METHODOLOGY FOR CALCULATING THE IMPACT OF FOREST MANAGEMENT ON ECOSYSTEM SERVICES: CARBON, WATER AND BIODIVERSITY

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Introduction

The great forest expansion of the second half of the last century has contributed to the sequestration of atmospheric carbon dioxide, which has been kept in the form of carbon in trees and forest soils. Today, however, the role of forests in climate change mitigation is at risk in Catalonia: the rate of CO_2 sequestration has declined by 17% over the last 25 years and the increasing frequency and intensity of disturbances threaten accumulated stocks. In 25 years, forest biomass in Catalonia has increased by 73% and forests today are 24% denser now than in 1990. As a consequence of global change, the health of forests has deteriorated, the risk of fire has increased and the provision of basic services such as water has been affected, with a 30% reduction in river flow in 25 years (Banqué *et al.,* 2020). Biodiversity has also been affected by forest expansion, especially biodiversity linked to scrub and open spaces. In the last 20 years, the populations of birds and butterflies associated with these environments have been reduced by 14% and 57% respectively (*State of Nature in Catalonia*, 2020).

In this scenario, multifunctional forest management is key to mitigating climate change and, above all, to adapting our landscapes to new climatic conditions. Beyond obtaining wood and cork products, in many cases a forest must be managed to prevent catastrophic fires or to increase the resistance of forest masses to longer periods of drought and more frequent disturbances, or so that they can recover more quickly. Management that integrates different objectives must promote the proper functioning of forests so that they provide those benefits —the forest ecosystem services— of greatest value for each territory, including fundamental aspects as the production of goods such as timber or cork, carbon sequestration, forest biodiversity, the supply of quality water and control of soil erosion.

This paper sets out a methodology for calculating the impact of multifunctional forest management on three key factors of climate change mitigation and adaptation in the Mediterranean: carbon, water and biodiversity. In this first version, the impact of management on the carbon balance has been obtained for the main conifers in Catalonia, the Aleppo pine, Scots pine and black pine, and for holm oak. The assessment of the impact of management on water resources (blue water) incorporates calculations for all forest typologies formations, while biodiversity is assessed for all types of forest management and formation.

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Impact of forest management on carbon balance

Chapter 1

Impact of forest management on carbon balance

1.1 Introduction

Forests have a key role to play in mitigating climate change: in Catalonia they sequester around 10% of total CO_2 emissions. Forests act as carbon fixers when they grow, i.e. when they capture more CO_2 through photosynthesis than they emit through respiration, influenced by climate, soil, other species and human intervention. As forests develop, the total amount stored, both in vegetation and in the soil, becomes greater, and it is a long time before the balance between sequestration and release is reached (Stephenson *et al.*, 2014). The moment when this equilibrium is reached depends on many factors, so the role of old-growth forests in carbon sequestration is still a matter of debate (see, for example, Gundersen *et al.*, 2021, questioning the results of Luyssaert *et al.*, 2008).

However, it is widely accepted that forests whose vitality is compromised (forests subject to a great deal of competition, experiencing water stress, etc.) can become emitters (Wang *et al.*, 2020). In the last decade, European forests have started to show signs of stagnation in their carbon sequestration capacity (e.g. Naaburs et al., 2013). Catalonia is no exception: the rate of CO_2 sequestration in Catalan forests has decreased by 17% over the last 25 years (Banqué *et al.*, 2020) and the increasing frequency and intensity of disturbances threaten the stocks accumulated over the years. In the Mediterranean, where water is a limiting factor, climate change is expected to have the opposite effect to the fertilising effect of increased CO_2 and rising temperatures expected in more northerly latitudes: growth, and thus the ability to sequester carbon, will be compromised by the reduced availability of water. Signs have already been detected in some forests in the north of Spain and in mountain areas, with species at the southern limit of their distribution, and in unmanaged forests (Vayreda *et al.,* 2012).

The only tool we have to increase the capacity of forests to mitigate climate change is sustainable forest management, as recognised in the Paris and Glasgow agreements. Some authors have estimated that forest management could not only maintain the rate of carbon sequestration but double it (Naaburs *et al.,* 2017). In Catalonia, an increase of up to 45% in carbon sequestration (from 1.24 to 2.25 tonnes/year) attributable to forest management has been documented (Vayreda *et al.,* 2012).

Forest management can contribute both to maintaining the rate of carbon sequestration in forests (increasing their vitality by regulating competition for resources and, locally, with plantations) and to conserving existing carbon stocks (e.g. through fire prevention measures, favouring the production of long-lived wood

products or taking care not to damage forest soils). Beyond the forest, the production of renewable and locally sourced raw materials, such as wood or cork, facilitates the replacement of fossil fuels and materials and contributes to targets for decarbonising the economy. Finally, management can help reduce the impact of forest fires and water stress, severe disturbances occurring frequently in the Mediterranean that can lead to both a loss of biodiversity and an uncontrolled increase in emissions.

1.2 IPCC methodology

Based on the IPCC (Intergovernmental Panel on Climate Change) (2006) guidelines for national greenhouse gas inventories (Chapter 4 on forest land in Volume 4 on agriculture, forestry and other land uses), the Forest Ownership Centre (CPF) and the Institute of Agrifood Research and Technology (IRTA) established a methodology for quantifying the effects of forest management on the carbon sink capacity of Catalan forests (Porras *et al.,* 2015; Porras *et al.,* 2017). The «gain-loss method» was applied, using the carbon pools and the formulas provided in the guidelines, in order to monitor carbon fluxes between pools and also to obtain carbon sequestration values in merchantable timber, standing timber, dead organic matter and roots, all at different times in the rotation (in the regeneration stages, when stands are more mature, and in the pre-regeneration stage) and at the end of the rotation (figure 1).

The impact of forest management was established by comparing silviculture regimed based on reference managed forests (included in the «Guidelines for Sustainable Forest Management in Catalonia» (ORGEST) (Piqué *et al.,*

2011; 2017) and the evolution of «unmanaged forests», over 150 years. This methodology showed that a managed stand fixed a greater amount of carbon than an unmanaged forest. Only in low-quality areas (site quality C) and in the case of some models where the main objective was fire prevention could values be lower. This methodology also allowed fixation to be compared in the different existing models (figure 2).

This methodology takes into account the carbon sequestration resulting from stand growth after forestry treatment has been applied, and over different cutting rotation periods depending on the species and model chosen. Trees standing after forestry intervention fix carbon as they grow, to a greater or lesser extent depending on the type of formation and the procedure followed at any given time. Calculation of the balance ended when timber was at the loader, destined for different industrial uses, depending on the species and the size of the product.

The calculation of the CO_2 sink took into account that each tonne of wood removes approximately 1.84 tonnes of CO_2 from the atmosphere and this is not released again until the end of the life cycle of the material (Correal *et al.*, 2017), which this study does not include, as it focuses on the forest. Another important issue in this methodoloty (of Porras *et al.*, 2007) is that the crown debris from managed sites and dead wood on the ground from unmanaged sites were included in the organic matter pool, but no allowance was made for the percentage of emissions they might account for, and the soil pool was not taken into account in the comparison.

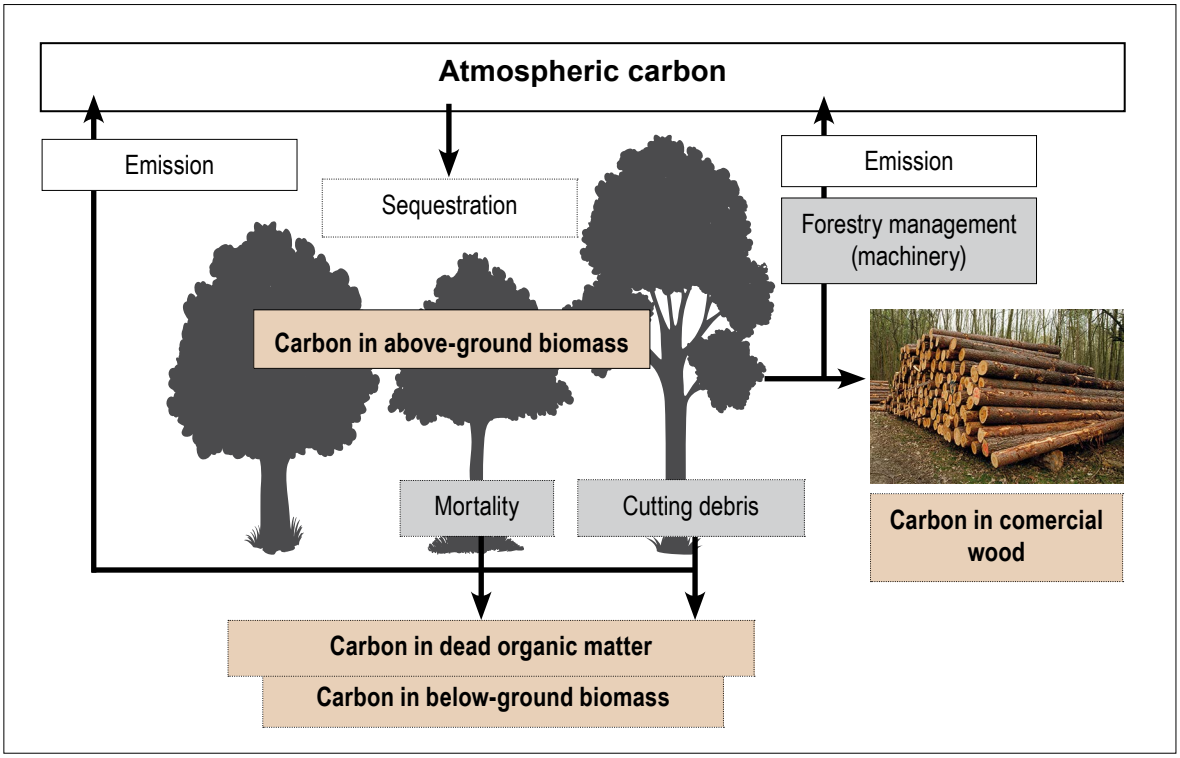


Figure 1. Carbon stocks and carbon transfer in a forest system.

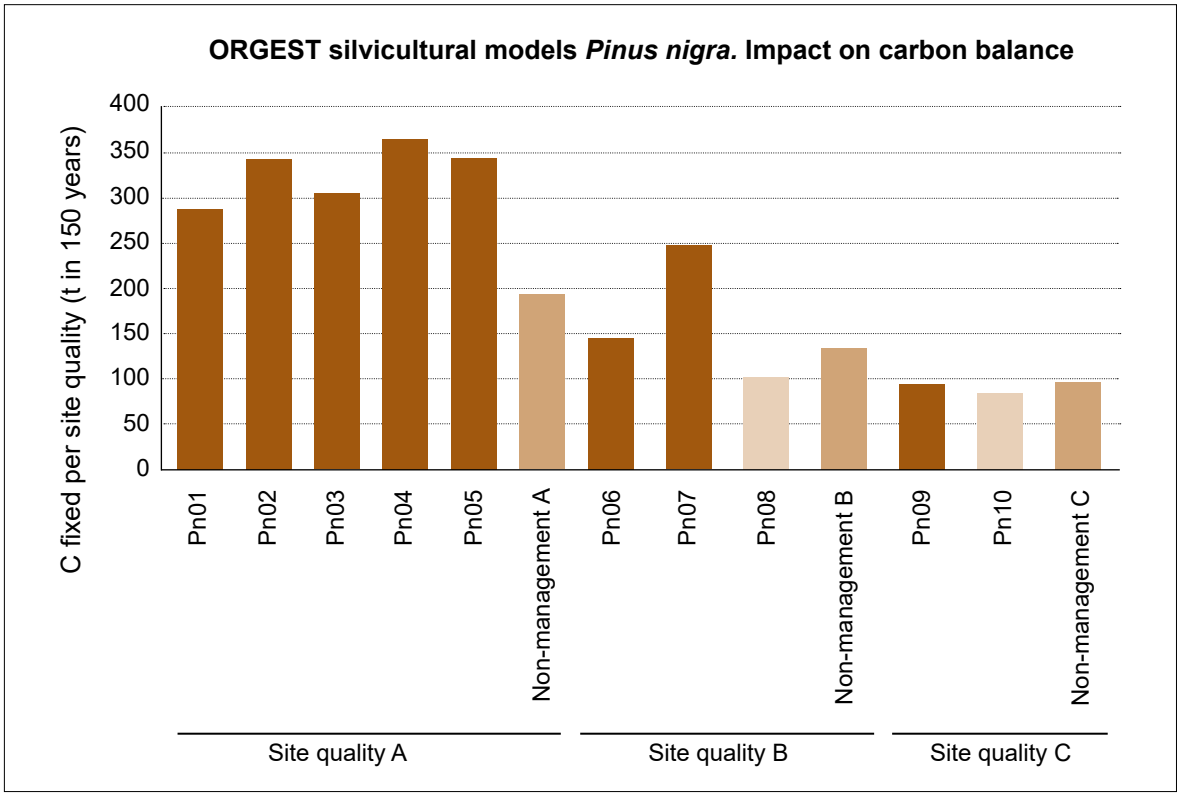


Figure 2. Tonnes of carbon fixed by black pine forests according to ORGEST models for unmanaged sites and for the three site qualities (high [A], medium [B] and low [C]).

1.3 Capsules and reference scenarios in the carbon balance

Following on from this first study, the proposed methodology goes a step further. Firstly, the impact of forest management on the **carbon sequestration** of the standing stock in a specific period according to model and pathway focuses on the period between two forestry interventions of one type and not the whole rotation. Sequestration includes **carbon emissions** from the use of machinery during forestry operations and in the transport of the product obtained to an industrial site (in the methodology used by Porras *et al.,* 2017, the process was restricted to the forest). The return to the atmosphere of carbon stored in shorter-lived products is also counted as emissions. Finally, the tonnes of **carbon avoided** are calculated by assessing what emissions could be avoided by the proposed management in the event of a major forest fire, according to the annual probability of its occurrence and the stock involved, and by assessing the positive effect of replacing non-renewable materials by wood or biomass.

Impact of management on CO_2 balance=

$$tCO_{2\ sequestered} - tCO_{2\ emitted} + tCO_{2\ avoided}$$

This calculation must take into account the scenario in which the impact is compared. **The reference scenario** varies according to whether forestry treatment is applied from natural regeneration, in plantations, or whether there

is a change from one management model to another:

a. **Non-management scenario:** simulations are executed showing the growth and mortality of the stand without any type of management from the same initial situation as that considered in the management scenario.

b. **BAU management scenario:** this is applicable when moving from a short rotation (BAU scenario) to a long rotation silvicultural model.

c. **Non-planting scenario:** crop field, wasteland, scrub or grassland forest where a tree plantation is possible but it is not executed. Carbon sequestration from previous land use is not taken into account.

In this methodology, the reference scenarios chosen were **no management** and **no planting**, according to the forestry measures proposed.

In terms of **time scale**, the most realistic approximation possible of what happens during the period following the intervention has been chosen. In order to delimit this period, no fixed interval is proposed. Its is preferable to choose a rotation period between interventions in accordance with the different forestry models applied, which can vary between 15 and 30 years. This moment is considered to coincide with the point at which the benefits of the intervention, in terms of stimulating growth, come to an end.

CAPSULE 1. CARBON SEQUESTRATION

Variations in carbon sequestration due to forest management

At the **management** level, the calculation of the **balance of carbon sequestered in the period considered,** tCO_2 sequestered, is obtained from the carbon fixed by the trees that remain standing after the implementation of forestry measures[1], compared to the unmanaged reference scenario. Bearing in mind that the unmanaged reference model is only available for the three commonest conifers in Catalonia, **the Scots pine, the Aleppo pine and the black pine (*Pinus sylvestris*, *Pinus halepensis* and *Pinus nigra*) and the holm oak (*Quercus ilex*)**, the analysis of the carbon balance in other forest formations is conditional on obtaining these models.

$$\mathbf{tCO_{2\ sequestered}} = tCO_{2\ \text{sequestered managed}} - tCO_{2\ \text{sequestered unmanaged}}$$

The carbon balance by forest formation, site quality, model and itinerary, comparing managed and unmanaged forests, can be found in annex.

In order to obtain the carbon content in trees and roots, in the products obtained, and in the harvest residues, different biomass fractions are previously calculated and a carbon factor (CF) per species is applied to them (Montero *et al.*, 2005), i.e. the percentage of carbon by weight contained in the dry matter for each species. The CF used for *Pinus sylvestris* and *Pinus nigra* is 0.509, for *Pinus halepensis* it is 0.499 and for *Quercus ilex* 0.475 (Table 1).

The biomass fractions calculated are total above-ground biomass (AGB), foliage and branch biomass (FBB) and total below-ground biomass (BGB). For the first two (AGB and FBB), we

1 "We use the conversion from tC to tCO_2 according the relation of the molecular weight of both compounds (33/12 g/mol).

Table 1. Percentage by weight of carbon content in dry matter applied to each species
Source: Montero *et al.* (2005)

Species	% Carbon	Species	% Carbon
Abies alba Mill.	50.6	*Pinus halepensis* Mill.	49.9
Abies pinsapo Boiss.	50.0	*Pinus nigra* Arn.	50.9
Alnus glutinosa L.	50.0	*Pinus pinaster* Alt.	51.1
Betula spp.	48.5	*Pinus pinea* L.	50.8
Castanea sativa Mill.	48.4	*Pinus radiata* D. Don	49.7
Ceratonia siliqua L.	50.0	*Pinus sylvestris* L.	50.9
Erica arborea L.	50.0	*Pinus uncinata* Mill.	50.9
Eucalyptus spp.	47.5	*Populus x euramericana* (Dode) Gulnier	48.3
Fagus sylvatica L.	48.6	*Quercus canariensis* Willd.	48.6
Fraxinus spp.	47.8	*Quercus faginea* Lamk.	48.0
Ilex canariensis Poir.	50.0	*Quercus ilex* L.	47.5
Juniperus oxycedrus L./*J. communis* L.	50.0	*Quercus pyrenaica* Willd.	47.5
Juniperus phoenicea L./*J. sabina* L.	50.0	*Quercus robur* L./*Q. petraea* Liebl.	48.4
Juniperus thurifera L.	47.5	*Quercus suber* L.	47.2
Laurus azorica (Seub.) Franco	50.0	Other conifers	50.0
Myrica faya Alt.	50.0	Other broadleaved	50.0
Olea europaea var. *sylvestris* Brot.	47.3	Other laurisylvae	50.0
Pinus canariensis Sweet ex Spreng.	50.0		

use allometric equations proposed by CREAF based on IEFC data, while for below-ground biomass (BGB), we use equations proposed by INIA (table 2). These equations are used to calculate the biomass fractions of both standing trees and trees affected by forestry intervention. A recent study (Ameztegui *et al.,* submitted) has verified the importance of choosing the right allometric equation for biomass estimates in forests, and has shown that it is always preferable to use equations developed from samples that are local or geographically close to the study area.

Table 2. Allometric equations for the calculation of biomass per species

Species	Var. D	Equation	a	b	c	Source
Pinus nigra	AGB	$a \cdot DBH^b \cdot Ht^c$	0.04	2.15	0.54	IEFC
Pinus halepensis	AGB	$a \cdot DBH^b \cdot Ht^c$	0.08	1.96	0.52	IEFC
Pinus sylvestris	AGB	$a \cdot DBH^b \cdot Ht^c$	0.06	2.05	0.56	IEFC
Quercus ilex	AGB	$a \cdot DBH^b \cdot Ht^c$	0.11	2.08	0.42	IEFC
Pinus nigra	FBB	$a \cdot DBH^b$	0.04	2.19		IEFC
Pinus halepensis	FBB	$a \cdot DBH^b$	0.08	1.95		IEFC
Pinus sylvestris	FBB	$a \cdot DBH^b$	0.06	2.03		IEFC
Quercus ilex	FBB	$a \cdot DBH^b$	0.11	2.00		IEFC
Pinus nigra	BGB	$a \cdot DBH^b$	0.02	2.39		INIA
Pinus halepensis	BGB	$a \cdot DBH^b$	0.03	2.30		INIA
Pinus sylvestris	BGB	$a \cdot DBH^b$	0.01	2.63		INIA
Quercus ilex	BGB	$a \cdot DBH^b$	0.55	1.79		INIA

AGB: total above-ground biomass (kg); FBB: foliage and branch biomass (kg); BGB: total below-ground biomass (kg); DBH: diameter at breast height (cm); Ht: total height (m).

The **carbon fixed between two intervetions** ($C_{ha/year}$) is the increase in the mass according to the period established, which varies according to each model and itinerary. It is calculated as the difference between the carbon stock at the end of the period and the carbon stock immediately after the intervention. These values are obtained by applying the allometric equations (Table 2) to the forest's structural parameters (diameter, height), as defined in the ORGEST forestry models for these two situations. From this value we subtract the proportion of carbon emitted into the atmosphere by the decomposition of cutting debris and dead roots from harvested trees. In the case of holm oak, the roots have not been considered as dead organic matter, since it is a regrowth species whose stump does not die.

$$C_{ha/year} = ((C.\ live\ trees_{tn} - C.\ live\ trees_{to}) + (C.\ live\ roots_{tn} - C.\ live\ roots_{to})) / (t_n - t_o) - (C\ wood\ debris + C\ dead\ roots) \times C\ emission\ ratio.$$

In the case of mixed stands, the reference model chosen for the main species must be taken into account when determining future growth. For secondary species, the average growth rates obtained in the reference forestry models should be used, depending on whether management is irregular or regular, and, in the latter case, on the stage of development of the species (young, adult or regenerating). The overall growth value will be obtained by applying the percentage of basal area (BA) represented by each species and the site quality of the species in the area where intervention is to be carried out.

Carbon emissions from the decomposition of cutting and roots

Through the decomposition of organic matter, much of the carbon is emitted into the atmosphere as CO_2, while a small proportion is absorbed in the soil in particulate form or dissolved in rainwater (Figure 1). To estimate the carbon content of coarse and medium debris (> 5 cm in diameter) in the forest diferent decomposition (k) rates (Olson, 1963) are obtained from literature (Table 3). Mattson *et al.* (1987) give an average rate of 0.083 for year^{-1} in different species in southern North Carolina (USA), while the rates given by Herrmann *et al.* (2015) for southern Germany are considerably lower. The latter authors estimate different rates according to species and do not find climate to be the factor explaining variations within species. Depending on the source of information and the species, debris is thus given a different weighting over time (Figure 3).

Mattson *et al.* (1987) estimate that about 2/3 of the carbon lost is emitted into the atmosphere and the rest is incorporated into the soil. If we apply over time the decomposition rates propo-

Table 3. Decomposition (k) rates (Olson, 1963) for different species according to the literature

Reference	Species	Diameter cm	Olson k rate y^{-1}	Location	Average P mm	Average T °C
Mattson *et al.*	Various	--	0.083	North Carolina	1,820	12.6
Herrmann *et al.*	*Fagus sylvatica*	<20	0.078	Southern Germany	657-1,700	8.5-11.0
	Pinus sylvestris	20-40	0.055	Southern Germany	657-1,700	8.5-11.0
		<20	0.050			
	Picea abies	20-40	0.030	Southern Germany	657-1,700	8.5-11.0
		<20	0.034			
		20-40	0.036			

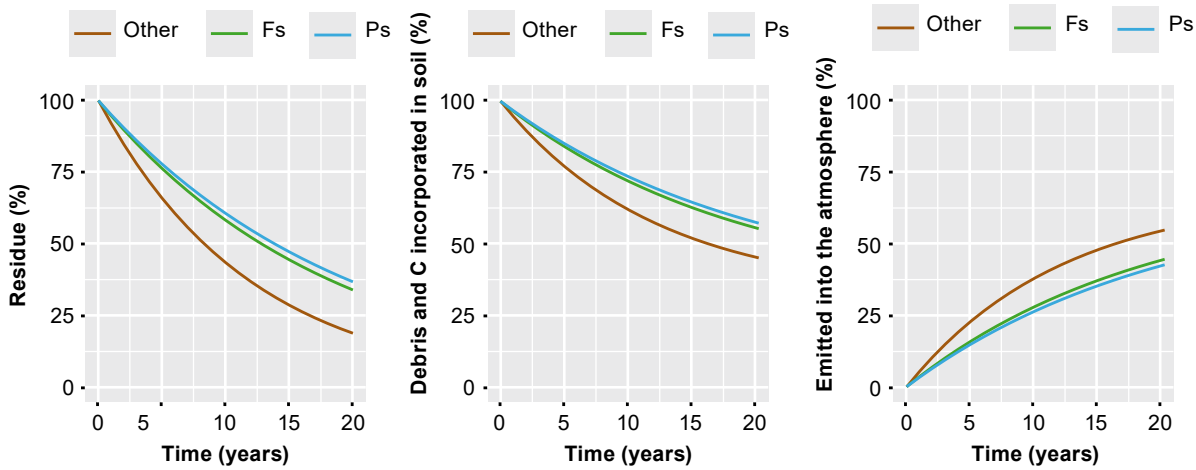


Figure 3. Changes in the percentage of C in debris according to species, held in decomposing remains (left, based on k = 0.083 yr^{-1}), plus C incorporated into the soil in particulate form or dissolved in rainwater (1/3 of decomposed debris, centre) and C emitted into the atmosphere (right). The diameter of debris was taken to be less than 20 cm. Figures have been obtained from the information given in Table 2. «Other» corresponds to the k rate of Mattsonn *et al.* (1987) and those for the species to Herrmann *et al.* (2015).

sed by authors and the assumption based on Mattson that only 1/3 of the C from decomposing remains is incorporated into the soil, an estimate can be made of the amount of C remaining in the forest (debris or soil) (Figure 3, center). C emitted to the atmosphere in time, would be its complementary. The C emitted into the atmosphere over time would be in addition to this amount. The percentages of C in the debris remaining in the forest, incorporated into the soil or emitted into the atmosphere over 10, 15 and 20 years are shown in Table 4. Medium and coarse roots are considered to have similar characteristics to debris.

In this methodology, to estimate carbon emissions to the atmosphere produced by the decomposition of cuttings and roots using this methodology, the reference value chosen was the rate of carbon emitted into the atmosphere

Table 4. Estimation of carbon over time after management for remaining weight (%), decomposing remains and that incorporated into the soil in particulate form of washed in according to rates estimated by different authors (Table 1) with debris diameter less than 20 cm.

Year	Remaining debris	Lost	Flow to ground	C in debris or soil	Emitted to atmosphere
Other					
10	43.5	56.5	18.8	62.4	37.6
15	28.7	71.3	23.8	52.5	47.5
20	18.9	81.1	27.0	46.0	54.0
Pinus sylvestris					
10	60.7	39.3	13.1	73.8	26.2
15	47.2	52.8	17.6	64.8	35.2
20	36.8	63.2	21.1	57.9	42.1
Fagus sylvatica					
10	58.3	41.7	13.9	72.2	27.8
15	44.5	55.5	18.5	63.0	37.0
20	34.0	66.0	22.0	56.0	44.0

indicated by Herrmann *et al.* (2015) for *Pinus sylvestris* for a period of 15 years (35.2%). To apply the rate indicated to estimates of carbon emissions from decomposition in the reference forestry models (ORGEST), an annualised value (2.35%) was taken and multiplied by the number of years between forestry interventions. The same rate was used for the different species considered in this methodology.

Carbon sequestration in unmanaged forests

In order to compare carbon sequestration in managed and unmanaged forests, different theoretical growth models were developed. The construction of such growth models for unmanaged stands is difficult, given the lack of available scientific studies. It is worth mentioning here the unpublished study «Silvicultural itineraries of non-management for *Pinus sylvestris* in Catalonia» produced by Fora Forest Technologies in 2019, presenting a theoretical growth model for *Pinus sylvestris*.

The methodology followed to construct the theoretical growth models used is based on the application of different studies and sources of information. Different models have been developed for the site qualities indicated in the ORGEST reference forestry models, and for black pine, Aleppo pine, Scots pine and holm oak formations.

In this methodology, the following steps have been followed:

1. Determining a dominant age-height adjusted to each site quality (site index).

   In the case of black pine and Scots pine formations, the formulas indicated in articles by Palahí *et al.* (2003 and 2004) on site index and growth of stems have been used, while for the Aleppo pine, the formulas indicated in the article by Montero *et al.* (2001) have been used. In order to adjust the age-height relationship to the different site qualities of the ORGEST models, reference values from these models have been used.

   A different approach was used for holm oak because, at the time of the study, there were no equations available to relate age to height. In this case, the height was obtained using the same equations as in the ORGEST reference models for holm oak, where the height is calculated according to the diameter. To determine how diameter evolves with stand age, a diametric growth factor for unmanaged forests, obtained from X. Mayor and F. Rodà (1993), was applied.

2. Determining changes in density and mortality.

   To do this, the stand density index (SDI), a density indicator originally developed in 1933 by Reineke (Reineke's stand density index), as used in the ORGEST reference models, was applied. The parameters used to calculate SDI were adjusted for each species, using studies carried out by Vericat *et al.* (2014) as a source for this adjustment.

   SDI can be defined as the equivalent number of trees per hectare of 25 cm quadratic mean diameter, formulated as:

$$SDI = N * \left(\frac{D_g}{25}\right)^{ß}$$

where, N is the number of trees per hectare, or density, D_g is the quadratic diameter and ß is the thinning slope.

   The same initial density as that used in the ORGEST models was used to obtain the density, adjusted for each reference age according to the SDI. To estimate the quadratic diameter, different equations according to species were used, devised by Piqué *et al.* (2017) based on National Forest Inventory (NFI) data and depending

on the height, age and density of the forest stand. To estimate mortality when applying the SDI, based on the SDI study (Vericat *et al.*, 2014), it was considered that mortality starts to occur when the SDI is 60% of the maximum and that when maximum SDI is reached or exceeded mortality is 100% of the excess trees (Table 5). In view of the difficulty of establishing a mortality gradient between 60% and 100% of the SDI, and considering the idea put forward in some studies (Palahí *et al.,* 2003) that the SDI is different according to the quality of the site, it was decided to adjust the onset of mortality to 100% of the maximum SDI in quality A, 72% in quality B and 53% in quality C.

Table 5. Parameters used in the SDI

Species	ß	Maximum SDI	Source
Pinus nigra	1.896	1,287	CTFC
Pinus halepensis	2.029	1,200	CTFC
Pinus sylvestris	2.071	1,339	CTFC
Quercus ilex	2,139	800	CTFC

Once the growth models for unmanaged forests had been obtained, the same allometric equations and species-specific carbon factors (Tables 1 and 2) were used as in the section «Variations in carbon sequestration due to forest management», and the same annual percentage of emissions from dead wood (2.35%).

$$\mathbf{C_{ha/year}} = ((\text{C. live trees}_{tn} - \text{C. live trees}_{to}) + (\text{C. live roots}_{tn} - \text{C. live roots}_{to})) / (t_n - t_o) - (\text{C wood debris} + \text{C dead roots}) \times \text{C emission ratio}$$

With these calculations, we obtain the carbon fixed in living organic matter, trees and roots, and the carbon emissions from decomposition, comparing the ORGEST management reference models and the non-management models, between two established cut-off periods.

Carbon sequestration in plantations

To determinate the carbon stored due to the planting of different species of interest, the allometric equations developed by CREAF with data from the Ecological and Forestry Inventory of Catalonia (IEFC), which are available at the Catalan Forestry Laboratory (https://laboratoriforestal.creaf.cat/), were used. The IEFC included the measurement of total above-ground

biomass (AGB) for a sub-sample of trees of all species. From these data, a series of allometric relationships were obtained for each species, for Catalonia as a whole, allow AGB to be estimated from other more easily obtained variables, such as species, normal diameter and, in some cases, height.

According to this methodology, the carbon storage capacity was calculated using the following steps:

1. Calculation, for each species, of the expected mean diameter at different ages: 0, 10, 20... up to 80 years, using the age-diameter ratios established in NFI1 and listed in the publications *Las coníferas en el Primer Inventario Forestal Nacional* (1980) and *Las frondosas en el Primer Inventario Forestal Nacional* (1979).

2. Calculation of the height corresponding to each diameter, using the ratios calculated in Ameztegui (2020) from NFI3 data in Catalonia. The height-diameter ratio was adjusted to the following non-linear model: $Ht = 1.3 + (H_1 - 1.3) * (1 - \exp(-B*DBH))$, where DBH is the diameter in cm, Ht is the height, in metres, and H_1 and B are two parameters to be estimated.

3. Calculation of total above-ground biomass for species of interest, based on the allometric ratios published in the IEFC and the diameter and height values calculated above.

4. Calculation of total biomass including roots (ratio of root biomass to above-ground biomass). As in the case of managed forests, this value was obtained from Ruiz-Peinado *et al.* (2012) for broadleaved trees and Ruiz-Peinado *et al.* (2011) for conifers. If a species had not been included in previous publications, a mean value for genus or functional group was assigned.

5. Conversion of biomass values to content of carbon, based on the proportion of carbon contained in the biomass of the different species (Montero *et al.,* 2005) and, subsequently, of CO_2.

This methodology allows us to obtain a table in which the CO_2 fixed by every tree is given, for the main tree species and by age-range (Annex). This methodology was chosen, rather than others based on the calculation of total stocks by volume, according to the National Forest Inventory, because the latter involve the indirect calculation of biomass from volume, by means of conversion factors. Although the latter methodology is the one recommended in the *Guía para la estimación de absorciones de dióxido de carbono*, published by the Spanish Ministry for the Ecological Transition and the Demographic Challenge, it has been found that these factors may vary not only according to the species, but also to the diameter class of the tree and even the volume equation used. There is, therefore, a degree of uncertainty about its suitability, which has led us to look for an alternative, even though both methodologies provide similar results for most of the species considered.

The carbon sequestration period would run from the time of planting to the first intervention, depending on the species chosen. The CO_2 fixed should not involve any other input or output; it will be the value sequestered and accumulated annually in the period set by the total number of trees being planted. The reference scenario is no planting, so only the carbon fixed by planted trees is taken into account, regardless of the pre-existing vegetation, which is considered to fix the same amounts in both scenarios.

$$\mathbf{tCO_2\ sequestered}_{\mathbf{Plantation}} = tCO_2 \text{ sequestered by 1 tree}_{tn} \times \text{N. trees planted/ha}$$

CAPSULE 2. CARBON DIOXIDE EMISSIONS FROM OPERATIONS AND PRODUCTS OBTAINED

Firstly, a calculation is made of emissions from the work of cutting and hauling timber, clearing brush where necessary, and transporting products to the relevant industrial site. Within the period considered, which is less than 30 years, emissions from biomass used for energy (2 years of shelf life) and/or biomass for pallets and packaging (10 years of shelf life) are also calculated.

$$\mathbf{tCO_{2\ emitted}} = tCO_{2\ \text{emissions from forestry work and transport}} + tCO_{2\ \text{product combustion emissions}}$$

Emissions from forestry work and transport to industrial sites

Information on forestry work has been obtained from the study by Porras *et al.* (2017), considering chainsaws, brush cutters and forestry tractors with winches as CO_2 emitting machinery. Although the emissions indicated in the study account for between 0.2 and 0.4% of the total carbon stock over 150 years, which is negligible with respect to the rate of carbon sequestration, they have been taken into account in this methodology. However, it is also considered that the management of a stand may involve the repair of forest trails, in the period between two interventions.

The average density of roads in forest estates in Catalonia is around 90 m/ha, while the average density of tracks for haulage in private forest management schemes is 47 m/ha (POCTEFA iForWOOD, 2020. *Project for the implementation of the dynamic network of roads in Catalonia*, VIACAT). If we do not have the length of the haulage roads in a stand, for a preliminary estimate the average density referred to above can be considered.

Based on the yield of each forestry operation (in h/ha or h/km) and the fuel consumption of the machinery used for each of these operations (environmental values obtained from the Forest Ownership Centre) (Table 6), the total emissions per hectare managed (tCO_2/ha) were calculated (Table 7). For this calculation, emission factors have been obtained for each energy source according to th *Guia de càlcul d'emissions de gasos amb efecte d'hivernacle* (GEH) (Catalan Ministry of Territory and Sustainability (DTES), 2021). During the execution of the work there may be a combination of different machinery and other methods of haulage; here we have focused on the most common situations.

$$\mathbf{tCO_{2\ emitted\ forestry\ work}} = \text{yield (h/ha)} \times \text{consumption (l/h)} \times \text{emission factor } (tCO_2/l)$$

For **emissions produced by the transport** of commercial wood to industrial sites, the same guide (DTES, 2021) gives values for emissions from mobile sources using fuel in different means of transport (related to the movement of people or goods). For a 14-20 t rigid lorry driven in a rural area a value of 0.549 kg CO_2/km is obtained. According to the amount of wood cut and left at the landing (in tonnes of green wood, density $t_{50\%}/m^3$), we need to know the total number of km to be driven to the industrial site, i.e. the number of journeys to be made by lorry. In general, it is considered that the transformation process is economically viable with journeys of up to 80 km to the sawmill and 40 km for the production of biomass for energy, average values that can be used if the final destination of the product is not known.

Table 6. Production, consumption and emission factors according to forestry operations

Machinery	Energy source	Consumption	Emission factor	Sapling thinning	Clearing undergrowth	Selective/ Low thin-nings	End of rotation seed thinnings	Road repairs
	l	l/h	$kgCo_2/l$	(h/ha)	h/ha	h/ha	h/ha	h/km
Chainsaw	Petro	0.625	2.24	48	32	40	32	-
Brushcutter	Petrol	0.625	2.24			-	-	-
Forestry tractor	Diesel	6.5	2.68	-	-	24	16	-
Fork-lift truck	Diesel	12	2.68	-	-	-	-	8

Table 7. Carbon emission values of machinery according to forestry operation

Forestry treatments	Emissions from machinery (tCO_2/ha)
Sapling thinning	0.0672
Clearing undergrowth	0.0448
Selective/mixed/low thinning	0.4741
End of rotation seed thinnings	0.3235
Repair of forest roads (47 m/ha)	0.0121

Combustion emissions from short life-cycle products

From dry above-ground biomass, excluding the crown that remains on the ground, we can obtain different products: biomass for energy, low- and medium-quality lumber for packaging and pallets, and high-quality wood for structures and other material with a life of more than 30 years (Figure 4). Depending on the type of forest (young stands, adult stands, stands with wood of greater or smaller dimensions) and the treatment applied, a certain quantity of each type of product is obtained (Table 8). The carbon fixed in these products can be re-emitted into the atmosphere within a few years if the type of product obtained has a short life cycle and is not recycled. The product biomass will be converted into carbon and produce CO_2 emissions at a rate of 3.67 tCO_2/tC.

In this methodology, a 100% CO_2 return has been considered for wood destined for heating purposes and a lower value for timber destined for pallets and packaging, taking a product rotation factor into account, assuming that it is used in multiple cycles or rotations for distribution (Waste Agency of Catalonia, 2006). In the latter case, a recycling factor (Fr) of 25% has been assumed, taking compliance with European recycling regulations into account.

tCO_2 product emissions = ((BM + BC) x CF) x (% wooe for energy + (% timber for pallets and packaging) * (1-Fr)) x 3.67 tCO_2/tC

Where:

BM: biomass of trunk crown
BC: biomass with bark
CF: percentage by weight of carbon in dry matter by species
Fr: Recycling factor

Table 8 shows the different products that can be obtained depending on the type of treatment carried out and the quality of the resources in the forest according to its condition and structure. Early thinning provides short-lived products and, as the stand becomes more mature, we can obtain a higher proportion of timber for structures, which helps to store the fixed CO_2 from the atmosphere for longer.

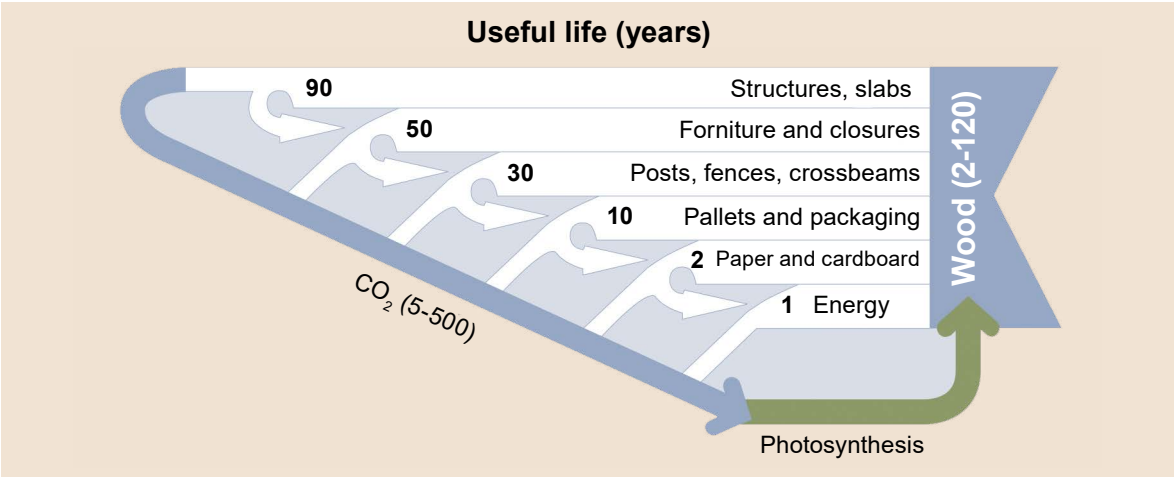


Figure 4. Shelf life of forest products. Source: Correal *et al.* (2017).

Table 8. Percentatge of products that can be obtained to the type of forestry treatments

Percentge (%) of marketable product obtained with ORGEST silvicultural treatment	**% long-life products (>30 years):** structures, furniture, fences, poles, crossbeams	**% medium-life products (<10 years):** pallets and packaging	**% short-life products (< 2 years):** paper, cardboard and energy
EA - Regulation of competition, sapling thinning/regrowth	0	0	0
EA - Regulation of competition, young pine forests (<30 years)	0	20	80
EA - Regulation of competition, adult pine forests - Thinning with low-quality products	10	40	50
EA- Regulation of competition, adult pine forests - Thinning with high-quality products	30	50	20
EA- Regulation of competition, broad-leaved/oak - low-quality products	0	0	100
EA- Regulation of competition, broad-leaved/oak - high-quality products	40	0	60
UEA- Regulation of competition, pine forests (selective cutting)	20	50	30
UEA- Regulation of competition, broadleaved/oak (selective cutting)	30	0	70
Long rotation forestry in pine forests	40	50	10
Long rotation forestry in broadleaved/ oak forests	40	0	60
Conversion to pasture in pine forests	20	50	30
Conversion to pasture in broadleaved/ oak forests and grazing land	0	0	100

EA: even-aged forest. UEA: UneEven-aged forest.
Low - and high - quality products are linked to size and form.

CAPSULE 3. CARBON DIOXIDE EMISSIONS AVOIDED

This capsule includes emissions that are avoided due to an improvement in stand structure that reduces the severity or intensity of forest fires, so that it does not emit part of the final fuel stock existing at a given time. For areas included in strategic fire prevention areas (PEG/AFG, others) established in landscape-scale plans, the emissions avoided can also be calculated according to the major forest fire prevention function of these infrastructures. On the other hand, emissions from fossil fuels or from the use of non-renewable materials avoided when they are replaced by biomass or wood are also considered (substitution effect).

$$\mathbf{tCO_{2\ avoided}} = tCO_{2\ \text{avoided by fire prevention}} + tCO_{2\ \text{avoided by substitution}}$$

Emissions avoided in fire prevention in a stand where preventive action has been taken

Large forest fires in Catalonia have average emissions (2006-2015) of 56,000 tCO_2 year-1 and are an essential modelling agent of Mediterranean cultural landscapes (Balde and Vega-García, 2019). Currently most fires are caused by human activity, and a small number that overcome extinction capacity under extreme weather conditions are responsible for most of the areas burnt (Rodrigues *et al.,* 2019). The CO_2 emissions expected from forest fires (tCO_2 ha^{-1} $year^{-1}$) can be determined from (i) the likelihood of burning and (ii) the most likely effects of fire in the stand affected (Alcasena *et al.,* 2021).

In order to determine the **annual probability of burning**, it is essential to simulate the behaviour and spread of thousands of forest fires (≥ 10,000 years) on a landscape scale. For this purpose, the minimum travel time (MTT) algorithm (Finney, 2002) has been used, calibrated at 150 m resolution to replicate the distribution of fires (burnt areas) observed during the period 1998 to 2015 (Alcasena *et al.,* 2018). In fire simulation, the most frequently observed weather scenarios are considered (e.g. synoptic conditions) in each macro-area (e.g. differentiated fire regime zone) and a distribution of fires based on occurrence (likelihood of ignition) models. By superimposing the burnt sections in the areas simulated we can calculate the frequency with which each stand (or pixel) in the landscape burns. Dividing this by the number of years simulated gives us the annual probability of burning. The annual probability of burning can be expressed as a value between 0 and 1, or a mean recurrence period in the burned area (e.g. p = 0.04, T = 25 years). So that this parameter can be used in the methodology presented here, a map has been drawn up with the annual probability of burning for the whole of Catalonia (Figure 5).

To **calculate the effects of or emissions from a fire** there are numerous procedures depending on the scale of application and the difficulty of obtaining input values for the calculations. First the biomass available must be determined, then the biomass consumption (fuel consumption or efficiency of burning) for each pixel or stand, and finally an emission factor is applied, relating the grams emitted of CO_2 per kilogram of dry matter consumed. It is not normally possible to have data for all fuel compartments for a given stand fuel model, unless destructive sampling is carried out, and other procedures more suitable for regional scales have to be applied. At the stand level, dry matter should be presented in t ha^{-1} for each of the fractions (fine, medium, coarse deadwood, organic soil matter, herbaceous, scrub, etc.) typified in the corresponding fuel model. Consumption or efficiency can be determined with models of burning conditions (fuel moisture for each fraction) (Prichard *et al.,* 2017) or by using response

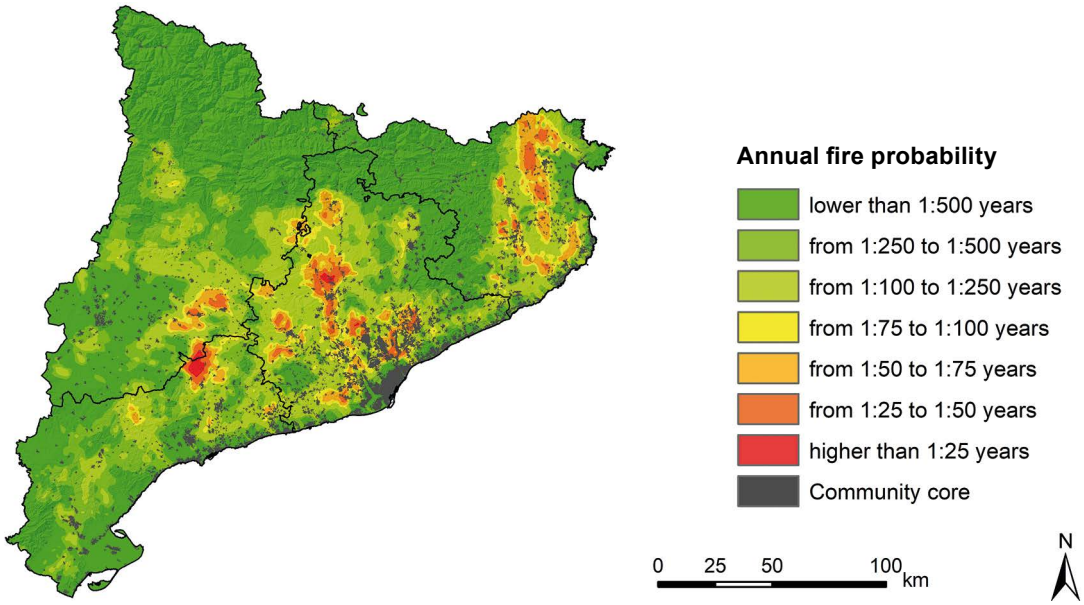


Figure 5. Map of annual fire probability. Source: Alcasena *et al.* (2018).

functions that estimate stand or pixel consumption based on the flame length obtained in the simulations (Ager *et al.*, 2010).

In our methodology, for the whole of Catalonia, the fraction of fuel burnt is obtained from published ratios of combustion factors or percentages of biomass consumed corresponding to vegetation classes typical of the Mediterranean environment (De Santis *et al.,* 2010) and fire severity classes based on simulations of conditional flame length (CFL) mapping (Table 9).

At the stand level, we need to know the value of the existing above-ground biomass and to apply the combustion factors for the relevant intensity and the average CO_2 emission factor for the vegetation type. Finally, it will be necessary to apply emission factors extracted from the literature for Mediterranean formations (Table 9), converted from gCO_2/kg dry a matter to tCO_2/t dry matter to facilitate calculations in forest units of measurement (Clinton *et al.,* 2006, 1396-1580 g/kg^{-1}).

The following considerations have been taken into account in defining management and non-management scenarios:

in **the unmanaged scenario**, it is assumed that a high-intensity fire will be generated on the existing above-ground biomass (AGB), ac-

Table 9. Combustion factors as a function of expected intensity (De Santis *et al.,* 2010) **and CO_2 emission factors** (Miranda *et al.,* 2005), **by vegetation type**

Vegetation type	**Combustion factor (CF)** (% biomass consumed)				CO_2 emission factor (EF) (tCO_2tms^{-1})
	fire severity				
	Low (CFL < 1.2 m)	Medium (CFL = 1.2-2.4 m)	High (CFL = 2.4-3.4 m)	Very high (CFL => 3.4 m)	
Scrub	0.71	0.84	0.89	0.95	1.477
Conifers	0.25	0.47	0.56	0.65	1.627
Boadleaved trees	0.25	0.40	0.48	0.56	1.393

cording to the annual probability considered. We assume that, without management, the expected intensity is high, thus avoiding the need to determine flame length, as the simulations have been performed under summer weather conditions (97th percentile) and for fires larger than 100 ha 100 ha. This methodology is applied to biomass before any measures are implemented to estimate what would be burnt in the absence of any management. This is a conservative value, as it only considers fires of a certain size (+100 ha), which produce the most emissions, and only emissions in the form of CO_2 (emissions of other C compounds such as CO, methane, particulate matter or black carbon, in lower orders of magnitude, are disregarded, e.g. CO emission factor = 0.093 tCO/t dry matter).

$$\mathbf{tCO_{2\ emitted\ unmanaged}} = (tAGB_{pre\text{-}treatment} \times CF_{(high\ intensity)} \times EF)$$

In **the managed scenario**, it is assumed that the fire's intensity will be lower, as the silvicultural treatment has reduced the above-ground biomass (AGB). Potential emissions from the stand are reduced, the intensity being lower due to management and the lower biomass available, but the probability of fire is estimated to remain the same, as it depends on the initiation and spread of fires around the stand, where no management measures are taken. The costs in CO_2 corresponding to management activities are already taken into account in the emissions from forestry work.

$$\mathbf{tCO_{2\ emitted\ management}} = (tAGB_{post\text{-}treatment} \times CF_{(low\ intensity)} \times EF)$$

The values obtained reflect the emissions if a fire were to occur next year, but the probability of its occurrence is equally distributed over the entire recurrence period for the pixel. It must, therefore, be multiplied by the annual probability of fire at that point (recurrence) and by the number of years to be considered in the emission avoidance project. It is proposed that this period should correspond to the number of years that management is effective in relation to the prevention of large fires, estimated, on average, to be 8 years, based on expert knowledge and data generated in the CPF network of demonstration plots.

The impact of management impact (CO_2 emissions avoided) will thus be the difference between the likely emissions without management and the likely emissions during the years that the forestry measures are effective, in the event of a fire according to the recurrence rate at that particular location:

$$\mathbf{tCO_{2\ fires\ avoided\ (stand)}} = (tCO_{2\ emitted\ unmanaged} - tCO_{2\ emitted\ managed}) \times \text{Anual probability of fire}_{stand} \times 8\ \text{years}$$

Emissions avoided in strategic fire prevention areas

If the stand where measures have been taken is intended to prevent large forest fires and is part of the approved fire prevention infrastructures, or is included in different landscape scale plans (technical plans for forest management and improvement, priority protection perimeters, etc.), in addition to reducing emissions in the managed plot itself, the prevention effect spreads to the surrounding (untreated) plots. To assess this additional impact, the methodology described in the previous section can be adapted and applied to the **area potentially protected** by this infrastructure. This calculation is, however, restricted to cases where the exact area protected by the strategic management point

is known, which requires careful delimitation by experts and is not always available.

A possible approach to delimiting the protected area, or the area affected by the infrastructure, is to estimate the effect that the measures taken have on the spread of fire or on the probability of burning in neighbouring areas, the «shadow effect», which varies according to meteorological conditions, the distribution of fuel in the landscape, orography and the location of ignition points. Treated plots interact with each other, with a synergistic effect, but the relationship between managed area and lower burn probability is not linear in the landscape but influenced by the size/shape of plots and their spatial distribution. This means that the use of fire simulators is necessary to assess the effects of preventive forest management. It is not, therefore, feasible to determine potentially protected areas for the whole of Catalonia, as it is not computationally possible to simulate thousands of fires for all the existing strategic stands or combinations of them.

The procedure we propose here for estimating protected areas is, therefore, not determined by *ad hoc*, but by the procedure proposed by Castellnou *et al.* (2019), where protected areas in the highest risk scenario are defined as the «polygon of fire potential that identifies a spatial unit in which the action of fire is expected to be homogeneous» which, in any case, requires delimitation by firefighting experts.

Furthermore, recognising that, beyond changing the effects of fire, strategic management points are, above all, opportunities for extinguishing it, we can introduce a change in **the management scenario** in which we assume that the probability of fire in a «protected area» is equal to zero during the 8 years in which the silvicultural treatment applied is effective. If a strategic point is managed, the fire is stopped at that point and the area protected by it does not burn, so the likely emissions of the management scenario are zero.

In the case on the **unmanaged scenario**, the methodology is applied as described in the previous section, considering that a fire will occur with an annual probability equal to that of the strategic point where forest management is applied.

Therefore, the emissions avoided, if a strategic point is managed, would be:

$$\mathbf{tCO_{2\ avoided\ fires\ (strategic\ point)}} = tCO_{2\ \text{emitted without management protected area}} \times \text{annual probability of fire}_{\text{strategic point}} \times 8 \text{ years}$$

Where: $\mathbf{tCO_{2\ emitted\ without\ management\ protect\ area}} = (tAGB_{\text{protected area}} \times CF_{\text{(high intensity)}} \times EF)$

The value of the existing above-ground biomass (AGB) in the protected area can be obtained from the information collected in individual or joint forest management plans, or from the maps of biophysical variables of trees in Catalonia for 2015-2016 drawn up by the ICGC and CREAF, to which we can apply the combustion factors for high-intensity fires and the average CO_2 emissions factor according to type of vegetation, as shown in Table 8. To avoid the need to determine values for different vegetation types, we can assume an average emission factor for any vegetation type of 1.650 tCO_2/t, an approximation frequently used for forested areas in the EU (Van der Werf *et al.,* 2017). In this case, an average combustion factor for high-intensity fires of 0.63 (CFL=2.4-3.4 m) is also used.

Substitution-effect emissions

The use of wood products displaces the use of other materials or energy sources, such as plastics, concrete, steel or fossil fuels, which leave a larger carbon footprint. The substitution (or displacement) coefficient measures the difference between the greenhouse gas emissions associated with the use of wood and an alternative material. If the coefficient is positive, it means that using wood products generates a smaller carbon footprint than using products made of other materials. This substitution effect is more or less beneficial depending on whether the type of treatment implemented leads to lower emissions in the end product, a greater substitution effect and lower emissions during the process of extracting the material from the forest (Agostini *et al.,* 2013; Martel, 2019). When wood is used as a fuel, it emits more CO_2 than gas or oil to produce same amount of heat, but less CO_2 is emitted in its production.

Valada *et al.* (2016) carried out an analysis of 51 comparative studies of wood products and products with equivalent functions made of other materials to calculate their substitution coefficients. The authors identify a wide range of results (between -0.7 and 5.1 kg C/kg C), depending on the life cycle stages and the substitute material considered. The average value of the substitution coefficient at the production stage is 0.8 kg C/kg C of a wood product, i.e. for every kilogram of carbon in wood products that displaces products made of other materials, there is an average reduction in emissions of about 0.8 kg. For structural timber, the figure is 1.3 kg C/kg C (Valada *et al.,* 2016). Other meta-analyses of the effect of substituting wood for other materials or fuels mention coefficients ranging from 0.59 to 3.47 tCO_2eq/m^3 wood (Rüter *et al.,* 2016, Roux *et al.,* 2017).

Other meta-analyses of the effect of substituting wood for other materials or fuels mention coefficients ranging from 0.59 to 3.47 «Bas Carbone» label (CNPF, 2019, based on Valada, 2016), the volume has been adapted to tonnes of product based on a life cycle of 150 years. Using the formula proposed by Sathre and O'Connor (2010), comparing emissions from non-renewable products with those from wood, and taking into account the life cycle values of Scots pine, which are comparable to other conifers, the following factors are obtained:

- **0.876 tCO_2** per tonne of **energy product** replacing diesel,
- **0.599 tCO_2** per tonne of **pallet product** as a replacement for disposable plastic pallets, and
- **1.069 tCO_2** per tonne of **structural product** as a replacement for reinforced concrete beams.

The index is calculated by taking into account both the emissions generated by the wood product and the emissions of the product substituted, subtracting what is generated from what is avoided. In the case of woodchips and diesel, all carbon contained in these materials is released into the atmosphere during combustion within the period analysed (100 years). In the case of pallets, this depends on the end-of-life scenario applied. In the case of wood, it is assumed that 38% of the product is landfilled and a further 32% is incinerated; therefore, 70% of the C will return to the atmosphere in the study period considered (100 years). In the case of plastic, it is assumed that 45% is incinerated and therefore only 45% of the C returns to the atmosphere (the remaining 55% goes to landfill and the C it contains does not reach the atmosphere in the 100-year period).

ANNEXES

Tables of carbon sequestration balances in managed forests, by silvicultural treatment, and non-managed forests, for the main forest typologies.

Table of carbon sequestration in tree plantations, by age range and for the maoin tree forest species

Table 1. Carbon sequestration balance for managed and unmanaged Scots pine forests

Main species	SQ	ORGEST Model	Forestry treatment	Carbon balance (t/ha/year)		
				Managed forests	Non-managed forests	Management -Non-management
Pinus sylvestris	A	Ps01	Sapling thinning	3.55	2.68	0.87
Pinus sylvestris	A	Ps01	Selective thinning (I)	4.36	2.91	1.45
Pinus sylvestris	A	Ps01	Selective thinning (II)	3.13	2.60	0.54
Pinus sylvestris	A	Ps01	Selective thinning (III)	2.63	2.09	0.54
Pinus sylvestris	A	Ps01	Selective thinning (IV)	2.14	1.59	0.55
Pinus sylvestris	A	Ps01	Preparatory cutting	0.07	0.01	0.06
Pinus sylvestris	A	Ps01	Seed cutting	0.11	-0.11	0.22
Pinus sylvestris	A	Ps01	Final cutting	0.52	-0.39	0.91
Pinus sylvestris	A	Ps02	Sapling thinning	2.36	2.16	0.20
Pinus sylvestris	A	Ps02	Low thinning (I)	3.94	3.02	0.92
Pinus sylvestris	A	Ps02	Low thinning (II)	3.48	2.68	0.80
Pinus sylvestris	A	Ps02	Low thinning (III)	2.67	1.43	1.24
Pinus sylvestris	A	Ps02	Low thinning (IV)	1.94	-1.01	2.95
Pinus sylvestris	A	Ps02	Preparatory cutting	-0.12	-0.14	0.02
Pinus sylvestris	A	Ps02	Seed cutting	-0.14	-0.16	0.02
Pinus sylvestris	A	Ps02	Final cutting	-0.03	-0.14	0.11
Pinus sylvestris	A	Ps03	Sapling thinning	2.71	2.31	0.41
Pinus sylvestris	A	Ps03	Low thinning (I)	3.86	3.05	0.81
Pinus sylvestris	A	Ps03	Low thinning (II)	2.78	2.57	0.21
Pinus sylvestris	A	Ps03	Preparatory cutting	1.16	2.03	-0.87
Pinus sylvestris	A	Ps03	Seed cutting	-0.33	0.82	-1.14
Pinus sylvestris	A	Ps03	Final cutting	0.05	-0.03	0.08
Pinus sylvestris	A	Ps04	Selective cutting	2.02	0.70	1.32
Pinus sylvestris	A	Ps05	Selective cutting	2.16	0.70	1.46
Pinus sylvestris	A	Ps07	Sapling thinning	3.78	2.73	1.05
Pinus sylvestris	A	Ps07	Low thinning (I)	4.24	2.82	1.41
Pinus sylvestris	A	Ps07	Low thinning (II)	4.25	2.37	1.88
Pinus sylvestris	A	Ps07	Low thinning (IV)	2.82	1.78	1.03
Pinus sylvestris	A	Ps07	Low thinning (V)	1.71	-0.58	2.30
Pinus sylvestris	A	Ps07	Preparatory cutting	0.16	-0.15	0.32
Pinus sylvestris	A	Ps07	Seed cutting	-0.20	-0.19	-0.01
Pinus sylvestris	A	Ps07	Final cutting	0.51	-0.26	0.77
Pinus sylvestris	B	Ps08	Sapling thinning	1.83	1.84	-0.01
Pinus sylvestris	B	Ps08	Selective thinning (I)	1.89	1.88	0.01
Pinus sylvestris	B	Ps08	Selective thinning (II)	1.20	-0.66	1.86
Pinus sylvestris	B	Ps08	Selective thinning (III)	1.12	-0.72	1.84
Pinus sylvestris	B	Ps08	Preparatory cutting	-0.30	-0.06	-0.24
Pinus sylvestris	B	Ps08	Seed cutting	0.07	-0.09	0.15
Pinus sylvestris	B	Ps08	Final cutting	0.49	-0.45	0.95
Pinus sylvestris	B	Ps09	Sapling thinning	1.37	1.51	-0.14
Pinus sylvestris	B	Ps09	Low thinning (I)	1.82	1.91	-0.09
Pinus sylvestris	B	Ps09	Low thinning (II)	1.30	-0.07	1.37
Pinus sylvestris	B	Ps09	Preparatory cutting	0.27	-0.26	0.53
Pinus sylvestris	B	Ps09	Seed cutting	-0.07	-0.30	0.23
Pinus sylvestris	B	Ps09	Final cutting	0.05	-0.02	0.07
Pinus sylvestris	B	Ps10	Selective cutting	1.13	0.76	0.37
Pinus sylvestris	B	Ps11	Sapling thinning	2.07	1.35	0.72
Pinus sylvestris	B	Ps11	Low thinning	1.87	1.44	0.43
Pinus sylvestris	B	Ps11	Low thinning	1.69	1.33	0.36
Pinus sylvestris	B	Ps11	Low thinning	1.08	-0.19	1.27
Pinus sylvestris	B	Ps11	Preparatory cutting	-0.11	-0.09	-0.02
Pinus sylvestris	B	Ps11	Seed cutting	0.04	-0.08	0.12
Pinus sylvestris	B	Ps11	Final cutting	0.29	-0.41	0.70
Pinus sylvestris	C	Ps13	Sapling thinning	0.87	0.81	0.05
Pinus sylvestris	C	Ps13	Selective thinning (I)	0.90	0.91	-0.01
Pinus sylvestris	C	Ps13	Selective thinning (II)	0.66	0.73	-0.07
Pinus sylvestris	C	Ps13	Selective thinning (III)	0.59	-0.70	1.29
Pinus sylvestris	C	Ps13	Preparatory cutting	0.14	-0.06	0.20
Pinus sylvestris	C	Ps13	Seed cutting	0.11	-0.06	0.17
Pinus sylvestris	C	Ps13	Final cutting	0.14	-0.54	0.67
Pinus sylvestris	C	Ps14	Selective cutting	0.55	0.33	0.23
Pinus sylvestris	C	Ps15	Sapling thinning	0.87	0.70	0.17
Pinus sylvestris	C	Ps15	Low thinning (I)	0.67	0.80	-0.13
Pinus sylvestris	C	Ps15	Low thinning (II)	0.76	0.81	-0.05
Pinus sylvestris	C	Ps15	Low thinning (III)	0.68	-0.55	1.23
Pinus sylvestris	C	Ps15	Preparatory cutting	0.16	-0.06	0.22
Pinus sylvestris	C	Ps15	Seed cutting	-0.02	-0.08	0.05
Pinus sylvestris	C	Ps15	Final cutting	0.03	-0.46	0.49

Table 2. Carbon sequestration balance for managed and unmanaged black pine forests

Main species	SQ	ORGEST Model	Forestry treatment	Carbon balance (t/ha/year)		
				Managed forests	Non-management model	Non-managed forests
Pinus nigra	A	Pn01	Sapling thinning (I)	5.28	3.37	1.91
Pinus nigra	A	Pn01	Selective thinning (I)	3.51	3.23	0.28
Pinus nigra	A	Pn01	Selective thinning (II)	2.44	2.68	-0.24
Pinus nigra	A	Pn01	Selective thinning (III)	1.23	1.30	-0.07
Pinus nigra	A	Pn01	Preparatory cutting	0.81	0.14	0.67
Pinus nigra	A	Pn01	Seed cutting	-0.02	0.22	-0.24
Pinus nigra	A	Pn01	Final cutting	0.63	0.06	0.57
Pinus nigra	A	Pn02	Selective cutting	1.71	1.38	0.33
Pinus nigra	A	Pn03	Sapling thinning	2.85	3.37	-0.52
Pinus nigra	A	Pn03	Thinning (I)	2.62	3.08	-0.46
Pinus nigra	A	Pn03	Thinning (II)	2.37	1.09	1.29
Pinus nigra	A	Pn03	Thinning (III)	1.29	0.03	1.26
Pinus nigra	A	Pn03	Preparatory cutting	1.01	0.10	0.91
Pinus nigra	A	Pn03	Seed cutting	-0.03	0.18	-0.21
Pinus nigra	A	Pn03	Final cutting	0.58	0.04	0.54
Pinus nigra	A	Pn04	Sapling thinning	2.83	2.73	0.09
Pinus nigra	A	Pn04	Low thinning	4.26	2.83	1.43
Pinus nigra	A	Pn04	Preparatory cutting	2.67	2.32	0.35
Pinus nigra	A	Pn04	Seed cutting	0.56	1.25	-0.69
Pinus nigra	A	Pn04	Final cutting	0.00	0.28	-0.28
Pinus nigra	A	Pn05	Selective cutting	1.59	1.38	0.21
Pinus nigra	B	Pn06	Sapling thinning	2.62	1.79	0.83
Pinus nigra	B	Pn06	Low thinning (I)	1.98	1.24	0.74
Pinus nigra	B	Pn06	Low thinning (II)	0.75	-0.13	0.87
Pinus nigra	B	Pn06	Low thinning (III)	0.05	-0.15	0.21
Pinus nigra	B	Pn06	Preparatory cutting	-0.11	0.06	-0.17
Pinus nigra	B	Pn06	Seed cutting	0.15	0.09	0.07
Pinus nigra	B	Pn06	Final cutting	0.54	-0.03	0.57
Pinis nigra	B	Pn07	Selective cutting	1.41	0.82	0.58
Pinus nigra	C	Pn09	Sapling thinning	1.39	0.72	0.67
Pinus nigra	C	Pn09	Low thinning	1.12	0.63	0.50
Pinus nigra	C	Pn09	Low thinning	0.69	-0.05	0.74
Pinus nigra	C	Pn09	Preparatory cutting	-0.02	0.08	-0.10
Pinus nigra	C	Pn09	Seed cutting	-0.09	0.07	-0.16
Pinus nigra	C	Pn09	Final cutting	0.02	-0.12	0.14

Table 3. Carbon sequestration balance for managed and unmanaged Aleppo pine forests

Main species	SQ	ORGEST Model	Forestry treatment	Carbon balance (t/ha/year)		
				Managed forests	Non-management model	Non-managed forests
Pinus halepensis	A	Ph01	Sapling thinning	5.55	3.86	1.69
Pinus halepensis	A	Ph01	Mixed thinning	3.39	3.77	-0.38
Pinus halepensis	A	Ph01	Low thinning (I)	1.69	1.64	0.05
Pinus halepensis	A	Ph01	Low thinning (II)	0.73	-0.42	1.15
Pinus halepensis	A	Ph01	Seed cutting	-0.09	-0.09	-0.01
Pinus halepensis	A	Ph01	Final cutting	1.48	-0.01	1.48
Pinus halepensis	A	Ph02	Sapling thinning	2.85	3.77	-0.92
Pinus halepensis	A	Ph02	Mixed thinning	2.08	3.30	-1.23
Pinus halepensis	A	Ph02	Low thinning	1.20	-0.32	1.52
Pinus halepensis	A	Ph02	Seed cutting	-0.12	-0.02	-0.11
Pinus halepensis	A	Ph02	Final cutting	1.61	0.14	1.48
Pinus halepensis	A	Ph03	Sapling thinning	4.67	3.87	0.80
Pinus halepensis	A	Ph03	Mixed thinning	3.51	3.62	-0.10
Pinus halepensis	A	Ph03	Seed cutting	-0.39	0.65	-1.04
Pinus halepensis	A	Ph03	Final cutting	1.41	-0.06	1.47
Pinus halepensis	B	Ph05	Sapling thinning	2.20	1.64	0.55
Pinus halepensis	B	Ph05	Mixed thinning	1.28	1.49	-0.21
Pinus halepensis	B	Ph05	Low thinning	0.98	0.97	0.01
Pinus halepensis	B	Ph05	Seed cutting	-0.26	-0.08	-0.18
Pinus halepensis	B	Ph05	Final cutting	0.48	-0.07	0.54
Pinus halepensis	B	Ph06	Sapling thinning	2.08	1.64	0.43
Pinus halepensis	B	Ph06	Mixed thinning	1.03	1.45	-0.42
Pinus halepensis	B	Ph06	Seed cutting	-0.07	1.20	-1.26
Pinus halepensis	B	Ph06	Final cutting	0.60	0.98	-0.38

Table 4. Carbon sequestration balance for managed and unmanaged holm oak forests

Main species	SQ	ORGEST Model	Forestry treatment	Carbon balance (t/ha/year)		
				Managed forests	Non-managed forests	Management -Non-management
Quercus ilex	A	Qii01	Selective cutting	1.28	0.51	0.77
Quercus ilex	A	Qii02	Selection of shoots	2.15	1.26	0.88
Quercus ilex	A	Qii02	C. to encourage regrowth	3.81	1.09	2.72
Quercus ilex	A	Qii03	Selection of shoots	2.74	1.22	1.52
Quercus ilex	A	Qii03	Thinnings (I)	3.84	1.18	2.66
Quercus ilex	A	Qii03	Thinnings (II)	0.50	-0.98	1.48
Quercus ilex	A	Qii03	C. to encourage regrowth	3.85	-0.14	4.00
Quercus ilex	B	Qii04	Selective cutting	0.76	0.20	0.56
Quercus ilex	B	Qii05	Selection of shoots	1.30	0.32	0.98
Quercus ilex	B	Qii05	C. to encourage regrowth	2.40	-0.21	2.62
Quercus ilex	B	Qii06	Selection of shoots	0.99	0.32	0.66
Quercus ilex	B	Qii06	Thinnings (I)	3.82	-0.21	4.03
Quercus ilex	B	Qii06	Thinnings (II)	0.46	-0.51	0.96
Quercus ilex	B	Qii06	C. to encourage regrowth	2.46	-0.17	2.62
Quercus ilex	C	Qii07	Selection of shoots	0.92	-0.34	1.26
Quercus ilex	C	Qii07	Thinnings (I)	1.08	-0.20	1.27
Quercus ilex	C	Qii07	Thinnings (II)	0.82	-0.26	1.09
Quercus ilex	C	Qii07	C. to encourage regrowth	0.57	-0.16	0.73

Table 5. Carbon sequestration by species and time periods

Species	CO_2 fixed per period (tCO_2/tree)														
	Year10	Year15	Year20	Year25	Year30	Year35	Year40	Year45	Year50	Year55	Year60	Year65	Year70	Year75	Year80
Abies alba	0.00	0.00	0.01	0.01	0.02	0.03	0.04	0.06	0.08	0.10	0.13	0.16	0.20	0.24	0.29
Acer campestre	0.00	0.02	0.05	0.10	0.17	0.26	0.36	0.48	0.62	0.77	0.94	1.13	1.33	1.54	1.77
Acer monspessulanum	0.00	0.02	0.05	0.09	0.15	0.23	0.31	0.42	0.53	0.66	0.80	0.95	1.11	1.29	1.47
Acer opalus	0.00	0.02	0.05	0.10	0.17	0.25	0.35	0.46	0.59	0.73	0.89	1.06	1.24	1.43	1.63
Alnus glutinosa	0.00	0.03	0.08	0.16	0.27	0.41	0.58	0.79	1.01	1.27	1.56	1.87	2.21	2.57	2.97
Arbutus unedo	0.00	0.02	0.05	0.10	0.16	0.24	0.33	0.44	0.56	0.69	0.83	0.99	1.15	1.32	1.51
Betula pendula	0.01	0.02	0.04	0.08	0.12	0.18	0.24	0.32	0.41	0.50	0.61	0.72	0.84	0.97	1.11
Castanea sativa	0.02	0.07	0.15	0.27	0.42	0.60	0.82	1.07	1.36	1.67	2.02	2.40	2.81	3.24	3.70
Celtis australis	0.00	0.02	0.05	0.09	0.15	0.23	0.33	0.45	0.58	0.73	0.90	1.08	1.28	1.50	1.73
Corylus avellana	0.00	0.02	0.05	0.10	0.16	0.24	0.33	0.44	0.56	0.69	0.83	0.99	1.16	1.34	1.53
Cupressus sempervirens	0.00	0.00	0.01	0.02	0.03	0.04	0.06	0.08	0.11	0.14	0.17	0.21	0.26	0.30	0.36
Eucalyptus globulus	0.03	0.13	0.32	0.61	1.03	1.61	2.35	3.30	4.46	5.86	7.53	9.51	11.80	14.46	17.50
Fagus sylvatica	0.00	0.00	0.01	0.02	0.03	0.05	0.07	0.10	0.14	0.18	0.23	0.29	0.35	0.41	0.49
Fraxinus angustifolia	0.00	0.02	0.05	0.10	0.17	0.26	0.37	0.50	0.64	0.80	0.98	1.17	1.38	1.61	1.85
Fraxinus excelsior	0.00	0.02	0.05	0.11	0.19	0.31	0.44	0.61	0.80	1.02	1.26	1.53	1.82	2.14	2.47
Ilex aquifolium	0.01	0.02	0.05	0.10	0.15	0.22	0.31	0.40	0.51	0.64	0.77	0.92	1.08	1.25	1.44
Juglans regia	0.00	0.02	0.05	0.10	0.16	0.24	0.34	0.46	0.60	0.75	0.92	1.10	1.30	1.52	1.75
Juniperus communis	0.01	0.01	0.02	0.02	0.03	0.04	0.05	0.06	0.08	0.09	0.11	0.14	0.16	0.19	0.22
Juniperus oxycedrus	0.01	0.01	0.02	0.02	0.03	0.04	0.05	0.06	0.08	0.09	0.11	0.13	0.16	0.18	0.21
Juniperus phoenicea	0.00	0.01	0.02	0.04	0.05	0.08	0.10	0.12	0.15	0.18	0.22	0.25	0.29	0.33	0.37
Pinus canariensis	0.00	0.01	0.03	0.05	0.08	0.12	0.17	0.22	0.28	0.36	0.44	0.53	0.62	0.73	0.84
Pinus halepensis	0.00	0.01	0.02	0.03	0.05	0.08	0.11	0.15	0.19	0.24	0.29	0.35	0.42	0.48	0.56
Pinus nigra	0.00	0.01	0.02	0.04	0.05	0.08	0.11	0.14	0.18	0.22	0.27	0.33	0.39	0.46	0.53
Pinus pinaster	0.01	0.03	0.06	0.11	0.16	0.23	0.31	0.41	0.51	0.63	0.76	0.91	1.06	1.22	1.39
Pinus pinea	0.01	0.02	0.04	0.07	0.11	0.17	0.23	0.30	0.39	0.49	0.60	0.72	0.85	0.99	1.14
Pinus radiata	0.04	0.18	0.46	0.93	1.62	2.59	3.87	5.50	7.55	10.04	13.05	16.61	20.79	25.65	31.24
Pinus sylvestris	0.01	0.03	0.05	0.07	0.11	0.15	0.20	0.25	0.32	0.39	0.47	0.55	0.64	0.74	0.84
Pinus uncinata	0.01	0.02	0.03	0.04	0.06	0.08	0.10	0.13	0.16	0.19	0.23	0.27	0.31	0.36	0.41
Platanus hispanica	0.01	0.03	0.07	0.13	0.22	0.32	0.46	0.61	0.79	0.98	1.20	1.44	1.69	1.97	2.26
Populus alba	0.06	0.19	0.41	0.71	1.10	1.55	2.08	2.66	3.29	3.96	4.67	5.41	6.17	6.94	7.73
Populus nigra	0.04	0.14	0.29	0.51	0.78	1.10	1.48	1.90	2.37	2.88	3.42	4.00	4.59	5.21	5.85
Populus tremula	0.05	0.20	0.46	0.84	1.33	1.92	2.61	3.37	4.19	5.07	6.00	6.96	7.94	8.95	9.96
Populus x canadensis	0.05	0.17	0.35	0.59	0.89	1.23	1.61	2.02	2.46	2.92	3.40	3.88	4.38	4.88	5.38
Prunus avium	0.00	0.02	0.05	0.10	0.17	0.26	0.36	0.49	0.63	0.79	0.97	1.16	1.37	1.60	1.84
Prunus spp.	0.00	0.02	0.05	0.09	0.15	0.23	0.32	0.44	0.56	0.71	0.87	1.05	1.24	1.44	1.66
Pseudotsuga menziesii	0.00	0.00	0.01	0.02	0.03	0.05	0.07	0.10	0.13	0.17	0.21	0.26	0.32	0.38	0.45
Pyrus spp.	0.00	0.02	0.05	0.09	0.15	0.22	0.31	0.41	0.53	0.66	0.81	0.97	1.14	1.33	1.53
Quercus canariensis	0.01	0.03	0.07	0.13	0.20	0.29	0.39	0.51	0.64	0.79	0.96	1.14	1.33	1.53	1.75
Quercus faginea	0.00	0.01	0.02	0.03	0.05	0.07	0.10	0.13	0.17	0.22	0.27	0.32	0.38	0.44	0.51
Quercus ilex	0.00	0.01	0.02	0.04	0.06	0.09	0.12	0.16	0.21	0.26	0.32	0.38	0.45	0.53	0.61
Quercus petraea	0.01	0.01	0.03	0.05	0.08	0.12	0.16	0.22	0.28	0.34	0.42	0.51	0.60	0.70	0.81
Quercus pubescens	0.00	0.01	0.03	0.05	0.07	0.11	0.15	0.20	0.26	0.33	0.40	0.48	0.57	0.67	0.78
Quercus pyrenaica	0.00	0.01	0.03	0.05	0.08	0.12	0.17	0.23	0.30	0.38	0.47	0.58	0.69	0.82	0.96
Quercus robur	0.00	0.02	0.04	0.08	0.14	0.21	0.30	0.40	0.52	0.65	0.81	0.98	1.16	1.36	1.58
Quercus suber	0.00	0.00	0.01	0.01	0.02	0.04	0.05	0.07	0.09	0.12	0.15	0.18	0.22	0.25	0.29
Salix alba	0.00	0.02	0.05	0.10	0.17	0.25	0.36	0.48	0.62	0.78	0.96	1.15	1.36	1.58	1.82
Salix atrocinerea	0.00	0.02	0.05	0.10	0.16	0.24	0.33	0.44	0.56	0.69	0.84	1.00	1.17	1.35	1.54
Salix caprea	0.00	0.02	0.05	0.10	0.17	0.25	0.34	0.46	0.58	0.72	0.88	1.04	1.22	1.41	1.61
Sorbus aria	0.00	0.02	0.05	0.10	0.16	0.24	0.33	0.44	0.57	0.71	0.86	1.02	1.20	1.39	1.59
Sorbus aucuparia	0.00	0.02	0.05	0.10	0.17	0.25	0.34	0.45	0.58	0.71	0.87	1.03	1.21	1.39	1.59
Sorbus domestica	0.01	0.03	0.06	0.11	0.18	0.26	0.35	0.45	0.57	0.70	0.85	1.00	1.16	1.34	1.53
Sorbus torminalis	0.00	0.02	0.05	0.10	0.16	0.25	0.35	0.47	0.60	0.76	0.93	1.11	1.31	1.53	1.76
Tilia cordata	0.00	0.02	0.05	0.09	0.15	0.23	0.32	0.44	0.57	0.73	0.90	1.10	1.32	1.55	1.81
Ulmus minor	0.01	0.03	0.08	0.15	0.25	0.37	0.51	0.68	0.87	1.08	1.31	1.55	1.82	2.10	2.39

References

Agència de Residus de Catalunya (ARC). 2006. Guia de bones pràctiques per al reciclatge i la recuperació de palets i embalatges de fusta. Legal deposit B-49146-2006.

Ager, A. A.; Finney, M. A.; McMahan, A. J.; Cathcart, J. 2010. Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. *Natural Hazards and Earth System Sciences*, 10: 2515-2526.

Agostini, A.; Giuntoli, J.; Boulamanti, A. 2013. Carbon accounting of forest bioenergy. JRC scientific and policy reports, EU 2014.

Alcasena, F. J.; Ager, A. A.; Salis, M.; Day, M. A.; Vega-Garcia, C. 2018. Optimizing prescribed fire allocation for managing fire risk in central Catalonia. *Sci. Total Environ*, 4, 872-885.

Alcasena, F.; Rodrigues, M.; Gelabert, P.; Ager, A.; Salis, M.; Ameztegui, A.; Cervera, T.; Vega-García, C. 2021. Fostering Carbon Credits to Finance Wildfire Risk Reduction Forest Management in Mediterranean Landscapes. Land, (10) 10: 1104. https://doi.org/10.3390/land10101104

Ameztegui, A.; Torné, G. 2020. Capacitat d'emmagatzematge de carboni de les principals espècies forestals de Catalunya. LIFE CLIMARK internal report. 8 pp.

Ameztegui, A.; Rodrigues, M.; Granda, V. Uncertainty of biomass stocks in Spanish forests: a comprehensive comparison of allometric equations. Submitted.

Andrews, P. L.; Heinsch, F. A.; Schelvan, L. 2011. How to generate and interpret fire characteristics charts for surface and crown fire behavior. Gen. Tech. Rep. RMRS-GTR-253. Fort Collins, CO: U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 40 pp.

Balde, B.; Vega-García, C. 2019. Estimación de emisiones de GEI y sus trayectorias en grandes incendios forestales en Cataluña, España. *Madera y Bosques*, 25, e2521764.

Banqué, M.; De Cáceres, M.; García-Valdés, R.; Martínez-Vilalta, J.; Valentín, J.; Vayreda, J. 2020. FOREStime. Canvi dels serveis ecosistèmics dels boscos de Catalunya al llarg dels darrers 25 anys (període 1990-2014). CREAF-OCCC-CTFC.

Beltrán, M.; Vericat, P.; Piqué, M.; Cervera, T. 2012. Models de gestió per als boscos de pinassa (*Pinus nigra Arn.*): producció de fusta i prevenció d'incendis forestals. Series: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST). Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya.

Beltrán, M.; Vericat, P.; Piqué, M.; Cervera, T. 2011. Models de gestió per als boscos de pi blanc (*Pinus halepensis L.*): producció de fusta i prevenció d'incendis forestals. Series: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST). Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya.

Castellnou, M.; Prat-Guitart, N.; Arilla, E., *et al.* 2019. Empowering strategic decision-making for wildfire management: avoiding the fear trap and creating a resilient landscape. **Fire ecol**,15, 31. https://doi.org/10.1186/s42408-019-0048-6.

Centre National de la Propriété Forestière (CNPF). 2019. Label Bas Carbone. Méthode conversion de taillis en futaie sur souches. Versió de 18/04/2019.

Clinton, N. E.; Gong, P.; Scott, K. 2006. Quantification of pollutants emitted from very large wild land fires in Southern California, USA. *Atmospheric Environment*, 40: 3686-3695.

Correal, E.; Labèrnia, C. 2017. Ús responsable dels productes fusters en elements urbans. Ajuntament de Barcelona.

De Santis, A.; Asner, G. P.; Vaughan, P. J.; Knapp, D. E. 2010. Mapping burn severity and burning efficiency in California using simulation models and Landsat imagery. *Remote Sensing of Environment*, 114(7): 1535-1545, DOI: 10.1016/j.rse.2010.02.008.

Departament de Territori i Sostenibilitat. 2001. Guia de càlcul d'emissions de gasos amb efecte d'hivernacle (GEH). Generalitat de Catalunya.

FAO and Ministerio de Medio Ambiente y Medio Rural y Marina de España. 2010. Evaluación de los recursos forestales mundiales. Informe nacional. España. FAO Departamento Forestal.

Gómez, M. 2020. Anàlisi de cicle de vida de productes de fusta derivats de diferents pràctiques de gestió forestal.

Gràcia, C.; Burriel, J. A.; Ibàñez, J.; Mata, T.; Vayreda, J. 2004. Inventari ecològic i forestal de Catalunya. Mètodes. Bellaterra: CREAF. ISBN: 84-932860-2-8.

Gràcia, C.; Sabaté, S.; Vayreda, J.; Sebastià, T.; Savé, R.; Alonso, M.; Vidal, M. 2010. Embornals. In: Segon informe sobre el canvi climàtic a Catalunya. Llebot, J. E. Institut d'Estudis Catalans i Generalitat de Catalunya. 1152 pp.

Gundersen, P.; Thybring, E.; Nord-Larsen, T.; Vesterdal, L.; Nadelhoffer, K. J.; Johannsen, V. K. 2021. Old-growth forest carbon sinks overestimated. *Nature*, 591(7851): E21, DOI: 10.1038/s41586-021-03266-z.

Herrmann, S., Kahl, T., Bauhus, J. 2015. Decomposition dynamics of coarse woody debris of three important central European tree species. For. Ecosyst. 2. https://doi.org/10.1186/s40663-015-0052-5

IPCC. 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Program. 2006. Eggleston, H. S.; Buendia, L.; Miwa, K.; Ngara, T.; Tanabe, K. (eds). Published by IGES, Japan.

Jandl, R. 2001. Secuestro de carbono en bosques. El papel del suelo. Taller Internacional sobre secuestro de carbono. IUFRO-RIFALC, Mérida, Venezuela.

Le Quéré, C.; Raupach, M. R.; Canadell, J. G. 2009. Trends in the sources and sinks of carbon dioxide. *Nature Geoscience*, DOI: 10.1038/ngeo689.

Long, J. 1985. A practical approach to density management. *Forest Chronicles*, 61: 23-27.

Luyssaert, S.; Schulze, E. D.; Börner, A.; Knohl, A.; Hessenmöller, D.; Law, B. E.; Ciais, P.; Grace, J. 2008: Old-growth forests as global carbon sinks. *Nature*, 455: 213-215.

Martel, S. 2019. Gestion, âge, CO_2 – que nous apprennent de récentes publications? [Management, age, CO_2 – what can we learn from recent publications?] *Forêt Entreprise*, 245: 59-61.

Martínez, E.; Alonso, L. 2015. Primeras experiencias en la cuantificación de la huella de carbono asociada a tratamientos silvícolas de mejora y prevención de incendios en montes mediterráneos en la provincia de Alicante. *Revista Montes*, 120: 6-11.

Mattson, K. G.; Swank, W. T.; Waide, J. B. 1987. Decomposition of woody debris in a regenerating, clear-cut forest in the Southern Appalachians. *Can. J. For. Res*, 17: 712-721. https://doi.org/10.1139/x87-114

Mayor X.; Rodà, F. 1993. Growth response of holm oak (*Quercus ilex L.*) to commercial thinning in the Montseny mountains (NE Spain).

Miranda, A. I.; Borrego, C.; Sousa, M.; Valente, J.; Barbosa, P.; Carvalho, A. 2005. Model of Forest Fire Emissions to the Atmosphere. Deliverable D252 of SPREAD Project (EVG1- CT-2001-00043). Department of Environment and Planning, University of Aveiro AMB-QA- 07/2005, Aveiro, Portugal. 48 pp.

Montero, G.; Cañellas, I.; Ruiz-Peinado, R. 2001. Growth and Yield Models for *Pinus halepensis Mill. Invest. Agr.: Sist. Recur. For*, 10(1).

Montero, G.; Ruiz-Peinado, R.; Muñoz. 2005. Producción de biomasa y fijación de CO_2 por los bosques españoles. Monografías INIA: Serie Forestal. Ministerio de Educación y Ciencia.

Naaburs, J.; Lindner, M.; Verkerk, P. J.; Gunia, K.; Deda, P.; Michalak, R.; Grassi, G. 2013. First signs of carbon sink saturation in European forest biomass. *Nature Climate Change*, 1-5.

Nabuurs, J.; Delacote, P.; Ellison, D.; Hanewinkel, M.; Hetemäki, L.; Lindner, M. 2017. By 2050 the Mitigation Effects of EU Forests Could Nearly Double through Climate Smart Forestry. (V8). DOI - 10.3390/f8120484

Nicholas, E.; Clinton, G. P.; Scott, K. 2006. Quantification of pollutants emitted from very large wild land fires in Southern California, USA. *Atmospheric Environment*, 40: 3686-3695.

Olson, J. S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology, 44: 322-331.

Palahí, M.; Grau, J. 2003. Site index model and individual-tree growth and mortality models for black pine (*Pinus nigra Arn.*) in Catalonia (Spain). *Investigación Agraria: Sistemas y Recursos Forestales*, 12(1): 137-148.

Palahí, M.; Tomé, M.; Pukkala, T.; Trasobares, A.; Montero, G. 2004. Site Index Model for *Pinus sylvestris* in north-east Spain. *Forest Ecology and Management*, 187(1): 35- 47.

Piqué, M.; Vericat, P.; Cervera, T.; Baiges, T.; Farriol, T. 2011. Tipologies forestals arbrades. Series: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST). Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya.

Piqué, M.; Vericat, P.; Beltrán, M. 2017. «Resource communication. ORGEST: Regional guidelines and silvicultural models for sustainable forest management». *Forest Systems,* 26 (2).

Porras, D.; Antón, A.; Savé, R. (unpublished). Actualización de la metodología de impacto ambiental de los bosques mediterráneos gestionados según los criterios ORGEST. Institut de Recerca i Teconologia Agroalimentàries. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya.

Porras, D.; Rabascall, X.; Cervera, T. 2015. Capacidad de sumidero de carbono de los modelos multifuncionales ORGEST. LIFE DEMORGEST internal report.

Porras, D.; Rabascall, X.; Cervera T. 2017. Efectes de la gestió forestal sobre la capacitat d'embornal de carboni dels boscos catalans. *Revista Silvicultura, 73*. Centre de la Propietat Forestal.

Prichard, S. J.; Kennedy, M. C.; Wright, C. S.; Cronan, J. B.; Ottmar, R. D. 2017. Predicting forest floor and woody fuel consumption from prescribed burns in southern and western pine ecosystems of the United States. *Forest Ecology and Management*, 405: 328-338.

Rodrigues, M.; Alcasena, F.; Vega-García, C. 2019. Modeling initial attack success of wild fire suppression in Catalonia, Spain. *Sci. Total Environ*, 2: 323.

Rodríguez, J.; Juanati, C.; Piqué, M.; Tolosana, E. 2005. Tècniques de desembosc en l'aprofitament forestal. Condicionants, mitjans i recomanacions. Exemple a Catalunya. Centre de la Propietat Forestal. Departament de Medi Ambient i Habitatge. Generalitat de Catalunya.

Roux, A.; Dhôte, J.-F. (coords), Achat, D.; Bastick, C.; Colin, A.; Bailly, A.; Bastien, J.-C.; Berthelot, A.; Bréda, N.; Caurla, S.; Carnus, J. M.; Gardiner, B.; Jactel, H.; Leban, J. M.; Lobianco, A.; Loustau, D.; Meredieu, C.; Marçais, B.; Mar-

tel, S.; Moisy, C.; Pâques, L.; Picart-Deshors, D.; Rigolot, E.; Saint-André, L.; Schmitt, B. 2017. *Quel rôle pour les forêts et la filière-bois françaises dans l'atténuation du changement climatique? Une étude des freins et leviers forestiers à l'horizon 2050. (What role can the French forestry and timber sector play in attenuating climate change? A study of forest obstacles and levers by 2050*). Study report for the Ministry of Agriculture and Food, INRA/IGN, 101 pp. + 230 pp. (Annexes).

Ruíz-Peinado, R. 2011. Los bosques y su efecto sumidero de carbono. CIFOR-INIA/IUGFS Situación actual de los bosques: retos y oportunidades. UIMP. Santander.

Ruiz-Peinado, R.; Del Rio, M.; Montero, G. 2011. New models for estimating the carbon sink capacity of Spanish softwood species. *Forest Systems*, 20(1): 176-188.

Ruiz-Peinado, R.; Montero, G.; Del Rio, M. 2012. Biomass models to estimate carbon stocks for hardwood tree species. *Forest Systems,* 21(1): 42-52.

Rüter, S.; Werner, F.; Forsell, N.; Prins, C.; Vial, E.; Levet, A. L. 2016. Climate benefits of material substitution by forest biomass and harvested wood products: Perspective 2030. Clim Wood2030. Final Report, Thünen Report 42, Braunschweig, Germany. http://dx.doi.org/doi:10.3220/REP1468328990000.

Sathre, R.; O'Connor, J. 2020. Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy* 13(2): 104-114.

Stephenson, N. L., *et al.* 2014. *Rate of tree carbon accumulation increases continuously with tree size. Nature*, 507: 90-93.

Valada, T.; Cardellini, G.; Vial, E.; Levet, A. L.; Muys, B.; Lamoulie, J.; Hurel, C.; Privat, F.; Cornillier, C.; Verbist, B. 2016. FORMIT Project - Deliverable 3.2 - LCA and Mitigation Potential from Forest Products. (FP7-311970).

Van der Werf, G. R.; Randerson, J. T.; Giglio, L.; van Leeuwen, T. T.; Chen, Y.; Rogers, B. M.; Mu, M.; Van Marle, M. J. E.; Morton, D. C.; Collatz, G. J.; Yokelson, R. J.; Kasibhatla, P. S. 2017. Global fire emissions estimates during 1997-2016. *Earth Syst. Sci.* Data: 9, 697-720. https://doi.org/10.5194/essd-9-697-2017

Vayreda, J.; Martínez-Vilalta, J.; Gracia, M.; Retana, J. 2012. Forest structure and management interact with recent changes in climate to determine the current forest carbon stock in Peninsular Spain. Global Change. *Biology*, 18: 1028-1041.

Vericat, P.; Beltrán, M.; Piqué, M. 2014. Competition and productivity of Mediterranean Pinus species in mixed forests: basing a particularised silviculture. MedPine5 Conference, comunicació oral.

Wang, S.; Zhang, Y. G.; Ju, W.; Chen, J.; Ciais, P.; Cescatti, A.; Sardans, J.; Janssens, I. A.; Sardans, J.; Fernández-Martínez, M.; Penuelas, J. 2020. Recent global decline of CO_2 fertilization effects on vegetation photosynthesis. Science, DOI: 10.1126/science.abb7772

2

Impact of forest management on water resources

Chapter 2

Impact of forest management on water resources

2.1 Introduction

The forest-water relationship has been studied over time in the framework of forest hydrology, mainly with the aim of regulating floods, preventing erosion and preserving water quality. It is only more recently that emphasis has been placed on the relationship between the area and structure of forests and the water resources provided by the ecosystem. In Catalonia, successive studies (Gallart and Llorens, 2004; Gallart *et al.,* 2011; Gallart *et al.,* 2013) have shown how the combined effects of climate change and increased water extraction on water availability have been intensified by changes in land use linked to rural abandonment. In particular, the spontaneous expansion of forests over former pastures and agricultural land has been identified as a major cause of the reduction of up to 20-30% in run-off into rivers and groundwater infiltration (blue water) observed in Catalonia over the last 25 years.

Forest management, by temporarily reducing forest cover, can influence blue water supply by reducing interception and evapotranspiration and increasing infiltration and run-off (Yan *et al.,* 2012). The reviews available estimate an increase in blue water attributable to thinning of 100-200 mm/year in the year when action is taken and up to 400 mm/year in permanent conversions to pasture (García *et al.,* 2017). However, an excessive or drastic reduction of forest cover (e.g. due to fire) can lead to increased erosion and consequent loss of water quality; when the right balance is achieved, forest management is able to supply timber, increase water resources, preserve water quality and regulate the seasonal nature of water flows.

In Mediterranean forests, where water is key, forest management must take the trade-offs between water and vegetation into account; this is known as «ecohydrological forest management» (Del Campo *et al.,* 2019). It means, among other things, managing forest biomass to shape tree-soil-water relationships by modifying tree cover and species composition in line with the local balance between water availability and consumption. Strategies such as canopy opening through thinning of different intensities, conversion to pasture and tree species selection are considered to be effective both in combating water scarcity by increasing river and aquifer recharge, and in increasing resilience and adaptation to climate change by improving the vitality of the remaining trees and the efficiency of their use of water (Bart *et al.,* 2021). Beyond the more or less significant potential impact on the availability of water for human consumption, the effects of forest management on water at the local level may be important for the maintenance of aquatic habitats in headwater stretches, ponds and wetlands that support species of great environmental interest.

Optimal intensities and strategies for ecohydrological forest management vary according to the characteristics of the ecosystem and the dominant tree species, even within the same catchment area or region. The methodology presented here focuses on indirectly assessing the impact of different intensities of thinning on blue water supply under different bioclimatic conditions.

2.2 Water modelling

Exported water (or blue water) corresponds to the difference between the volume of water from precipitation and the volume that the vegetation evapotranspires or intercepts in its canopy. 4 major components of total precipitation can be distinguished:

1. The part intercepted by the canopy and trunks.
2. The part taken up and evapotranspired by vegetation.
3. Water flowing downhill in the form of direct run-off.
4. Sub-surface infiltration and groundwater.

The first two components are used by vegetacion (green water) and the second two are what we call blue water (*FOREStime*, Banqué *et al.,* 2020).

As part of the LIFE DEMORGEST project (2012-2017), the water balance was simulated for different forest management and non-management scenarios, using the GOTILWA+ simulator (CREAF, 2005), which simulates water flow in a forest stand over time under changing environmental conditions, such as those due to management or climate change. This model was applied to the 5 most common tree species found in Catalonia and the period analysed was 150 years. Overall the results indicated **an average increase in blue water of 14% in managed pine forests**, (figure 1).

According to this study, it could be observed that the type of management and the site quality (an indicator of the productive capacity of a site for a given forest species and product) affected this variable, as can be seen in the ORGEST models of black pine (figure 1), but the results are always favourable compared to unmanaged sites.

In 2020, within the framework of the LIFE CLIMARK project, and based on the **MEDFATE simulator** (De Cáceres *et al.,* 2015) the impact of management was calculated, applying different silcivultural regimes, defined in the SFM guidelines of Catalonia (ORGEST) to several forest sites, according to objective and site quality. Significant differences were observed before and after the intervention in terms of water resources released downstream (work carried out by De Cáceres in 2019, *Summary of simulations on CLIMARK forest plots*, as part of the LIFE CLIMARK project, unpublished). The MEDFATE model is a mechanistic stand-scale model that measures the water balance by simulating daily interception, infiltration, run-off, percolation, and evaporation from the soil, and transpiration via trees and shrubs. The simulations in these stands have been carried out with historical climate data from the period 1990-2018, applied to structural data for each stand, before and after intervention. For each year, a calculation was made of blue water (run-off and infiltration) and the number of days with water stress, when demand for water was greater than the quantity available (more than 50% per species). The application of different procedures increased the amount of blue water by between 26 and 367 mm/year (an average increase of 80% in the 12 project stands) due to a decrease in interception and more infiltration. The simulated

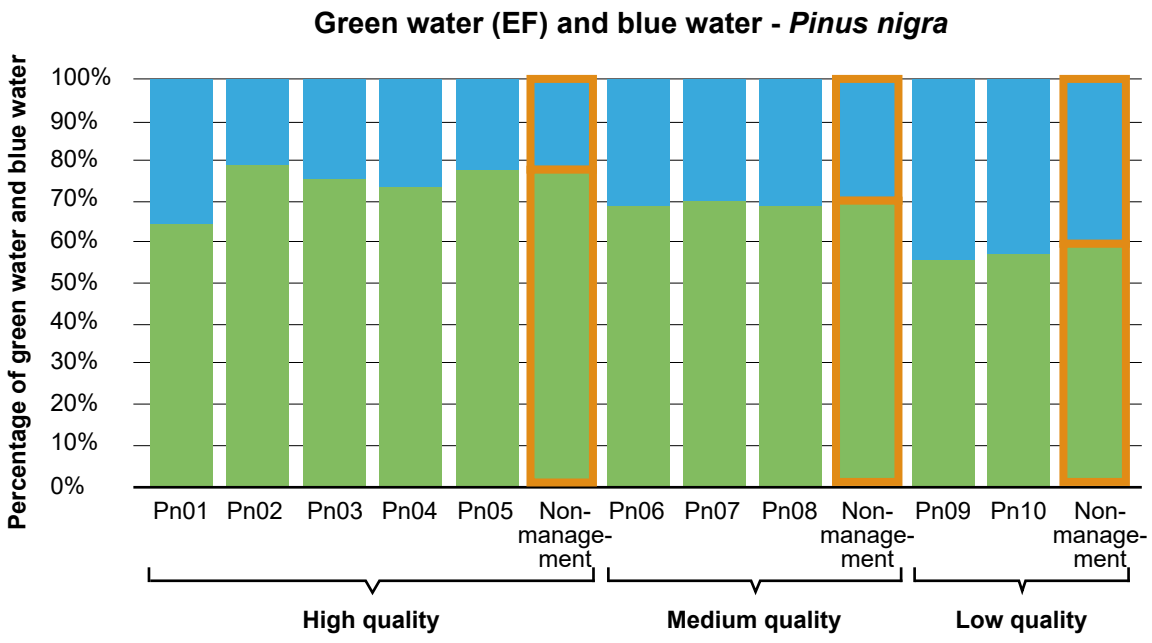
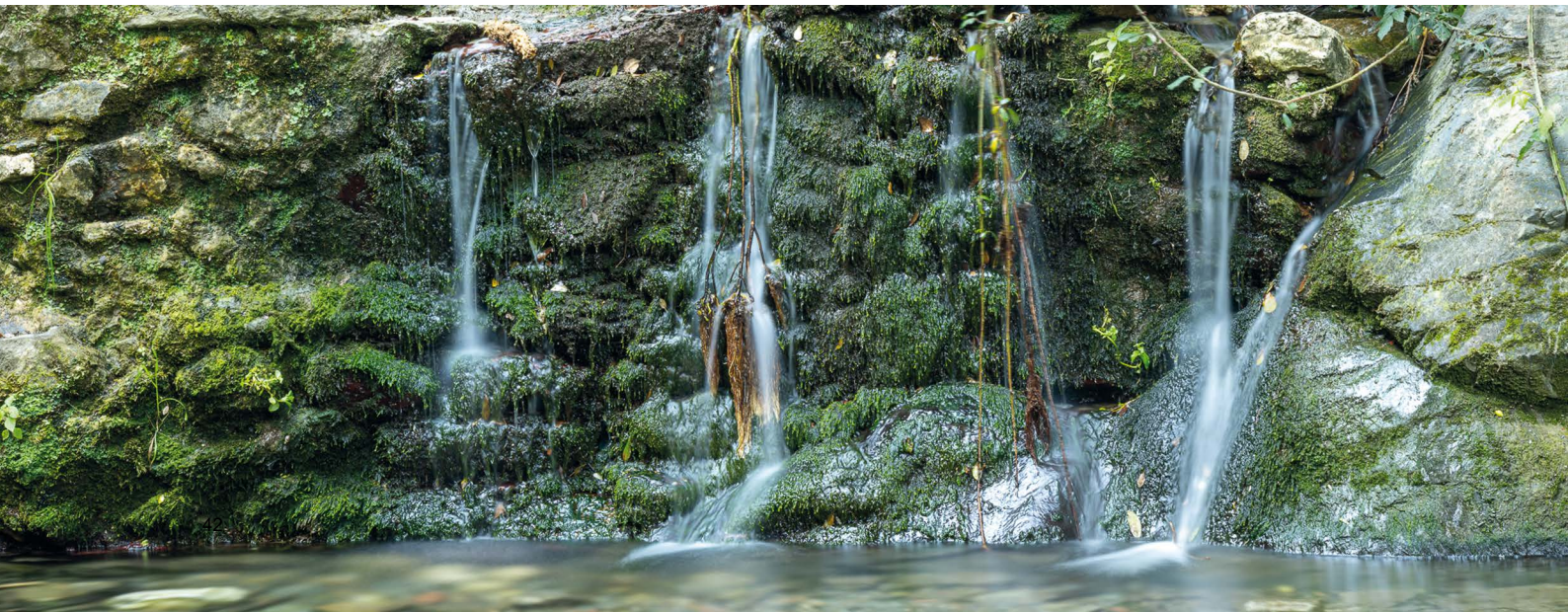


Figure 1. Example of the results obtained with the GOTILWA+ simulator in various management and non-management scenarios, applying the ORGEST reference models, in a stand ob black pine.

benefit was higher for more intensively thinned Scots pine and in areas with higher rainfall.

Through the use of these methods, it was observed that the ORGEST thinning regimes can have a positive impact in terms of increasing blue water, and work has been undertaken to find an easy, transparent methodology to calculate the benefits of this ecosystem service at any given time. It should be noted, however, that, while gains can be clearly seen in improved soil moisture, it is more difficult to detect them in circulating flows, in benefits for springs or in increased phreatic zones, especially when the impact of management is at the stand scale. Generally, when we talk about more blue water resources, we do not specify how much of it is circulating water.



2.3 Methodology to estimate the impact of management on water exported

The methodology developed here is based on the study described in the *FORESTime* report (Banqué *et al.,* 2020) on the evolution of ecosystem services in Catalonia over the last 25 years. In this report, the amount of blue water generated in each plot of the Catalan National Forest Inventory (NFI) is calculated by using the MEDFATE simulator, developed by De Cáceres *et al.* (2015), to compare changes in this variable between 2 periods separated by 25 years, 1988-1992 and 2021-2016.

Based on overall water export data (m^3/ha and year) compiled indistinctly for the 2 periods studied, a new methodology was developed, relating the water export values obtained in the NFI plots to environmental variables, such as precipitation (P) and potential evapotranspiration (PET), to determine variations by area, and relating them to variables involving structure of the stand, such as basal area (BA), which is indicative of forest management, and forest type, which reflects the different responses of species tree to water use.

Concerning environmental variables, the humidity index (HI) is used, based on the P/PET ratio, with values obtained from the Digital Climate Atlas of Catalonia (Meteorological Service of Catalonia, 2001). There are different methodologies for calculating PET. Here we have opted for that proposed by Hargreaves and Samani (1985), which is the one used in the *FORESTime*, report, in order to obtain comparable values. Both in this methodology and in the report, data for P and PET come from the same sources (from State Meteorological Agency (AEMET) stations). A spatial interpolation process is carried out on the data to give continuous results in the form of a map. However, they are obtained at different times and via different processes, which may lead to some differences in values, especially in areas with few stations.

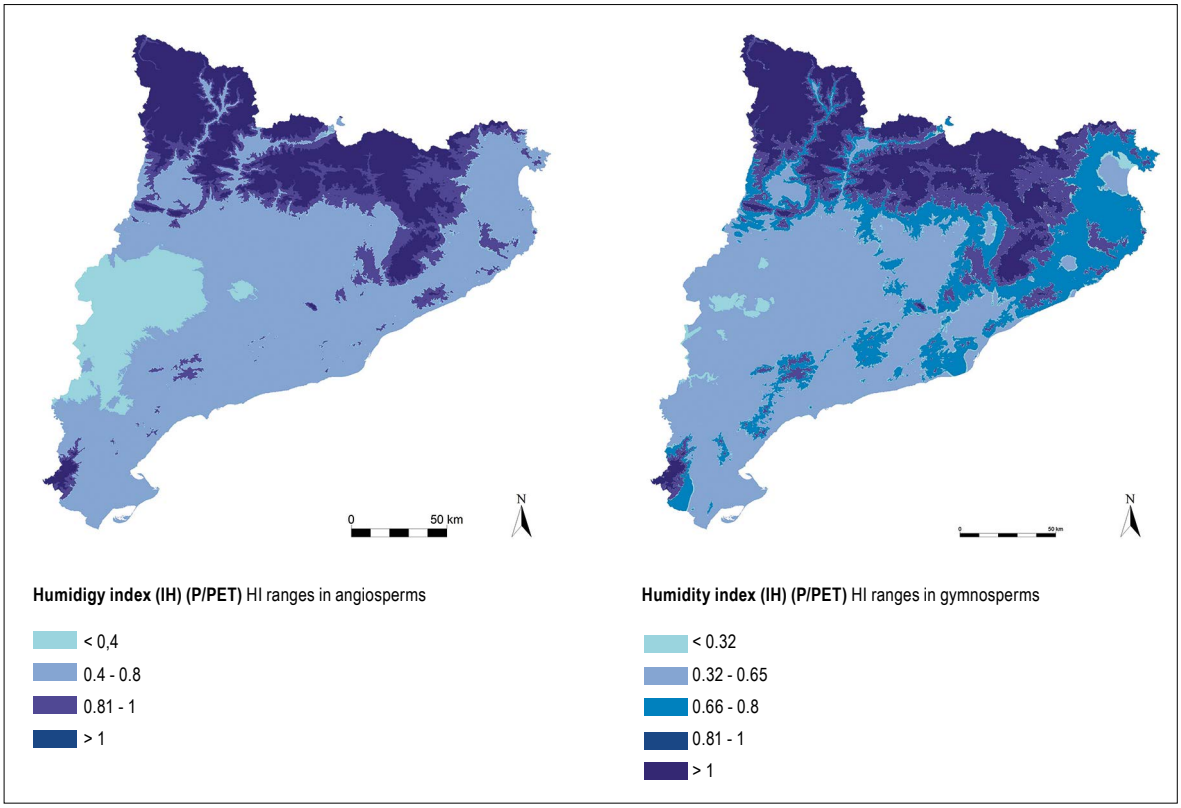


Figure 2. Maps of humidity index (IH) categories for Catalonia according to ranges of angiosperms and gymnosperms.
Source: prepared by the authors based on P and PET values given in the Meteorological Service of Catalonia's Digital Climate Atlas of Catalonia (2001).

Table 1. Logarithmic regressions to estimate water exported (Y) in m³/ha by value of HI and BA (m²/ha) in angiosperms

Group	Angiosperms		
Forest species	*Quercus ilex*, *Quercus suber*, *Quercus* sp, *Fraxinus* sp, *Fagus sylvatica*, *Castanea sativa*, *Corylus avellana*, *Acer* sp, *Alnus glutinosa*, *Betula* sp.		
IH 0.4 - 0.8	y = -559.3ln(x) + 2275.4	R^2= 0.9419	n= 525
IH 0.81 - 1	y = -812.8ln(x) + 3096.6	R^2= 0.9577	n= 452
IH 1.01 - 1.94	y= -1000ln(x) + 4456.1	R^2= 0.7375	n= 365

Table 2. Logarithmic regressions to estimate water exported (Y) in m³/ha by value of HI and BA (m²/ha) in gymnosperms

Group	Gymnosperms		
Forest species	*Pinus nigra, pinus sylvestris, Pinus halepensis, Pinus pinea, Pinus uncinata, Abies aba, Larix decidua, Pseudotsuga menziesii*		
IH 0.32 - 0.65	y = -546.5ln(x) + 1678.4	R^2= 0.9212	n= 621
IH 0.66 - 0.8	y = -781.3ln(x) + 2644.8	R^2= 0.9569	n= 525
IH 0.81 - 1	y= -682.6ln(x) + 3897.9	R^2= 0.8820	n= 521
IH 1.01 - 1.13	y= -920.2ln(x) + 6190.1	R^2= 0.8243	n= 352

HI values below 1 indicate that average evaporative demand exceeds average precipitation. According to the UNEP-FAO classification (Nicholas *et al.,* 1997), a value below 0.65 indicates a dry-sub-humid zone and a value above that indicates a humid zone. These thresholds have been used to define the HI categories considered, adjusting them in some cases according to the sample obtained (n) (values below 0.65, between 0.66 and 0.8, between 0.81 and 1, and above 1). As part of this methodology, a map has been drawn up with this classification (figure 2).

Regarding structural variables, although the variable most directly related to water use by vegetation is the leaf area index (LAI), given the difficulty of measuring it and its limited availability, in some practical applications it has been replaced by the basal area (BA). This structural variable, which is easier to measure and is commonly used in forestry, also correlates with LAI. For example, Winckler *et al.* (2010), in a compilation of hydrological studies carried out in Canada, describe the relationship found between BA and water exported, where the water yield of a catchment increased by 3 mm for each percentage point of BA removed in selective cutting. For the analysis, the NFI BA data have been grouped into categories at 5 m^2/ha intervals (< 10, 10-15, 15-20, 20-25, 25-30, 30-35, 35-40 and > 40) and separated into two groups, angiosperms and gymnosperms.

The results have shown different logarithmic regressions from these two groups, according to HI and BA (Tables 1 and 2 and Annexes). In angiosperms, HI values below 0.65 account for 18% of the total first sample (95 plots between 0.4 and 0.8). In gymnosperms, they form a distinct group, a sample of 621 values. In terms of HI values above 1, in angiosperms the mean value is 1.17, with one value reaching 1.94, while in gymnosperms the maximum value does not exceed 1.13.

2.4 Calculation of the water provision indicator

The impact of **forest management** on water supply is obtained by calculating the difference in water exported in an unmanaged scenario compared with a managed scenario. Thus, the water exported in a forest with a given BA (initial BA = unmanaged value) is compared with the water exported in the same forest after thinning (final BA = managed forest value). The values (m^3/ha and year) are obtained by applying the corresponding regression equation, according to tree species and HI, to each BA.

$$\textbf{Impact of management on } H_2O \textbf{ exported} = (m^3\ H_2O_{\text{post-thinnig}} - m^3\ H_2O_{\text{initial}}) \times n \text{ years}/2$$

However, the new forest structure created is not fixed, but evolves over time as the trees grow, reaching a point at which the initial BA values are restored. Here, we consider that this evolution is linear and that the end point coincides with the time when it is advisable to intervene again in the forest according to the ORGEST silvicultural regimes.

Therefore, to obtain the volume of additional water generated (m^3/ha) as a result of management over the whole period, the difference between the amount of water exported after intervention and the amount exported previously is calculated, and multiplied by half the number in years of the period considered (n years/2), since it is considered that in the year of intervention (year 0) the increase will be maximum, while in year n (the end of the period) the increase in blue water with respect to the initial situations is zero.

As an example, applying the regressions to a pure Aleppo pine forest with a HI of o.66, an initial basal area of 24 m^2/ha and a final post-treatment basal area of 14.4 m^2/ha would result in a gain of 399 m^3/ha per year (=39.9 mm in the first year). If the period assessed is 12 years, then the overall impact of management would be 399 m^3/ha*12/2 = 2394 m^3/ha total (=239.9 l/m^2=239.4 mm), representing an average of 20 mm/year for the 12 years assessed (39.9 mm in the first year and 0 mm in the last).

In the case of **mixed stands**, the regression is applied for each species with the data of the whole stand, weighted for the % of basal area of each species, to obtain the final sum. When the initial blue water values are negative (a situation of water stress), only positive volumes (above zero) are considered as blue water generated, even if the intervention compensates for pre-existing water stress.

In the case of plantations, most sources consulted, mainly from Chile and Australia, indicate that in the first years of planting there is no effect on flow rates (e.g. Prado, 2015), even in fast-growing species. It is from the age of 5-10 years and up to 30-35 years that we find maximum growth and maximum impact on blue water. A review of China and the USA (compiled in Yan, 2021) suggests average blue water reduction values of 10-50 mm/year in semi-arid areas (50%) and up to 300 mm/year in humid areas (30%), taking into account that they are more significant in conifers than in broadleaved trees.

In the absence of references for slow-growing species in Europe, the conservative option below is proposed. It assumes a standard blue water reduction at year n (= just before thinning, estimated at 15-20 years) as follows:

- **For enrichment plantings in already forested areas** (approx. 100 trees/ha in areas already wooded) assuming a 5% increase in BA (1 BA point on an initial BA of 20) = **100 m^3/ha** (=10 mm in years 15-20).
- **For agroforestry plantations** (100 trees/ha in crop fields) = **250 m^3/ha** (= 25 mm in years 15-20).
- **For forest plantations** (400-900 trees/ha on non-forested land) = **500 m^3/ha** (= 50 mm in years 15-20).

The values are approximate and are based on the analysis of *FOREStime* data (from which it is estimated that for every 1% increase in BA, water consumption increases by 1 to 2 mm [10-20 m^3/ha]) and on the literature available.

The same logic would be applied to these values as in the case of thinning, but assuming that until year 10 there is no effect on blue water (according to the literature), i.e. t_{10} = zero reduction and t_n = maximum reduction. To obtain the overall value of blue water reduction due to planting over the whole period considered, the annual figure would thus be multiplied by (n-10)/2.

ANNEXES

Logarithmic regressions for water exported, by BA variations and Humidity Index, in angiosperms and gimnosperms

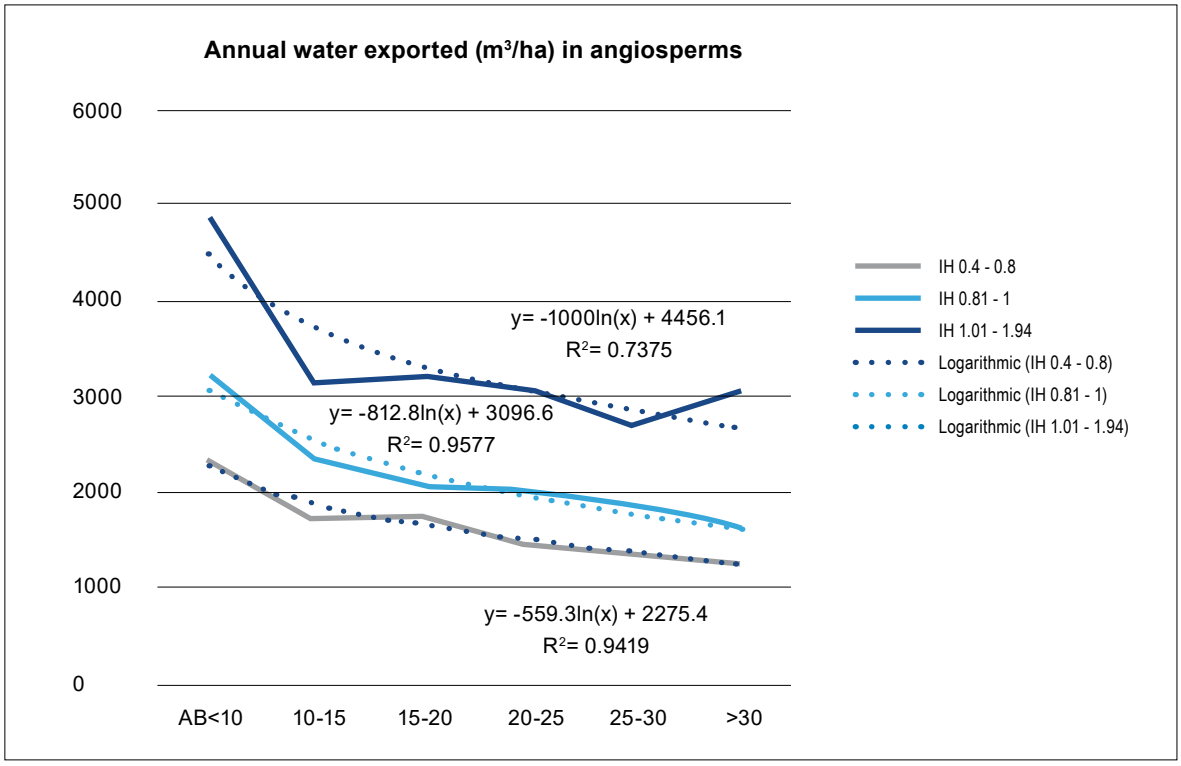


Figure 1. Logarithmic regressions for HI and BA in angiosperms.
Source: own figures based on the *FORESTime* studi (Banqué *et al.,* 2020; De Cáceres, 2015)

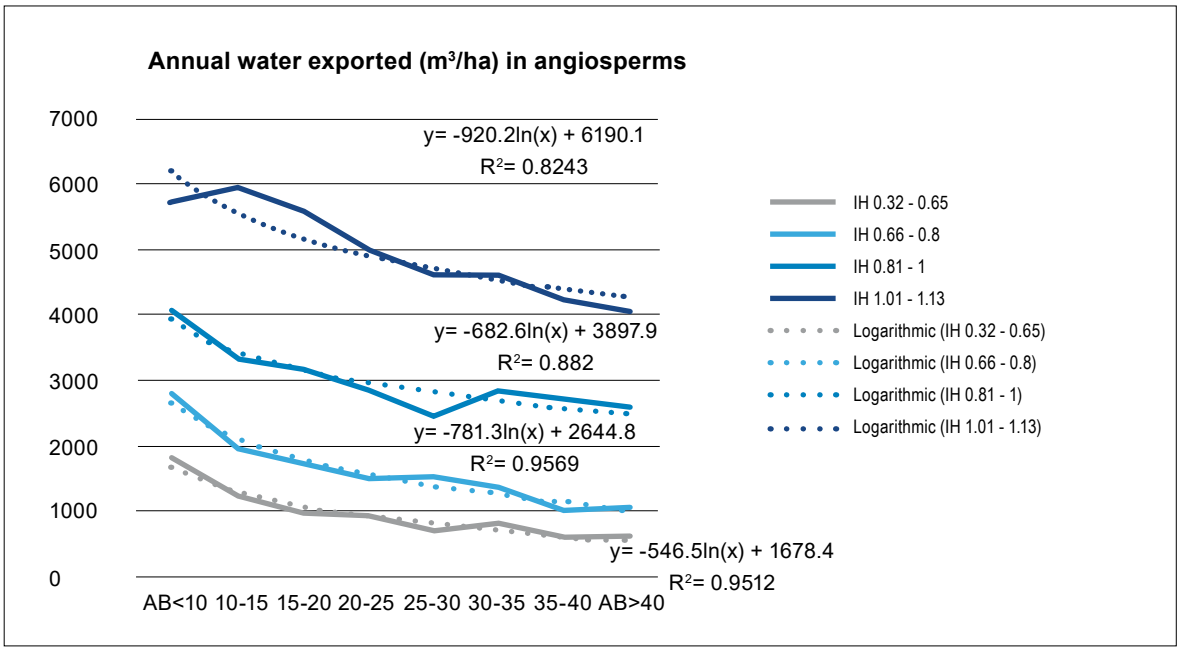


Figure 2. Logarithmic regressions for HI and BA in gymnosperms.
Source: own figures based on the *FORESTime* studi (Banqué *et al.,* 2020; De Cáceres, 2015).

References

Banqué, M.; De Cáceres, M.; Ainsa, García-Valdés, R.; Martínez-Vilalta, J.; Valentín, J.; Vayreda, J. 2020. *FOREStime*. Canvi dels serveis ecosistèmics dels boscos de Catalunya al llarg dels darrers 25 anys (període 1990-2014). CREAF-OCCC-CTFC.

Bart, R.; Ray, R.; Conklin, M.; Safeeq, M.; Saksa, Ph.; Tague, C.; Bales, R. 2021. Assessing the effects of forest biomass reductions on forest health and streamflow. https://doi.org/10.1002/hyp.14114

De Cáceres, M.; Martínez-Vilalta, J.; Coll, L.; Llorens, P.; Casals, P.; Poyatos, R.; Pausas, J. G.; Brotons, L. 2015. Coupling a water balance model with forest inventory data to predict drought stress: The role of forest structural changes vs. climate changes. *Agric. For. Meteorol*, 213: 77-90, DOI:10.1016/j.agrformet.2015.06.012.

Del Campo, A. D.; González-Sanchis, M.; Ilstedt, U.; Bargués-Tobella, A.; Ferraz, S. 2019. Dryland forests and agrosilvopastoral systems: water at the core. *Unasylva*, 251: 27-35.

Gallart, F.; Llorens, P. 2004. Observations on land cover changes and water resources in the headwaters of the Ebro catchment, Iberian Peninsula. *Physics and Chemistry of the Earth*, 29: 769-773.

Gallart, F.; Delgado, J.; Beatson, S. J. V.; Posner, H.; Llorens, P.; Marcé, R. 2011. Analysing the effect of global change on the historical trends of water resources in the headwaters of the Llobregat and Ter river basins (Catalonia, Spain). *Physics and Chemistry of the Earth*, 36: 655-661.

Gallart, F.; Llorens, P.; Manzano, A. 2013. Gestión de la cuenca del Cardener para garantizar los recursos de agua. Conservar aprovechando. Cómo integrar el cambio global en la gestión de los montes españoles. CREAF: 93-97.

Garcia-Chevesich, P. A.; Neary, D. G.; Scott, D. F.; Benyon, T. R. (eds). 2017. Forest management and the impact on water resources: A review of 13 countries. IHP - VIII/Technical Document No. 37. Paris, France: United Nations Educational, Scientific, and Cultural Organization (UNESCO), International Hydrological Program. 203 pp.

Nadal-Sala, D.; Sabaré, S.; Garcia, C. 2014. GOTILWA+: un modelo de procesos que evalúa efectos del cambio climático en los bosques y explora alternativas de gestión para su mitigación. *Revista Ecosistemas*, 22: 29-36.

Nicholas, J.; Middelton, N. 1997. Atlas Mundial de desertificación. Programa de las Naciones Unidas para el Medio Ambiente.

Prado, J. A. 2015. Plantaciones forestales, más allá de los árboles. Colegio de Ingenieros Forestales de Chile (CORMA). 167 pp.

Servei Meteorològic de Catalunya. 2001. Atles Climàtic Digital de Catalunya (www.meteo.cat/climatologia/atles_climatic).

UNECE-FAO. 2018. Forests and water. Valuation and payments for forest ecosystem services.

Winkler, R. D.; Moore, T. E.; Redding, D. L.; Spittlehouse, B. D.; Smerdon, B.; Carlyle-Moses, D. E. 2010. In: Pike, R. G.; Redding, T. E.; Moore, R. D.; Winkerand, R. D.; Bladon, K. D. (eds). 2010. Compendium of forest hydrology and geomorphology in British Columbia. B. C. Min. For. Range, For. Sci. Prog., Victoria, B. C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B. C. Land Manag. Handb., 66. www.for.gov.bc.ca/hfd/pubs/Docs/Lmh/Lmh66.htm

Yan, H.; Wang, S. Q.; Billesbach, D. P.; Oechel, W.; Zhang, J. H.; Meyers, T.; Martin, T. A.; Matamala, R.; Baldocchi, D. D.; Bohrer, G.; Dragoni, D.; Scott, R. 2012. Global estimation of evapotranspiration using a leaf area index-based surface energy and water balance model. Remote Sensing of Environment, 124, 581-595, DOI: 10.1016/j.rse.2012.06.004.

Yang, Z.; Hou, F.; Cheng, J.; Zhang, Y. 2021. Modeling the Effect of Different Forest Types on Water Balance in the Three Gorges Reservoir Area in China, with CoupModel. *Water*, 13: 654. https://doi.org/10.3390/w13050654

3

Impact of forest management on biodiversity

Chapter 3

Impact of forest management on biodiversity

3.1 Introduction

Forests are the most biodiverse ecosystems on the planet, comprising thousands of species from three major groups: animals, plants and fungi. Beyond its indisputable intrinsic value, forest biodiversity is the basic support for the rest of the ecosystem services provided by the forest. In relation to climate change mitigation, the role of certain species is key to ensuring continued CO_2, sequestration, through natural regeneration (in the case of pollinating insects), maintaining soil fertility and tree growth (in the case of fungi) or pest control. A biodiverse forest system is more resilient and resistant to the effects of climate change, as it promotes different adaptive responses to cope with the uncertainty associated with this change (Messier *et al.,* 2019).

According to recent reports, overall forest biodiversity shows a certain stability both at European level (Pötzelsberger, 2021, based on EEA 2020, Forest Europe 2020, Maes *et al.,* 2020 and IPBES 2018 reports) and in Catalonia (*State of Nature in Catalonia,* 2020, IENC). For example, there is a stable or increasing trend for most forest birds and there are increases in the amount of dead wood and the diversity of tree species However, rare or endemic species do show a more critical or declining state, and, in the case of Catalonia, so do species linked to open spaces such as scrubland and pastureland. Another question is when the condition of forests is analysed using maturity criteria, in which case 62% of Catalan forests are in an unfavourable position (IENC, 2020), given that they are mainly young forests. Within European protected areas, forests are considered the habitat most in need of improvement, and forestry activities are still seen as a threat, especially in terms of the removal of dead wood and old trees or clear-cuttings (EEA, 2020).

The conservation of forest biodiversity can be approached on the basis of two strategies: the *segregation* strategy, which distinguishes between forests for conservation (or «natural evolution») and forests for production (or «managed» forests), and the *integration* strategy, which provides for the sustainable production of multiple goods and services from the same forest (Kraus and Krumm, 2013). It is generally accepted that an appropriate combination of the two strategies provides the best habitat quality (Krumm *et al.,* 2020). In any case, managed forests, because of their role as biological corridors and because of the area they occupy (95% of European forests are considered semi-natural, with some degree of human intervention), are considered to play a key role in the conservation of global biodiversity, which must continue to be guaranteed and enhanced in the context of climate change (Mauser, 2020).

Forestry and biodiversity conservation

The biodiversity found in a forest is the result of a combination of complex dynamics in which disturbances play a fundamental role. Natural disturbances (storms, fires, pests, etc.) affect the forest structure and generate a landscape-scale mosaic of habitats that largely determine the plant and animal communities present.

Forestry is an anthropogenic disturbance that seeks to accelerate or control processes that would naturally take many more years to occur (e.g. regulation of competition or regeneration) in order to achieve a certain objective such as timber production. Through the process of selecting the species and individuals to be felled, planted or favoured, forestry not only has an impact on that species of tree but also alters the functioning of the forest and the biodiversity it supports, given the numerous host-tree associations that exist and the different environmental conditions generated.

Any forestry intervention thus has effects on biodiversity, but the degree and direction of the impact may vary depending on the intensity and type of management (Chaudhary *et al.,* 2016). Generally, even-aged, monospecific forests, with trees of the same age range, managed by clear-felling (which could be associated with large-scale disturbances), are generally considered less biodiverse than those that maintain greater heterogeneity and complexity in the stand, with trees of different ages and managed by selective cutting, where only a few individuals are removed at a time and the presence of tree cover and regeneration is permanent. However, a recent review by Nolet *et al.,* (2018) has concluded the opposite, when the analysis takes into account a scale larger than that of the stand where intervention takes place.

The application of complementary techniques grouped under the name of «retention» or «integrative» forestry has been shown to contribute to increasing overall diversity, especially that of rarer species that require more time to colonise a habitat (Kraus and Krumm, 2013; Aggestam *et al.,* 2020). These techniques, which can be applied in any approach to forestry (Gustafsson *et al.,* 2010; 2020), consist of actively maintaining or restoring legacies and other attributes of mature forests, such as large old trees, trees

with unique features, large dead wood or open areas, with the aim of providing continuity in forest structure, function and composition over generations. Their adaptation to Mediterranean sites is currently being tested in the framework of the lifebiorgest.eu project.

The objective of this methodology is to quantify *ex ante* the impact that integrative management has on forest's capacity to support biodiversity in the short term.

3.2 The potential biodiversity index: a quick reliable measurement of a stand's capacity to host biodiversity

In order to assess the effect of management on total biodiversity in terms of cost-efficiency, the potential biodiversity index (IBP) methodology was chosen. This is a proxy indicator of the capacity of a forest to host biodiversity, based on the field assessment of 10 key factors.

The index came originally from France, where, since 2008, the CNPF and INRAE have been working with it as part of the implementation of the French national biodiversity strategy (Larrieu and Gonin, 2008; *Larrieu et al.,* 2012). Since 2011, they have had a version adapted to Mediterranean conditions, which has already been used in different regions (Gonin, Larrieu and Deconchat, 2017). In parallel, in Catalonia, the Forest Ownership Centre (CPF) began work in 2010 on a biodiversity index (Fernández *et al.,* 2013), to bring to a close a decade of study into forest management and biodiversity in the framework of the Biodiversity Monitoring Programme for the Forests of Catalonia – BIBOCAT. After a period of testing and improvement, the first version of the IBP for Catalonia, compatible with international Mediterranean IBP standards, was finalised in 2019: the IBP_Cat (Baiges *et al.,* 2019), coordinated by the CPF and the CNPF, and agreed with the different agents involved in the management and conservation of forests in Catalonia.

The IBP involves observation of 10 key factors (Figure 1): the first seven factors, which we can call stand factors (stand IBP), are those whe-

re management has most impact. They refer to key elements, present in mature forests, which have been found to be the attributes that contribute most to greater diversity: large trees, dead wood, forest microhabitats, clearings... If these elements can be maintained in managed forests, their ability to host biodiversity is greatly increased, especially for those species that depend on them at some point in their life cycle. The last three are the context factors (context IBP): the continued existence of the forest, i.e. the age of the site as undisturbed forest land and the presence of aquatic and rocky environments in the stand, which have their own specific diversity.

For each of these 10 factors are defined different states (between little and very favorable to host biodiversity) and is given to each one numerical value (fron 0 to 5). The application of the IBP is to evaluate in the field in which of these states falls the studied stand, for each factor, until you fet a final overall score. The results are given in the form of absolute value of IBP, disaggregated into IBP_rodal (first 7 factors) and IBP_context (last 3 factors) and in percentage.

The IBP was originally intended for use by non-experts to facilitate the integration of biodiversity conservation criteria in forest management, but it also has important pedagogical and monitoring aspects. Recently, along with payment schemes for ecosystem services or environmental impact assessments, it has also been increasingly used as a tool to assess the impact of forest management on biodiversity. Thus, for

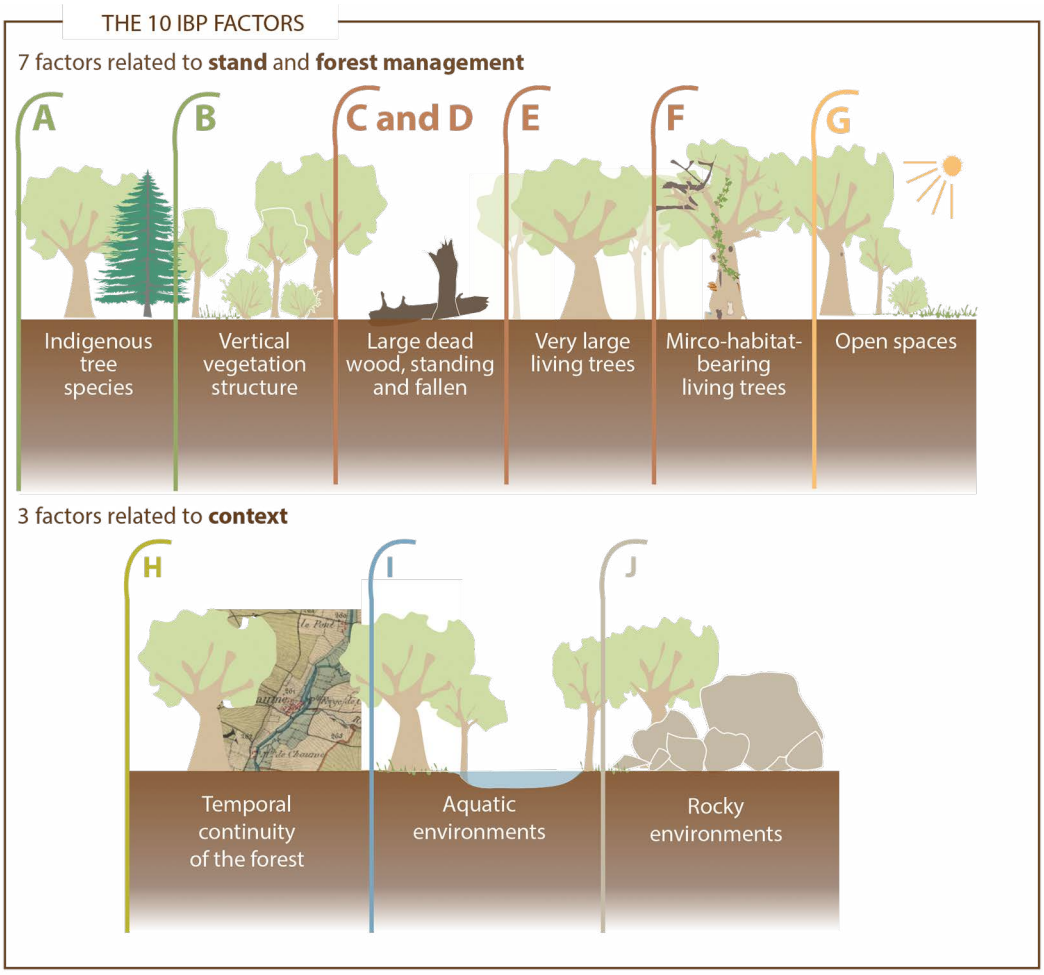


Figure 1. Factors considered in the IBP methodology. Source: Baiges *et al.* (2021), adapted from Emberger *et al.* (2013).

example, in the voluntary carbon credit market developed in France under the «Bas Carbone» label (Decree no. 2018-1043, of 28 November 2018), the IBP is used as an indicator in the *ex post* measurement of forest management intervention previously assessed to define the measures to be carried out.

The IBP methodology includes a proposal for its ex post application to assess the impact of a specific measure, based on the combined use of the IBP result and the raw data collected (Baiges *et al.,* 2021).

What has not been documented is its use in *ex ante* assessments, i.e. in estimating the potential impact of a given forestry intervention in the short and medium term. This is what is proposed in the methodology presented here.

3.3 Estimation of the impact of forestry treatments on biodiversity

The methodology proposed is applicable at the stand scale (landscape scale is not considered) and focuses on the impact of management on the total diversity of species in the stand (not rare or threatened species), i.e. ordinary taxonomic biodiversity.

For the ex ante assessment of the impact of intervention on biodiversity, the following two scenarios have been considered:

- **Integrative management scenario:** this considers management methods that take into account criteria for conserving and enhancing biodiversity in the definition of management itineraries. **Conservation measures** are a priority and include the identification, marking and retention of elements that are key to the presence of a multitude of species and particularly of those that are not common in managed forests, the loss of which is very difficult to reverse in the short to medium term: large trees, large standing or fallen dead trees, or trees with significant micro-habitats. Dependint on the stand characteristics, additional **improvement measures** can also be carried out to increase the carrying capacity of the stand in the short term: the generation of large dead wood, standing (maintenance of tall stumps or banding) and fallen, enrichment tree planting and the creation of small openings to favour tree regeneration, flowering species or a new stratum. It is assumed that if integrative forest management is applied, the biodiversity carrying capacity of the stand should be at least equal to or greater than before the intervention.

- **Non-management scenario:** the biodiversity carrying capacity of an unmanaged stand is assumed to neither increase or decrease in the short term, but to remain the same.

Impact on biodiversity=
=Biodiversity hosting capacity$_{\text{managed stand}}$ -
Biodiversity hosting capacity$_{\text{unmanaged stand}}$

The short-term impact of management on biodiversity is estimated as the difference between the biodiversity carrying capacity of a stand where integrative management has been applied and that of a stand where no management has been applied. This difference will be due only to improvement measures, as it is assumed that conservation, while an essential requirement of integrative management, does not imply any difference with respect to a non-

management scenario, where the existing key elements would also be conserved[1].

The impact expressed as the **percentage improvement in biodiversity carrying capacity due to management**, based on the IBP, and is calculated as follows:

Biodiversity impact

(% improvement in PBI due to management)=

$$=(PBI\_stand_{managed} - IBP\_stand_{unmanaged})/PBI\_stand_{unmanaged}$$

Where:

$IBP\_stand_{unmanaged}$, is the initial IBP_stand diagnosis, carried out in the field prior to intervention.

$IBP\_stand_{managed}$, is the initial IBP_stand value (assumed to be maintained because elements of value have been retained) to which is added the sum of the gains associated with each of the imprvement measures proposed by the manager (gain in IBP points, according to Table 1).

The IBP factors likely to be affected by management are factors A, B, C, D and G. Although the retention of trees with diameters below but close to the thresholds considered in the IBP may increase the value of the E-factor (large trees) in the medium term, it is not taken into account in the methodology proposed, as it focuses on active measures that are easy to quantify.

Likewise, only measures that generate a *significant* improvement in the biodiversity carrying capacity of a stand (= change in IBP valuation) are considered. In other words, it would exclude measures that, although they may improve biodiversity quantitatively, do not do so qualitatively (for example, if small dead wood is generated but there was already sufficient dead wood in the stand to guarantee the maximum possible diversity of species linked to it, based on the thresholds established in the IBP). These measures could be integrated using the raw data as a complement to the IBP assessment (e.g. the total number of trees banded), similar to that proposed in the IBP methodology (Baiges *et al.,* 2021), but this has been discarded here to simplify calculation and to give more weight to the impact on species diversity and less to the number of individuals of the same group.

There are two important considerations to bear in mind:

- A 0% improvement score does not mean that no integrative management has been carried out. It indicates that all existing key biodiversity elements in the stand have been conserved, even if no new ones have been generated. The 0% result shows that there is no significant difference between the biodiversity carrying capacity of a forest that is unmanaged and one where integrative management has been applied.

- Estimating the IBP involves an imprecision of measurement that has not been taken into account in this methodology. It is estimated that the observer effect can vary the IBP score obtained in the field by up to 10% (Gosselin and Larrieu, 2020).

The maximum improvement in IBP scores that could be achieved by management in the short term would be 14 points (Table 2) and would correspond to a monospecific, closed forest with large trees and no dead wood, where, in addition to conserving all valuable elements, mixed enrichment planting is carried out, a lot of large dead wood is generated, standing and fallen, and small openings are created for flowering plants and new species to colonise. The minimum room for improvement is generally at the extremes, either because the score is already very high and does not allow for much more qualitative improvement or because, when it

[1] *Conservation measures should be considered when comparing* **the «integrative» management scenario** *with a* **«normal» management scenario**. *In this case, we suggest giving more weight to forests where the effort to conserve key elements is greater, by applying the ratio of the current IBP_stand value and the maximum possible IBP_stand value (35)..*

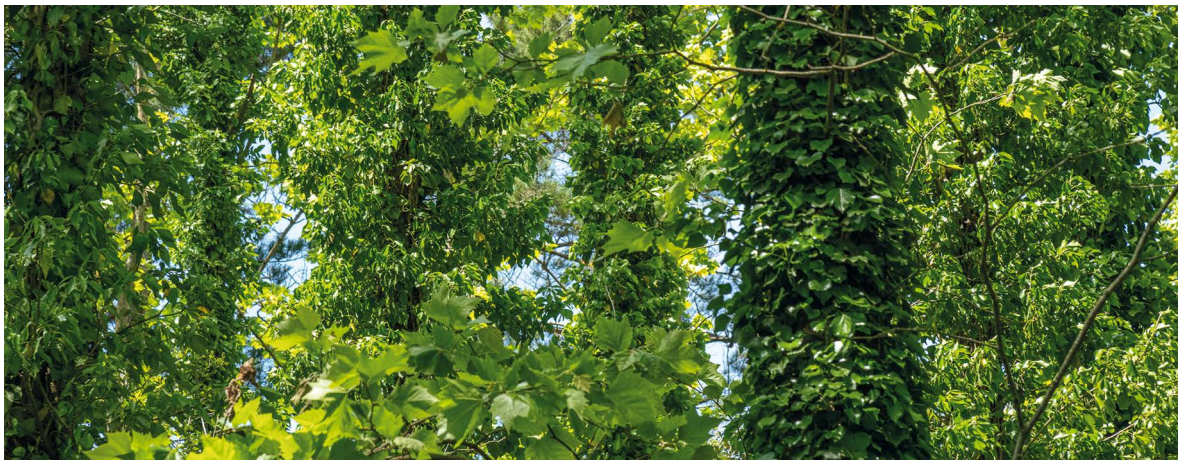


Table 1. Estimated gain in IBP factors (Mediterranean region) that are affected by the improvement measures implemented

PBI Factors	Initial IBP score	Measure proposed	Gain in IBP points
FACTOR A Indigenous tree species	0	Planting (1)-2 new genera	1
		Planting 3-4 new genera	2
		Planting 5 or more new genera	5
	1	Planting 1-2 new genera	1
		Planting 3 or more new genera	4
	2	Planting (1)-2 or more new genera	3
FACTOR B Diversity of strata	0, 1	New stratum obtained (e.g. by opening up space for scrubland)	1
FACTOR C Large standing dead wood	0	Generation ≥ 1 tree/ha standing dead wood, 17.5-27.5 cm	1
		Generation ≥ 1-2 tree/ha standing dead wood, ≥ 27.5 cm	2
		Generation ≥ 3 tree/ha standing dead wood, ≥ 27.5 cm	5
	1	Generation 1-2 tree/ha standing dead wood, ≥ 27.5 cm	1
		Generation 3 tree/ha standing dead wood,. ≥ 27.5 cm	4
	2	Generation 3 tree/ha standing dead wood, ≥ 27.5 cm	3
FACTOR D Large dead wood, fallen	0	Generation ≥ 1 tree/ha dead wood, fallen ≥ 17.5-27.5 cm	1
		Generation 1-2 tree/ha dead wood, fallen ≥ 27.5 cm	2
		Generation ≥ 3 tree/ha dead wood, fallen ≥ 27.5 cm	5
	1	Generation 1-2 tree/ha dead wood, fallen ≥ 27.5 cm	1
		Generation ≥ 3 tree/ha dead wood, fallen ≥ 27.5 cm	4
	2	Generation ≥ 3 tree/ha dead wood, fallen ≥ 27.5 cm	3
FACTOR G Open spaces	0	Opening/clearing spaces for flowering vegetation. < 5% area	5
	2 (when < 1%)	Opening/clearing spaces for flowering vegetation < 5% area	3

* **Factors E and F** (large living trees and micro-habitat-bearing trees) are not affected by the active management applied.

Table 2. Maximum achievable gain through the implementation of active biodiversity enhancement measures, according to the initial carrying capacity of the stand (in IBP points)

Value IBP_stand initial (IBP points)	Maximum achievable score IBP_stand (IBP points)	Maximum gain (IBP points)
0 (forestation)	5	**5**
0	12	**12**
1	12	**11**
2	12	**10**
3	12	**9**
4	12	**8**
5	12	**7**
6	12	**6**
7	12	**5**
8	20	**12**
9	20	**11**
10	20	**10**
11	25	**14**
12	25	**13**
13	26	**13**
14	26	**12**
15	26	**11**
16	30	**14**
17	30	**13**
18	30	**12**
19	30	**11**
20	30	**10**
21	35	**14**
22	35	**13**
23	35	**12**
24	35	**11**
25	35	**10**
26	35	**9**
27	35	**8**
28	35	**7**
29	35	**6**
30	35	**5**
31	35	**4**
32	35	**3**
33	35	**2**
34	35	**1**
35	35	**0**

is very low, there are often fewer elements on which to base improvement.

For plantations in non-forested areas, the maximum gain is 5, if 5 different genera of species are planted, and the minimum gain is 0, if only 1 species is planted, which would not happen in integrated management (mixed plantations are always recommended).

Generally, the percentage improvement due to management will not exceed 100% except in very low initial IBP scores (< 13 points), where the margin for improvement is high.

References

Aggestam, F.; Konczal, A.; Sotirov, M., et al. 2020. Can nature conservation and wood production be reconciled in managed forests? A review of driving factors for integrated forest management in Europe. Journal of Environmental Management, 268, 110670. https://doi.org/10.1016/j.jenvman.2020.110670

Baiges, T.; Cervera, T.; Palero, N.; Gonin, P.; Larrieu, L. 2019. Posada a punt de l'índex de biodiversitat potencial: un termòmetre per a mesurar la capacitat d'allotjar biodiversitat dels boscos (gestionats) de Catalunya. *Silvicultura*, 80: 26-36.

Baiges, T.; Cervera, T.; Palero, N.; Gonin, P.; Larrieu, L. 2021. Mètodes de camp i aplicacions de l'IBP. In publication.

Brotons, L.; Pou, N.; Herrando, S.; Bota, G.; Villero, D.; Garrabou, J.; Ordóñez, J. L.; Anton, M.; Gual, G.; Recoder, L.; Alcaraz, J.; Pla, M.; Sainz de la Maza, P.; Pont, S. i Pino, J. 2020. Estat de la Natura a Catalunya 2020. Departament de Territori i Sostenibilitat. Generalitat de Catalunya. Barcelona

Chaudhary, A.; Burivalova, Z.; Koh, L., *et al.* 2016. Impact of Forest Management on Species Richness: Global Meta-Analysis and Economic Trade-Offs. *Sci Rep* 6, 23954. https://doi.org/10.1038/srep23954

EEA. 2020. State of nature in the EU. Results from reporting under the nature directives 2013-2018 EEA Report n. 10. https://www.eea.europa.eu/publications/state-of-nature-in-the-eu-2020

Emberger, C.; Larrieu, L; Gonin, P. 2013. Dix facteurs clés pour la biodiversité des espèces en forêt: Comprendre l'indice de biodiversité potentielle (IBP). Document technique. Centre national de la propriété forestière.

Fernández, M.; Campeny, R.; Baiges, T.; Cervera, T.; Abián, J. L. 2013. Índice de evaluación del potencial de los bosques para la biodiversidad. Actas del VI CFE, 10-14, June, Vitoria-Gasteiz.

Gonin, P.; Larrieu, L.; Deconchat, M. 2017. Index of Biodiversity Potential (IBP): How to extend it to Mediterranean forests? 5th Mediterranean Forest Week – Agadir; Forêt méditerranéenne t. XXXVIII, 3: 343-350.

Gosselin, F.; Larrieu, L. 2020. Developing and using statistical tools to estimate observer effect for ordered class data: The case of the IBP (Index of Biodiversity Potential). Ecological Indicators 110 (2020), 105884.

Gustafsson, L.; Kouki, J.; Sverdrup-Thygeson, A. 2010. Tree retention as a conservation measure in clear-cut forests of northern Europe: a review of ecological consequences, Scandinavian Journal of Forest Research, 25: 4.

Gustafsson, L.; Bauhus, J.; Asbeck, T., et al. 2020. Retention as an integrated biodiversity conservation approach for continuous-cover forestry in Europe. Ambio, 49: 85-97. https://doi.org/10.1007/s13280-019-01190-1

Kraus, D.; Krumm, F. (eds). 2013. Integrative approaches as an opportunity for the conservation on forest biodiversity. European Forest Institute. 284 pp.

Krumm, F.; Schuck, A.; Rigling, A. (eds). 2020. How to balance forestry and biodiversity conservation – A view across Europe. European Forest Institute (EFI); Swiss Federal Institute for Forest, Snow and Landscape Research (WSL), Birmensdorf. 640 pp.

Larrieu, L.; Gonin, P. 2008. L'indice de Biodiversité Potentielle (IBP): une méthode simple et rapide pour évaluer la biodiversité potentielle des peuplements forestiers; Rev. For. Fr. 06-: 727-748.

Larrieu, L.; Gonin, P.; Deconchat, M. 2012. Le domaine d'application de l'Indice de biodiversité potentielle (IBP). Rev. For. Fr 5-, 64(5): 701-710.

Mauser, H. (ed). 2021. Key questions on forests in the EU. Knowledge to Action 4, European Forest Institute. https://doi.org/10.36333/k2a04

Messier, C.; Bauhus, J.; Doyon, F.; Maure, F.; Sousa-Silva, R.; Nolet, P.; Mina, M.; Aquilué, N.; Fortin, M. J.; Puettmann, K. 2019. The functional complex network approach to foster forest resilience to global changes. *Forest Ecosystems*, 6: 21.

Nolet, P.; Kneeshaw, D.; Messier, C.; Beland, M. 2018. Comparing the effects of even and uneven-aged silviculture on ecological diversity and processes: a review. *Ecol Evol.,* 8(2): 1217-26.

Pötzelsberger, E.; Bauhus, J.; Muys, B.; Wunder, S.; Bozzano, M.; Farsakoglou, A-M.; Schuck, A.; Lindner, M.; Lapin, K. 2021. Forest biodiversity in the spotlight – what drives change? European Forest Institute. https://doi.org/10.36333/rs