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RESEARCH ARTICLE

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The multi-objective Spanish National Forest Inventory

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Abstract

Aim of study: To present the evolution of the current multi-objective Spanish National Forest Inventory (SNFI) through the assessment of different key indicators on challenging areas of the forestry sector.

Area of study: Using information from the Second, Third and Fourth SNFI, this work provides case studies in Navarra, La Rioja, Galicia and Balearic Island regions and at national Spanish scale.

Material and Methods: These case studies present an estimation of reference values for dead wood by forest types, diameter-age modeling for *Populus alba* and *Populus nigra* in riparian forest, the invasiveness of alien species and the invasibility of forest types, herbivore preferences and effects on trees and shrub species, the methodology for estimating cork production , and the combination of SNFI4 information and Airborne Laser Scanning datasets with the aim of updating forest-fire behavior assessment information with a high degree of accuracy.

Main results: The results show the suitability and feasibility of the proposed methodologies to estimate the indicators using SNFI data with the exception of the estimation of cork production. In this case, additional field variables were suggested in order to obtain robust estimates.

Research highlights: By broadening the variables recorded, the SNFI has become an even more important source of forest information for the development of support tools for decision-making and assessment in diverse strategic fields such as those analyzed in this study.

Additional keywords: invasive species; forest modelling; dead wood; browsing impact; fire hazard; cork production

Abbreviations used: AIC (Akaike´s information criterion); ALS (airbone laser scanning); CBD (canopy bulk density); CBH (canopy base height), CFL (canopy fuel load); DWV (dead wood volume); NFI (National Forest Inventory); SNFI (Spanish National Forest Inventory)

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Introduction

There has been a shift in the aims of forest policy and forest management in Europe from wood production to sustainable ecosystem management, which should consider all the goods and services provided by the forest. To address these increasing, new demands for information, intensive monitoring of the status of forests is required. Forest information systems are needed in order to formulate and implement global forest policy and forest management and in turn, depending on the aims of the forest policy, forest inventory systems are modified to meet the demands for information (Fig. 1). The main international processes or requirements demanding forest information are the United Nations Framework Convention On Climate Change (UNFCC) and the Kyoto protocol, the Global Assessment of Forest Resources (FRA), required by the United Nations Food

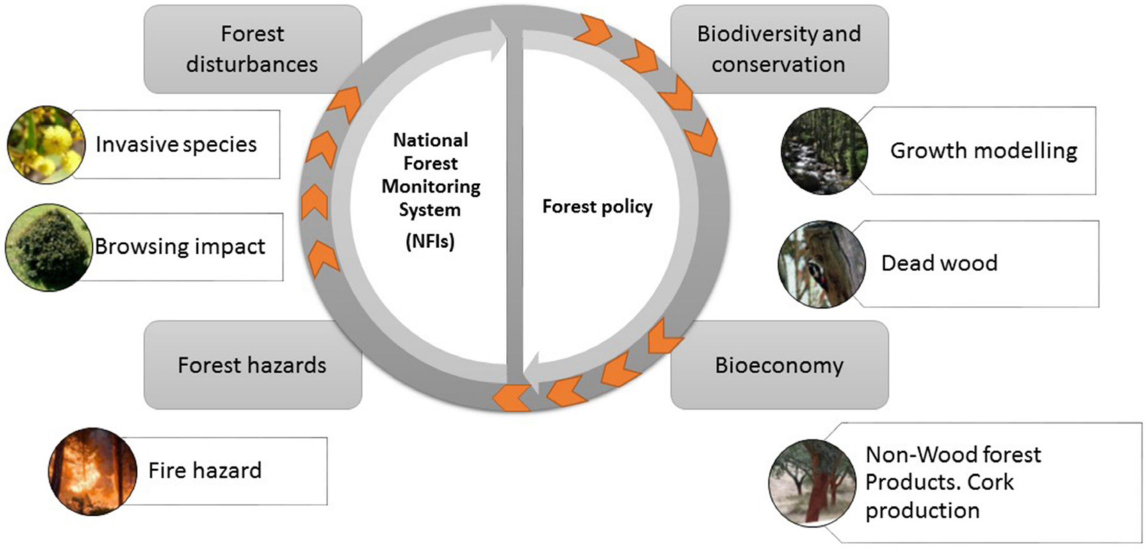

Figure 1. Diagram of the role of forest information systems in forest policy processes where each case study topic analyzed in this article was framed.

and Agriculture Organization (FAO) and the Criteria and Indicators for Sustainable Management, reported on the State of European forests (SoEF) requested by the pan-European process for the protection of forests in Europe (FOREST EUROPE) (Vidal *et al.*, 2016). These processes require comprehensive forest information, covering aspects as varied as growing stock, carbon pools and non-wood forest products related with the green economy as well as information on forest biodiversity, forest risks and disturbances, or social indicators.

Additionally, according to the European forest strategy (EC, 2013), the Commission and the Member States should set up of the Forest Information System of Europe by collecting harmonised Europe-wide information on the multifunctional role of forests and forest resources integrating several modules, *e.g.* on forests and natural disturbances, forest and the bioeconomy, forests and climate change and forest and ecosystem services. Therefore, there is a continued need for assessment and estimation of forest indicators at EU level to support the development and implementation of a number of European environmental policies as well as to identify appropriate forest management practices (Vidal *et al.*, 2016). National forest inventories (NFIs) are the primary source of forest data for national and large-area assessments due to the high degree of monitoring and the diversity of variables recorded at national level. The scope of NFIs has been broadened to include new variables to meet these increasing information requirements (Tomppo *et al.*, 2010).

In Spain, the necessity for homogenous and objective forest statistics for decision making at national level provided the impetus for undertaking the First National Forest Inventory (SNFI1) between 1965 and 1974. Due to circumstances related to the national infrastructure, the Second Spanish National Forest Inventory (SNFI2) did not commence until 1986. Since the NFI2, there has been a continuous inventory with permanent plots in most of the Spanish regions operating over a cycle of approximately 10 years. From the second cycle onwards the plots are permanent, located at the nodes of a 1×1 km grid, thus enabling comparisons. Since the Third National Forest Inventory (SNFI3, 1997-2007), land cover classification and forest area estimation are described prior to the NFI using the National Forest Map, (E 1:50,000; NFM50). The Fourth National Forest Inventory (SNFI4) is currently ongoing and the area of each NFI4 forest strata is estimated using the NFM25 (E 1:25,000), adding the tessera (*i.e.* basic unit, having a specific land use with homogeneous forest structure and forest type) belonging to each stratum (Vallejo & Sandoval, 2013).

The main aim of this paper was to present applications (case studies) of the SNFI as a multi-objective inventory capable of responding to the demand for forest information regarding forest biodiversity and conservation, bioeconomy, forest hazards and forest disturbances (as highlighted in the European Forest Strategy) (Fig. 1). The specific objectives of the case studies, chosen for their relevance as well as novelty, were as follows: i) to quantify and characterize dead wood volume by forest type; ii) to model the relationship between age and diameter for riparian species; iii) to characterize and analyze the spread of invasive species; iv) to estimate the impact of livestock and wildlife

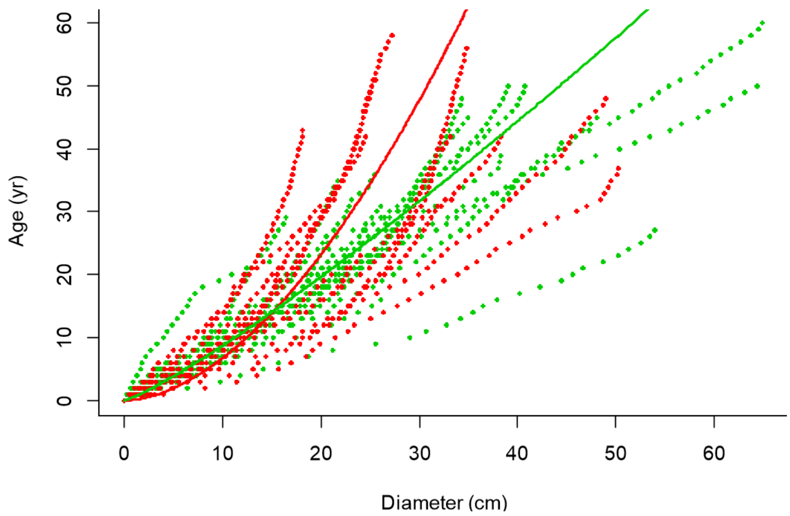

Figure 2. Relationship between age (yr) and diameter (cm) for the entire dataset. *Populus alba* values are represented by red dots and its model by a red line, and *Populus nigra* values are represented by green dots and its model by a green line.

browsing; v) to estimate cork production at national level, and vi) to develop models to estimate the main canopy fuel complex characteristics needed to assess crown fire potential using airbone laser scanning (ALS) metrics as regressors.

Material and methods

In Spain, land cover classification and forest area estimation are described prior to the SNFI using the National Forest Map (Vallejo & Sandoval, 2013). The SNFI covers all forest land in Spain. From the second cycle onwards, the plots are permanent, enabling growth comparisons and stratification to be undertaken post-sampling. Sample plots are established at the intersections of a 1×1 km UTM grid (Alberdi *et al.*, 2010). Field plots consist of four concentric circular fixed areas with radii of 5, 10, 15 and 25 m (Alberdi *et al.*, 2016).

All the different datasets of the case studies were extracted from SNFI2, SNFI3 and overall, from SNFI4, which started in 2008 and is currently ongoing, with field data collection and data processing work progressing simultaneously.

Dead wood volume quantification

Data from 2,396 SNFI4 plots in Navarra province located in Northern Spain were used in this study. In this area, the three bio-geographical regions, Mediterranean, Atlantic and Alpine, are well represented.

Dead wood components data were recorded in the SNFI 15 m radius subplot. The dead wood components considered were as follows: dead standing and downed trees; dead standing and downed saplings; downed coarse wood pieces; stumps and accumulations. Trees as well as shrub species were recorded and the five decay classes proposed by Hunter (1990) and Guby & Dobbertin (1996) were considered. Downed trees and saplings are recorded when the stump or thickest end is within the plot. Other dead wood components are recorded when more than 50% of the piece is inside the plot (Alberdi *et al.*, 2014). The methodology to estimate tree volume (either standing or downed) was described in Crecente-Campo *et al.*, (2015). The aim of this study was to present the dead wood volume reference values in different forest ecosystems.

Species growth models

Cores of the dominant trees of the three main species per plot are extracted in the SNFI4 plots (25 m radius). With this information, two nonlinear mixed age-diameter models have been developed for the species *Populus alba L.* and *Populus nigra L.* (Eq. [1]) in order to estimate the tree age depending on its diameter. They incorporate a random effect for each individual tree. Cores were air dried, mounted and sanded and measured with a LINTAB measuring table (Rinntech, Heidelberg, Germany) with an accuracy of 0.01 mm. A total of 1,233 records of a total of 32 trees have been used (Fig. 2) to fit the mixed models with the statistical package *nlme* (nonlinear mixed-effects models) of the *R software program* (R Development Core Team, 2014). These trees were distributed in the Mediterranean biogeographical region in plots located in Central-Northern Spain (La Rioja and Madrid regions).

Table 1. Invasive species list selected from the additional biodiversity inventory of Galicia SNFI4 (2008). The functional type (FnT), mean density (individuals/ha) of the plots in which they are present, total number of plots where they occur and the total percentage per province in the whole forest area studied (Galicia) are detailed by species.

Species	FnT [1]	Density	Provinces				Galicia (%)
			La Coruña	Lugo	Orense	Pontevedra	
			Total number of plots				
Acacia dealbata Link	T	4,226.0	37	18	70	42	2.76
Acacia longifolia (Andr.) Willd.	T	318.3	1	-	-	-	0.02
Acacia mearnsii De Wild.	T	2,562.3	-	-	-	3	0.05
Acacia melanoxylon R.Br.	T	1,456.0	54	23	3	111	3.15
Arundo donax L.	H	9,549.3	-	-	1	1	0.03
Baccharis halimifolia L.	H	-	-	-	-	-	-
Cortaderia selloana Schult. & Schult. f.	H	11,5652.6	3	1	-	1	0.08
Phyllostachys spp.	H	-	2	1	-	2	0.08
Phytolacca americana L.	H	7,957.7	1		5	-	0.10
Prunus laurocerasus L.	T	294.4	37	12	1	7	0.94
Reynoutria japónica Houtt.	H	38,197.2	-	1	-	1	0.03
Tradescantia fluminensis Vell.	H	-	-	-	-	2	0.03
Tritonia crocosmiflora (Lemoine) N.E.Br.	H	99,471.8	4	1	-	-	0.08
Total (%)	-	-	6.01	3.72	7.53	14.74	7.36

[1] T: trees; H: herbaceous

$$t_{ij} = (\beta_0 + b_{0i}) \times d_{ij}^{\beta_1} + \varepsilon_{ij}$$
$$b_{0i} \sim \mathcal{N}(0, \Psi_1) \text{ and } \varepsilon_{ij} \sim \mathcal{N}(0, \sigma^2) \quad [1]$$

where t_{ij} is the response variable (age expressed in years), j is the observations for every ring of the tree, i is the tree level, d_{ij} is the predictive variable (diameter expressed in cm), β is the fixed estimated parameter, b_i is the mixed estimated parameter and ε_{ij} is the error term. Ψ_1 is the error correlation structure and σ^2 is the variance.

The significance level considered was 0.05 and the nonlinear mixed models were fitted through maximum likelihook. The Akaike´s information criterion (AIC) was used to compare alternative nonlinear mixed models (Zhao *et al.*, 2005; Wang *et al.*, 2007) and select the better models for each species (data not shown). The suitability of the models was also evaluated through the significance of the parameters of the models, the analysis of the model residuals and to evaluate the goodness of fit of the model we also used the root mean square error (RMSE) according to Willmott (1982).

Invasiveness and invasilibility

This case study was based on SNFI4 information for the four provinces which comprise the region of Galicia, in the north-west Iberian Peninsula. This region presents a dominant Atlantic climate, acidic soils and a complex topography with altitudes ranging from sea level up to 2,124 m.

To study this indicator, prior to the field work, a list of invasive species likely to be found in forested areas of the monitored province is drawn up for the additional biodiversity monitoring plots (Table 1; Sanz Elorza *et al.*, 2004; Fagúndez & Barradas, 2007; Romero Buján, 2007). The invasive tree, shrub and herbaceous species previously listed were then recorded in 10 m, 5 m and 1 m radius subplots respectively. In addition, their presence in the 25 m radius SNFI plot was registered.

To analyze the characteristics of the invasive species in the study area, the alien species were classified by functional type and their mean density in the study area assessed based on SNFI field plot data. To examine the rate of spread of each species, we then assessed the degree of invasion of the forested study area as a whole and by forest type, based on the presence/ absence information from the SNFI records for Galicia (5,993 plots). Finally, the vulnerability to the invasion of different forest types was analyzed through the total number and proportion of plots invaded as well as by the number of the different species hosted.

Browsing impact

Data from 900 plots in the SNFI4 corresponding to Balearic Islands, located to the East of Spain, were used

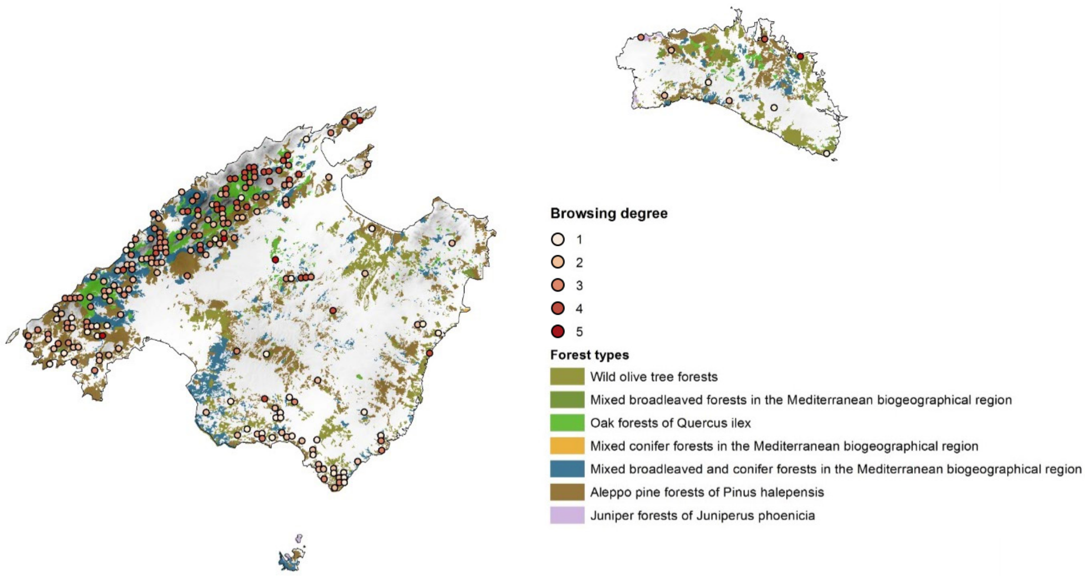

Figure 3. Degree of browsing impact per NFI plot and the distribution of forest types in the Balearic Islands (from Spanish Forest Map 1:50000).

in this study (Fig. 3). In the SNFI4, browsing impact data are recorded in the 10 m radius subplot for trees, saplings and shrub species and in the 5 m radius subplot for tree regeneration. For each species, crown cover was estimated as a proxy of browse availability. Average browsing degree, indicating browse utilization, was also recorded by species according to the classification by Fernández-Olalla *et al.* (2006), based on Etiènne *et al.* (1995) and Aldezábal & Garín (2000). The 6-rank browsing degree classification was as follows: (1) no browsing evidence; (2) slight browsing evidence: only a few twigs browsed; (3) low browsing intensity: plenty of twigs browsed but clearly under 50% of potential browsing biomass; (4) intense, although sustainable, browsing: plenty of twigs browsed, around 50% of potential browsing biomass; (5) high browsing intensity: consumption over 50% of potential browsing biomass and clear shaping of the original plant form; and (6) maximum browsing intensity: no or almost no green parts remain.

To study browsing preferences in the study area, the utilization of each woody species was compared with its availability through the forage ratio index (Krebs, 1999; Fernández-Olalla *et al.*, 2006):

$$w_{ij} = \frac{o_{ij} * p_{ij} \Big/ \sum_{i=1}^{n} o_{ij} * p_{ij}}{p_{ij} \Big/ \sum_{i=1}^{n} p_{ij}} = \frac{o_{ij} * \sum_{i=1}^{n} p_{ij}}{\sum_{i=1}^{n} o_{ij} * p_{ij}} \qquad [2]$$

where w_{ij}: forage ratio or preference (selection) index for species *i* in plot *j*; o_{ij}: browsing utilization of species *i* in plot *j; pij*: browsing availability of species *i* in plot *j*, and *n*: number of woody species present in plot *j*.

Cork production

To estimate cork production at national level, plots from the SNFI2 and SNFI3 were used. Since cork-related variables inventoried in SNFI changed from the SNFI2 to the SNFI3, two different approaches were performed to obtain cork production. In the SNFI2 the variables recorded were debarking height and cork thickness. Debarking height was measured in all the trees in the plot and cork thickness was inventoried on 1 to 6 trees per plot. The cork thickness of the rest of the trees in the plot was assumed to have the same cork thickness which was equal to the arithmetic mean of the cork thickness of the trees measured. With these data the volume of cork for an individual tree (V_{cork}) was estimated through the following equation:

$$V_{cork} = dh \cdot ct \cdot pbh \qquad [3]$$

where *dh* is the debarking height (cm), *ct* is the cork thickness (cm) and *pbh* is the stem perimeter at breast height (cm). By multiplying *Vcork* (cm^3) by the average cork density, the weight of cork for every tree was obtained. The cork of each plot was calculated as the sum of the cork weight of all the trees in the plot. The number of plots considered was 2109, with a total of 10,543 cork oaks.

In the case of SNFI3, only the debarking height was measured. Since no direct estimation of the cork volume was possible with these data, a different approach was adopted based on the application of a diameter increment model to data from SNFI2 plots. Hence, to estimate the cork thickness in the trees from the SNFI3, the variables used were the diameter under cork and the cork thickness of all the cork trees from the SNFI2. Using this information, the annual diameter increment of these trees was estimated through the model developed by Sánchez-González *et al.* (2006). All the variables included in this model were directly given for SNFI plots except for site index, which is estimated by the dominant height model developed by Sánchez-González *et al.* (2007). With this increment data, the theoretical diameter under cork at the SNFI3 was estimated by adding the annual diameter increment in SNFI2 multiplied by the number of years elapsed between SNFI2 and SNFI3 to the recorded diameter at breast height under cork in SNFI2. The difference between the estimated diameter at breast height under cork in SNFI3 and the diameter over cork given for the plots in the SNFI3, is the cork thickness (*ct*) for the trees in the SNFI3 plots. Using the estimated cork thickness data and the *dh* reported in the SNFI3 plots, the weight of cork per plot was estimated according to Eq. [3]. In order to allow comparisons between SNFIs, the same plots were analyzed in each. In accordance with this criterion 2109 plots (10,543 cork oaks).

Canopy fuel modeling

In this study, models to estimate the main canopy fuel complex characteristics in maritime pine (*Pinus pinaster* Ait.) and radiata pine (*Pinus radiata* D. Don) plantations in Galicia (NW Spain) were obtained by combining 554 sample plots from the SNFI4 with information based on ALS over the same plots. These variables, related to the available fuel in the aerial layer, were canopy base height (CBH), canopy fuel load (CFL) and canopy bulk density (CBD). CBH is considered the most important variable in estimating the potential of surface fires to transition to crown fires, and CBD in estimating the potential for active crown fire and crown fire intensity. CFL is used to estimate the amount of canopy material consumed in a crown fire and it is useful not only in fire behavior simulations but also in fire effects simulations (Keane, 2015).

ALS information was provided by the Spanish National Aerial Photography Program (Plan Nacional de Ortofotografía Aérea, PNOA). In total, 39 metrics related to canopy cover and height distribution were extracted from ALS pulses and used as regressors for statistical analyses (see Tables 2 and 3). For further details of the procedure used to obtain the ALS metrics, see the steps outlined in González- Ferreiro *et al.* (2013).

In this study, needles and fine twigs (up to 5 mm at the thick end) were considered as available fuel (*i.e.* the fuel that is consumed within the flaming front of a crown fire), and the "load over depth method" first proposed by Van Wagner (1977) was used to calculate CBH and CBD. According to this approach, CFL (the dry mass of available canopy fuel per unit ground area) was assumed to be homogeneously distributed

Table 2. Potential airbone laser scanning (ALS) explanatory variables related to height distribution

ALS metrics related with height distribution (m)	Description
h_{max}	Maximum
h_{mean}	Mean
h_{mode}	Mode
h_{median}	Median
h_{SD}	Standard deviation
h_{CV}	Coefficient of variation
h_{skw}	Skewness
h_{kurt}	Kurtosis
h_{ID}	Interquartile distance
h_{AAD}	Average absolute deviation
$h_{MADmedian}$	Median of the absolute deviations from the overall median
$h_{MADmode}$	Mode of the absolute deviations from the overall mode
h_{L1}, h_{L2}, ..., h_{L4}	L-moments
h_{Lskw}	L-moment of skewness
h_{Lkur}	L-moment of kurtosis
h_{05}, h_{10}, h_{20}, ..., h_{90}, h_{95}, h_{99}	Percentiles
h_{25} and h_{75}	First and third quartiles

Table 3. Potential airbone laser scanning (ALS) explanatory variables related to canopy closure

ALS metrics	Description
PFRAhmean	Ratio of the number of the first laser returns above hmean to the number of first laser returns for each plot
PFRAhmode	Ratio of the number of the first laser returns above hmode to the number of first returns for each plot
PARAhmean	Ratio of the number of the all laser returns above hmean to the number of all laser returns for each plot
PARAhmode	Ratio of the number of the all laser returns above hmode to the number of all laser returns for each plot
PFRA4	Ratio of the number of the first laser returns above 4 m height to the total number of first laser returns for each plot
PARA4	Ratio of the number of the all laser returns above 4 m height to the total number of first laser returns for each plot
CRR	Canopy relief ratio: $(h_{mean} - h_{min})/(h_{max} - h_{min})$

Table 4. Descriptive statistics for the main tree and stand variables corresponding to the sample plot used in this study. *SD*, standard deviation; *d*, tree diameter; *h*, total tree height; *cl*, crown length; *N*, stand density; *dg*, quadratic mean diameter; *G*, basal area; and *H*, dominant height.

	Statistics	*d* (cm)	*h* (m)	*cl* (m)	*N* (stems/ha)	*dg* (cm)	*G* (m^2/ha)	*H* (m)
Maritime pine (n = 436)	Minimum	7.50	1.50	0.10	14	4.33	0.49	3.80
	Maximum	92.00	45.80	23.70	4615	53.47	83.83	28.75
	Mean	27.87	15.90	6.28	911	19.69	21.88	15.09
	SD	12.70	5.94	2.91	735.02	8.30	14.59	5.66
Radiata pine (n=118)	Minimum	7.50	1.30	0.60	15	4.57	0.91	3.90
	Maximum	107.00	39.30	23.30	3191	84.72	78.28	30.37
	Mean	26.58	17.35	8.13	784	21.45	22.55	16.98
	SD	11.28	6.20	3.86	532.86	9.86	13.40	5.70

throughout the aerial layer of the stand; CBH was the vertical distance between the ground surface and the height of the mean crown base of the stand; and then CBD (the amount of burnable canopy fuel by canopy volume) was estimated by dividing the CFL by the canopy length (CL), which was estimated as the difference between the mean height ($\overline{h}$) and CBH. These definitions of CBH and CBD are compatible with the canopy fuel stratum characteristics used in the crown fire initiation and propagation model developed by Van Wagner (1977) and Cruz & Alexander (2010, 2012), which has been implemented in most wildland fire simulation systems. The compatible systems of tree biomass equations developed for maritime pine and radiata pine in Galicia (reported in Diéguez-Aranda *et al.*, 2009) were used to estimate the dry weight of the fine aboveground tree fractions. The systems require measurements of tree diameter and height as input variables. The mean, maximum and minimum values and the standard deviation (SD) for the main tree and stand variables are shown in Table 4.

Potential and linear models to estimate the canopy variables for each species from different combinations of ALS metrics were fitted and the

Table 5. Dead wood (DW) volume in the different biogeographical regions and forest types (with a number of plots greater than 30) of Navarra. SD: standard deviation

	Volume (m³/ha)					Growing stock (m³/ha)	Nplot
	DW	SD	Max DW	Min DW	% DW		
Region							
Alpine	14.05	23.95	231.11	0.00	5.61	236.62	416
Atlantic	10.55	20.07	312.81	0.00	5.36	186.19	839
Mediterranean	5.88	12.75	143.74	0.00	5.37	103.69	1140
Forest type							
Mixed mautochtonous broadleaved and coniferous forest of the Alpine biogeographical region	20.56	39.01	231.11	0.00	258.06	7.38	42
Flood plain forest	19.31	22.38	113.97	0.00	108.77	15.07	81
Mixed forest in the Atlantic biogeographic region	14.01	31.65	312.81	0.00	139.99	9.10	123
Scots pine forest	10.95	16.55	143.74	0.00	191.45	5.41	334
Beech forest	10.29	17.04	168.99	0.00	229.97	4.28	723
Oak forest of *Quercus robur* and/or *Quercus petraea*	10.07	13.20	56.07	0.00	139.22	6.75	81
Mixed autochtonous broadleaves and coniferous forest of the Mediterranean biogeographical region	7.91	16.64	134.83	0.00	84.31	8.58	92
Austrian pine forest	7.18	17.01	117.72	0.00	174.42	3.95	96
Oak forest of *Quercus humilis*	3.51	6.82	36.62	0.00	121.53	2.81	44
Oak forest of *Quercus faginea*	3.25	7.08	46.66	0.00	76.03	4.10	119
Mixed broadleaves forest in the Mediterranean biogeographical region	3.10	4.79	24.28	0.00	72.62	4.09	67
Productive stands of poplar tree and *Platanus hispanica*	2.34	8.25	47.95	0.00	98.65	2.32	33
Aleppo pine forest	2.28	5.47	28.64	0.00	37.77	5.69	120
Oak forest of *Quercus ilex*	1.32	2.72	21.85	0.00	67.10	1.93	175
Juniper species association	0.34	1.30	7.49	0.00	3.40	9.05	37

best models were selected by comparing the values of the RMSE and the model efficiency (ME) (Willmott, 1982).

Results

Dead wood

The average dead wood volume (DWV) for Navarra province was 8.83 m³/ha, presenting a high SD due to its high amount of variation (18.11 m³/ha). This volume represents 6.34% of all the wood volume of forests.

Standing and downed trees present in Navarra account for an equal percentage of DWV (35% each), while the percentage of standing saplings is slightly higher than downed saplings (3% and 2% respectively). Furthermore, the large percentage of branches (18%) in this region should also be mentioned. The dead wood was highly decompose, as classes 4 and 5 account for 60% of the total DWV.

The DWV in the three ecoregions of Navarra differed; from 14.05 m³/ha (SD=24.10 m³/ha) in the Alpine region to 10.55 m³/ha (SD=20.18 m³/ha) in the Atlantic region and 5.88 m³/ha (SD=12.97 m³/ha) in the Mediterranean region (Table 5). However, the average percentage of dead wood for the three ecoregions represents between 5.4 and 5.6% of the total wood volume.

Mixed forest types generally presented higher average DWV. This is the case of "Mixed autochthonous broadleaf and coniferous forests in the Alpine biogeographical region" (20.56 m^3/ha), "Flood plain forest" (19.31 m^3/ha) and "Mixed forests in the Atlantic biogeographic region" (14.01 m^3/ha). In contrast, in forests dominated by Mediterranean species, such us "Juniper forests (*Juniperus* spp.)" (0.34 m^3/ha), "Oak forests of *Quercus ilex*" (1.32 m3/ha) or in managed forests, the figures are lower (Table 5).

Species growth models

For the *P. alba* the best model (Table 6) among the tested ones had an AIC=2,767.09, being the number of observations 554. For the *P. nigra* the best model presented (Table 6) had an AIC=3,290.31 and the number of observations 679. The residuals analysis shows a RMSE of 2.66 years for the *P. alba* model and 2.50 years for the *P. nigra* model. The simplicity of both models and the fact that they showed a good distribution of residuals (Fig. 4) were considered for their selection.

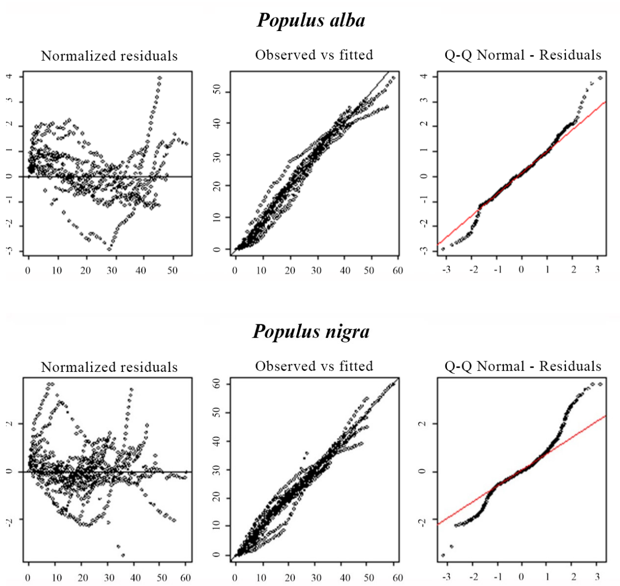

Figure 4. Residuals plots for the *P. alba* and *P. nigra* nonlinear mixed models: normalized residuals, observed vs fitted, and Q-Q (quantile-quantile) normal – residuals.

Invasiveness and invasibility

Even though in terms of composition there were a similar number of invasive herbaceous (7) and invasive tree (5) species recorded, our findings suggests that there is a greater potential for invasion by invasive tree species (mainly *Acacia* spp.) in forest ecosystems of NW Spain (Table 1). *A. melanoxylon* and *A. dealbata* displayed the greatest invasiveness, being present in 3.2% and 2.8% respectively of the total forested area of the study region and exhibiting a high number of trees/ha. Among herbaceous species, *Cortaderia selloana, Phyllostachys* spp. and *Phytolacca americana* were the most widespread in NW forest ecosystems with a high number of individuals/ha. This trend according to functional type was patent if we examine the total percentage of forest area invaded (7.4%) by functional type, with 6.9% corresponding to alien tree species and 0.5% to herbaceous species.

As regards the level of invasion per province, Pontevedra, with a high incidence of fires, showed considerably higher rates of invasion than the other regions. As regards forest vulnerability to invasion, the highest rates of invasion in native forests were found in riparian (11.3%) and *Quercus robur* forests (6.2%) (Fig. 5). Accordingly, riparian forest, together with mixed native forests, presented the highest number of different invasive species of both trees and herbaceous plants. However, the highest degree of invasion was found in altered forests such as non-native mixed forests and *Eucalyptus* spp. plantations.

Browsing impact

Fig. 3 shows the degree of browsing impact for SNFI plots in the Balearic Islands. The greatest browsing damage was found in the northern part of the Tramuntana Mountain range on the island of Mallorca, while moderate browsing was also present on the island

Table 6. Parameter estimates and standard errors (SE) for the models (Eq. [1]) of *Populus alba* and *Populus nigra*. d: diameter at breast height (cm).

Parameter	*Populus alba*			*Populus nigra*		
	Estimate	**SE**	***p*-value**	**Estimate**	**SE**	***p*-value**
Intercept	0.1140938	0.01710143	<0.0001	0.5877771	0.04932257	<0.0001
d	1.7746872	0.02542582	<0.0001	1.1721764	0.01446236	<0.0001
	Estimate	**Lower**	**Upper**	**Estimate**	**Lower**	**Upper**
SD (tree)	0.05480264	0.03691658	0.08135448	0.1741427	0.1242732	0.2440242
SD (residual)	2.689166	2.533235	2.854696	2.525992	2.393343	2.665993

Table 7. Herbivore preferences for tree and shrub species specified by forest types in the Balearic Islands. W represents the forage ratio in the forest type and species determined.

Forest type	Nplot (>20)	Tree species	W	Nplot (>20)	Shrub species	W
Oak forest of *Quercus ilex*	45	*Olea europaea*	2.37	44	*Smilax aspera*	2.91
	106	*Quercus ilex*	0.58	26	*Ruscus aculeatus*	0.45
	21	*Arbutus unedo*	0.32	60	*Pistacia lentiscus*	0.11
	27	*Phillyrea latifolia*	0.24			
Aleppo pine forest	282	*Olea europaea*	0.91	121	*Smilax aspera*	0.89
	96	*Quercus ilex*	0.55	42	*Chamaerops humilis*	0.50
	48	*Phillyrea latifolia*	0.22	23	*Lavandula* spp.	0.47
	76	*Arbutus unedo*	0.08	27	*Myrtus communis*	0.32
	303	*Pinus halepensis*	0.03	165	*Phillyrea angustifolia*	0.22
	177	*Juniperus oxycedrus*	0.02	54	*Cistus salvifolius*	0.20
				213	*Cistus albidus*	0.16
				67	*Rhamnus alaternus*	0.14
				43	*Globularia alypum*	0.14
				43	*Ruscus aculeatus*	0.10
				29	*Calicotome spinosa*	0.09
				408	*Pistacia lentiscus*	0.06
				263	*Erica multiflora*	0.05
				225	*Rosmarinus officinalis*	0.04
				45	*Lonicera* spp.	0.04
				144	*Asparagus* spp.	0.03
				26	*Dorycnium pentaphyllum*	0.03
				104	*Cistus monspeliensis*	0.03
Mixed broadleaved forest in the Mediterranean biogeographical region	40	*Olea europaea*	0.87	25	*Pistacia lentiscus*	0.10
Wild olive tree forest	143	*Olea europaea*	1.45	38	*Smilax aspera*	0.44
				115	*Pistacia lentiscus*	0.05
				57	*Asparagus* spp.	0.01
Mixed autochthonous broadleaved and coniferous forest of the Mediterranean biogeographical region	175	*Olea europaea*	1.42	63	*Phillyrea angustifolia*	2.14
	89	*Quercus ilex*	0.41	61	*Smilax aspera*	1.34
	31	*Arbutus unedo*	0.37	41	*Rhamnus alaternus*	1.25
	39	*Phillyrea latifolia*	0.05	34	*Ruscus aculeatus*	0.33
				71	*Cistus albidus*	0.15
				35	*Rosmarinus officinalis*	0.10
				153	*Pistacia lentiscus*	0.08
				69	*Asparagus* spp.	0.02

of Menorca. Specifically, there were 695 plots in which the browsing degree was greater than 3 for any species, indicating areas that suffer significant difficulties to achieve natural regeneration.

Table 7 shows the preferences of herbivores for tree and shrub species according to forest type in the Balearic Islands.

Non-wood forest product production (cork)

The cork oak trees that produce cork presented larger dimensions in the SNFI3 than in the SNFI2 (Table 8). However, *ct* was significantly smaller, hence the volume of cork was much lower in SNFI3 than in SNFI2. The estimation of *ct* was based on the assumption that the age of

Table 8. Statistics for the cork weight production and the main stand variables selected for the plots analysed; *dh:* debarking height; *ct:* cork thickness; V_{cork}: volume of cork; W_{cork}: weight of cork.

SNFI	Variables	N	Min	Max	Mean	SD
2	*d* (cm)	2109	13.00	127.30	37.96	149.67
	h (m)	2109	4.00	25.00	8.46	2.34
	dh (m)	2086	0	10.00	2.35	1.70
	ct (mm)	2038	1.00	70.00	24.92	11.49
	V_{cork} (m³/ha)	2109	0	35.02	3.35	3.90
	W_{cork} (kg/ha)	2109	0	8790.61	841.99	979.48
3	*d* (cm)	2083	9.10	135.60	41.38	16.12
	h (m)	2050	0	25.00	8.78	2.65
	dh (m)	1987	0	10.00	2.34	1.68
	ct (mm)	1981	-90.20	87.17	7.14	12.28
	V_{cork} (m³/ha)	2109	-3.83	31.08	1.20	2.24
	W_{cork} (kg/ha)	2109	-962.67	7801.48	301.49	562.93

Table 9. System of equations obtained for each species using ALS metrics as predictors. Canopy fuel load (CFL) in kg/m²; Canopy base height (CBH) in m; mean height ($\bar{h}$) in m; h_{99} is the 99th percentile of ALS height distribution and PFRAhmean is the ratio of the number of the first laser hits above mean height to the number of first laser hits for each plot. ME is the model efficiency; RMSE is the root mean square error and%RMSE is the percentage of *RMSE* over the mean value of the dependent variable. All coefficients were significant at $p < 0.05$.

Species	Var.	Model	b_0	b_1	b_2	ME	RMSE	%RMSE
Maritime pine	*CFL*	$CFL = b_0 h_9^{b_1} PFRA_{hmean}^{b_2}$	0.0441	0.8991	0.0733	0.4134	0.3145	55.19
	CBH	$CBH = b_0 h_9^{b_1}$	0.4665	0.9429	---	0.4876	2.5559	43.56
	$\bar{h}$	$\bar{h} = b_0 + b_1 h_9$	4.5684	0.5368	---	0.6216	2.7192	18.01
Radiata pine	*CFL*	$CFL = b_0 h_9^{b_1} PFRA_{hmean}^{b_2}$	0.0472	0.8508	0.1215	0.4527	0.2938	43.19
	CBH	$CBH = b_0 h_9^{b_1}$	0.0929	1.4303	---	0.6777	1.7978	34.96
	$\bar{h}$	$\bar{h} = b_0 + b_1 h_9$	3.8151	0.6573	---	0.6505	2.9105	17.14

the cork (number of years since the last debarking) was the same for all the trees inventoried in the SNFI2 and SNFI3. As a result of this assumption together with the approach followed to estimate cork thickness in the SNFI3, very high negative values for cork thicknesses were obtained.

Canopy fuel modeling

In a first step, linear and potential models were fitted to estimate the three canopy variables related to crown fire (*i.e*, CFL, CBD and CBH), using different combinations of ALS metrics. However, the results obtained for CBD were highly inaccurate for both species. Therefore, we decided to model the mean height ($\bar{h}$) and then estimate CBD indirectly, by dividing the estimations of CFL by the crown length, obtained as the difference between the estimates of mean height and CBH (van Wagner, 1977). The best set of equations for each species is shown in Table 9.

Linear models provided better goodness-of-fit statistics than potential models for mean height in both species, whereas potential models showed better performance for the other two independent variables (CFL and CBH).

The observed variability explained by the models ranged from 41% to 45% for CFL; from 49% to 68% for CBH and from 62% to 65% for $\bar{h}$, with better results for radiata pine stands in all cases.

The 99th percentile (h_{99}) was the only predictor in all the models where the dependent variable was, directly or indirectly, related to canopy heights (CBH and $\bar{h}$). However, the models of CFL, indirectly related to canopy fuel biomass, also included the ratio of the number of the first laser hits above mean height to the number of first laser hits for each plot (PFRAhmean).

Discussion

SNFI constitute one of the best forest information systems, providing robust and reliable information

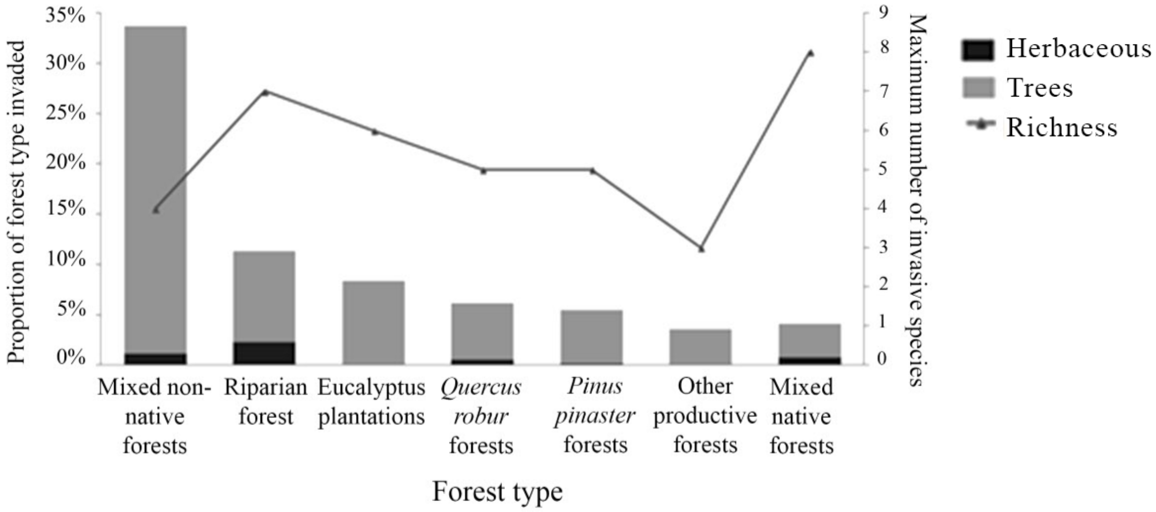

Figure 5. Proportion of the forest types of Galicia invaded by exotic species with different functional type and maximum number of invasive species found in each forest type.

for forest management and policy making. The results of this paper evidence the fact that SNFI are a relevant source of forest information for the development of support tools for decision-making and assessment in diverse strategic fields. Therefore the continuation of SNFI project is crucial to both the forestry and environmental sectors. The increasing requirement for new information combined with the potential of NFI's to provide this data has led to them becoming multifunctional inventories (Vidal *et al.*, 2016).

Deadwood is a complex but highly important aspect of forest management. It has been identified as one of the most important components of forest ecosystems, both functionally and structurally (Harmon *et al.,* 1986; McComb & Lidenmayer, 1999). Moreover, deadwood data is recorded in most NFIs given its potential role as a forest carbon pool (IPCC, 2006). The mean quantity of deadwood found in the Navarra region was 8.83 m^3/ha, which is in line with amounts reported at European level (Forest Europe, 2015) by several countries such as Ireland, Hungary, Sweden and Norway, which present average values of 7.63, 7.67, 7.8 and 9.40 m^3/ha, respectively. Despite the high dispersion found in deadwood stock records evidenced by the high SDs, reference values have been provided for different forest types and bioregions. Although the percentage of dead wood by volume was similar in the different ecoregions, the DWV value was lower in monospecific Mediterranean forest areas. Lombardi *et al.* (2008) analyzed DWV values in Italy, obtaining values between 4.8 m^3/ha (managed forest) and 27.2 m^3/ha (unmanaged forest) for montane beech forest; 4.0 m^3/ha for *Pinus nigra* plantations and 5.8 m*3*/ha for *Q. ilex* stands. These results were slightly higher than those obtained in our study (Table 5). Böhl & Brändli (2007) assessed the deadwood stock in the Swiss NFI and distinguished four different forest types: pure conifer forest, mixed conifer forest, pure broadleaf forest and mixed broadleaf forest (mean values of 20.0, 27.0, 25.7 and 32.1 m^3/ha, respectively). Although these values were higher than those obtained in this study (Table 5), they reveal the same tendency between monospecific and mixed forests.

The assessment of tree age based on diameter related data provides an effective decision-making tool for forest managers based on ecological principles. In the case of riparian forests, it is of particular interest to determine the age-class distribution in order to assess whether the number of saplings is adequate to replace the ageing population or if regeneration of the forest is threatened. Age is a key variable in inventories (Bisoffi *et al.*, 1987) as it can contribute to the development of appropriate conservation strategies as regards indigenous species and the identification of stands with older trees requiring conservation. Methodologies for the development of age-diameter models for riparian forests have identified various sources of error attributable to different causes associated with sampling design (deposit of sediment which hides a part of the base of the tree) or with the dynamics of riparian ecosystems in which regeneration occurs both from seeds and through vegetative regeneration from roots (Mahoney *et al.*, 1991).

The introduction, establishment and spread of invasive alien species is one of the most serious nature conservation issues currently faced at European and global scales (UNEP, 2010; EC, 2011). In this context, the recent inclusion of non-native species data in periodic permanent sample units in National Forest Inventories (Corona *et al.*, 2011) constitutes a valuable tool for monitoring the broad-scale evolution of plant invasions in forest ecosystems (Hernández *et al.*, 2014). The degree of regional invasion depends on the time since introduction, the potential of the species to invade (*i.e.* invasiveness) and ecosystem resistance to invasion (*i.e.* invasibility) (Vilà *et al.*, 2008). The two *Acacia* species showed the highest potential to invade in forest ecosystems of NW Spain. Due to certain attributes such as its rapid growth rate, prolific production of high-longevity seeds, germination stimulated by fire, allelopathic effects and the absence of natural enemies (Marchante *et al.*, 2003), these *Acacia* species are catalogued as some of the most widespread and aggressive invasive species in Europe. The highest degree of invasion was found in altered forests, which supports a previous hypothesis suggesting that habitats might be more susceptible to invasion when there is an increase in the amount of unused resources resulting from natural or human disturbances (Davis *et al.*, 2000). The findings of this case study highlight the suitability of using detailed NFI information to identify invasive species rates and invasiveness as well as the invasibility of forest types.

In the Balearic Islands, where the case study concerning the impact of browsing was carried out, the Mallorcan wild goat (*Capra aegagrus hircus* L.), the feral goat (*Capra hircus* L.) and domestic goat are the main browsers (Seguí *et al.*, 2005). Goats which have been introduced to the islands and other ungulates are known to cause significant damage to the flora (Perea *et al.*, 2015), particularly the endemic flora of the islands (Garzón-Machado *et al.*, 2010). However, in part of the Balearic Islands, these goat populations could partially occupy an important ecological niche, as reported for other species in Mediterranean areas (Fernández-Olalla *et al.*, 2016). In this case, the goat populations could occupy the niche vacated by the extinct Balearic Island cave goat (*Myotragus Balearicus*, Bate) (Bover & Alcover, 2003), supporting the co-evolution of the island flora and browsing herbivores (Alcover *et al.*, 1999). Long-term monitoring of browsing is necessary to ensure the prevention of damage to endemic and other flora in these Islands. Exhaustive studies into the impact of browsing have been undertaken recently in the area to facilitate the management of the goat population (Rivera-Sánchez *et al.*, 2014). However, the SNFI provide the possibility to systematically monitor the consequences of animal and plant interaction over time. Our results agree with those of Rivera-Sánchez *et al.* (2014) and suggest that *Olea europaea* is the most preferred species and that *Pistacia lentiscus* is one of the least preferred species; both of these species being present in all forest types studied. However, forest types and goat population density can affect the relative preferences between species.

The case study concerning non-wood forest products provides evidence that production can be estimated by using data from the SNFI. However, the case study also illustrates the importance of choosing the appropriate variables to be inventoried in order to assure accurate estimations of the production of this type of forest product. When cork thickness and debarking height were reported (SNFI2 plots), cork production could be estimated with the required level of accuracy and robustness. However, when cork thickness was not measured (SNFI3 plots) it needed to be estimated based on data from the SNFI2, a diameter increment model and the assumption that the age of the cork (number of years since the last debarking) was the same for all the trees that were inventoried in the SNFI2 and the SNFI3. The results accomplished with this approach (where very high negative values for cork thicknesses were obtained, Table 8) proved that cork production can only be estimated when the appropriate cork-related variables are inventoried. In short, cork thickness and age should be monitored to obtain cork production estimates with an appropriate level of accuracy at national scale. This recommendation has been taken into account in the SNFI4 and sampling of cork variables will now be implemented in subsequent provinces.

Existing cartographic-based wildland fire simulation systems, such as FARSITE (Finney, 2004) or FlamMap (Finney, 2006), estimate the crowning potential based on fire behavior models. The use of these systems requires canopy raster layers at landscape level. These canopy variables cannot be directly measured, but can be modeled from other supplementary tree or stand characteristics, and the SNFI provides periodic information that could be used for this purpose (*e.g.*, Fernández-Alonso *et al.*, 2013a,b). Wall-to-wall countrywide ALS coverage can be used in combination with the proposed models to derive the spatially explicit maps of the canopy variables. Maritime and radiata pine forests should first be identified by using the strata of the Spanish Forest Map (http://www.mapama.gob.es/es/biodiversidad/servicios/banco-datos-naturaleza/informacion-disponible/mfe50.aspx). These maps will be useful to program silvicultural treatments in the context of fuel management decision making.

The SNFI has proven to be a valuable source of information for the development of tools to aid decision-making and the assessment of forest fire behavior. The main shortcoming of the SNFI is the long period between measurements. However, if used in combination with other sources of information, such as countrywide ALS datasets, the information can be updated with a high degree of accuracy and wildland fire simulators will benefit from this synergy, which could provide an essential tool for forest fire hazard management.

Future challenges of the Spanish National Forest Inventory

The demands for forest information have been substantially increased and therefore the scope of NFIs has broadened introducing new variables requiring assessments (Tommpo *et al.*, 2010). NFIs are facing different challenges to provide strategic information for forest related policies.

One of the main challenges for the SNFI is to facilitate reliable forecasts of wood availability, which are highly important in order to define strategies for climate change mitigation or to develop strategies for promoting the use of wood as a renewable energy source (Barreiro *et al.*, 2016). Although most of the countries in Northern and Central Europe have developed nationwide projection tools based on empirical growth models (Barreiro *et al.*, 2016), some of the characteristics of Southern European forests hinder the development of nationwide projection systems. As regards Spain, the high variability of the forests, both in terms of composition and structure, makes the development of national scale growth models particularly challenging. Nevertheless, a large number of growth models have been developed in Spain (Bravo *et al.*, 2011), although they are mainly regional scale models which focus on productive species in monospecific even-aged stands. Hence, these models are not suitable at national scale or for use with SNFI data, which still do not record age. SNFI data are said to have four major weaknesses for forest modelling purposes (Álvarez-González *et al.*, 2013): excessively long inventory cycles for fast growing species; systematic inventory not allowing comparison between silvicultural alternatives in small areas; the smallest radius plot is too small for ingrowth models; and the absence of stand age.

Sampling optimization is always an important concern for SNFI. It is important to continuously evaluate the use of new technology such as ALS, the satellite constellation Sentinell of the European Union (EC, 2014) or even Unmaned Aerial Vehicles (UAV), although all these technologies cannot replace fieldwork (McRoberts & Tomppo, 2007). It is also crucial adequately communicate the extensive forest information to society through accessible databases, comprehensive reports and publications aimed at the different groups involved (stakeholders, managers, policy makers, etc.).

In the context of different national and international forest policy information requirements, further variables are currently being evaluated for monitoring in the ongoing SNFI cycles such as those related to non-wood products, plot soil characteristics or shrubs and trees in non-forest areas.

Additionally, European NFIs should provide comparable results as European information is obtained by aggregation (Tomppo *et al.*, 2010). Although many harmonization processes have already been carried out, such as the assessment of forest area (Vidal *et al.*, 2008) or above ground biomass (Vidal *et al.*, 2016), further studies are required, especially as regards new recorded variables.

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