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ORIGINAL PAPER

Does forest biomass harvesting for energy reduce fire hazard in the Mediterranean basin? a case study in the Caroig Massif (Eastern Spain)

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Abstract The Mediterranean basin is a fire-prone area and is expected to continue being so according to projected climate and socioeconomic changes. Sustainable exploitation of forest biomass could have a positive effect on wildfire hazard mitigation. A modelling approach was used to compare how four different Scenarios for biomass collection for energy use affect fire behaviour and potential burnt area at landscape level under extreme meteorological conditions in a typical Mediterranean Massif. A case study of *Pinus halepensis* stands in Valencia (Eastern Spain) was conducted. The FARSITE simulator was used to evaluate the burnt area and fire behaviour parameters. Simulations predicted a significant increase in the burnt area and the values of most fire behaviour parameters in a Scenario of rural abandonment, relative to the current situation. Biomass management through thinning reduced canopy bulk density; however, no differences in the values of the main fire behaviour parameters were detected. Thinning and understory clearing, including biomass collection in large shrub fuel model areas, significantly reduces fire hazard. Forest biomass sustainable harvesting for energy is expected to reduce fire hazard if management includes intense modification of fuel models, comprising management of shrub biomass at the landscape level. Strong modification of forest fuel models requires intensive silvicultural treatments. Therefore, forest biomass collection for energy in the Mediterranean basin reduces fire hazard only if both tree and shrub strata are managed at landscape level.

Keywords Wildfire prevention · Forest fuel management · Fuel model · Fire behaviour · FARSITE · Aleppo pine

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Introduction

The Mediterranean basin is a fire-prone area and is expected to continue being so according to climate change predictions (Bedia et al. 2014; Moriondo et al. 2006; Turco et al. 2014). Although wildfires do not always spell ecological disaster in the Mediterranean basin (Pausas et al. 2008), a decline in the regeneration capacity of some forests that are affected by recurrent fires has been detected (e.g. Espelta et al. 2008; Rodrigo et al. 2004). In rural areas of Southern Europe, the effects of socioeconomic changes (which started at the beginning of the twentieth century, but have accelerated since the 1950s) have been exacerbated in the last few decades by the effects of climate change. The long duration of this process has led to the accumulation of fuel biomass, the increase in fuel connectivity as well as the generation of new forest–agriculture interfaces and the presence of forest fuels on abandoned agricultural land (Fernandes 2013; Martínez et al. 2009, Martínez-Fernández et al. 2013; Moreira et al. 2011; Ortega et al. 2012).

One of the pillars of wildfire prevention is the reduction in fire hazard by decreasing the amount and continuity (horizontal and vertical) of forest fuels (e.g. Agee and Skinner 2005; Graham et al. 2004; Schwilk et al. 2009; Reinhardt et al. 2008; Omi 2015). However, implementation of this preventive silviculture practice is (like many prevention practices) subject to fluctuations in financial resources and policy priorities. The lack of stable financial resources hinders the effectiveness of fire prevention. In the current context of global change, in which forests are increasingly threatened by fire, the success of prevention strategies is critical. The challenge is to establish how this can be done when the urgent need for fire prevention coincides with an economic crisis. Experts in Mediterranean Europe consulted within the framework of the FIRESMART project (http://cordis.europa.eu/result/rcn/56645_en.html) agree that this dilemma could be, if not resolved, at least greatly mitigated by the sustainable exploitation of forest biomass as a bioenergy resource.

The above reasoning has been emphasized in different national and international forums. However, a number of structural problems (legal, social, economic obstacles) hamper the use of the forest biomass resource in Mediterranean ecosystems. Mediterranean forests characterized by low timber growth and high slope (thus restricting the use of mechanized treatments) are usually not profitable. Various European R&D and LIFE+ projects (i.e. FIRESMART, BIOENERGY & FIRE PREVENTION, FORRISK, ENERBIOSCRUB: www.cordis.europa.eu) highlight that the sustainable exploitation of forest biomass for energy purposes represents an opportunity for promoting development and employment in rural areas of Mediterranean countries. Such exploitation would ensure provision of some income that could be used to finance management activities (Marino et al. 2014b).

Forest management has a positive effect on wildfire severity mitigation (e.g. Agee and Skinner 2005; Raymond and Peterson 2005; Reinhardt and Holsinger 2010; Stephens et al. 2009a, b; Stephens and Moghaddas 2005; Waldrop et al. 2008), and numerous studies have demonstrated that silvicultural treatments can prevent large fires and crown fires at stand level (e.g. Cram et al. 2006; Dailey et al. 2008; Graham et al. 2009; Martinson and Omi 2008; Prichard et al. 2010). Some studies carried out at landscape level (e.g. Finney 2003; Finney et al. 2007; González et al. 2006, 2007; Lehmkuhl et al. 2007; Schmidt et al. 2008; Wimberly et al. 2009) have used fire spread simulators to assess the efficacy of fuel management and logging operations in reducing fire hazard including the Mediterranean basin (e.g. Arca et al. 2007, Duguy et al. 2007, González-Olabarría and Pukkala 2011, González-Olabarría et al. 2012; Oliveira et al. 2016). Nevertheless, little is known about the extent to which forest biomass extraction for energy use reduces fire hazard (Barbour et al. 2008; Hudiburg et al. 2011; Iversen and Van Demark 2006; Mitchell et al. 2009; Neary and Zieroth 2007; Sacchelli et al. 2013). Moreover, although some studies have investigated the effectiveness of shrub fuel treatments in reducing fire risk at stand level (Marino et al. 2011, 2014a), studies on the use of shrub biomass for energy purposes are scarce (Nuñez-Regueira et al. 2004; Pérez et al. 2014; Regos et al. 2016). Given the persistent uncertainty due to a lack of quantitative assessments of how forests managed for biomass extraction decrease fire risks, in a recent study Regos et al. (2016) suggested the development of new studies at finer scales to clarify this linkage. Furthermore, the lack of specific biomass growth models for estimating shrub biomass (Pasalodos-Tato et al. 2015) hinders biomass management planning in Mediterranean ecosystems.

According to Stephens (1998), a reduction in fire risk at landscape level can only be brought about by significant changes in fuel structure and load. Such changes imply changes in available-to-fire shrub biomass (surface fuel) and canopy biomass (crown fuel). The bioenergy industry currently does not usually demand shrub biomass as a raw material, because of technical and economic limitations. However, technological advances will probably be made in the near future (e.g. ENERBIOSCRUB project http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction = search.dspPage&n_proj_id = 5000). The following three points should therefore be considered in relation to fuel management and fire prevention that may change the present situation: (i) whether the expected fire behaviour and load could be significantly changed by extracting biomass from trees only; (ii) whether the collection of shrub biomass could satisfy the needs for extraction of biomass for energy purposes and wildfire prevention; and (iii) whether evaluation of fire behaviour is required in relation to biomass management Scenarios under extreme meteorological conditions at landscape level. Moreover, extreme meteorological conditions (high temperatures and drought events) are expected to occur more frequently in the future and this will affect wildfire hazard, fire behaviour and fire severity in the Mediterranean basin (Bedia et al. 2014).

The aim of this study was to compare, using a modelling approach, how four different ways of exploiting biomass for energy purposes will affect fire behaviour and potential burnt area at landscape level under extreme meteorological conditions in a typical Mediterranean Massif.

Materials and methods

Study site

The study site is located in Eastern Spain (Fig. 1a) and comprises 18 municipalities (Fig. 1b) covering an area of 130,542 ha. Landscape is defined by the following UTM coordinates (ETRS85, zone 30): ($X = 665{,}068$; $Y = 4{,}355{,}699$), ($X = 687{,}620$; $Y = 4{,}295{,}207$), ($X = 664{,}680$; $Y = 4{,}354{,}885$), ($X = 704{,}101$; $Y = 4{,}317{,}277$). In most of the study area, the potential vegetation is a *Quercus ilex* L. forest; however, at present most of the study area is dominated by *Pinus halepensis* Miller stands derived from plantations or post-fire regeneration (Online resource, Table S1-1). Shrublands are dominated by *Erica* spp. mixed with *Ulex parviflorus* Pourr., *Rosmarinus officinalis* L., *Quercus coccifera* L., perennial grasses (*Brachypodium retusum* (Pers.) Beauv.) and other annual herbaceous species (Gramineae, Fabaceae, Asteraceae Fam.). The study site is a typical meso-Mediterranean bioclimatic stage, with mean annual temperature between 13 and 18 °C and annual precipitation between 350 and 700 mm.

Several wildfires affected the study area during 1969–2012 period, burning approximately 75 % of the surface analysed (Fig. 1c). Ten of the wildfires were megafires that affected areas of more than 5000 ha. These data correspond to a fire recurrence of less than 20 years and occurrence of a large wildfire (more than 500 ha) every two years. The area is dominated by *Pinus halepensis* forest stands of different ages, according to fire recurrence: stands not affected by wildfires for more than 50 years (50–100 years old); even-aged stands affected by wildfires between 20 and 50 years ago (20–50 years old); and post-fire regenerated stands mixed with shrubland (<20 years old) recently originated from wildfire. Land vegetation cover comprises shrubland, grassland and agricultural land (Online resource 1, Table S1-1).

Simulations and geodata processing

The *FARSITE Fire Area* simulator (www.firelab.org) was used to predict the burnt area and fire behaviour parameters at landscape level. We assume the limitations of FARSITE to simulate extreme fire behaviour (Cruz and Alexander 2010, Fernandes et al. 2011, Cruz et al. 2012, Alexander and Cruz 2013, Benali et al. 2016). Therefore, we consider this study as a modelling approach to compare different management Scenarios in extreme weather conditions, assuming that outputs could highlight the relative importance of the vegetation on fire hazard. This simulator requires three typical fire triangle inputs and different thematic layers from geodatabases (Online Resources 1 and 2): (1) topography (elevation, slope and aspect layers), (2) fuel (vegetation cover, surface fuel model, tree height, height of crown and tree canopy bulk density layers) and (3) meteorological Scenario.

(1) Topographical data were obtained from a digital elevation model developed by the Spanish Geographical Institute (resolution 5 m). The model was developed using LiDAR data (0.5 pulses/m^2),

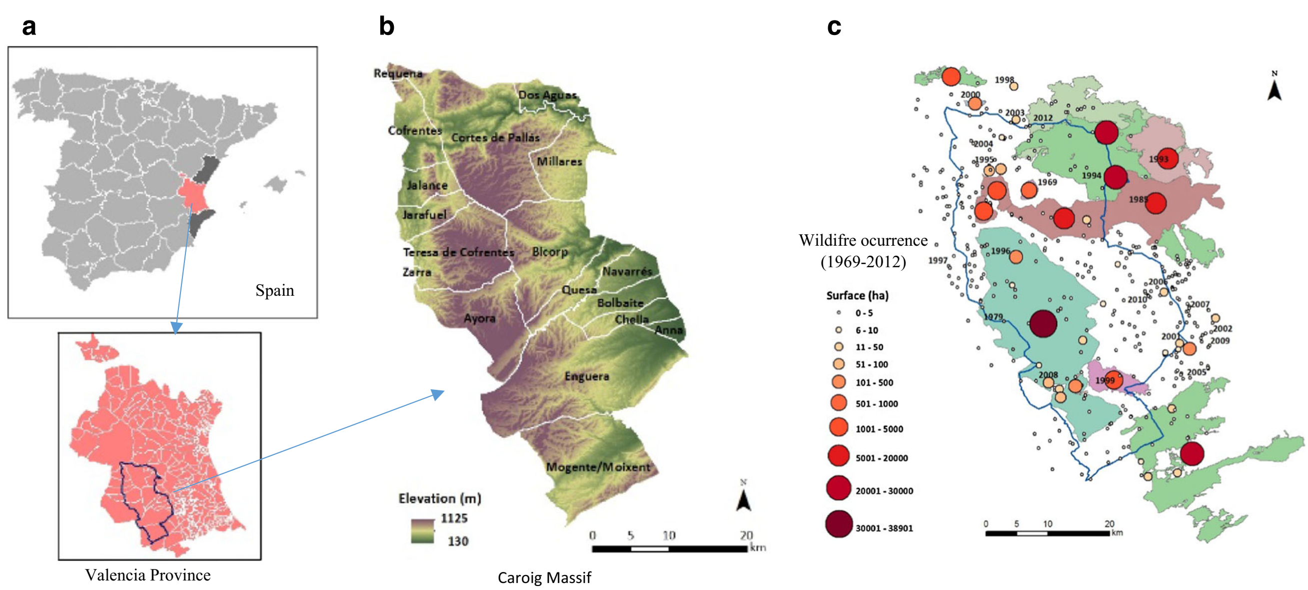

Fig. 1 **a** Geographic location of the Caroig Massif. The autonomous region of Valencia is highlighted in Spanish map, and the province of Valencia is shown in *pink* and the Caroig Massif area in *blue*. **b** Elevation map showing the different municipalities. **c** Fire surface and ignition points (1969–2012) as part of the PNOA project (technical characteristics available at http://pnoa.ign.es/caracteristicas-tecnicas).

(2) Vegetation data (Online Resource 1). LiDAR files (Lass) from the PNOA project (2010) were used to estimate forest fuel cover (FCC), stand height (H) and stand density (N), according to the methodology proposed by González-Olabarría et al. (2012): The raw LiDAR data were processed using the FUSION system (McGaughey and Carson, 2003) to produce structured statistical information about the laser returns. A predefined height of 2 m was used as a threshold to model mature trees. The LiDAR data were aggregated in data subsets using a square network, of area 900 m^2 (30 m $\times$ 30 m pixel size), for computing metrics at that spatial scale. LiDAR pulse returns from aboveground vegetation (>2 m from the ground) were processed using FUSION software (http://forsys.cfr.washington.edu/fusion/fusionlatest.html), to generate different percentiles (1st, 5th, 10th, 20th, 25th, 30th, 40th, 50th, 60th, 70th, 75th, 80th, 90th, 95th, 99th) of the height and intensity of the pulses, the mean height and intensity values of pulses, and the interquartile range of the intensity values. The forest canopy cover was then estimated from the ratio between the number of first returns above 2 m and the total number of first returns. A field inventory of 100 plots (30 m x 30 m) in the study area included 3521 trees (Table 1). The biomass equation proposed by Ruiz-Peinado et al. (2011) was used to estimate fine fuel biomass of each tree. Data inventory and LiDAR data suggest a low variability in height of crown above ground (CBH = 2 $\pm$ 0.5 m, average $\pm$ standard error) for the three types of *Pinus halepensis* stands. For that reason, we decided not to model this variable and to assume a constant value of CBH = 2 m. The bulk density (CBD, kgm^{-3}) was estimated using the "load over depth method" (Van Wagner 1977). Finally, a model to predict CBD using basal area (G) was fitted ($CBD = 0.0143 + 0.0235G$; $r^2 = 0.80$; SEE = 0.04; $N = 100$). Predictions were expanded to the whole study area fitting a model between stand height average (H) and basal area ($H = 4.534G^{0.255}$; $R^2adj = 0.61$; RMSE = 1.25; $N = 100$) and using processed LiDAR data ($H = 0.95LH\_90^{0.86}$; $R^2adj = 0.85$; RMSE = 1.31; $N = 100$; being LH_90 the 90th percentile height of the canopy pulses). Forest fuel surface biomass (Online Resource 1, Table S1-2, S1-3) was obtained through field inventory (Quílez and Chinchilla 2012) adapting the fuel models proposed by Scott and Burgan (2005) to local conditions to yield more accurate wildfire simulations (Arca et al. 2007). This layer was generated by the *Consorcio de Bomberos de Valencia* (Valencian Firefighters Consortium) based on information collected before 2006 and updated during 2010 through field validation of classified models. The FARSITE calibration proposed by Duguy et al. (2007) was used. The pixel size was set to 30 m for all geographic information layers. The fuel break network (Online Resource 1, Fig.S1-1) was obtained from the Valencian Forest Service and was taken into account in the simulations by assuming suitable management in order to maintain buffers for different types of fuel breaks (Online Resource 1, Table S1-4). *FlamMap* simulations carried out with the minimum travel time (MTT) and the treatment optimization module (TOM) algorithms (Finney 2006) were used to assess the buffer proposed by the Valencian Forest Service. The ARCGIS 10.1® ModelBuilder tool was used to process all geodatabases described (Online Resource 1, Fig. S1-2).

(3) The meteorological conditions (Online resource 2) were established by considering an extreme Scenario that occurred during the Cortes de Pallás wildfire (August 2012), a burnt area close to the study site. This wildfire burnt more than 20,000 ha and involved extreme convective fire behaviour and a large number of crown fires and spotting events. This Scenario (Online resource 2, Table S2-1, Table S2-2) was assessed using the climate prediction models CGCM2 and ECHAM4 (www.ipcc-data.org) and is expected to be repeated at least twice before 2040 (Online Resource 2, Fig. S2-1).

Table 1 Characteristics of *Pinus halepensis* stands (average and range) in the Caroig Massif according to field inventory ($n = 100$ plots), conducted within the framework of the European LIFE+ project "Bioenergy and Fire Prevention" (http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction=search.dspPage&n_proj_id=3653, Life 09 EN-E.000450)

Age (years)	Canopy cover fraction (%)	Density of trees (trees/ha)	Dominant height (m)	Basal area (m^2/ha)	Mean annual increment (m^3/ha year)	Total biomass (Mg/ha)
<20	80 (50–100)	3500 (2000–5000)	4.5 (3–6)	31 (1–64)	0.42 (0–3.9)	6.7 (0–155)
20–50	62 (10–100)	654 (500–1000)	8 (6–15)	16.4 (2–32)	1.9 (0–5.3)	33 (0–103)
>50	49 (10–100)	278 (100–500)	12 (8–18)	9.8 (1–35)	5.1 (0–1.39)	137 (0–432)

Systematic ignition points were selected by ensuring the following: (1) wildfire simulated areas must not overlap, to enable simulation of independent events; (2) in each fuel model, the number of ignitions was weighted by the area occupied by each model relative to the whole study area; (3) the simulation area and perimeter were included in the study area taking into account the wind Scenario selected (west 206°–280°, see Online Resource 2, Table S2-2); and (4) a randomized process was carried out to locate the coordinates of ignition points in areas previously selected by applying (1), (2) and (3). This yielded 40 ignition points distributed homogeneously over the study area (Fig. 2). During the Cortes de Pallás wildfire, the extreme forest fire behaviour made direct attack impossible and the suppression activities carried out during the first 6 h of the event were barely effective. Therefore, in order to detect the initial propagation and potential behaviour under extreme meteorological conditions, the duration of simulation was fixed at 6 h and suppression activities were not taken into account. Fuel moisture contents were fixed for each class of dead fuel using predictions from models proposed by Ruiz et al. (2009) and according to the selected meteorological Scenario: 3 % for the 1-hour time lag (diameter of dead fuel <6 mm), 5 % for the 10-hour time lag (diameter of dead fuel between 6 and 25 mm), 7 % for the 100-hour time lag (diameter of dead fuel between 25 and 75 mm) and 5 % for necromass of aerial fuels. Herbaceous fuels were assumed to be dry, and the FMC was set at 5 % (Scott and Burgan 2005). The FMC of live fuels was fixed at 50 % according to a very low level of live moisture content cited by Viegas et al. (2001) in Mediterranean ecosystems. These values assume extreme meteorological conditions and are common in the study area during some summers in the Iberian Peninsula (Cardil et al. 2015) as in the Cortes de Pallás event. Crown fire and spotting options of the FARSITE program were selected to relate surface and crown fires.

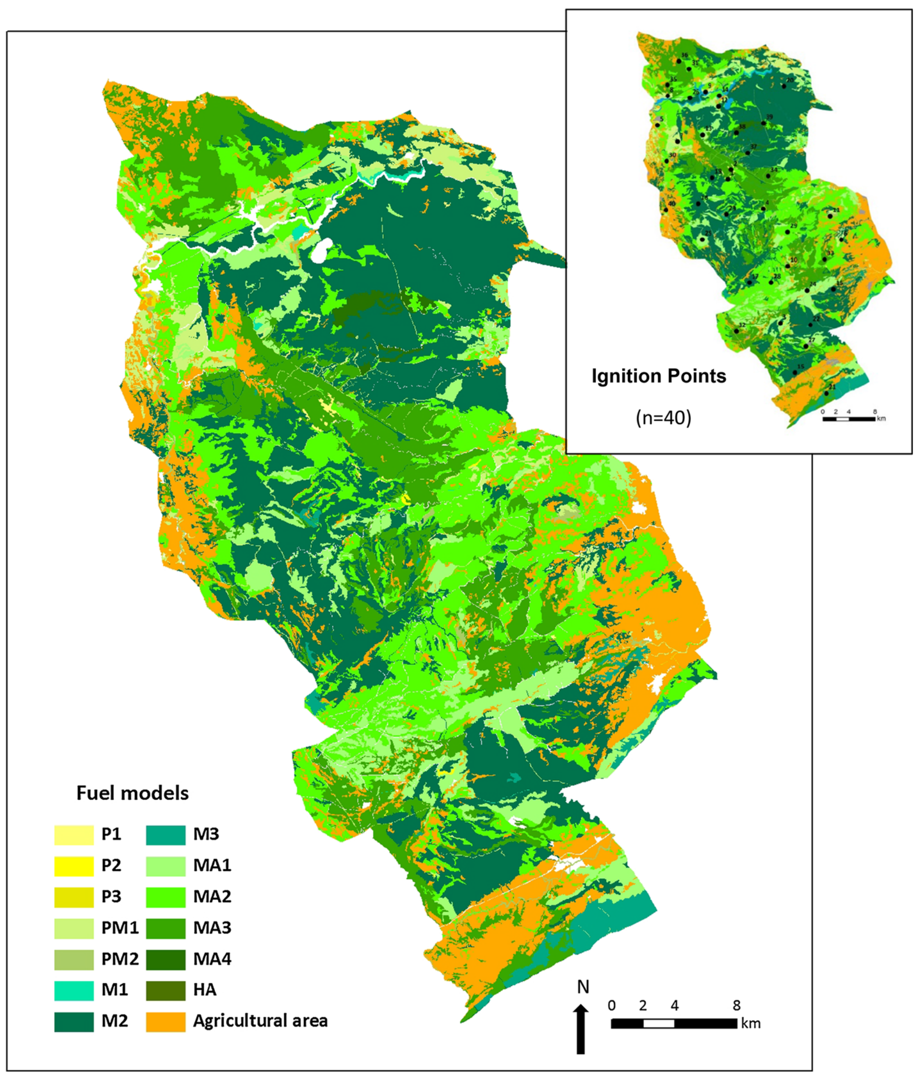

Fig. 2 Current forest fuel model Scenario (A) showing surface models (local firefighters code, Online resource 1, Table S1-2) and selected ignition points

Management Scenarios

Forest fire behaviour and burnt area were assessed by considering different Scenarios (Online Resource 3): (1) Scenario A: current forest fuel model (i.e. unmanaged); (2) Scenario B: rural abandonment in which current agricultural land is converted into pasture and shrubland forest fuel models; (3) Scenario C: biomass management through thinning; (4) Scenario D: biomass management through forest stands thinning and understory clearing.

Scenarios B, C and D were established according to the results of the European LIFE + project "Bioenergy and Fire Prevention" (http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction = search.dspPage&n_proj_id = 3653, Life 09 EN-E.000450). This project was developed in the study area (the Caroig Massif).

Scenario B is an ongoing process in the study area and is considered the most likely future Scenario. It was assessed using field inventories (Online resource 3, Fig. S3-1). These data ($n = 200$ plots) showed that a high proportion of agricultural areas have been abandoned, giving rise to fuel load available to burn in extreme meteorological conditions, generating greater continuity between agricultural and forest land (Ortega et al. 2012). Field inventories suggested transformation of cereals into grasslands models (P1 and P2) and tree agricultural plantations into model PM1 because shrubs such as *Cistus* sp. and *Thymus* sp. were observed in abandoned plots. Scenario C was established by taking into account that production of bioenergy from wood is currently the most common system for managing biomass for energy purposes in forests. Shrub is therefore managed in this Scenario only in maintaining fuel breaks. Field inventory data and analysis of forest stand variables suggested the suitability of *P. halepensis* for bioenergy purposes. Silvicultural treatment (thinning) was proposed for sustainable management (Online resource 3, Fig. S3-2) by prioritizing the stability and future regeneration of the stands studied (Montero et al. 2001). This Scenario would be realistic if wood for bioenergy purposes is considered in a future rural development policy (Marino et al. 2014b). An additional Scenario D was established by considering tree thinning and mechanical understory removal in forest and mechanical clearing in grassland and shrubland areas to reduce fire hazard. Although this Scenario is not realistic at present, increased innovation in biomass management (new machinery and boiler systems) could make the management of shrubs for energy purposes profitable (e.g. LIFE+ project ENERBIOSCRUB). Assuming a dynamic process of shrub regeneration after clearing (Pasalodos-Tato et al. 2015), we consider that a reasonable treatment for reducing ecological impacts at landscape level (every 5–8 years) would be to reduce the surface fuel load in order to transform models MA1 (11.51 Mg/ha) and MA3 (5.79 Mg/ha) into models MA2 (4.44 Mg/ha) and additionally M3 (4.1 Mg/ha) and model M2 (2.91 Mg/ha) into model M1 (1.76 Mg/ha) (see details in Online Resources 1 and 3, Table S1-3 and Table S3-1).

Output variables and statistical analysis

The following output variables were considered in the simulations: total burnt area (BA, ha); burnt perimeter (BP, km); fire line intensity, according to Byram (1959) (FLI, kW/m); flame length (FML, m); rate of spread (ROS, m/min); heat per unit area (HPA, MJ/m^2); reaction intensity, according to Rothermel (1972) (RCI, kW/m^2); and crown fire activity, according to Van Wagner (1977) (CFR, 1 = surface fire; 2 = passive fire; 3 = active fire). FARSITE provides an output simulation map (30 m × 30 m grid) for every variable studied. Simulations ($n = 40$) were analysed per pixel (30 m × 30 m) by calculating the mean value, maximum value and standard deviation of predicted output variables (FLI, FML, ROS, HPA and RCI) for each simulation. The relative percentage of pixels classified as passive and active fire was calculated for each simulation (CFR, %).

Nonparametric Wilcoxon tests were used to compare differences between current (A) and simulated Scenarios B, C and D for all variables (mean, mean of maximum and mean of standard deviation). A partial least squares (PLS) model was developed (Eq. 1) in order to explore the effect of the most intensive forest biomass management Scenario (D) in reducing fire hazard (two levels, A vs. D), the effect of the forest fuel model in which wildfire is initiated and their interactions. The average fire behaviour parameters (FBP) simulated outputs (BA, BP, FLI, FLM, ROS, HPA, RCI, CFR) were used as dependent variables. Scenario, fuel model and their interactions were used as predictors. The PLS technique enables a simultaneous linear fit by using autocorrelated variables with a small number of data ($n = 40$) relative to the number of variables (in this case 25 categorical predictors and 8 dependent variables). The model was developed by establishing three PLS components to explain the variability of dependent variables (simulation outputs FBP). The PLS output is easy to interpret because the algorithm generates linear models for each FBP (BA, BP, FLI, FML, ROS, HPA, RCI and CFR):

$$\mathrm{FBP} = a + b\,\mathrm{Scenario} + \sum_{i=1}^{12} c_i \mathrm{Fuel\ Model}_i + \sum_{j=1}^{12} d_j (\mathrm{Scenario} \times \mathrm{Fuel\ Model}_i) \quad (1)$$

where Scenario is a categorical dummy variable that defines the Scenario (A vs. D), Fuel Model$_i$ is a categorical variable that defines the 12 fuel model ($i = 12$ levels) according to the fuel model of selected ignition points against the lowest risky fuel model, litter under canopy ($Model_i$ vs. HA). The interaction between *Scenario* and every *Fuel Model* is also included as predictor (*Scenario x Fuel Model$_i$*). Scaled model coefficients (a, b, c_i, d_j) were used to interpret the results according to their absolute value (a higher value implies a higher relative importance of Scenario, model or interactions to explain FBP values) and sign (a positive value implies a direct relationship between independent variable and FBP variable, and the relation is inverse for a negative value). The model fit was evaluated by the R^2Y (similar to R^2-adjusted of a parametric model), and R^2X statistics provides information about the autocorrelation of predictors variables (Wold 1985). The statistical analysis was carried out using the Statistica 10.0 package©.

Results

The results showed an increase in the average predicted burnt area (BA), fire line intensity (FLI), flame length (FML), reaction intensity (RCI) and activity of crown fire events (CFR) in Scenario B relative to Scenarios A, C and D (Table 2). All values varied widely (see standard deviation, Table 2), indicating the high variability of local conditions in the simulated events. Wilcoxon tests showed a significant increase in the average values for forest fire behaviour parameters in Scenario B (BA, BP, CFR, FLI_MAX, FML_MAX) relative to the current Scenario A (Table 3). Biomass management through thinning as the main silvicultural treatment (Scenario C) reduced canopy bulk density (Online resource 3, Table S3-1); however, no differences in the average values of the main parameters were detected, relative to the current situation (Table 3). All parameters studied are significantly reduced in Scenario D, relative to the Scenario A (Table 3).

According to these results, the significant reduction in surface fuel load in Scenario D relative to Scenario C is more effective for reducing fire hazard in the study area. The burnt area by land cover type (Table 4) shows the fuel type effects on fire hazard on the different land covers. Scenario B increased BA regarding Scenario A in cover types P (grasslands) and PM (grass and shrubs) because agricultural lands are converted into pasture and shrubs fuel models. Scenario C was quite similar to Scenario A; therefore, thinning in forest cover types (MA and HA) did not show changes on BA. Scenario D showed a strong effect on burnt area in cover types M (shrubs) and MA (forests and shrubs under canopy), with a reduction of 50 % of the total burnt area regarding Scenario A. This reduction in BA in Scenario D agrees well with the reduction in fire behaviour parameters by cover type (Online resource 4, Table S4-2).

The partial least squares (PLS) model coefficients including Scenarios A and D are shown in Fig. 3. Scenario, Fuel Models and their interactions explained more than 50 % of the FBP variability (model fit: $R^2Y = 0.54, R^2X = 0.25$, 3 components). Therefore, the combination of local topography and their interactions with vegetation and meteorological conditions (fire triangle) would explain the rest of the variability in all FBP. Scenario D had a greater effect than Scenario A (highest scaled coefficient for most of the dependent variables), confirming the results of the Wilcoxon tests (Table 3) and highlighting the positive effect of the intense modification of fuel load at landscape level under extreme meteorological conditions. Examination of the fire simulations showed that thinning and understory clearing (Scenario D, transformation of MA1, MA3 and MA4 into MA2 fuel models), including biomass reduction in extensive shrub fuel model areas (transformation of M2 and M3 into the M1 forest model), significantly reduce fire hazard relative to the current Scenario A (Table 3; Fig. 3).

Significant effects of fuel models and the interaction between Scenario and fuel models in which ignition point was included were detected for some FBP (Fig. 3). Burnt area (BA), burnt perimeter (BP) (Fig. 3a) and average rate of spread (ROS_MEAN, Fig. 3b) increased significantly when the ignition point was included in models PM1 and P3 (mixed grass and shrub models), and they decreased significantly in MA2, MA3 and MA4 (forested areas with low understory shrubs biomass). Nevertheless, flame length (FML), fire line intensity (FLI) (Fig. 3b), heat per unit area (HPA), reaction intensity (RCI) (Fig. 3c) and passive crown fire activity (CFR) (Fig. 3d) were significantly higher when wildfire started in forest areas with high amounts of available understory fuel (model MA1). The interaction between fuel model and Scenario was significant in models M2 and M3 (Fig. 3a, 3b), corresponding to shrub areas with high biomass fuel load. In summary, the results show that the model in which fire is initiated has a significant influence on the average FBP parameters which affect adjacent forest fuels. The initial forest fire behaviour affects burnt area and average FBP values in each simulation. The simulations highlight the importance of the connectivity between areas covered by shrubs and grasses (higher rate of spread and fire line intensity) and the forest with high shrubs fuel load under canopy (higher fire line intensity, heat per unit area) which increase the crown fire activity.

Discussion

Results show the difficulties in reducing wildfire propagation and hazard by managing vegetation under extreme meteorological Scenarios (Brotons et al. 2013) and indicate that weather could be more important than fuel in

Table 2 Average values, standard error (in brackets) and range of fire behaviour parameters for 40 simulations ($n = 40$) carried out to characterize potential fire behaviour in the four different Scenarios considered: A (current unmanaged Scenario), B (abandoned agricultural land Scenario), C (management Scenario applying thinning for energy uses) and D (management Scenario applying thinning and bush clearing for energy uses)

	BA (ha)	FLI (kW/m)	FML (m)	ROS (m/min)	HPA (MJ/m^2)	RCI (kW/m^2)	CFR (%)
A	483.13 (312.1)	596	1.3	4.3	7.5	570	9.3
	28.5–1171	(705) 121–21175	(0.6)	(3.5)	(3.1)	(239)	(11.8)
			0.7–45	1.1–126	4.8–150.6	345–2550	0–50
B	850.57 (431.1)	1072	1.6	4.4	7.8	580	9.7
	28.5–1437	(1683)	(0.9)	(5.4)	(5.3)	(212)	(11.6)
		121–21175	0.7–45	1.1–127	4.8–150.6	345–2550	0–50
C	480.94 (326.4)	507	1.3 (0.63)	4.3	7.5	575	9.2
	28.5–1171	(456)	0.7–45	(2.4)	(2.8)	(210)	(11.6)
		63–21,175		1.1–126	4.8–150.6	345–2550	0–50
D	217.25 (189.3)	364	1.1 (0.52)	2.9	6.5	460	5.3
	59.1–980.5	(337)	0.7–8.7	(2.5)	(1.7)	(153)	(10.2)
		121–5919		1.1–36	4.5–85.7	271–1432	0–36

BA burnt area, *FLI* flame intensity, *FLM* flame length, *ROS* rate of spread, *HPA* heat per unit area, *RCI* reaction intensity, *CFR* activity of crown fire (passive and active)

Table 3 Comparison between current Scenario (A) and simulated Scenarios (B, C and D) for selected FBP variables: mean (_MEAN), mean of maximum values (_MAX) and mean of standard deviation values (_SD)

Variable	A versus B		A versus C		A versus D	
	Z test	*p*	*Z* test	*p*	*Z* test	*p*
BA	2.80	0.00,506	0.36	0.71,500	5.06	0.00,000
BP	2.70	0.00,511	0.44	0.65,472	5.23	0.00,000
FLI_MEAN	1.07	0.28,450	1.34	0.17,962	4.58	0.00,000
FML_MEAN	1.58	0.11,413	1.39	0.17,971	4.17	0.00003
ROS_MEAN	0.25	0.79886	0.11	0.98732	4.14	0.00003
HPA_MEAN	1.27	0.20262	1.07	0.28505	2.29	0.03612
RCI_MEAN	0.46	0.64646	0.12	0.91008	3.67	0.00024
CFR_SURFACE	2.80	0.00506	0.11	0.98539	1.88	0.06037
CFR_PASSIVE	2.55	0.01086	0.12	0.91118	4.82	0.00000
CFR_ACTIVE	0.45	0.65472	0.01	0.99999	2.66	0.00768
FLI_MAX	2.02	0.04311	0.05	0.99989	4.37	0.00001
FML_MAX	2.10	0.03569	0.04	0.99995	4.68	0.00000
ROS_MAX	1.57	0.11585	0.10	0.99851	0.32	0.00088
HPA_MAX	1.01	0.31049	0.09	0.99931	4.82	0.00000
RCI_MAX	2.03	0.04252	0.09	0.99923	4.86	0.00000
FLI_SD	0.87	0.38627	0.45	0.65472	2.23	0.00000
FML_SD	0.46	0.64646	0.45	0.65472	2.44	0.02566
ROS_SD	0.66	0.50762	0.12	0.90777	2.78	0.00548
HPA_SD	0.76	0.44458	1.60	0.10881	3.08	0.00205
RCI_SD	0.46	0.64646	0.11	0.98427	4.29	0.00002

The values of the Wilcoxon test statistic and the *p* values are shown

BA burnt area (ha), *BP* burnt perimeter (km), *FLI* flame intensity (kW/m), *FLM* flame length (m), *ROS* rate of spread (m/min), *HPA* heat per unit area (MJ/m^2), *RCI* reaction intensity (kW/m^2), *CFR_SURFACE* activity of surface fire (%), *CFR_PASSIVE* passive crown fire (%), *CFR_ACTIVE* active crown fire (%)

determining fire severity (Bradstock et al. 2010; Lydersen et al. 2014). Nevertheless, Fernandes et al. (2016) stated the complexity of the problem and the difficulty to generalize. Carbon accumulation in forest stands is a dynamic process (Pasalodos-Tato et al. 2015), and thinning and understory clearing must therefore be programmed in

Table 4 Burnt area (BA) by cover types in the simulated Scenarios (A, B, C, D). Mean of burnt area, total area burnt (Tot) and standard deviation (Sd) for FARSITE simulations outputs ($N = 40$) are shown

Scenario	A				B				C				D			
Cover type	Mean	Tot	Sd	%	Mean	Tot	Sd	%	Mean	Tot	Sd	%	Mean	Tot	Sd	%
Grasslands	6.38	255.15	16.73	0.20	212.02	1448.15	86.73	1.11	6.38	255.15	16.73	0.20	6.48	259.20	17.69	0.20
Grass and shrubs	48.75	1949.83	125.12	1.49	197.90	3268.82	230.11	2.50	48.75	1949.83	125.12	1.49	39.18	1567.00	104.80	1.20
Shrubs	209.47	8378.73	268.77	6.42	217.20	8687.90	268.34	6.66	209.47	8378.62	268.77	6.42	45.88	1835.16	43.13	1.41
Forests and shrubs	207.33	8293.06	150.40	6.35	222.10	8884.13	149.41	6.81	207.40	8296.06	150.62	6.36	120.33	4813.06	112.02	3.69
Forest	0.05	1.93	0.30	0.00	0.05	1.93	0.30	0.00	0.05	1.93	0.30	0.00	0.05	1.93	0.30	0.00
Agricultural	8.32	332.63	14.11	0.25	0.00	0.00	0.00	0.00	8.40	335.82	14.18	0.26	4.21	168.53	8.93	0.13
Riparian vegetation	2.85	23.35	15.02	0.02	1.30	15.47	2.56	0.01	0.50	4.53	1.44	0.00	1.13	12.68	5.61	0.01
Total	483.13	19,234.67	312.10	**14.73**	850.57	22,306.40	431.10	**17.09**	480.94	19,221.94	326.40	**14.72**	217.25	8657.56	189.30	**6.63**

Percentage of burnt area (%) regarding area of study (130, 542 ha) is shown for each cover type. Percentage of simulated burnt area regarding area of study in *bold*

order to maintain low canopy bulk density and low surface fuel load to make this fuel structure consistent with the sustainability of ecological processes such as regeneration of vegetation and wildlife habitats (Moritz et al. 2014, Malico et al. 2016). Fuel reduction treatments in many Mediterranean ecosystems only reduces fire hazard in the short term because shrubs recover quickly (Fernandes 2015; Fernández and Vega 2014; Marino et al. 2011, 2014a). Maintenance of low shrub biomass levels at landscape level by mechanical bush clearing is currently unrealistic (not profitable for biomass industry). Other treatments have been suggested for reducing fuel loads, such as controlled grazing (Ruiz-Mirazo et al. 2011) and prescribed burning (Fernandes 2013). However, these treatments could generate problems related to social acceptance and biological impacts, mainly in protected areas. Nevertheless, simulation outputs show that only large modifications in forest fuel models require intensive silvicultural treatments repeated over time (Stephens and Moghaddas 2005) and their efficacy varies greatly in different ecosystems (Price et al. 2015).

The absence of significant differences between Scenarios A and C is consistent with the results found by Oliveira et al. (2016). Comparison of Scenarios A and B revealed significant differences in some variables analysed (Table 3) and comparison of Scenarios A and D revealed significant differences in all variables analysed including the reduction in crown fires (Table 3). Therefore, *P. halepensis* stands characterized by a regular structure with a single layer of large trees, low tree density and low understory due to higher canopy closure (Scenario D) are the least vulnerable to crown fire (Álvarez et al. 2012; Jiménez et al. 2016). The limitations of the simulation approach using FARSITE (Cruz and Alexander 2010; Alexander and Cruz 2013, Benali et al. 2016) could modulate the results presented. Most of the mentioned authors advise about the underestimation of crown fire activity (CFR) of Van Wagner (1977) model regarding field data. For that reason, the significant decrease in CFR predicted in Scenario D could be negligible using other crown fire models (e.g. Cruz et al. 2012). Therefore, the commented limitation to predict crown fire events could overestimate the efficacy of Scenario D to reduce crown fire activity. Some authors suggest that active crown fire during extreme meteorological conditions does not depend on the forest fuels present (Bradstock et al. 2010, Jiménez et al. 2013). The climatic change Scenario could, consequently, limit the efficacy of fuel load reduction in preventing crown fires (Miller et al. 2009, Moritz et al. 2014).

The results also show that the increase in grassland and mixed shrub and grassland (P and PM areas) leads to higher fire hazard in Scenario B (transformation of abandoned agriculture land) than in Scenario A (Table 3). In

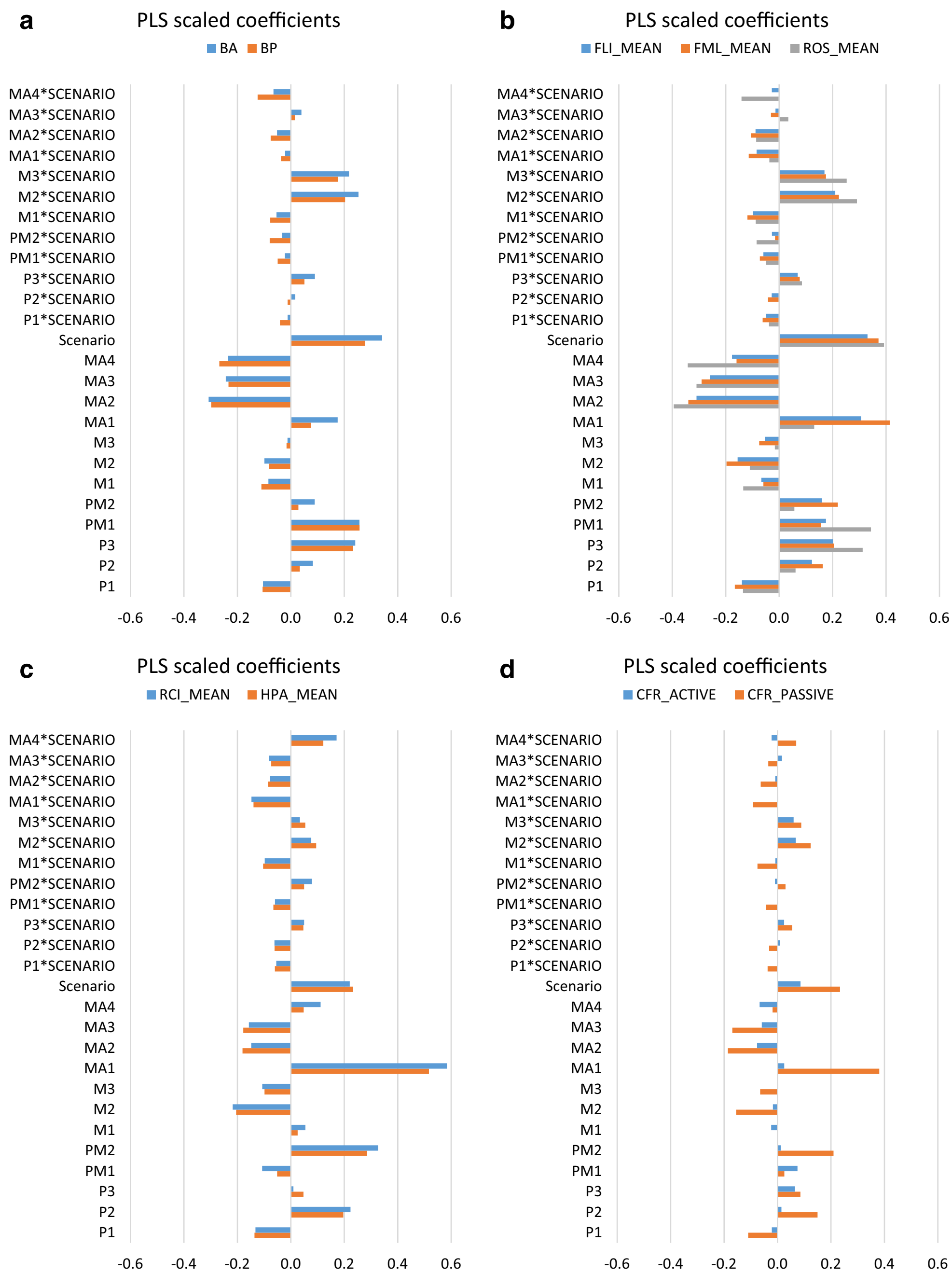

Fig. 3 Scaled coefficients for partial least squares (PLS) models (*coloured bars*) for predicting average fire behaviour parameters (FBP): **a** burnt area (BA), burnt perimeter (BP), **b** fire line intensity (FLI), flame length (FL), rate of spread (ROS), **c** reaction intensity (RCI), heat per unit area (HPA) and **d** crown fire activity (CFR) using the following categorical predictors: *forest Fuel Model* in which wildfire is initiated (all models vs. HA model), *Scenario* (A vs. D) and their interactions (*Scenario* × *Model*). Scaled coefficient signs indicate the positive or negative effect of each predictor on FBP parameters, and the absolute value is a measure of the relative importance of each predictor in generating the PLS model (model fit $R^2Y = 0.54$, $R^2X = 0.25$, 3 components)

addition, these models showed high and positive values of scaled coefficients for predicting FBP variables on comparing Scenarios A and D (Fig. 3). Models P and PM are considered high risk because of the associated high ignitability and rate of fire spread of these types of vegetation (Marino et al. 2011, 2014a). The area covered by these types of vegetation is increasing due to rural abandonment (Martínez et al. 2009; Ortega et al. 2012) and fire-prone adaptive traits (Pausas et al. 2008), thus increasing the fire hazard and potential burnt area (Scenario B, Table 2). This process has been recognized in the rest of the Mediterranean basin (Moreira et al. 2011). Shrublands in the study area composed by highly flammable species and fire-prone ecosystems comprise potential fuel, generating high energy, high suppression constraints and safety problems for firefighters (Fernández and Vega 2014). Agroforestry activities (e.g. extensive grazing, wood plantations for energy purposes) in models P and PM and shrub clearing for bioenergy in these large areas (in the case of available technology) therefore significantly reduce fire hazard at landscape level (Fernandes 2013, Marino et al. 2014b). The effectiveness of agroforestry activities and shrub clearing depends on many factors (type of activity, extension, spatial distribution) and their socioeconomical impact (Duguy et al. 2007). Surprisingly, this effect may also be important for assessing crown fire activity in forest areas derived from grassland and shrubland (high and positive values of scaled coefficients for models P2, PM2, Fig. 3d). This may be explained by the combined effect of high FLI and FML and low crown base height (average of 2 m). This generates high susceptibility to passive crown fires (high FLI and low CBH, according to Van Wagner 1977) in many areas where grasslands and shrublands are in contact with forest stands. Brotons et al. (2013) pointed out that grasslands, mixed grass and shrubland areas may also be important for predicting the probability of crown fire activity in forest stands at landscape level in the Mediterranean basin characterized by high fuel connectivity. Fuel connectivity is higher in unmanaged or rural abandonment Scenarios (A, B), and climate change will further increase fire risk (Flannigan et al. 2009). Therefore, this study confirms the susceptibility of unmanaged Mediterranean areas to wildfire in the context of climate change (Moritz et al. 2014) and also the low level of suppression opportunities expected in the future. This pessimistic view contrasts with the conclusions reached in this and other studies (e.g. Álvarez et al. 2012, Regos et al. 2016), suggesting that reduction in fuel load will at least reduce fire severity, providing the opportunity of fire-prone ecosystems to regenerate after fire (Pausas et al. 2008), reducing ecological impacts (Bond and Keeley 2005) and increasing the resilience of Mediterranean ecosystems (Collins et al. 2009; Moritz et al. 2014; Parks et al. 2014).

Overall, the simulations and comparisons of the different Scenarios suggest that the removal of forest biomass for energy uses will only reduce fire hazard at landscape level under extreme meteorological conditions when fuel load is managed in forest stands (canopy and understory) and in large shrubland areas and grasslands around forests (Fig. 3). Brotons et al. (2013) identified the interactions between fire regime and fire suppression via human-affected landscape patterns (such as the Caroig Massif) and assessed the need for intense suppression activities to increase the efficacy of preventive actions at landscape level in the context of climate change. An intermediate Scenario in which part of fuel load would be treated is more realistic (Regos et al. 2016). Moreover, although it would probably reduce fire severity (Stephens et al. 2009a, b), it would not reduce the average values of potential fire behaviour parameters (Boer et al. 2009; Fernandes 2015). Regos et al. (2014) demonstrated that post-fire regenerated areas (shrubland and mixed areas of seedling shrubland) close to forests are strategic areas for wildfire suppression, as they modulate the shape and surface of subsequent wildfires in NE Spain. Therefore, a Scenario in which less intensive treatments are carried out in strategic areas to increase suppression activities may be more realistic and promising option (e.g. prioritizing strategic areas in P2, PM2, M3 and MA1 forest models according to PLS results, Fig. 3) (Fernandes 2013; Calkin et al. 2014; Regos et al. 2016).

Conclusions and management implications

Forest biomass collection for energy is expected to reduce fire hazard in the Mediterranean basin if management and land use strategies include important changes in fuel models, including management of shrub biomass at the landscape level. Shrubs could be harvested for energy in the context of a future forest biomass management Scenario (Pérez et al. 2014), thus contributing to reducing fuel load in such areas. However, this Scenario is not realistic at present, while the demand for biomass products is mainly focused on wood harvesting (Scenario C). Therefore, aerial biomass removal from the tree stratum should be complemented with controlled grazing, agroforestry activities and prescribed burning (Scenario D) to reduce surface fuels. Such integrated fire management actions represent a more efficient tool than the present Scenario in relation to reducing fuel load and decreasing fire hazard. A strategic plan in which only 10 % of the landscape is treated would reduce fire hazard at a much lower cost than associated with suppression and structure (North et al. 2012). It is generally agreed that large investments are required to manage fuels or to prevent fuel ignition over entire landscapes and the financial resources available are always limited. Lafortezza et al. (2015) highlighted the need to prioritize fuel removal in densely populated landscapes in order to maximize the number of people affected by wildfire suppression per dollar spent on fuel removal in the Mediterranean basin.

Activities aimed at promoting rural development would also slow down rural abandonment, which might increase fire hazard as a result of the transformation of agriculture land into shrubland and forest fuel models. However, subsidies are required to enable mechanical bush clearing to be carried out and thus prevent unwanted forest fires.

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