands, agroforestry systems, broadleaved forests, coniferous forests, mixed forests, shrublands) and non-burnable (i.e., urban areas, agricultural areas, open areas with either scarce or no vegetation, and water bodies). Then, burnable land covers were assigned to five standard fuel types from the Canadian Fire Behaviour Prediction system (Forestry Canada Fire Danger Group, 1992; Wotton et al., 2009): C6 – Conifer plantation, for the coniferous forests and the conifer dominated mixed forest; M2 – Boreal Mixewood-Green 25% conifers, for the broadleaved forest and the broadleaved dominated mixed forest; M2 – Boreal Mixewood-Green 50% conifers, for the broadleaved and the coniferous mixed forest; O1 – Grass, for grasslands and shrublands; and O1b – Grass, for agroforestry systems. Weather data used in baseline simulations were retrieved from the Bragança weather station for the year 2007. A complete fire season (i.e., 1 year) containing noon daily records was used to compute the previously mentioned Canadian Forest Fire Weather Index (FWI) System codes. Information on daily fire ignitions for the year 2007 was retrieved from the Portuguese Fire Database (ICNF, 2021) and used to feed the BFOLDS-FRM model. Data entries registered as false alarms, as well as data entries for ignitions that originated fires smaller than 0.06 ha were excluded.

For model evaluation, we constrained the model to simulate one fire

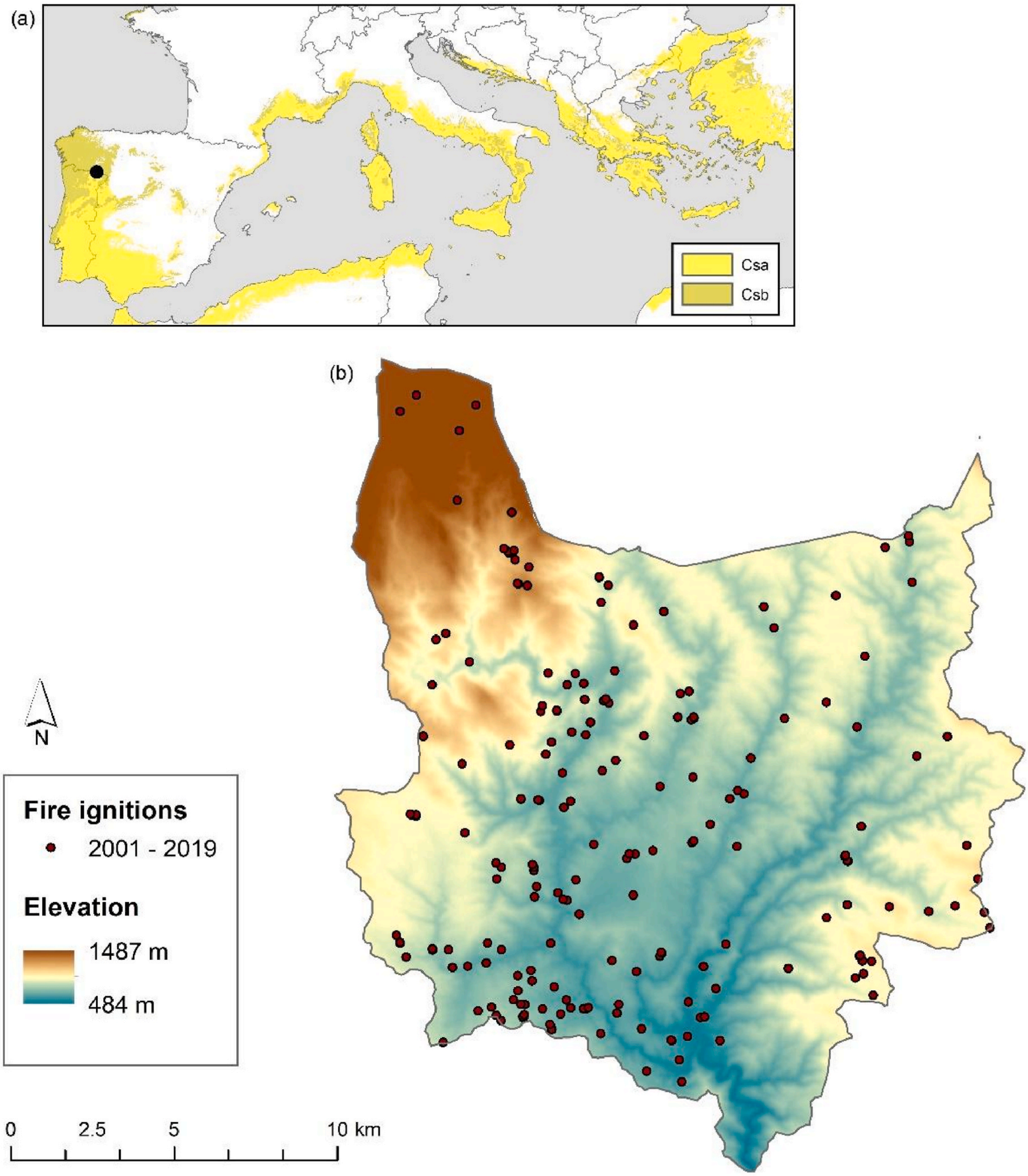


Fig. 2. (a) Location of the Sabor River upper basin in the context of the Mediterranean basin, including also the distribution of the Csa (Temperate, dry and hot summer) and Csb (Temperate, dry and warm summer) climate-types in southern Europe (Beck et al., 2018); (b) digital elevation model of the study area and location of fire ignitions recorded between 2001 and 2019 (ICNF, 2021).

season at a time (i.e., time step = 1) using the ignition and fire weather data of the correspondent year, while the random number seeds parameter of the LANDIS-II core model was kept constant, thus preventing the model from determining ignition success in a fully stochastic manner (Appendix B). As such, we run the model one time (one simulation) assuming that the results represent the model reference behaviour under the defined set of conditions.

2.4.2. Model sensitivity assessment

The One-At-a-Time (OAT) sensitivity analysis method was applied varying one input factor at a time, while keeping all other constant (Pianosi et al., 2016). BFOLDS-FRM, previously parameterized in the baseline simulation step, was run (n = 31) to assess model sensitivity to variation in the quality of spatial data (i.e., spatial resolution: 25 m vs. 100 and 300 m cell size; and thematic resolution: 1 ha vs. 25 ha Minimum Map Unit) and fire weather data (i.e., local weather station vs. ERA5-Land database), as well as in the user assumptions regarding the spatial distribution of fire ignitions (i.e., unweighted vs. weighted fire ignition seed probability surface), the spatialization of the fire weather data (i.e., weather point data vs. splined weather surface), land cover to fuel type conversion (i.e., standard vs. custom fuel types), and model built-in parameters (i.e., ignition and fire spread DMC thresholds; smouldering fire interval; crown base height of forest fuel types; and fuel load and curing of grass fuel types) (Appendix C). The outputs analysed (annual number of fires, annual mean fire size, and annual mean fire intensity - Appendix C) were compared against the outputs of the baseline simulation.

We applied a statistical analysis and an index-based analysis to evaluate model sensitivity. The statistical approach evaluated the sensitivity of model outputs, i.e., annual mean fire size and annual mean fire intensity, to changes in the spatial and non-spatial components of the model. The non-parametric Wilcoxon-Mann-Whitney test (Neuhäuser, 2011) was applied to test for difference in medians between baseline and sensitivity simulations outputs, and the stochastic superiority measure (A) (Vargha and Delaney, 2000), a nonparametric statistic of effect size that informs about the magnitude of the differences between outputs from tested (t) and baseline (d) simulations by computing the probability that a random observation of *t* is higher than a random observation of *d*. The A-measure statistic was ranked in four classes (Vargha and Delaney, 2000): No difference ($0.44< A <0.56$), Small ($0.36< A \leq 0.44$ or $0.56\leq A <0.64$), Medium ($0.29< A \leq 0.36$ or $0.64\leq A <0.71$), Large ($A \leq 0.29$ or $A \geq 0.71$). All statistical analyses were conducted using the software R (R Core Team, 2020). The sensitivity index approach was applied to evaluate the sensitivity of the model outputs, i.e., annual number of fires, annual mean fire size, and annual mean fire intensity, to changes in the model built-in parameters. A dimensionless sensitivity index (I) (Lenhart et al., 2002) was computed through variation of the parameter range by ±20% and ±40% from their default value tested in the baseline simulation step. The sensitivity of each parameter was assessed by ranking the sensitivity index in four classes (Lenhart et al., 2002): Class I - Small to negligible ($0.00 \leq I < 0.05$), Class II - Medium ($0.05 \leq I < 0.20$), Class III - High ($0.20 \leq I < 1.00$), and Class IV - Very high ($I \geq 1.00$).

2.4.3. Model calibration

We carried out the calibration procedure by manually adjusting model parameters used in the baseline simulation (Appendix D) either one parameter at a time or multiple parameters at once (n = 318). The calibration was informed by the results of the sensitivity analysis in a trial-and-error exercise (Jorgensen and Fath, 2011; Waveren et al., 1999). An evaluation of calibration results was applied to minimize the error (differences) between simulated and observed data by comparing outputs from the calibration exercise against fire regime attributes (Krebs et al., 2010) of annual number of fires, annual total burned area, and annual mean fire size observed in the Sabor River upper basin for 2007, using these as reference criteria (Appendix D). For that we calculated the relative error of simulated data in the study area for the selected attributes of the fire regime using Equations (1)–(3):

$$RE_{NF} = [(NF_{sim} - NF_{obs})/NF_{obs}] \times 100 \qquad \text{(Equation 1)}$$

$$RE_{BA} = [(BA_{sim} - BA_{obs})/BA_{obs}] \times 100 \qquad \text{(Equation 2)}$$

$$RE_{MFS} = [(MFS_{sim} - MFS_{obs})/MFS_{obs}] \times 100 \qquad \text{(Equation 3)}$$

where RE_{NF} is the relative error for the annual number of fires, RE_{BA} is the relative error for the annual total burned area, and RE_{MFS} is the relative error for the annual mean fire size, NF_{sim}, BA_{sim}, and MFS_{sim} are the simulated annual number of fires, total burned area, and mean fire size, respectively, for year 2007, and NF_{obs}, BA_{obs}, and MFS_{obs} are the observed annual number of fires, burned area, and mean fire size, respectively, for the same year. Simulations with different parameter settings whose outputs showed a relative error ranging between ±35% were assumed as acceptable for model prediction (Cruz and Alexander, 2013), and selected for the following model evaluation step.

2.4.4. Model validation

We carried out model validation by applying BFOLDS-FRM to the Sabor River upper basin to years 2015 and 2018 (Appendix E). We used the Portuguese Fire Database (ICNF, 2021) to retrieve observed data on the annual number of fires and the annual burned area for the Sabor River upper basin in these years. Based on results of model calibration tests in 2007, we identified different model parameters settings (n = 13) whose outputs resulted in a relative error within ±35% and applied them to the 2015 and 2018 (intermediate simulations). Therefore, we run BFOLDS-FRM (n = 1) independently for each year analysed and model settings (Appendix E). Then, we compared three selected attributes of the fire regime in the study area for 2015 and 2018 against simulated outputs (Appendix E). We evaluated model accuracy by applying Equations (1)–(3) to quantify model deviation from observed data, assuming as acceptable an error of up to ±35% in model predictions (Cruz and Alexander, 2013). Finally, from the intermediate simulations, we kept the model parameter configuration that generated the outputs with the lowest relative error.

2.5. Assessment of the fire regulation capacity and of the climate regulation ecosystem service

We modelled FRC based on five fire regime attributes (proxies) derived from simulations with BFOLDS-FRM: total annual burned area, annual mean fire size, annual mean fire intensity and number of fires with size above 100 ha and mean fire intensity above 500 kW m^{-1} in years 2007 (reference year), 2015, and 2018. We used two main criteria to assess FRC: (1) landscape capacity to regulate overall fire regime attributes and (2) landscape capacity to regulate large and potentially severe fires, i.e., fires larger than 100 ha, the official size threshold for large fires in Portugal (ICNF, 2019), and mean fire intensity above 500 kW m^{-1}, which is generally accepted as the limit for direct fire control by firefighting crews using hand tools (Hirsch and Martell, 1996) and the threshold for tree injury by fire (Van Wagner, 1973). We assumed that the Sabor River upper basin landscape was able to regulate the spatiotemporal attributes of fire if (1) the total annual burned area, the annual mean fire size, and the mean fire intensity decreased, compared to the year 2007, and/or (2) the number of fires larger than 100 ha and the mean fire intensity above 500 KW m^{-1} remained at 2007 levels.

Besides, we investigated the effect of FRC dynamics on the supply of the climate regulation ecosystem service. CRES is the avoided release of carbon to the atmosphere resulting from the capacity of terrestrial ecosystems to act as carbon sinks (Haines-Young and Potschin, 2018; Keith et al., 2021; Sil et al., 2017) and was modelled by applying the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) carbon storage and sequestration module (Sharp et al., 2020) to the Sabor River upper basin in 2007, 2015, and 2018 using data from previous work in the area (Sil et al., 2017) (Appendix F). We assumed the carbon stored in aboveground biomass (AGB) as a proxy for the climate regulation ES supply (Haines-Young and Potschin, 2018; Keith et al., 2021). Then, we used the total annual burned area simulated with BFOLDS-FRM in each year to estimate the potential losses of carbon stored in the landscape, assuming that all carbon stored was released into the atmosphere. Therefore, we expressed the effect of FRC on CRES as the difference between the estimated maximum potential supply of the CRES and its potential loss resulting from simulated fires in the study area. We used 2007 as the reference year from which we compared the corresponding carbon losses due to fire.

Additionally, to improve results interpretation, we carried out an ancillary analysis on land use and land cover changes for the study area, and the distribution of the number of days per fire weather index (FWI) class for each of the years analysed (Appendix G).

3. Results

3.1. BFOLDS-FRM model evaluation

3.1.1. Sensitivity analysis

Statistical analysis showed that the distribution of BFOLDS-FRM outputs, when tested for sensitivity to changes in the quality and assumptions of spatial, fuel type and weather inputs, only differed significantly for changes to spatial grid resolution (cell size from 25 to 300 m) and thematic resolution (1–25 ha) (Table 1 and Appendix C).

Table 1

Sensitivity assessment of BFOLDS-FRM variables (annual number of fires, annual mean fire size, and annual mean fire intensity) for model inputs and parameters based on statistical analysis and a sensitivity index-based analysis. Lenhard's sensitivity index (LSI) classes: Class I – Small to negligible ($0.00 \leq I < 0.05$), Class II – Medium ($0.05 \leq I < 0.20$), Class III – High ($0.20 \leq I < 1.00$), Class IV – Very high ($I \geq 1.00$); Vargha-Delaney A-Measure of Stochastic Superiority (VD-A) to assess the magnitude of the effect size: No difference (nd): $0.44 < A < 0.56$, Small: $0.36 < A \leq 0.44$ or $0.56 \leq A < 0.64$, Medium: $0.29 < A \leq 0.36$ or $0.64 \leq A < 0.71$, Large: $A \leq 0.29$ or $A \geq 0.71$; Wilcoxon-Mann-Whitney (WMW) test for levels of significance: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$; na = not applicable.

Model component	Target model input/parameter	Tested values	Model outputs: Annual nr. fires	Annual mean fire size			Annual mean fire intensity		
			LSI	LSI	VD-A	WMW	LSI	VD-A	WMW
Spatial input	Initial communities map Slope layer	25m vs. 100 m cell size	na	na	nd	ns	na	Small	ns
	Aspectlayer Ecozone layer	25m vs. 300 m cell size	na	na	Small	*	na	Large	***
Spatial input	Initial communities map	1ha vs. 25 ha Minimum Map Unit	na	na	Small	ns	na	Medium	*
Spatial input	Spatial fire ignitions seed	Unweighted vs. Weighted fire ignition seed probability surface	na	na	nd	ns	na	nd	ns
Weather input	Fire weather data	Local Weather Station vs. ERA5-Land database	na	na	Small	ns	na	nd	ns
Weather input	Fire weather data spatialization	Weather point data vs. Splined weather surface	na	na	Small	ns	na	Small	ns
Fuel type input	Fuel types	Standard vs. Custom fuel types	na	na	nd	ns	na	nd	ns
Built-in parameter	Ignition DMC threshold	20%	III	III	nd	ns	III	nd	ns
		−20%			nd	ns		nd	ns
		40%	III	III	nd	ns	II	nd	ns
		−40%			Small	ns		Small	ns
Built-in parameter	Spread DMC threshold	20%	I	I	nd	ns	III	nd	ns
		−20%			nd	ns		nd	ns
		40%	III	III	nd	ns	II	nd	ns
		−40%			nd	ns		nd	ns
Built-in parameter	Smoulder interval (hours)	20%	III	III	nd	ns	III	nd	ns
		−20%			nd	ns		Small	ns
		40%	I	I	nd	ns	I	nd	ns
		−40%			nd	ns		nd	ns
Built-in parameter	Forest fuel types: crown base height (m)	20%	I	I	nd	ns	III	nd	ns
		−20%			nd	ns		nd	ns
		40%	I	II	nd	ns	II	nd	ns
		−40%			nd	ns		nd	ns
Built-in parameter	Grass fuel types: fuel load (kg m^{-2})	20%	I	I	nd	ns	III	nd	ns
		−20%			nd	ns		Small	ns
		40%	I	I	nd	ns	III	Small	ns
		−40%			nd	ns		Small	ns
Built-in parameter	Grass fuel types: fuel curing (%)	20%	III	IV	nd	ns	III	nd	ns
		−20%			Medium	*		Small	ns
		40%	III	IV	nd	ns	IV	Small	ns
		−40%			Large	****		Large	****

Still, all these inputs had meaningful effects on the annual mean fire size and mean fire intensity, showing differences that ranged from small to large, except when tested for changes to *Fire weather data spatialization* and *Fuel types* parameters (Table 1 and Appendix C).

Regarding the relative influence of the tested built-in parameters on output variables, significant differences only were found when varied *Fuel curing* parameters (decreased by 40% and 20%), showing a small to large effect on both the annual mean fire size and mean fire intensity outputs. Also, both the *Ignition DMC threshold* and *Smoulder interval* parameters showed a small effect on model outputs (Table 1 and Appendix C).

Nevertheless, sensitivity index-based analysis indicates that all output variables were sensitive to changes in *Spread DMC threshold* and *Smoulder* interval parameters. Also, all output variables showed very high sensitivity to changes in *Ignition DMC threshold* parameter and *Fuel curing* parameters (Table 1 and Appendix C). In turn, changes made to *Crown base height* and *Fuel load* parameters showed a more pronounced sensitivity for the annual mean fire intensity output than for the annual number of fires and mean fire size (Table 1 and Appendix C).

3.1.2. Model calibration

Results from the baseline simulation indicate that the fire regime attributes simulated with BFOLDS-FRM deviated substantially from the data observed in the study area in 2007, overestimating total burned area and mean fire size and underestimating the number of fires (Table 2). Conversely, the estimated relative error (RE) of the fire regime attributes simulated after model calibration (Fig. 3, Appendix D) became within the range of ±35%, indicating satisfactory model predictions (Table 2).

Table 2

Observed and simulated fire regime attributes in baseline and calibration steps for the Sabor River upper basin in 2007, and respective relative errors (RE) of BFOLDS-FRM simulations.

Fire regime attribute	Observed	Simulated: Baseline	RE (%)	Calibration	RE (%)
Number of fires	52	34	−34.62	45	−13.46
Total burned area (ha)	277.30	15526.81	5499.26	269.31	−2.88
Mean fire size (ha)	5.33	456.67	8463.58	5.98	12.23

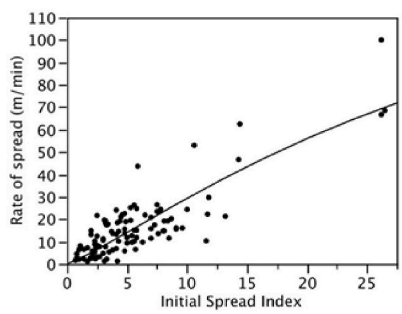
3.1.3. Model validation

Overall, BFOLDS-FRM was able to emulate observed trends of selected attributes of fire regime for the three years analysed with an estimated mean relative error within the ±35% interval considered as reasonable for model predictions in the validation step (Table 3). On average, BFOLDS-FRM overpredicted annual number of fires and annual total burned area, and underpredicted annual mean fire size (Table 3). Despite the satisfactory overall model performance, the area burned in 2018 was overpredicted with a relative error outside the range accepted

(a)

Inputs and parameters	Initial	Modified
Spatial fire ignitions seed	Unweighted mode	Weighted mode
Fuel types	Standard FBP fuel types: C6 – Conifer plantation M2 – Boreal Mixed wood (25%) M2 – Boreal Mixed wood (50%) O1a - Matted grass O1b - Standing grass	Standard FBP fuel types: C6 – Conifer plantation M2 – Boreal Mixed wood (25%) M2 – Boreal Mixed wood (50%) Custom fuel types: O1a – Seminatural areas O1b - Agroforestry areas
Ignition DMC threshold	20	5
Spread DMC threshold	20	44
Smoulder interval (hours)	3	22
O1a - Curing (%) in fall	75	62

(b)



Eq.1 - Custom O1a: $RoS = 130\left[1 - e^{(-0.0344 \times ISI)}\right]^{1.21}$

Eq.2 - Custom O1b: $RoS = 205\left[1 - e^{(-0.035 \times ISI)}\right]^{1.7}$

Grass type	Spring curing	Spring GFL	Summer curing	Summer GFL	Fall curing	Fall GFL
O1a	25	0.25	50	0.5	75	0.75
O1b	30	0.3	60	0.3	80	0.3

Fig. 3. (a) Initial and modified BFOLDS-FRM inputs and parameters used in baseline and calibration steps for the Sabor River upper basin; (b) Custom fuel types: adaptation of the standard grass fuel types (O1a and O1b) of the Canadian Fire Behaviour Prediction system to best fit the vegetation types in the Sabor River upper basin. Fire rate of spread as function of Initial Spread Index (ISI), and fire rate of spread (RoS) equations (Eq. (1) and Eq. (2)) for the custom fuel types, as well as the grass fuel load (GFL) (kg/m^2) and fuel curing (%) values by season are shown.

Table 3
Observed and simulated fire regime attributes for the Sabor River upper basin and relative errors (RE) of simulations in 2007, 2015, 2018 and mean for the three years analysed.

		Fire regime attribute		
		No. of fires	Total burned area (ha)	Mean fire size (ha)
2007	Observed	52	277.30	5.33
	Simulated	45	269.31	5.98
	RE (%)	−13.46	−2.88	12.23
2015	Observed	18	301.05	16.73
	Simulated	24	297.06	12.38
	RE (%)	33.33	−1.32	−25.99
2018	Observed	29	110.35	3.81
	Simulated	33	161.19	4.88
	RE (%)	13.79	46.07	28.36
Mean	Observed	33	229.57	8.62
	Simulated	34	242.52	7.75
	RE (%)	3.03	5.64	−10.12

as reasonable, although predictions for both the number of fires and mean fire size were within that interval (Table 3). The predicted burned area for 2015 had a substantially lower relative error, although the simulated annual number of fires was greater than the number of fires observed, which increased the associated relative error, in turn, underpredicting the mean fire size (Table 3).

3.2. Assessment of modelled fire-related functions and services

3.2.1. Fire regulation capacity

Overall, most of the simulated fires in the Sabor River upper basin were small and of low intensity, although differences were observed among the three dates analysed (Fig. 4). Simulated fire regime attributes increased in 2015 compared to 2007, with fires exceeding 100 ha and the 500 $kW\ m^{-1}$ mean fire intensity (Table 4). On the other hand, overall simulated attributes in 2018 decreased compared to 2007 (Table 4). In 2018, simulated fires were smaller and less intense than in 2007, but one attained more than 100 ha, with mean fire intensity >500 $kW\ m^{-1}$ (Fig. 4 and Table 4).

3.2.2. Impacts of fire on the carbon storage balance and the climate regulation ecosystem service

The maximum potential for carbon storage in the landscape increased from 2007 on. Compared to 2007, our estimates indicate that carbon stocks in 2015 increased by 40%, while in 2018 increased by 70% (Table 5). On the other hand, the losses in the maximum potential carbon stored in the landscape due to the simulated fires increased over time (Fig. 5 and Table 5). In 2015, losses represented 1.2% of the maximum potential for carbon storage in the landscape, indicating an increase of 80% compared to 2007 (Table 5). In 2018, losses represented 0.7% of the maximum potential for carbon storage in the landscape, increasing 20% compared to 2007.

4. Discussion

4.1. Model evaluation

Changes made to the model spatial inputs did affect its outputs (see Table 1), which is in line with findings reported for other modelling platforms that simulate fire mechanistically (Cary et al., 2006). As an ecological process, fire creates landscape heterogeneity and responds to the spatial patterns of fuel composition and configuration, which are crucial for its regulation (Turner, 2010; Turner et al., 2012). Fire spread is simulated in BFOLDS-FRM as a mechanistic process where, among

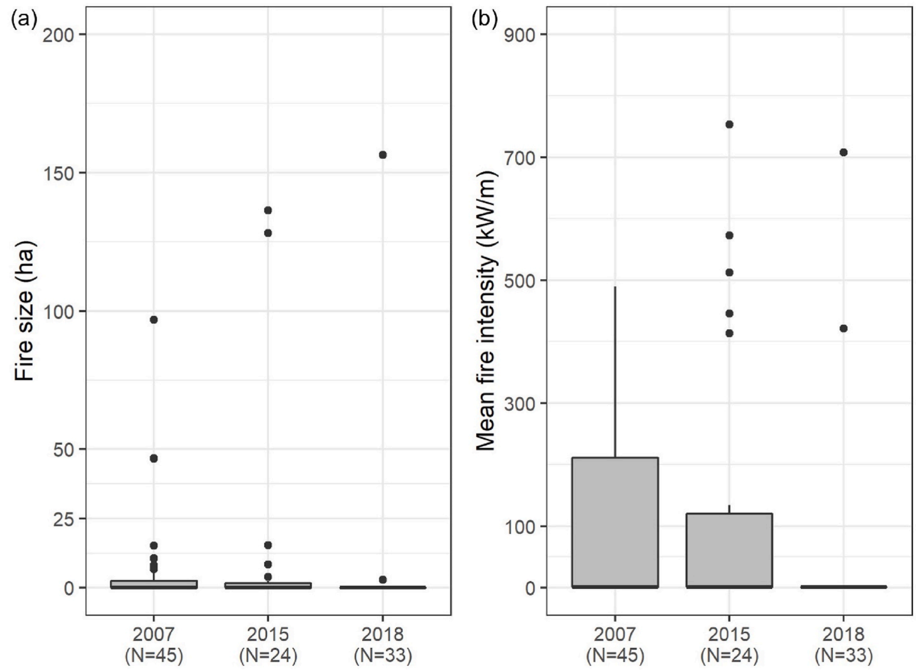


Fig. 4. Simulated attributes of fire regime in the Sabor River upper basin in 2007, 2015, and 2018: (a) fire size (ha), and (b) mean fire intensity (kW m^{-1}). Values in brackets indicate the number of simulated fires in each year. Boxplots show median, quartiles, and outlier values.

Table 4
Simulated fire regime attributes used as indicators of the fire regulation capacity of the Sabor River upper basin in 2007, 2015 and 2018.

Fire regime attribute	2007	2015	2018
Total annual burned area (ha)	269.31	297.06	161.13
Annual mean fire size (ha)	5.98	12.38	4.88
Annual mean fire intensity (kW/m)	92.19	126.26	36.22
Nr. fires (Fire size >100 ha)	0	2	1
Nr. fires (Mean fire intensity >500 kW m^{-1})	0	3	1

Table 5
Estimates of the maximum potential for the total carbon stored (Tg C) and the mean carbon density (Mg C ha^{-1}) in the landscape and the corresponding impact (losses) due to simulated fires in the Sabor River upper basin in 2007, 2015, and 2018.

Year	Carbon Max. Potential		Carbon Supply after Fire		Carbon Losses	
	(Tg C)	(Mg C ha^{-1})	(Tg C)	(Mg C ha^{-1})	(Tg C)	(Mg C ha^{-1})
2007	0.54	23.292	0.535	23.065	−0.005	−0.227
2015	0.757	32.729	0.748	32.356	−0.009	−0.373
2018	0.919	39.523	0.913	39.25	−0.006	−0.273

other factors, spatial patterns of pixel-based fuel types and topographic layers are crucial (Perera et al., 2008). In this regard, it is important to ensure that the data used in the parameterization of model spatial inputs has the best quality possible in terms of spatial and thematic resolution to describe dynamics accurately since they affect modelling and simulation of landscape fire processes (Saura, 2002; Taneja et al., 2021; Turner and Gardner, 2015). On the other hand, although our indicators did not show a clear effect of the spatial weighting pattern based on historical fire ignition records on model outputs (see Table 1), similar to other studies (Bar Massada et al., 2011; Perera et al., 2009), we consider that applying this information to fire simulations may be relevant, particularly in the context of mountains in the Mediterranean region since fire ignition patterns in these areas are influenced by the use of fire as a management tool (Catry et al., 2009; Sequeira et al., 2020). Our results suggest that the quality of spatial data inputs is important since it affects the model outputs. As such, the user must be aware of the trade-offs that may arise when selecting spatial inputs for model parameterization. For example, calibrating the model with the most detailed data available benefits the accuracy of model predictions, but may also increase the computational costs to perform the simulations (Taneja et al., 2021). Besides, integrating the fire ignition patterns in the simulations is useful to represent real-world dynamics in mountainous landscapes and thereby to characterize the fire regime in these areas (Fernandes et al., 2014; Keeley et al., 2012).

As expected, BFOLDS-FRM output variables were also sensitive to changes made to weather inputs (see Table 1), since these comprise the most important information for simulating fire patterns (Perera et al., 2008). Our results agree with findings from other fire modelling platforms where climate variables were very important for fire simulation (Cary et al., 2006; Hummel et al., 2013). Inconsistencies in the rainfall variable between observed (Bragança weather station) and estimated (ERA5-Land dataset) (Appendix A) can partially explain the variation found in our results, with the latter underestimating the fire weather indices in the study area and thereby decreasing both burned area and fire intensity. The availability of fire weather data is a critical point in the application of fire simulation models (Riley and Thompson, 2016). Despite recent advances in the supply of ready to use FWI (and sub-indices) data at the global scale (e.g. ERA5 reanalysis products; Vitolo et al., 2020), their application at the local scale remains limited due to their spatial resolution (e.g. 28–56 km grid cell size). As for prediction of the future fire danger conditions using meteorologically based indices (e.g. the Canadian Fire Weather Index), uncertainties regarding the use of weather data (e.g. regional climate models) have been described in the scientific literature, indicating either a potential negative bias in fire predictions (Herrera et al., 2013) or acceptable agreement between weather information and fire predictions (Amatulli et al., 2013). Ultimately, our results stress that the weather data quality is important for estimates of fire weather conditions since those are critical for fire simulations in BFOLDS-FRM.

Regarding the influence of modifications in the FBP standard fuel types, although BFOLDS-FRM has responded to those changes, the

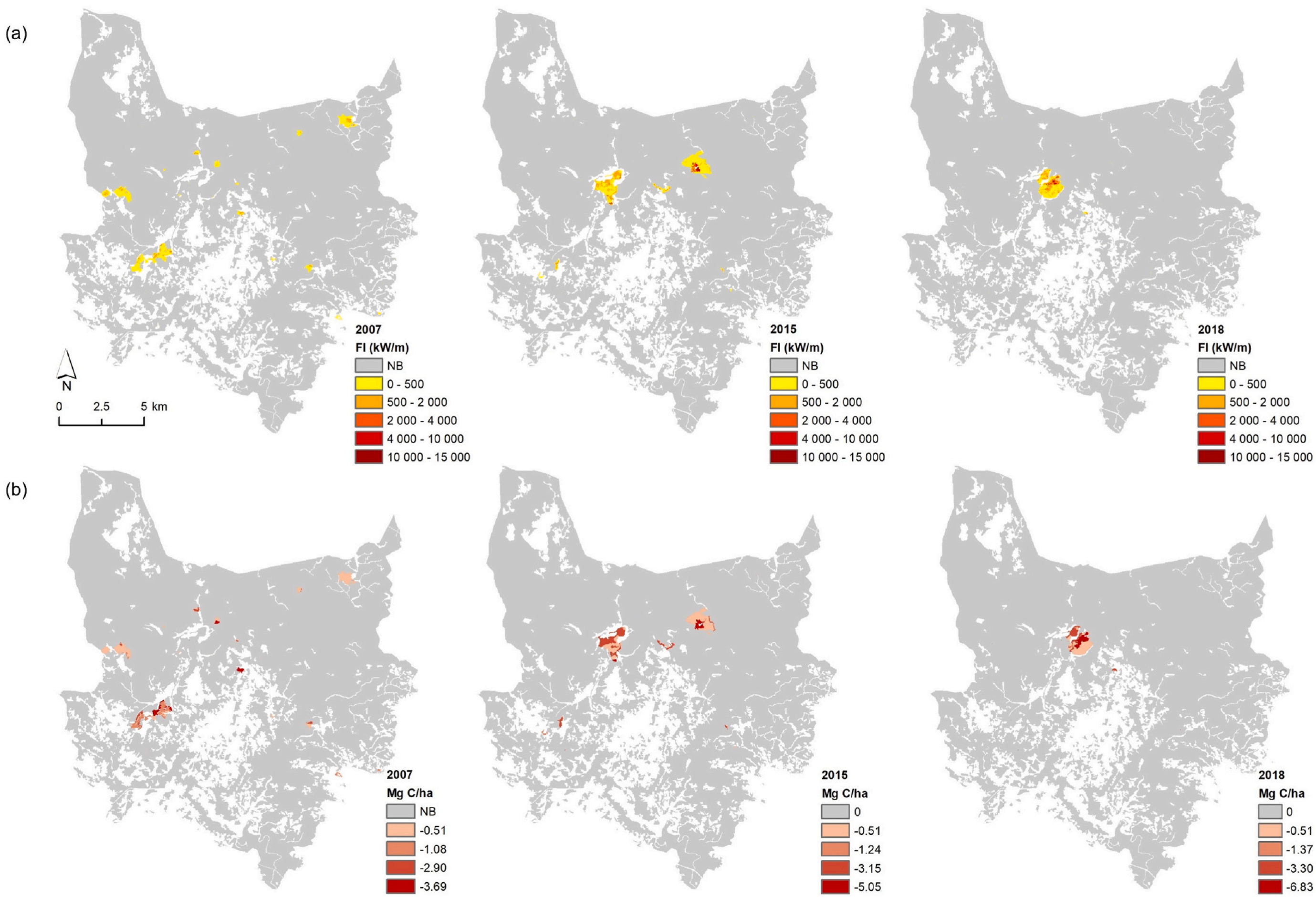


Fig. 5. Simulated fires in the Sabor River upper basin in 2007, 2015 and 2018, showing: a) burned area classified by fire intensity (kW m^{-1}), and the respective b) impact (losses) on carbon stored (Mg C ha^{-1}), expressed as the difference between the simulated maximum potential for carbon storage, and the carbon stored after the simulated fires. NB = unburned.

indicators did not show a relevant influence on model outputs (see Table 1), which partially supports the results reported by Sturtevant et al. (2009). The recalculation of the fire spread rate (RoS) for the standard FBP O1 fuel type reduced the potential fire spread rate, in turn restricting the size of the simulated fires. However, more substantial RoS differences would have been observed if fire weather conditions used in simulations were more severe, especially in terms of wind speed, as it affects the computation of the Initial Spread Index (ISI) (Forestry Canada Fire Danger Group, 1992; Wotton et al., 2009). On the other hand, outputs were highly sensitive to changes made to the degree of grass curing (see Table 1) since RoS is greatly enhanced by fully cured grass (Forestry Canada Fire Danger Group, 1992; Wotton et al., 2009). The replacement of FBP O1 fuel subtypes with custom fuel types allowed to improve the representation of the vegetation found in the study area, namely semi-natural and agroforestry areas, since O1 fuel subtypes are limited to grasslands, either matted (O1a) or standing dead (O1b) (Forestry Canada Fire Danger Group, 1992). Also, in shrublands, which are abundant in the study area, RoS is mostly controlled by vegetation height (Anderson et al., 2015), whereas grassland RoS varies mainly with the degree of grass curing (Cruz et al., 2015) and fuel load (Cruz et al., 2017) thus underlining the need for adjustments (see Tables 2 and 3). Studies carried out elsewhere have emphasized the need to either develop or adapt fuel models to improve fire behaviour estimates for local vegetation types (Clark et al., 2008; Cruz and Fernandes, 2008; Fogarty et al., 1998).

Number of fires and mean fire intensity and size variables were very sensitive to changes made to parameters based on the duff moisture code (DMC) (see Table 1). This behaviour was expected since the DMC is an indicator of fuel consumption in boreal forests (Wotton, 2008). Hence, our results reflect how BFOLDS-FRM applies the principles underlying the Canadian Forest Fire Danger Rating System to simulate fire ignition, spread and extinguishment (Perera et al., 2008). Calibration of the DMC based parameters indicated improvements in model ability to emulate the fire regime patterns observed in the study area (see Tables 2 and 4), which agrees with other studies carried out in the Mediterranean region (Dimitrakopoulos et al., 2011) that suggest the need to adapt and/or modify assumptions related to the DMC to improve the predictions of both fuels moisture and burned area. On the other hand, the need for adjusting DMC-based parameters may reflect, in part, the variability associated with the prediction of burned area and number of fires when the FWI indices are applied across the Mediterranean basin (Amatulli et al., 2013). Such differences may be partially related to the type of dominating vegetation, i.e. forest or shrubland (Cruz et al., 2003a; Dimitrakopoulos et al., 2011; Fernandes, 2016) or to the different characteristics of the duff layer in Mediterranean ecosystems and boreal areas in terms of quantity, depth and moisture (Dimitrakopoulos et al., 2011) particularly when related to shrubland (Fernandes, 2005, 2016). Also, strategies of full fire suppression in Mediterranean countries (Rigolot et al., 2009) may hamper the FWI-based simulation of fire. Nevertheless, the DMC is a useful predictor of fuel consumption in maritime pine stands (Fernandes and Loureiro, 2013), while the moisture content of the duff layer (variable used in the calculation of the DMC code) has been correlated to re-ignition events and the occurrence of smouldering in Aleppo pine stands (Xifré-Salvadó et al., 2020). Given the importance of these parameters to simulate fire in BFOLDS-FRM and the limitations and uncertainties regarding the use of FWI indices in the Mediterranean region, particularly the role of DMC in shrubland fires, careful calibration of the DMC-based parameters in BFOLDS-FRM should be considered.

4.2. Patterns in fire-related functions and services

Overall, our results indicate that FRC in 2015 decreased compared to 2007, while it increased in 2018, although the capacity to regulate potential large and intense fires decreased from 2007 onward (see Table 4 and Figs. 3 and 4). Land cover changes that took place in the Sabor River upper basin landscape over time (see Appendix G) together with the BFOLDS-FRM outputs can explain, in part, our results. Forest expansion between 2007 and 2015, mostly at the expenses of seminatural areas, but also of some non-burnable areas (e.g., agriculture), led to increasing fuel continuity and fuel hazard in the landscape, allowing fire to spread more easily over larger areas and with high intensity (see Table 4 and Figs. 3 and 4). On the other hand, between 2015 and 2018, although the vegetated area continues to grow, the transition from vegetated to non-burnable areas may have enabled, in some cases, the disruption of fire propagation in the landscape. Also, the conversion of coniferous forests, mainly to deciduous forests, may have balanced the general fire intensity, although such changes have not prevented the occurrence of a large fire in the simulations (see Table 4 and Figs. 3 and 4). Similar patterns were observed in previous studies carried out in the study area concerning the effect of LULC changes on the dynamics in the capacity of the landscape to regulate fire (Azevedo et al., 2011; Sil et al., 2019b), as well as for other areas the Mediterranean basin regarding the effects of changes in LULC on fire hazard and fire regime (Moreira et al., 2011; San-Miguel-Ayanz et al., 2012) and fire regulating functions (Depietri and Orenstein, 2019).

Landscape changes between 2007 and 2018 promoted carbon storage in the Sabor River upper basin landscape (see Table 5), particularly through the expansion of forest areas, together with vegetation growth and increase of carbon stocks in aboveground biomass (see Appendices F and G). Our results are in line with previous work assessing the CRES carried out in the area (Sil et al., 2017), as well as for other mountainous areas, where similar patterns of landscape changes tend to increase the supply of CRES (Locatelli et al., 2017; Pais et al., 2020). On the other hand, the observed changes have modified the landscape capacity to regulate fire in the years analysed, threatening the supply of CRES differently, as suggested by the variation in carbon losses (see Table 5 and Fig. 5). For example, in 2015, the growth of carbon losses by 80% compared to 2007 can be partially related to the decrease in FRC due to important changes in landscape structure and composition that were reflected in the increase in the simulated total burned area. In 2018, although FRC increased, as suggested by the reduction in the burned area, carbon losses due to fires increased by 20% compared to 2007. Such an increase reflects a large fire, partially driven by increasing forest fuels continuity in the landscape, together with an increase of carbon stocks over time, which resulted in greater carbon losses. Our results are in line with findings reported by Thom and Seidl (2016) concerning the impact of [fire] disturbances on carbon storage, as well as the potential risks associated with the increase of forest areas e.g., to balance carbon emission envisioned in the European Green Deal (European Comission, 2019) discussed in Hermoso et al. (2021).

In addition to the effects of landscape changes on the FRC dynamics and the supply of CRES, our results also reflect the influence of annual fire weather conditions. For example, although 2015 showed the lowest number of fire ignitions among dates, it accounted for more days with very high and extreme fire danger (FWI>45) compared to 2007 and 2018 (see Appendix G). In turn, the growing fuel continuity and fuel hazard in the landscape overtime, along with more severe weather conditions, may have increased susceptibility for large and intense fires in 2015 simulations increasing the impact of fires on CRES supply (see Fig. 5). Conversely, 2018 accounted for more days with low fire danger (FWI<23), which may have limited the conditions for fire spread and the reduction of the overall fire size and fire intensity (see Table 4 and Figs. 3 and 4), despite the number of simulated ignitions exceeded 2015. However, when weather conditions worsened, during the summer season, and fuel continuity and fuel hazard in the landscape were relatively high, a large and intense fire was simulated that impacted high-carbon stock area (see Fig. 5). Our results agree with findings that show a relationship between FWI thresholds and fire size (Fernandes, 2019). However, a more detailed analysis should be carried out to reduce uncertainty regarding the effect of landscape structure and fire weather conditions in driving fire regime attributes (Fernandes et al., 2016).

4.3. Modelling strengths, limitations, and recommendations for future research

Overall, BFOLDS-FRM is a valuable tool for informing sustainable planning and management of mountainous areas prone to fires in the Mediterranean wherein, similarly to the Sabor River upper basin, increasing fire hazard in the landscape is an ongoing process driven by rural depopulation and vegetation encroachment in abandoned areas (Azevedo et al., 2011), lack of effective forest management (Pérez-Rodríguez et al., 2018), and potential intensification of afforestation activities due to demand for natural resources, e.g. bioenergy (Pérez-Rodríguez and Azevedo, 2020) or actions to cope with climate change (Hermoso et al., 2021). The high predictive capacity of the model allowed to estimate the selected fire regime attributes of the Sabor River upper basin within an acceptable range of error regarding the observed data (see Table 3), which in turn enabled characterizing the dynamics of the landscape capacity to regulate fire (see Table 4 and Fig. 4), as well as to estimate potential impacts on the supply of the climate regulation ecosystem service (see Table 5 and Fig. 5). Ultimately, BFOLDS-FRM is particularly useful for testing fire-smart management alternatives that aim fire hazard mitigation while benefiting the supply of ecosystem services at the landscape scale (Damianidis et al., 2021; Fernandes, 2013; Pais et al., 2020), enabling Mediterranean mountains to cope with challenges related to their high vulnerability to global changes (Schroter et al., 2005).

Although BFOLDS-FRM was able to emulate the observed pattern of the relative dominance of small fires in the study area (see Fig. 4), after comparing simulated outputs with fire records, it was clear that the model was not able to accurately capture the occurrence of large fires, except in year 2015 (Appendix E). Such deviation may be related to oversimplification of fuel types used in the simulations after aggregation of land cover data in major LULC classes, as well as to simplification of the fire weather conditions based on daily rather than hourly estimates for the whole study area, since these are of particular relevance concerning large fires (Fernandes et al., 2016). In addition, the conceptualization and mechanics of the model itself can explain these differences because, unlike other models, e.g. MEDFIRE (Brotons et al., 2013), the size and extinction of the simulated fires in BFOLDS-FRM are neither predefined nor depend on modelled fire suppression efforts, but rather are emergent properties of the simulation process, arising from the interaction between the theoretical assumptions of the model, the assumptions of the users, and the inputs provided (Perera et al., 2014).

Notwithstanding our effort to assess and apply BFOLDS-FRM, we acknowledge some limitations in our modelling framework. Model evaluation results were obtained based on model runs without replicates, as we set the parameter "random number seeds" as a constant, that is, preventing the model from having a stochastic behaviour. This approach is common in this type of assessment and useful when testing the sensitivity of several model parameters (Grant and Swannack, 2008) because it allows evaluating the relative effect of each parameter, avoiding the "noise" that comes from the model stochasticity. Besides, it decreases the number of simulations and the computational costs and time spent to complete the model evaluation steps (Grant and

Swannack, 2008). However, we acknowledge that this approach fails to simulate and assess the uncertainty in model predictions that can derive from variability in the frequency and the location of fire ignitions (Riley and Thompson, 2016). Therefore, we recommend that future work should consider model output variability as a way to optimize and improve model predictions for its implementation as a management and decision support tool (Uusitalo et al., 2015).

Sensitivity analysis allowed a deeper understanding of the relative influence of BFOLDS-FRM inputs and parameters on model outputs (see Table 1). However, we recognize that the approach used, i.e., One-At-a-Time (OAT) method, has some limitations, for instance, it does not allow to effectively explore the interaction of inputs and parameters, and thus, quantifying their combined influence on model outputs (Saltelli et al., 2019). Although more comprehensive approaches do exist, e.g., All-at-a-time (AAT) methods (Pianosi et al., 2016) these were not considered due to their implementation complexity, which particularly increases when the model relies heavily on spatial inputs (Pianosi et al., 2016), as well as due to the high number of simulations and time required to process model results and proceed with model evaluations (Uusitalo et al., 2015).

Moreover, we carried out our analysis considering only three years: 2007, 2015, and 2018. We acknowledge that these years may not represent the full range of conditions existing in the study area, which may restrain model response during the calibration step and raise uncertainties regarding model behaviour validation. On the other hand, complete and compatible data to parameterize the model and perform the calibration and validation steps are limited for the study area. Although statistical methods available in the scientific literature, e.g. cross-validation (Refaeilzadeh et al., 2009) can be useful in model training and validation when available data is scarce, their application would be unfeasible due to the model mechanics. To overcome these issues, at least partially, we used data from the same system but for different years (Waveren et al., 1999), which introduced some variability in the environmental conditions, such as the daily weather conditions, as well as in the spatial composition and configuration of land cover classes/fuel types, or the daily fire ignition data. Although our approach allowed us to evaluate the behaviour and capabilities of the model, we acknowledge that improving the model evaluation process with more model applications is needed. Either testing the model in the study area using independent data or applying it to similar systems outside the study area will strengthen the applicability and usefulness of this tool in fire-related research in mountain Mediterranean landscapes.

Our modelling approach captured the influence of landscape dynamics on fire behaviour (see Table 4 and Figs. 4 and 5), but it was not able to capture the effect of fire on the landscape. Simulating such interaction is essential to represent feedbacks between disturbances and landscape dynamics (Turner, 2010). As such, we acknowledge that uncertainties may persist regarding whether the model can effectively characterize the fire regime and the FRC in the study area, despite its ability to emulate the selected fire regime attributes in the years evaluated. Therefore, future application of BFOLDS-FRM may benefit from coupling the model with, e.g., a succession extension of the LANDIS-II platform (Scheller et al., 2007), allowing to simulate interactions and feedbacks between landscape and fire. Besides, it would enable continuous spatiotemporal outputs to explore future changes in fire regime.

Lastly, our approach to assess the impacts of fire on carbon stored simplifies the potential interactions between fire and carbon dynamics. As such, carbon stocks represent mean values for generic LULC classes, and those are only for the aerial biomass (Appendix F) while disregarding other carbon pools in terrestrial ecosystems, such as litter and soil (Lorenz, 2013) that can also be affected by fires (Garcia-Hurtado et al., 2013). We assumed that fire fully consumed aerial biomass, in turn releasing all the stored carbon into the atmosphere, which is seldom the case since part of the carbon resulting from biomass burning can incorporate dead organic matter and soil pools as pyrogenic carbon (Jones et al., 2019). Therefore, future work on the effects of fire on carbon balance should consider a more comprehensive assessment of carbon stocks assigned to each LULC class, and sub-processes that are part of carbon dynamics as a result of fire activity (Thom and Seidl, 2016).

5. Conclusions

The evaluation of BFOLDS-FRM contributed to a more in-depth understanding of the relative influence of inputs on model behaviour, and to improve the identification of key parameters for fire simulation. Moreover, adjustments made to model inputs and parameters improved BFOLDS-FRM accuracy and emulated fire regime attributes in the Sabor River upper basin with a relative error within acceptable bounds. In this regard, we emphasize that the application of BFOLDS-FRM in the context of the fire regime of Mediterranean mountainous areas must consider: (i) supplying the model with accurate and precise data to properly characterize the spatial patterns of fuels and fire weather conditions; (ii) carefully calibrating parameters based on fire weather conditions due to the high model sensitivity to these parameters; (iii) customizing standard fuel types to meet the characteristics of the existing vegetation; and (iv) including ignition patterns associated with human activities. Our work underlined that BFOLDS-FRM can provide useful outputs to support the characterization of fire-related ecosystem functions and services in a Mediterranean mountainous area. Still, we have identified potential limitations that may arise when applying the model to different areas, which should thus be considered in future applications.

Software name

Boreal Forest Landscape Dynamics Simulator – Fire Regime Module (BFOLDS-FRM) for LANDIS-II.

Programming language

C#

Developers

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Available at

https://www.landis-ii.org/extensions.

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Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envsoft.2022.105464.

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