TRADITIONAL FIRE USE IMPACT IN THE ABOVEGROUND CARBON STOCK OF THE CHESTNUT FORESTS OF CENTRAL SPAIN AND ITS IMPLICATIONS FOR PRESCRIBED BURNING

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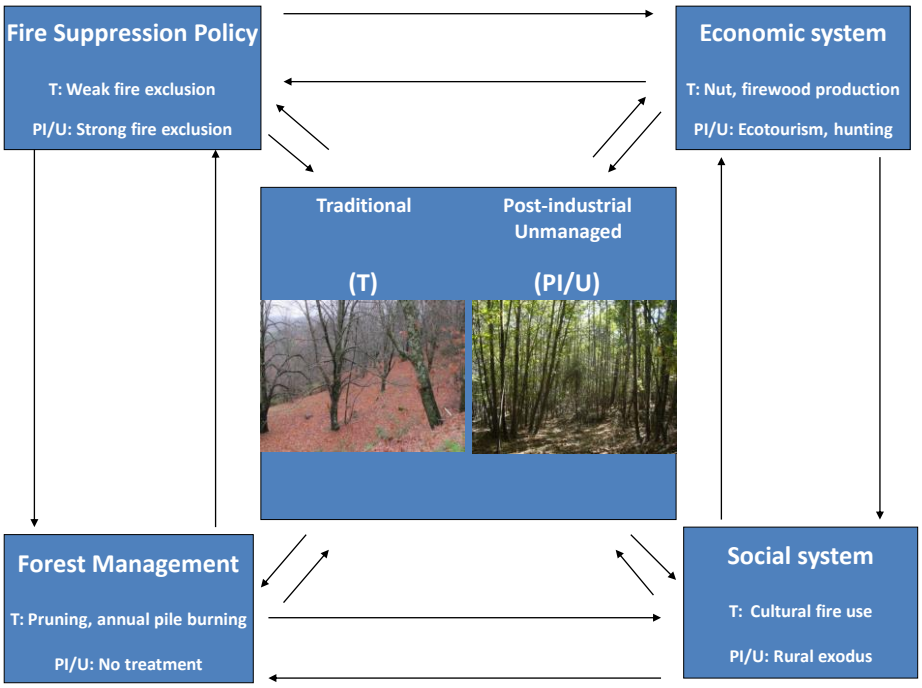
GRAPHICAL ABSTRACT



ABSTRACT

Chestnut forest ecosystems have a complex fire ecology; a result of centuries of co-evolution with pre-industrial era, cultural fire use by local communities based on Traditional Ecological Knowledge (TEK). As the “forest transition” unfolds throughout Europe however, and the traditional role of chestnut forest ecosystems as producers of edible nuts and firewood declines, chestnut forest resilience may be endangered due to disturbance regime changes driven by transformations in land use linked to the rural exodus, state fire exclusion policies and climate change. In this study we compared the aboveground carbon stocks of two chestnut forests located in Central Spain which can be considered representative of divergent Europe-wide management trends. In the first site of Casillas traditional burning and forest management practices are still widespread and their impacts on forest stand structure can be characterized

as maintaining “open canopy”, low density stands dominated by old growth chestnut trees. In the second site of Rozas de Puerto Real traditional fire use has declined and natural ecological succession processes have resumed resulting in high density, “closed canopy” stands dominated by young chestnut tree saplings and increasing pine, oak and shrub encroachment. For both sites we used in-the-field monitoring methods to estimate aerial carbon stock using allometric equations. Our results suggest that carbon sequestration and species richness is greater in the traditionally managed chestnut forest stands. Since present demographic trends present a bleak picture for the maintenance of traditional fire use by local communities, we argue that future fire management of unmanaged chestnut stands and maintenance of traditional forest stands ought to be implemented through surrogate prescribed burning plans that replicate the seasonal timing and ecological effects of TEK based controlled burning.

1. Introduction

The Forest Transition Model (FTM) theorizes that forest cover tends to expand as societies undergo industrialization and urban development (Mather, 1992). This expectation arises from the untested hypothesis that as rural population density declines forests expand and biomass per hectare and carbon uptake should also necessarily increase (Mather, 1992; Mather and Needle, 1998; Walker, 1993; Houghton *et al.*, 2000; Rudel *et al.*, 2005; Angelsen and Rudel, 2013). On the basis of the conventional wisdom generated by theories such as FTM, policy networks linked to European climate change mitigation efforts have sought to incentivize reforestation and afforestation policies. These policies have become firmly established as one of the key “low cost” strategies in the global climate change mitigation regime for the

foreseeable future though empirical evidence supporting the FTM has been obtained from only a few European and Asian experiences (Rudel *et al.*, 2005; Stern, 2007; Barbier *et al.*, 2009; Angelsen and Rudel, 2013). European countries, in particular, experienced substantial expansions in forest cover as they industrialized and urbanized thus seemingly corroborating the main tenets of the FTM (Fuchs *et al.*, 2014; European Environmental Agency, 2015). Though, there is no doubt that the FTM provides valuable insights into the relationship between large-scale socio-economic development processes and forest cover trends it also seems to show some limitations regarding the potential increases in carbon sequestration to be derived from net expansions in forest cover (Oliveira *et al.*, 2017). These limitations are often related to the omission of the role played by fire as a disturbance regime in many of the Earth's terrestrial ecosystems (Chapin *et al.*, 2000; Hurteau *et al.*, 2008; Oliveira *et al.*, 2017). Some studies, in fact, have begun to empirically test the hypothesis that as forest cover expands so does the frequency and intensity of fires which may result in unintended positive feedback loops with climate change, increased landscape fuels and state led fire exclusion policies (Seijo and Gray, 2012; Stephens *et al.*, 2014). These changing feedbacks could potentially contribute to an increase in greenhouse gas emissions from the forestry sector in spite of net expansions in forest cover (Chapin *et al.*, 2008; Hurteau *et al.*, 2008; Batllori *et al.*, 2013; Stephens *et al.*, 2014; Oliveira *et al.*, 2017).

Chestnut forests in Spain, and by extension Europe, are currently experiencing a forest transition (Conedera *et al.*, 2016). During the 20th century many chestnut forests were abandoned as a result of the gradual decline of pre-industrial forms of land use and

management that require continued cultural, labour intensive inputs (Grund *et al.*, 2005; Krebs *et al.*, 2012; Pezzatti *et al.*, 2013; San Roman *et al.*, 2013; Zlatanov *et al.*, 2013; Seijo *et al.*, 2015; Conedera *et al.*, 2016; Seijo *et al.*, 2017). Chestnut forest cover however, as the FTM predicted, expanded overall, though old growth formations are increasingly encroached upon by younger saplings from both chestnut and other deciduous and conifer tree species growing in dense, closed canopy formations as natural ecological succession processes resume (San Roman *et al.*, 2013; Zlatanov *et al.*, 2013; Conedera *et al.*, 2016; Seijo *et al.*, 2017). As a result researchers and managers have become concerned with the continued ability of chestnut forest ecosystems to provide valuable ecosystem services, such as the conservation of biodiversity and carbon sequestration, under the new circumstances generated by the rural exodus, land use transformations, climate change and changing fire regimes (Pezzatti *et al.*, 2013; San Roman *et al.*, 2013; Seijo *et al.*, 2017).

Much of this uncertainty centers around the potential resilience, or lack thereof, of chestnut forest ecosystems to changing disturbance regimes, especially fire (Zlatanov *et al.*, 2013; Conedera *et al.*, 2016 Seijo *et al.*, 2017; López-Sáez *et al.*, 2017). Like many other Mediterranean species, chestnuts have a complex fire ecology often linked to annual, low intensity, non vegetative season, anthropogenic traditional fire use (Grove and Rackham, 2000; Pausas and Keeley, 2009; Seijo *et al.*, 2015, 2017; Lopez Saez *et al.*, 2017). These pre-industrial era forms of cultural burning based on traditional ecological knowledge maximizing the exploitation of food products and firewood, may have in fact augmented chestnut tree resilience to fungal and insect pests while modifying some key fire regime attributes that, in

turn, may have helped prevent large fires and conflagrations by reducing landscape fuel beds and ladder fuels (Seijo et al., 2015, 2017).

The half a century long application of state policies of fire exclusion, climate change and recent transformations in land use may however lead to important fire regime transformations in Mediterranean type ecosystems where fire events may be getting larger and more severe (Moreira *et al.*, 2011; Fernandes *et al.*, 2013; Stephens *et al.*, 2014; Oliveira *et al.*, 2017). Studies based on the evolution of land use and land cover trends during the last 20th century have confirmed that traditional activities (i.e. agriculture and animal husbandry) were abandoned in Mediterranean rural areas when they were depopulated (Grove and Rackham, 2000; Viedma *et al.*, 2006; Moreira *et al.*, 2011; Viedma *et al.*, 2015) . These socioeconomic changes along with the expansion of wildland-urban interface areas and the aforementioned factors may be accentuating the coupling of fire regimes with climate (Pausas and Fernández-Muñoz, 2012). These general fire regime trends could have important consequences for the ability of chestnut forest ecosystems to continue sequestering carbon and provide other key ecosystem services in the near future in spite of their relatively recent net expansion.

In this article we will explore these hypotheses about the chestnut forest transition through the empirical evidence obtained from one such process taking place in two chestnut forest ecosystem sites in Central Spain that can be considered representative of Europe-wide trends (Conedera *et al.*, 2016). Specifically we compared carbon aboveground vegetation stocks and

species richness in two contrasting chestnut forest ecosystem with diverging fire regimes and management strategies associated with human systems exhibiting markedly different levels of economic development (Seijo and Gray, 2012). We make the case that our findings can inform and offer valuable insights into the unintended ecological consequences of the FTM in terms of carbon balance, species richness and, indirectly, fire risk. These insights could in turn provide useful social-ecological criteria for the formulation of climate change mitigation and adaptation strategies that take into consideration the potential use of prescribed burning inspired by traditional fire use to help abate landscape fuels while simultaneously favouring chestnut forest ecosystem resilience to fire and insect disturbances, conserving biodiversity and enhancing local rural economies through high added value nut production (Seijo *et al.*, 2015, 2017).

2. Methods

2.1 Study area, site characteristics and fire regimes

This study was conducted in sweet chestnut (*Castanea sativa* Mill.) forest stand sites of Casillas and Rozas located in the foothills of the mountains of Gredos (central Spain) (Fig.1). Data collection was carried out at 900-1100 m.a.s.l. range of elevation hillside forests, with south-southeast orientation and the same slope ranges in Casillas (40°19'N and 4°34'W), autonomous community of Castilla y León, and on the limit between Navahondilla and Rozas de Puerto Real (40°18'N and 4°29'W) autonomous community of Madrid in the summer of 2016. These municipalities are contiguous geographically but separated by a political boundary between

autonomous communities in the Spanish state and exhibit contrasting forest and fire management strategies. Site selection was determined theoretically by previous studies (Seijo *et al.*, 2015, 2017) based in the contrasted economic development levels of both municipalities and, especially, the forest stand structure that represented the different forest and management strategies applied during the last century (late 1800s in the less developed site of Casillas and 1986 in the more developed site of Rozas de Puerto Real – from now on referred to as Rozas) as well as the uneven intensity of implementation of autonomous community fire exclusion policies (Seijo *et al.*, 2015,2017).

Based on these previous studies´ findings, we selected for analysis “traditional” stands (defined as open canopy, old growth, clear understory chestnut forest stands for nut production maintained through annual literfall pile burning by local communities) located in Casillas, whose stand structure resembles those of “dehesas” or Mediterranean savannas (grassland or scrubland communities, associated with big diameter, old growth scattered trees). In contrast, in Rozas, our selected stands were characterized as being “unmanaged” since the forest structure was dominated by very dense closed canopy chestnut groves, composed of young trees re-sprouting from root systems and associated coppiced chestnut stands. In these “post-industrial” era chestnut forest stands natural forest succession processes have resumed either due to a lack of active management or to foster the creation of hunting real estates and ecotourism developments where “wilder” looking chestnut forest stands are preferred (Geist *et al.*, 2001; Council of Europe, 2007).

Despite differing significantly in economic development levels and state fire exclusion investments and policy implementation, the municipalities share, as noted, similarities in their biophysical conditions. The potential vegetation of the area corresponds to the *Luzulo-Quercetum pyrenaicae* series (Rivas-Martínez, 1987). The soils are of granitic origin, and can be classified as humid brown soils (humic cambisols according to the FAO nomenclature, FAO, 1981) that can contain an important amount of humus. Both sites are characterized by a dry-summer Mediterranean climate, with precipitation concentrated in the autumn, spring and winter months. Mean annual temperature for Rozas is 12.1 °C and 13.4 °C for Casillas, and mean annual precipitation is 831 mm in Rozas and 978 mm in Casillas based on data from meteorological stations located in each village (Seijo *et al.*, 2015).

Local communities in the less economically developed site of Casillas still carry out TEK based traditional fire use practices in order to promote an open canopy, old growth chestnut forest stand structure. In this site state authorities, for a variety of reasons, have applied fire exclusion policies with less intensity than in the adjacent more developed site of Rozas (Seijo and Gray, 2012; Seijo *et al.*, 2015 and 2017). In the more economically developed site of Rozas traditional fire use has, consequently, practically disappeared. Additionally, state fire exclusion policies are applied with more intensity in Rozas due to the autonomous community of Madrid's greater financial resources (Seijo *et al.*, 2015). Chestnut forest use has also shifted away from traditional, pre-industrial era nut and firewood production to emphasize eco-tourism and recreational hunting estate activities which are based on "wilder", unmanaged landscape

aesthetic preferences (Seijo *et al.*, 2017). Previous studies have shown that changes in vegetation, fuel continuity and forest stand structure at the landscape level associated with these latter management goals may be associated with changes in fire regime attributes for both sites (Seijo *et al.*, 2015, 2017). In these studies divergent fire regimes were characterized for the traditionally managed and "unmanaged" chestnut forest stands of Casillas and Rozas respectively (Seijo *et al.*, 2015, 2017). Wildfire event incidence as recorded in official autonomous community records was greater in Casillas than in Rozas from 1984 to 2009 with the former experiencing 52 fire incidents in contrast to 31 in the latter. Median annual burned areas per fire event were similar for both municipalities (Rozas 0.41 ha, Casillas 0.42 ha) but once differences in landscape area were accounted for, burnt surface per year was larger in Rozas than in Casillas by a factor of 10 (2.12 ha km^2 $yr.^{-1}$ compared to 0.22 ha km^2 $yr.^{-1}$). These remarkable fire regime differences seem to be mainly driven by the large fire Rozas experienced in 1985 (1,257 ha : ie. official statistical definition in Spain: >500 ha) whereas none of these events took place in Casillas during the recorded period. The plots we sampled in Rozas were all, in fact, affected by this large fire which in turn has decisively affected present stand structure. In Casillas fire incidents in early spring and autumn months account for a much greater proportion (53%) than summer fires (47%). This contrasts with Rozas where summer months account for the vast majority of fire events (71%) with early spring and autumn months fire events amounting to far less (29%). Finally, according to a pilot fire history study carried out in both locations, more fire events appear in fire scarred chestnut trees in Rozas than in Casillas (Seijo *et al.*, 2017). These findings suggest that fire severity could possibly be greater, in spite of

less fire incidence, in Rozas than in Casillas but more studies would be needed to further ascertain this hypothesis (Table 1).

2.2 Studied species

The sweet chestnut (*Castanea sativa* Mill.) is a deciduous, hard-wood angiosperm tree species native to Western Europe and the Iberian Peninsula, where several pre-Holocene glacial refugia have been identified (Lopez Saez et al., 2017). It has been widely cultivated throughout the Mediterranean Basin, in areas with abundant precipitation and its geographical range is closely associated with the activities of pre-industrial traditional agrarian societies and their targeted products and services (Conedera *et al.*, 2004; López-Sáez *et al.*, 2017). Recent studies in the area (Prada *et al.*, 2016) have pointed to the importance of improving the management and economic potential of sweet chestnut, and also, called for the quantification of its role in mitigating climate change through its storage of carbon in long and medium-term products.

We complementarily used data obtained by Seijo *et al.* (2015 and 2017) for chestnut forest management types and their impact on fire regime changes and stand structure in the Gredos mountain range of Central Spain. These studies revealed connections between economic and political development processes and forest stand structure in these ecosystems that is not only relevant to this specific study area but is also emblematic of current chestnut forest transitions taking place throughout all of Europe (Grund *et al.*, 2005; Krebs *et al.*, 2012; Pezzatti *et al.*,

2013; San Roman *et al.*, 2013; Zlatanov *et al.*, 2013; Seijo *et al.*, 2015; Conedera *et al.*, 2016; Seijo *et al.*, 2017).

2.3 Sampling design

In the summer of 2016 we sampled monospecific chestnut forest stand structures shaped by the different management strategies applied to them (therefore referred to as traditional and unmanaged) in two different unevenly developed municipalities; Casillas and Rozas (Fig 1).

At each of the two sites, efforts were made to derive biometric estimates of total carbon storage in the aboveground vegetation as one of the most important carbon retention pools, and to compare the effects of the different forest stand structures derived from alternative management strategies associated with different economic development levels as identified in previous chestnut forest ecosystem studies in both sites (Seijo *et al.*, 2015, 2017). For each site, 3 circular plots (15 m radius) were established, their GPS coordinates registered for the plot center and, within them, individual tree variables were obtained for the biomass study. We estimated the richness of species in the understory for each management strategy by counting the established species in the sampling plots.

Within each plot, we measured the perimeters at breast height (1.3 m) of all trees with a diameter greater than 5 cm using a measuring tape. We then reconstructed the average forest

structure following the Spanish National Inventory's diametric classes (5cm range diametric classes) so as to determine tree density per hectare. The heights of all trees were measured by combining the angular measurements from a clinometer and the distance to the plot center measured with a laser telemeter, when this was feasible. Since the high density of the stands in the Rozas plots rendered the viewing of the canopy crown and even the crown base height impossible, heights were finally estimated using LiDAR data. The LiDAR data were downloaded from the National Airborne Orthophotography Plan (PNOA), hosted and performed by the National Geographic Institute, Ministry of Infrastructure of the Spanish Government. The LiDAR cloud data from the PNOA have a point density of 0.5 first return per square meter, enough to derive the dominant height for homogenous forests such as the ones studied here. In order to assure the comparability of the two sets of plots, the tree heights were derived from the LiDAR data processing for both sets. From the normalized tree heights, that is, the difference between the crown top's elevation and the ground elevation, the 95th percentile was used as the most reliable value for tree heights for each spot.

2.4 Total aboveground biomass and carbon storage

The application of mathematical allometric equations has been widely used in forest biomass estimates to assist managers with decision making and policy development. The estimation of aboveground biomass weight provides a suitable management tool for use by forest managers and researchers (carbon cycle studies, nutritional balances of the forest system, etc.) at different scales (e.g. individual tree or stand level). In spite of the existence of these non-

destructive methodologies, there are only few studies focused on hardwood species in Europe. *Castanea sativa* Mill., and by extension chestnut forests, have been the focus of some studies aimed at the monitoring of biomass, forest productivity, and carbon dynamics under different forest management activities (Leonardi *et al.*, 1996; Montero *et al.*, 2005; Salazar *et al.*, 2010; Ruiz-Peinado *et al.*, 2012; Menéndez-Miguélez *et al.*, 2013; Prada *et al.*, 2016). For these species, many of the applied nonlinear allometric models allow to calculate values for different plant sections (trunk, branches of different thickness), and also total root and aboveground biomass for each individual tree. Each model has its own specific requirements, but the most common explanatory tree variable correlated with the biomass is the diameter at breast height (dbh). The accuracy of the biomass estimates is usually increased through the inclusion of tree height, and some authors have also considered stand variables (such as age, basal area, site index or dominant or mean height) for improving the accuracy of the estimations (Menéndez-Miguelez *et al.*, 2013). The biometric measures presented in this study were focused specifically on carbon storage in aboveground vegetation and do not include possible carbon stocks present in soils.

We estimated total aboveground biomass by applying five different individual tree allometric equations considering as explanatory variables dbh and, in two cases, LiDAR derived tree height as well (Ruiz-Peinado *et al.*, 2012 and Menéndez-Miguélez *et al.*, 2013) depending on the characteristics of each model (Table 2). These equations have been most commonly used in the literature for estimating large scale inventory-based forest carbon biomass budgets across regional boundaries in Southern Europe for chestnut forests (Leonardi *et al.*, 1996). Some of the models have been widely applied in the last 20 years in projects related with biomass

accumulation and carbon dynamics as in, for instance, the one employed by the Spanish state's Third Forest National Inventory (Ruiz-Peinado *et al.*, 2012; Montero *et al.*, 2013) carbon budget that served as the base for the "Evaluation Report on Impacts, Vulnerability and Adaptation in the Forests and Biodiversity of Spain in light of Climate Change" (Herrero and Zavala, 2015). In the present study, the total biomass for each plot was measured by summing up all the individual tree aboveground biomass estimates within it. The carbon content was set at 48.4% of the total aboveground biomass estimates (Montero *et al.*2005).

The perimeter measured on all trees at breast height (1.3m) was transformed into diameter at breast height values (dbh, cm) which were then used as a data input for the allometric equations applied together with the LiDAR-derived tree height (m). Of the five allometric models we applied (Table 2), only two use tree height as an explanatory variable for biomass calculation. Total aboveground carbon storage by plot, tree and hectare were also calculated following the five different allometric equations.

2.5 Statistical analysis

It is well-known that carbon estimates can vary considerably depending on the different micro environmental and local characteristics of specific study areas. However, to extrapolate and strengthen our results and expand our conclusions with respect to the management practices, total carbon storage under different management treatments (traditional and unmanaged) were calculated and compared for the two different sampled plots through a straightforward

one-way ANOVA analysis. P values were considered statistically significant when $P \leq 0.1$ and displayed in Table 3. All statistical analyses were performed with the R statistical software (R Development Core Team, 2008). The graphical comparison of the yield of the several allometric equations applied is presented as the mean for the 3 plots for each type of management strategy. The error bars refer to the standard error, since the value for each plot represents the sum of all trees' biomass.

3. Results

3.1 Forest structure

A total of 84 trees were measured in the 3 traditional plots in the Casillas site and 1039 in the 3 unmanaged plots of the Rozas site. A completely different forest stand structure for both sites is derived from this data (Figure 2 and Table 1) resulting from these divergent management strategies. Forest management based on TEK is expressed as a low tree density in the stands as opposed to the unmanaged plots. In the traditionally managed plots of Casillas we obtained a mean of 80.2 ± 8 trees per ha while in the unmanaged plots of Rozas we measured 5074.1 ± 829 trees per ha. Size diameter classes were also significantly different for both locations. In the Casillas plots, tree diameters ranged from 16 to 159 cm but most of the individual trees were very large. Trees were smaller in the unmanaged plots of Rozas, where the minimum tree diameter observed (although not considered) was 1 cm for the smallest tree, while the biggest tree was only 24 cm in diameter. These differences in forest stand structure promote

noticeable changes in the crown density that clearly influences the micro environmental conditions and the presence of other species in the understory. Thus, the maximum richness of species registered in the open traditional forest plots amounted to 35 different species (mainly herbaceous), while less than 8 species appeared in the closed canopy unmanaged plots. Figures for the open stands in our study corresponded to the mean tree density values registered for traditional management areas. However, unmanaged plots showed high tree densities, in the upper ranges or even exceeding those values registered in the management scenarios described in the specialized literature on the Iberian Peninsula even considering those stands dedicated to intense, industrially oriented timber production (Prada *et al.*, 2016). However, these values are in the same range that those registered for abandoned chestnut coppice areas in Mediterranean mountain ranges (Leonardi *et al.*, 1996; Menéndez-Miguélez *et al.*, 2013). LIDAR height estimation varied between 17 m and 22 m in Casillas, and 14m to 26m in Rozas. However, mean tree height rose to 19.9 m in Casillas and only 15.3 m in Rozas probably due to the tree age differences. As noted by Seijo *et al.*, 2017, differences in forest stand structure translate into significantly different fuel loads and ignition patterns. Similar findings have been obtained in other studies looking at forest stand structure and its influence on fire regime attributes (Agee, 1996; Fernandes, 2009).

3.2 Carbon storage

Allometric equations resulted in a huge variability of results in terms of aboveground carbon storage in the different plots and trees, depending on the management type (Fig. 3 and 4).

Values ranged from a minimum of 13 to 24 Mg/ha and maximum accumulations of 34 to 59 Mg/ha for Rozas and Casillas respectively depending on the allometric model. Carbon storage was higher in the traditional plots of Casillas than in the unmanaged plots of Rozas based on the results of four of the five applied models, resulting in an average of 2.6 times more carbon stocks in the traditional stands than in the unmanaged ones. Despite the large variability of the biomass estimates among plots subjected to the same forest management strategy, three of the five allometric models applied showed statistically significant higher carbon stocks in the traditional stands of Casillas than in the unmanaged stands of Rozas (Table 3). The equation by Ruiz-Peinado *et al.* (2012) was the only one that showed an opposite result to the rest of the models, overestimating the accumulation of carbon in unmanaged plots in Rozas and overshooting by more than twice the values obtained with the rest of the models. In addition, this model particularly underestimated the storage capacity of the traditionally managed stands when compared with the other models.

4. Discussion

4.1 Chestnut forest stand structure in the "traditional" and "unmanaged" plots

The significant differences found in forest structure (tree density and carbon storage per tree) in our study, is closely linked to the diverging fire regime trends in the traditional plots of Casillas and the unmanaged plots in Rozas which experienced a large fire in 1985 (Seijo *et al.* 2017). Our results seem to replicate the findings of Minnich (1983) on fire regime divergence between unevenly economically developed Southern and Baja California. It is very likely that

divergent fire regimes in Casillas and Rozas are partly driven by the uneven decline of TEK based fire management practices (Seijo *et al.*, 2017). Previous studies have shown that TEK based fire management practices - mainly the non vegetative season, annual burning of litterfall and ladder fuels - has been steadily declining throughout Gredos though this trend seems to have been sharper in Rozas than in Casillas (Seijo *et al.*, 2015, 2017). Official fire statistics indicate that fire incidence is greater in Casillas than in Rozas but burnt surface per incident and in the aggregate is relatively smaller (See Table 1). Wildfire incidents in Casillas seem to be also more closely linked with accidental escapes taking place in the traditional annual pile burning of chestnut leaves and understory fuels during the non vegetative season, while in Rozas fire incidence increases during the summer, vegetative season months (Seijo *et al.*, 2017).

Seijo et al (2015, 2017) suggest that historically traditional chestnut stand fire management practices linked to nut and firewood production based on TEK have influenced considerably the stand structure and fire regime of chestnut forest ecosystems in both municipalities but that, due in all likelihood to the rural exodus and uneven application of fire exclusion policies in both municipalities, fires seem to be becoming larger, less frequent and uncharacteristic (ie. taking place during the vegetative rather than the non-vegetative season as in TEK managed areas) in the unmanaged chestnut forest stands of Rozas. Conditions in Rozas, indeed, seem to be shifting towards what Stephens *et al.* (2014) denominate the “mega-fire” triangle; meaning that antecedent disturbances (the large fire of 1985 in Rozas), the implementation of state fire exclusion policies and climate change could be hypothetically driving the emergence of more

403 infrequent but larger fire events with important ecological and human system consequences
404 (Stephens *et al.*, 2014). This higher "large fire" risk would compound with the lesser efficiency
405 in aboveground carbon storage of unmanaged chestnut forest stands suggesting that
406 traditional, pre-industrial era fire management practices and their impact on stand structure
407 increases carbon stocks while diminishing large fire risk (Table 1).

408 ***4.2 Carbon stocks in "traditional" vs. "unmanaged" chestnut forest stands***

409 Although there is an increasing abundance of studies on biomass measurement that have been
410 improving the accuracy of the allometric equations for *Castanea sativa* Mill. (Ruiz-Peinado *et*
411 *al.*, 2012) there still exists an enormous variability in the outcome result for aboveground
412 biomass estimation using the simplest and more accessible independent variable; dbh. In spite
413 of the limitations of some of the applied models (i.e. Ruiz-Peinado *et al.*, 2012 is only applicable
414 with a minimum of 12.cm diameter for thick branches biomass and the maximum diameter
415 used for model construction was 50.6 cm) and of our study due to the lack of a site specific
416 allometric equation to estimate root biomass and other carbon pools (mainly soil carbon
417 stocks) the majority of the models concurred that traditional open stands managed according
418 to TEK criteria accumulated higher amounts of carbon in aboveground vegetation stocks than
419 the unmanaged plots of Rozas. Managing chestnut forests with traditional cultivation practices
420 such as grafting and thinning with frequent, low intensity seasonal, litterfall pile burning,
421 increased the total aboveground carbon stock in the vegetation.

422

Higher tree density in the unmanaged plots of Rozas decreased the amount of carbon storage in the aboveground vegetation in the system compared to the traditionally managed plots, particularly, in the long term if we factor in increased large fire occurrence (see Table 1). Previous dendroecological studies in the area (Seijo *et al.*, 2017) registered faster annual growth in the unmanaged stands in terms of mean annual tree-ring width (2.8 ±1.1 mm and 4.3 ± 1.2 mm in traditional and abandoned stands respectively) thus suggesting enhanced carbon accumulation in the short term. Our study suggests, however, that traditional chestnut forest management increases the stable carbon pool in the aboveground vegetation overall and that short term gains in the unmanaged plots are offset by temporal and environmental dynamics (driven by disturbance regimes) since forests are not infinite, exponentially growing carbon stocks.

4.3 Implications for future chestnut forest management: A need for prescribed burning based on TEK based traditional fire practices?

The existing literature on chestnut forest ecosystems suggests that similar natural succession and fire regime dynamics to those observed in Casillas and Rozas are emerging throughout Europe though more case studies would be needed to confirm this hypothesis (Conedera *et al.*, 2016; Seijo *et al.*, 2017). This study's findings suggest therefore that in locations where TEK-based traditional fire management still exists strategies for adaptation and mitigation to climate change could be conceivably implemented at a minimal economic cost to the state by local communities that have both the TEK and the adequate social, economic and cultural incentives to continue using annual, pile burning of litterfall in chestnut forest landscapes, particularly

when taking into account disturbance regime transformations likely to be induced by anthropogenic climate change (Stephens *et al.*, 2014; Seijo *et al.*, 2017).

In fire management it is widely understood that open stands increase solar radiation and wind movement in the understory which results in warmer temperatures and drier fuels throughout the fire season. While open stand structures can encourage a surface fire to spread, such fires do little ecological damage if the tree species are adapted to this specific type of historical fire regime (Pausas and Fernández-Muñoz, 2012). These types of frequent, low severity, surface fire events are relatively easy to control and less likely to support a crown fire even under severe weather conditions. However, the vast tree density and closed tree canopy registered in the unmanaged stands of Rozas facilitates the influence of ladder fuels in fire behavior, allowing fires to climb upward into the crowns thus helping to sustain crown fires once they are started (Agee, 1996; Fernandes, 2009). Rapidly moving crown fires typically consume nearly all the fine fuels in a forest canopy when wind and a sloping topography, as in the Gredos mountain range, are thrown into the mix. Nowadays, crown fires caused by excessive fuel accumulation seem to pose a severe threat to ecological and human infrastructure and represent a major challenge for contemporary fire management (Stephens *et al.*, 2014). Despite the variable effects of decreased fire occurrence on fuels and that fire behavior may change subtly from site to site, the higher incidence of crown fires in areas with historically frequent, surface fire regimes is generally occurring all over the world but especially in arid and semiarid areas with possibly deleterious ecological and human system consequences (Stephens *et al.*, 2014).

466

467 The results of this study suggest that the traditionally managed chestnut forest stands of
468 Casillas sequester carbon more efficiently than the abandoned chestnut forest stands of Rozas
469 in the aboveground vegetation carbon pool. When this evidence is coupled with the greater
470 large fire risk present in these latter plots as well as the lower species richness in their
471 understory some important insights for the future management of these forest ecosystems
472 emerge. If managers find it desirable to maximize carbon sequestration, understory species
473 richness and decrease large fire risk it may be necessary to maintain the annual, non-vegetative
474 season, low intensity, surface fire regime that traditionally managed chestnut forest stands
475 have co-evolved with. Paradoxically, the rural exodus and the ageing of the population in
476 Casillas and Rozas may make this management goal difficult, if not impossible, to accomplish in
477 the future. Casillas much like many other rural locations throughout Europe exhibits overall
478 population loss and an inverted demographic pyramid (Seijo *et al.*, 2017). Given that much of
479 the TEK based burning is carried out by the older members of the community this does not
480 bode well for their continuity.

481

482 Prescribed burning by trained professionals replicating the ecological and human system goals
483 of TEK based burning may therefore constitute one of the few alternatives under the new
484 climate change conditions in the Mediterranean of modulating carbon assimilation and
485 maintaining ecosystem productivity and biodiversity in the “new normal” characterized by
486 more frequent droughts, decreasing precipitation and higher average temperatures (Nemani *et*

487 *al.*, 2003; Batllori *et al.*, 2013; Fernandes *et al.*, 2013; Khabarov, 2014; Vicente-Serrano *et al.*,
488 2014; Seijo *et al.*, 2017). The use of prescribed fire should probably be implemented by, or
489 under the guidance of professionals, since current conditions (fuel buildup as the rural exodus
490 proceeds) usually present a high fire escape probability. Nonetheless, mechanical treatments
491 around old growth trees (in combination with prescribed burning in open areas) and thinning in
492 the abandoned plots could conceivably offset these risks. The creation of a Protected
493 Geographical Indication plus an organic certification would also facilitate selling the nuts at a
494 better market price and incentivize continued management of these forests (Xunta de Galicia,
495 2017; MAPAMA, 2017). From the large fire risk mitigation point of view, old growth forest
496 stands might represent very interesting stand structures for Strategic Management Points by
497 increasing overall landscape heterogeneity (EFI, 2017). When treating the entire landscape is
498 not feasible; efforts should be focused on specific portions of the landscape where large fire
499 risk can be efficiently mitigated. Traditionally managed old growth chestnut forest stands, in
500 sum, may not only represent efficient aboveground carbon stock and biodiversity conservation
501 areas but also interesting wildfire prevention areas already embedded in the landscape.

502

503 ***5. Conclusions***

504 Chestnut forest transitions leading to greater forest cover may, indeed, bolster carbon
505 sequestration but only when and if they effectively take into account important ecological
506 disturbance processes such as fire (Hurteau *et al.*, 2008 and 2014; Batllori *et al.*, 2013). In an
507 effort to augment economic productivity and design mitigation policies for greenhouse gases

(GHG) emissions through reforestation and afforestation states, donors, the modern timber industry and forest managers promote younger and densely wooded forest structures - for hunting estates, logging or recreational eco-tourism purposes - though multiple harvesting options and alternative management strategies exist to harmonize all of these forms of land use (Angelsen and Rudel, 2013; Prada *et al.*, 2016). Chestnut forests in Rozas are an example of these current forestry practices. By directly or indirectly fomenting denser chestnut forests populated with younger, smaller diameter trees , carbon uptake may be incentivized in the short to medium term harvesting length periods (Alvarez *et al.*, 2014; Prada *et al.*, 2016). However, in the medium to long term the traditionally managed chestnut forests of Casillas are more effective in performing this function given their greater resilience to disturbances such as fires, pests and drought as well as the added value of their greater biodiversity which may also indirectly bolster ecosystem resilience through as yet poorly understood ecosystem feedback mechanisms (Chapin *et al.*, 2000; Oliver *et al.*, 2015). This could have significant implications for forest management of chestnut forest ecosystems throughout Europe which are currently subject to strict state fire exclusion policies and are being allowed to re-wild in part due to the propagation of economic incentives linked to European and global scale climate change mitigation initiatives (Stern, 2007; European Commission, 2011).

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Figures

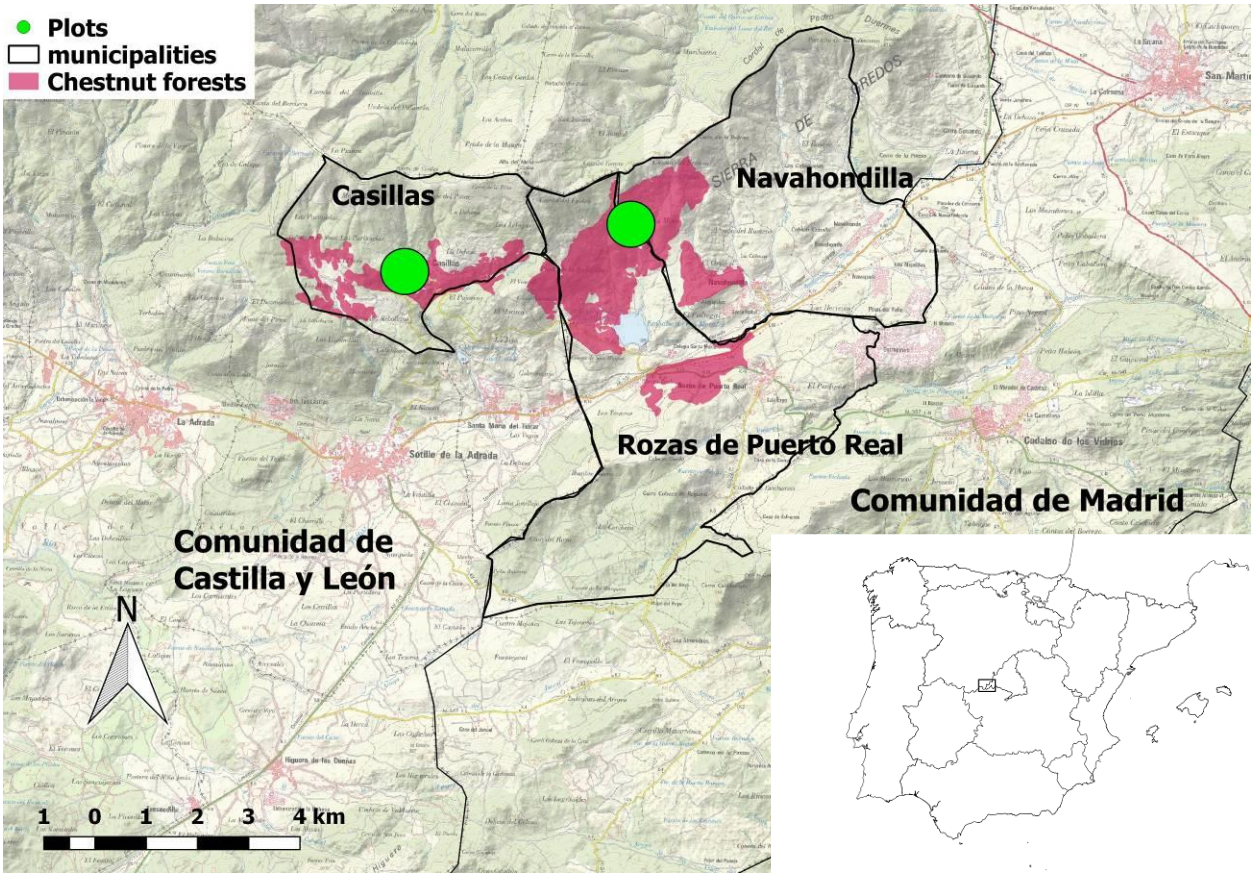

Figure 1. Map indicating the location of the study sites in central Spain - Casillas and Rozas - within the Iberian Peninsula. Red color indicates chestnut forest extent, green dots the sampled plots for each management strategy and black lines boundaries between municipalities and autonomous communities.

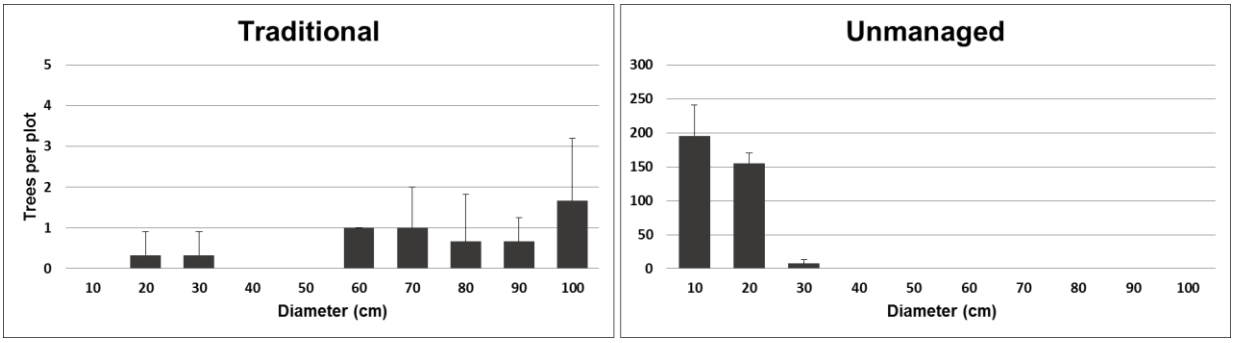

Figure 2. Mean tree density and distribution of diameter classes of traditional and unmanaged stands in the chestnut forest of Casillas and Rozas respectively

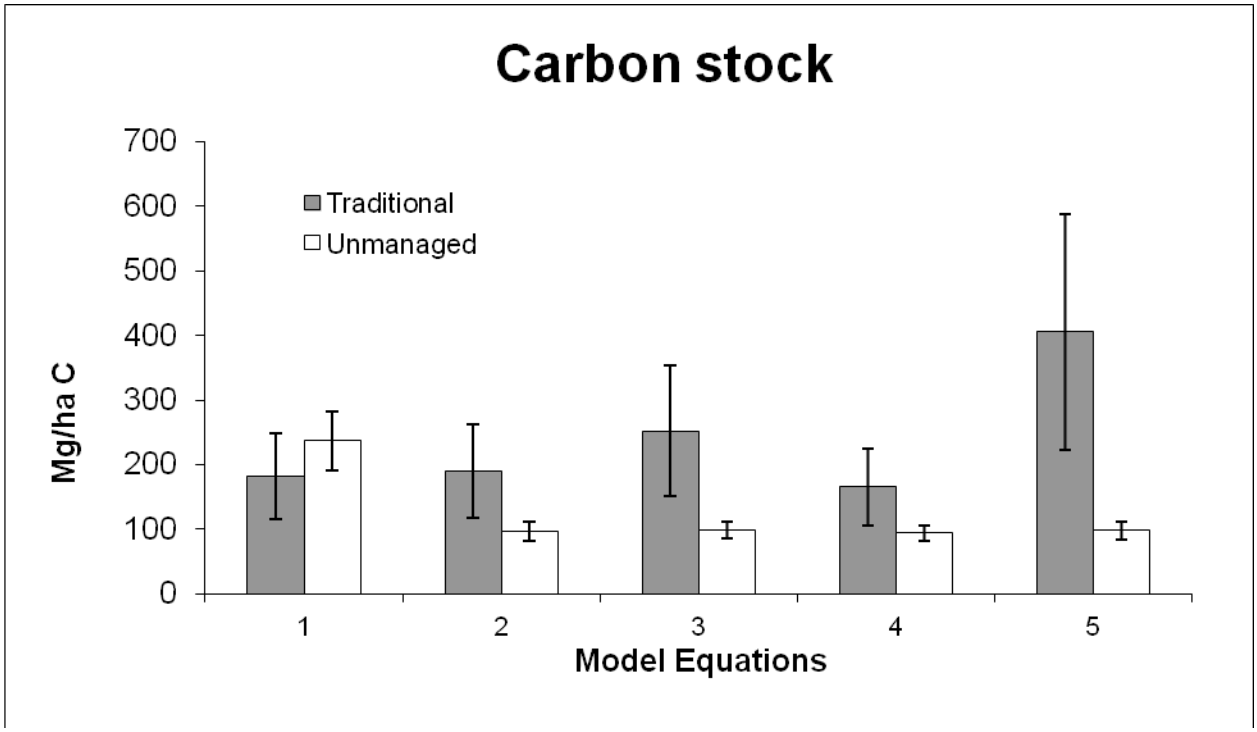

Figure 3. Average aboveground carbon stock estimates of the plots for the two management types. Error bars indicate standard deviations for each Group. The allometric equations results are: 1. Ruiz-Peinado *et al.* 2012; 2. Menéndez-Miguélez *et al.* 2013; 3. Patrício *et al.* 2005; 4. Montero *et al.* 2005; 5. Leonardi *et al.* 1996

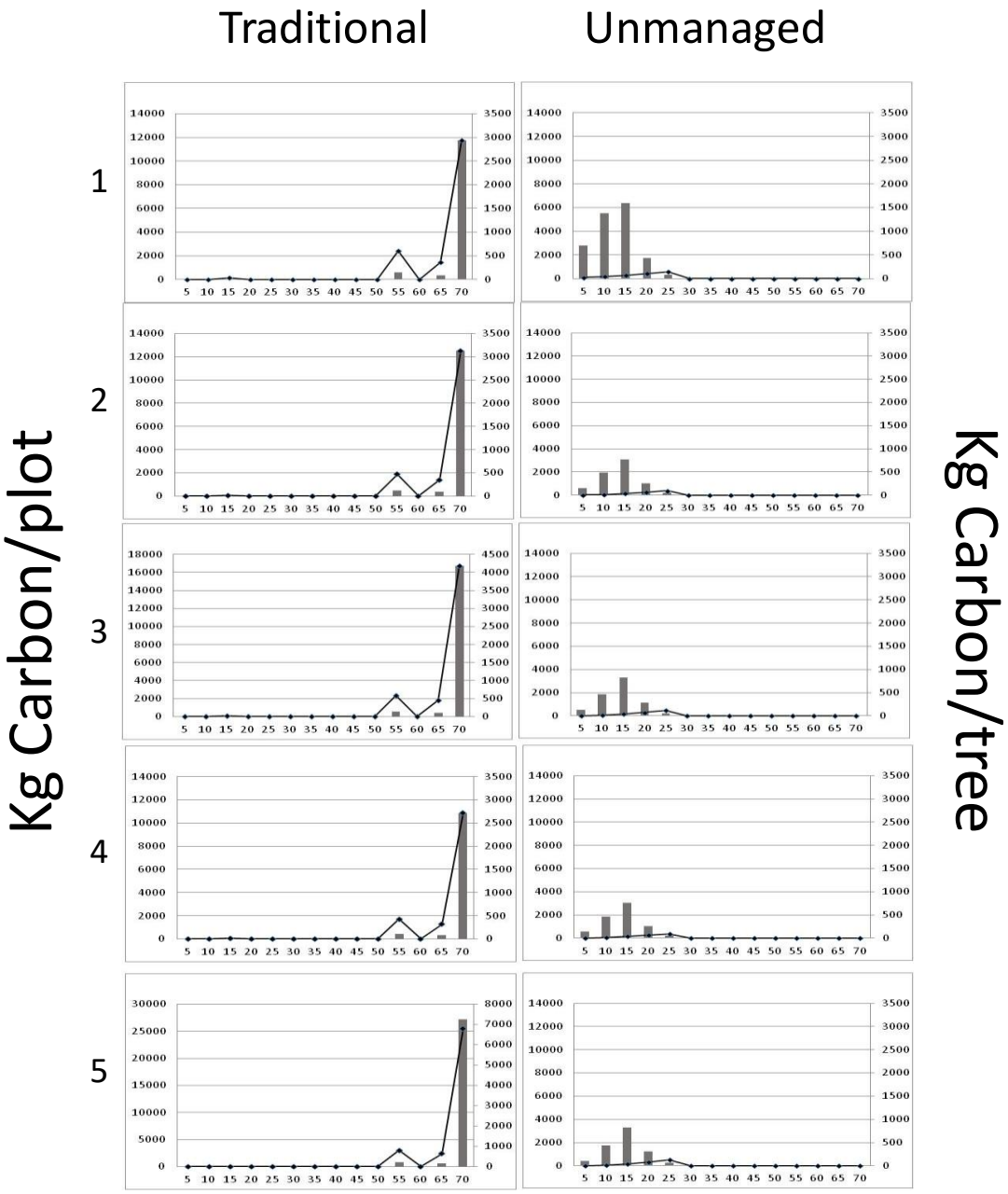

Figure 4. Average carbon stock estimates in the aboveground vegetation of the plots for the two management types per diametric classes (cm). The numbers correspond to the allometric equations models: 1. Ruiz-Peinado *et al.* 2012; 2. Menéndez-Miguélez *et al.* 2013; 3. Patrício *et al.* 2004; 4. Montero *et al.* 2005; 5. Leonardi *et al.* 1996

Fire Management	Fire incidents	Severity	Season	Burnt Surface	Stand structure
Traditional fire use	52	Low severity	53% Non-vegetative	0.22 HA KM2YR-1 No large fire events (>500 hectares)	
Post-industrial unmanaged	31	Medium severity	71% Vegetative	2.12 HA KM2YR-1 Large fire event 1985 1,267 hectares	

Table 1 Divergent fire regime attributes in Casillas and Rozas de Puerto Real and matching chestnut forest stand structure 1984-2009

Authors	Equations	Ref. C content	Place
Ruiz-Peinado *et al.* 2012 W_{tot} = stems + thick + medium + thin branches	$W_s = 0.0142 \cdot d^2 \cdot h$	Mean total C: 344.5 (kg/tree)	Central Mountain Range and Sierra de Ronda (Spain)
	$W_{th} = 0.223 \cdot (d-12.5)^2$		
	$W_{mb} = 0.230 \cdot d \cdot h$		
	$W_b = 0.221 \cdot d \cdot h$		
Menéndez-Miguélez *et al.* 2013 W_{tot} = wood + bark + crown	$W_{wood} = 0.01391 \cdot (d^2 \cdot h)^{1.006}$	-	Asturias (Spain)
	$W_{bark} = 0.004119 \cdot h^{1.086} . (d^2)^{0.7889}$		
	$W_{crown} = 0.5408 \cdot h^{-1.439} \cdot (d^2)^{1.386}$		
Patrício *et al.* 2004	$W_{tot} = 0.1236\ d^{2.3929}$	106.85 – 228.75 (Mg/ha)	Northern Portugal
Montero *et al.* 2005	$W_{tot} = e^{-1.70831} \cdot d^{2.21544} \cdot e^{(0.223169^2/2)}$	-	Cáceres and Málaga (Spain)
Leonardi *et al.* 1996	$W_{tot} = 0.066 \cdot d^{2.628}$	60 (Mg/ha)	Sierra de Gata (Cáceres, Spain)

Table 2. Summary of the allometric equations used for the study, with reference to the authors publishing them, equation formulae, reference C stock value (if available) and study sites of their adjustment (Place): biomass; biomass codes: W_{tot} : total; W_s: stem; W_{th} : thick branches; W_{mb} : medium branches; W_b :thin branches.

		Df	Sum Sq	Mean Sq	F value	Pr(>F)	signif.
Ruiz-Peinado *et al.* 2012	Treatment	1	4567	4567	1.384	0.305	
	Residuals	4	13198	4567			
Menéndez-Miguélez *et al.* 2013	Treatment	1	12971	12971	4.693	0.0962	*
	Residuals	4	11056	2764			
Patrício *et al.* 2005	Treatment	1	35051	35051	6.739	0.0603	*
	Residuals	4	35051	35051			
Montero *et al.* 2005	Treatment	1	7543	7543	4.007	0.116	
	Residuals	4	7530	7530			
Leonardi *et al.* 1996	Treatment	1	142388	142388	8.485	0.0436	**
	Residuals	4	67124	16781			

Table 3. Analisys of variance (ANOVA) for the two managements. Significance coding: * 0.1; ** 0.05