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**Long-term recovery of multifunctionality in Mediterranean forests depends on restoration strategy and forest type**

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**ABSTRACT**

1. Forest area is increasing in temperate biomes through active and passive restoration of old fields. Despite the large extension of restored forests, the success of contrasting restoration strategies (active – planted forests - vs. passive – secondary forests -) over time has never been evaluated before in Mediterranean forests.

2. We studied how restoration strategy determined forest restoration success. Firstly, we evaluated which restoration strategy resulted in forests more like references (i.e. forests with continuous canopy cover since at least the 1940s) in terms of structure, diversity, functional composition, and dynamics. Secondly, we assessed whether active restoration accelerated forest recovery compared to passive restoration.

3. We studied a chronosequence of recovery in four forest types (mountain and Mediterranean pine forests and mesic and Mediterranean oak forests) using the data of the Spanish Forest Inventory in central Spain. Each plot was classified as planted, secondary or reference forest. We modelled the response ratios of 11 forest attributes and a multifunctionality index as a function of restoration strategy, forest age, and abiotic and biotic constraints.

4. Secondary forests showed a greater likeness to references than planted forests in oak forests while minor differences between secondary and planted forests were found in pine forests. The recovery speed of most forest attributes in secondary and planted forests was similar. Multifunctionality was higher, and increased more rapidly, in planted than in secondary forests in Mediterranean oak forests. However, multifunctionality was similar for both restoration strategies in the other forest types.

5. *Synthesis and applications.* The long-term assessment of forest recovery in Mediterranean abandoned fields indicated that both planted forests and natural forest succession are successful restoration strategies, depending on the aim and the forest type. In our research, restoration strategy did not influence the magnitude and speed of forest recovery in pine forests. However, in oak forests, natural forest succession led to forests

more alike to references, but planted forests can maximise and accelerate recovery of forest multifunctionality.

## RESUMEN

1. La superficie forestal está aumentando en zonas templadas gracias a la restauración activa y pasiva de campos agrícolas abandonados. A pesar de la extensión de los bosques restaurados, el éxito de la restauración activa – bosques plantados - vs. la restauración pasiva – bosques secundarios - no se ha evaluado a largo plazo en ambientes mediterráneos.

2. En este trabajo se analizó (1) qué estrategia condujo a bosques más parecidos a las referencias (zonas forestales con una cubierta arbórea constante desde al menos los años 40) atendiendo a la estructura, la diversidad, la composición funcional y la dinámica forestal; y (2) si la restauración activa aceleró la recuperación respecto a la restauración pasiva.

3. Establecimos una cronosecuencia de recuperación con los datos del Inventario Forestal Nacional para cuatro tipos de bosque del centro de España (pinares de montaña y mediterráneos, y bosques de quercíneas mésicos – rebollares - y mediterráneos – encinares -). Las parcelas se clasificaron en bosques plantados, secundarios o referencias. Modelizamos los ratios de respuesta a la restauración de 11 atributos forestales y un índice de multifuncionalidad en función de la estrategia de restauración, la edad del bosque y los condicionantes ambientales.

4. Los bosques secundarios fueron más parecidos a las referencias que las plantaciones en los rebollares y encinares. Las diferencias entre bosques plantados y secundarios fueron menores en los pinares. La velocidad de recuperación de la mayoría de los atributos forestales fue parecida en ambas estrategias. La multifuncionalidad fue mayor y aumentó más rápidamente en bosques plantados que en bosques secundarios en los encinares, pero fue similar para ambas estrategias en los otros bosques.

5. *Síntesis y aplicaciones.* El análisis de la recuperación a largo plazo de bosques en campos abandonados mediterráneos mostró que tanto las plantaciones como la sucesión natural pueden ser estrategias exitosas según el objetivo y el tipo de bosque a restaurar. La estrategia de restauración no determinó la magnitud y la velocidad de recuperación en los pinares. Sin embargo, la sucesión natural generó bosques más parecidos a las referencias en bosques de quercíneas, mientras que las plantaciones en encinares favorecieron la recuperación de la multifuncionalidad.

Keywords: abandoned fields; forest age; forest inventory; forest recovery; Mediterranean; multifunctionality; restoration strategy; forest succession.

Palabras clave: campos abandonados; edad del bosque; estrategia de restauración; inventario forestal; Mediterráneo; multifuncionalidad; recuperación forestal; sucesión forestal.

## INTRODUCTION

Over the last decades the amount of agricultural area has decreased and forest area has increased in both temperate and boreal zones (FAO, 2016), directly helping to meet Aichi Targets and the UN Sustainable Development Goals (Martinez & Maximilian, 2010). Forest restoration strategies vary from passive, such as secondary succession or natural forest recovery after disturbance, to active restoration where human interventions accelerate and influence the successional trajectory of recovery (Stanturf, Palik, Williams, Dumroese, & Madsen, 2014). The success of forest restoration depends on a range of factors including the abiotic and biotic drivers (e.g. Andivia, Villar-Salvador, Tovar, Rabasa, & Rey-Benayas, 2017) and the forest age, the ecological function(s) evaluated and their trade-offs (Montoya, Rogers, & Memmott, 2012), the reference forests (White & Walker, 1997), and the restoration strategy (Crouzeilles et al., 2016; Meli et al., 2017).

Climate and soil are major environmental drivers of forest recovery as they determine recruitment, plant growth (Andivia et al., 2017; Squeo et al., 2007), and vegetation structure (Nadal-Romero, Petrlic, Verachtert, Bochet, & Poesen, 2014; Robledano-Aymerich et al., 2014). Forest recovery depends also on the distance and relative position of the propagule sources (Ruiz-Benito, Gómez-Aparicio, & Zavala, 2012; Sloan, Goosem, & Laurance, 2015) whose effect is influenced by the individual or aggregated size of forest patches (Andivia et al., 2017; Crouzeilles et al., 2016; Sloan et al., 2015). Forest age is another important predictor of diversity and biomass recovery (Crouzeilles et al., 2016; Meli et al., 2017) and increases the similarity between restored and reference forests (Jones et al., 2018). Most studies address the effect of forest age on forest recovery at local scales (i.e. data from small number of plots under similar environmental conditions). However, studies of ecosystem recovery at larger spatial scales are needed because decision-making in ecosystem restoration and biodiversity conservation planning must be done at the landscape or regional scale (Montoya et al., 2012), where environmental differences can affect ecosystem recovery.

The perception of restoration success is highly dependent on the ecological function used to determine success (Montoya et al., 2012). For instance, planted forests in arid and semi-arid areas can increase soil carbon sequestration and reduce soil erosion but they can also increase water shortage and reduce water yield (Cao, Chen, & Yu, 2009; Chang, Fu, Liu, & Liu, 2011). The use of multiple indices (e.g. Gatica-Saavedra, Echeverría, & Nelson, 2017) or multifunctionality indices that integrate several ecological functions (e.g. Byrnes et al., 2014; Manning et al., 2018) is advisable to avoid a biased perception of restoration success. The success of forest restoration also depends on the reference forests used for comparisons. References are usually past or least-disturbed contemporary landscapes (Stanturf et al., 2014), but reference forests may have evolved under different environmental conditions and have different legacies than restored ecosystems. Traditionally, restoration studies have used a small number of reference sites, but using multiple reference sites captures contingent variation of ecosystems and partially solves the problem of site specificity that is inherent to the use of few references (White & Walker, 1997). Finally, global studies have not found a consistent effect of restoration strategy on restoration outcomes (Jones et al., 2018; Meli et al., 2017). Conversely, an effect may be expected at finer scales since dynamics and plant diversity in planted and natural forests varies across gradients of biotic and abiotic conditions (e.g. stand structure and precipitation; Gómez-Aparicio, Zavala, Bonet, & Zamora, 2009; Ruiz-Benito, Gómez-Aparicio, & Zavala, 2012).

Europe is a hotspot of rural abandonment and rewilding (Navarro & Pereira, 2015) and the Mediterranean Basin has undergone large-scale afforestation programs initiated in the late 19$^{th}$ century (Vadell, de-Miguel, & Pemán, 2016; Vallauri, Aronson, & Barbero, 2002). However, the success of active and passive restoration in achieving the attributes of reference forests has never been evaluated over time in Mediterranean forests. To meet this challenge, we aim to evaluate how restoration strategy determines forest restoration success using a chronosequence at regional scale (i.e. a space-for-time substitution approach), with multiple references for each restored plot, by comparing 11 forest attributes or functions (hereafter attributes) and multifunctionality. Specifically, our objectives were to: (1) quantify if planted and secondary forests reach references in terms of forest structure, diversity, functional composition and dynamics; (2) determine if the recovery of these attributes and forest multifunctionality is accelerated in planted forests as compared to secondary forests; and (3) identify if the restoration strategy determines forest multifunctionality under different biotic and abiotic conditions. We tested three hypotheses: (1) secondary forests are more similar to references than planted forests once accounting for age, abiotic factors and landscape context; (2) the initial assistance for recovery in planted forests will speed up some restoration outcomes, such as vegetation cover, compared to secondary forests; and (3) active restoration will benefit forest multifunctionality more intensively in low productive environments with a poorly-preserved forest matrix. Our results point at the ecological factors that should be considered to decide between active and passive forest restoration in Mediterranean regions, while contributing to close the science-practice gap.

## MATERIALS AND METHODS

### Study area

The study area encompasses most of the Community of Madrid, which covers *c.* 8,000 $km^2$ in central Spain (see **Appendix S1** in Supporting Information, **Fig. S1.1**). Climate is continental Mediterranean, with mean annual temperature and annual rainfall between 6 and 16 ºC and 400 and 1500 mm, respectively, and soils are entisols and inceptisols with similar acid chemical composition (references for climate and soil of the study area are in **Appendix S1**). In the study region, there is a marked transition from *Pinus sylvestris* L. forests at the highest (1500-1900 m a.s.l.) and wettest sites, to *Quercus ilex* L. forests in the lowest elevations (500-1000 m a.s.l.) and driest sites. The intermediate altitude and precipitation sites are dominated by *Quercus pyrenaica* Willd. forests. In addition, mixed forest of the Mediterranean pines *Pinus pinea* L. and *Pinus pinaster* Aiton occupy the South-West area at 550-1650 m a.s.l. (**Fig. S1.1**).

### Forest inventory data

We used the Spanish Forest Inventory (Alberdi, Sandoval, Condés, Cañellas, & Vallejo, 2016; SFI hereafter) to evaluate recovery after abandonment of agricultural fields (cropland and pastureland). The SFI is a network of permanent plots over forests and scrublands at a density of 1 plot per $km^2$, surveyed in 1990-1994 (SFI II), 2000 (SFI III) and 2012-2013 (SFI IV) in the Madrid region (see **Appendix S2** for details on inventory methodology). We selected plots with (1) at least one alive tree, recruit or shrub, and (2) < 25% of basal area corresponding to riparian species (i.e. *Alnus glutinosa*, *Fraxinus angutifolia*, *Populus* sp., *Salix* sp. and *Tamarix* sp.) to exclude riparian forests.

The SFI plots were assigned to their potential forest type according to Sainz, Sánchez, & García-Cervigón (2010): (1) mountain pine forest of *P. sylvestris*; (2) Mediterranean pine forest of *P. pinaster* and/or *P. pinea*; (3) mesic or submediterranean oak forest of *Q. pyrenaica*; and (4) Mediterranean oak forests of *Q. ilex* (**Table S1.1; Fig. S1.1**). The potential forest types were delimited using recent land use maps, theoretical successional series of vegetation corrected by paleogeographic data, and predictive models (Sainz-Ollero, Sánchez, & García-Cervigón, 2010). We discarded SFI plots in other potential forest types over basic soils because of the small number of plots (113 or 4.4% of the sample) and the heterogeneity of dominant species including shrubs. This reduced soil composition differences to acid and mainly poor developed soils across the study area.

**Predictors of forest recovery**

We considered five potential predictors of forest recovery: (1) restoration strategy (i.e. active vs. passive restoration corresponding to planted and secondary forests, respectively), (2) forest age, (3) annual rainfall, (4) maximum slope, and (5) forest cover around each plot. We classified each SFI plot as planted, secondary or reference forest using the SFI coordinates and series of aerial photographs and land use maps dating from 1933 to 2009 (**Table S2.1**). For each series and map, we classified the past land use of SFI plots as (1) agricultural or (2) shrubland or natural/planted forest, identifying a sequence of land uses for each plot, which was used to distinguish different restoration strategies (see **Appendix S2** for details on photo-interpretation and plot classification criteria). We estimated forest age based on the year the plot was abandoned. Abandonment year was set as the central year of the period when the shift from agricultural to shrubland or forest was observed. For the plots classified as shrubland/forest along the entire sequence of land uses, 1935 was assigned as the abandonment year. Then, we calculated forest age as the difference between the year when the SFI plot was sampled (i.e. 1992 in SFI II, 2000 in SFI III, and 2012 in SFI IV) and the abandonment year.

Mean annual rainfall (mm) for each plot was calculated from Gonzalo's (2010) map (spatial resolution: 1 $km^2$; temporal resolution: monthly; survey period: 1950 to 1999), and maximum slope (º) was obtained from SFI field data. Soil erodibility can govern restoration outcomes in semi-arid environments (Robledano-Aymerich et al., 2014). Therefore, we used maximum slope as a proxy of soil erodibility (i.e. structure) and soil depth (Martz, 1992). Forest cover in a 1-km-radius area around the plot centre was calculated as the percentage of forest and shrubland in different land use maps contemporary to the different SFI surveys (**Table S2.1**), using ArcMap 10.0 (ESRI Inc., Redlands, CA, USA).

**Recovery of forest attributes**

We used 11 forest attributes to assess recovery of forest structure (tree biomass and coefficient of variation of tree height), diversity (woody species richness and functional dispersion), functional composition (functional identity of maximum height, functional identity of leaf mass per area (LMA), functional identity of seed mass, and cover of frugivore-dispersed shrubs), and dynamics (recruitment of reference species, tree growth, and dead wood) (see a description of each attribute in **Appendix S3**). We checked these variables for extreme values (**Figure S3.1 to S3.11**) and one tree growth outlier was excluded from the analyses.

We compared each attribute in planted and secondary forests with the levels in reference forests. Firstly, we identified the reference plots and calculated reference values for each attribute according to the methodology described in **Appendix S2**. Secondly, we quantified the standardised mean effect size of each forest attribute between restored and reference forests using response ratios (Rey-Benayas, Newton, Diaz, & Bullock, 2009; see **Appendix S2** for response ratio calculation and **Appendix S4** for exploratory analysis).

**Multifunctionality**

We calculated a multifunctionality index (see calculation in **Appendix S2** and exploratory analysis in **Appendix S5**) based on eight forest attributes related to global forest functioning (Jax, 2005): tree biomass (resource use), coefficient of variation of tree height (resource use and habitat provision), woody species richness (habitat provision and resilience), functional dispersion (resilience), cover of frugivore-dispersed shrubs (habitat provision), recruitment of reference species (regeneration), tree growth (productivity), and dead wood (habitat provision and nutrient cycling; see descriptions of each function in **Appendix S3**). We assumed that an increase in each attribute value was desirable. We did not include the attributes related to functional composition (i.e. functional identity of maximum height, LMA and seed mass) in the multifunctionality index because we could not rank the desirable outcome for each forest type or link them to single specific functions (e.g. Ruiz-Benito et al., 2014).

**Statistical analyses**

We fitted mixed linear models for each potential forest type separately for the response ratios of the 11 attributes and for the multifunctionality index. All statistical analyses were performed using the package "lme4" (Bates et al., 2017) in R 3.4.3 (R Core Team, 2017). Plot identity was considered as a random effect to account for non-independence due to successive surveys in the same plots. The predictor variables were restoration strategy, forest age, annual rainfall, maximum slope, and forest cover (see correlation analysis in **Fig. S6.1**). The numerical predictor variables were standardised, enabling the interactions to be tested and compared (Schielzeth, 2010). We used a normal error distribution and identity link (**Fig. S6.2**). The linearity between each predictor and the response variable was checked through partial residual plots (Schielzeth, 2010; **Fig. S6.3-14**). The predictor variables were log-transformed in cases with non-fulfilled linearity (**Table S6.1**).

We fitted models including all predictors to test whether restoration strategy had a significant effect on the response variables, while considering confounding factors such as landscape configuration and abiotic constraints. Secondly, we fitted models including all predictors and the pairwise interaction restoration strategy × forest age. We compared the full model with a reduced model in which the interaction term was dropped, using ΔAIC > 2 as an indicator for supporting the more complex model (AIC means Akaike Information Criterion; Burnham & Anderson, 2002). Finally, we fitted multifunctionality models including the three-way interaction restoration strategy × forest age × annual rainfall or restoration strategy × forest age × maximum slope or restoration strategy × forest age × forest cover. We compared the multifunctionality models with reduced models without the triple interaction, using ΔAIC as in the former comparison. We could not test all possible pairwise interactions simultaneously, as well as the three-way interaction term for potential mountain pine forests, due to lack of data (**Fig. S5.1**).

## RESULTS

### Attributes of restored forests in comparison to reference forests

Overall, restoration strategy influenced structure, diversity, functional composition and dynamics of forests, and the relationship was stronger in potential oak forests than in potential pine forests (**Fig. 1**). Models marginal $R^2$ varied between 1% and 36% while conditional $R^2$ varied from 6% to 99% (**Table 1;** see complementary results in **Appendix S7**).

Restoration strategy affected the forest structure in both potential oak forest types (mesic and Mediterranean) (**Fig. 1a**). Tree biomass of planted forests was more similar to references and had higher values than secondary forests (**Fig. 1a**). Conversely, secondary forests had higher coefficients of variation of tree height than planted forests (**Fig. 1b**).
Secondary forests had higher woody species richness than planted forests in potential pine forests, but no differences were observed in functional dispersion (**Fig. 1c, 1d**). In potential

mesic oaks forests, functional dispersion was higher in planted than in secondary forests (**Fig. 1d**). Functional composition was closely linked to the restoration strategy, with clear differences among forest types (**Fig. 1e-h**). In potential oak forests, secondary forests were more similar to references for all functional identity traits (**Fig. 1e-g**). In potential Mediterranean pine forests, planted forests had higher functional identity of maximum height than secondary forests (**Fig. 1e**). We did not observe differences between restoration strategies in the cover of frugivore-dispersed shrubs (**Fig. 1h**).

When differences in forest dynamics (i.e. tree growth, recruitment, dead wood) between restoration strategies were found, secondary forests were generally more similar to references than planted forests (**Fig. 1i-k**). Secondary forests had more recruitment of reference species in potential oak forests and lower tree growth in three of four forest types than planted forests (**Fig. 1i-j**).

**Speed of forest recovery**

Differences in the speed of forest recovery between planted and secondary forest (represented by the interaction restoration strategy × forest age) were small and only supported in potential oak forests (**Table 1; Fig. 2**). The recovery time was different depending on the restoration strategy for structure, functional composition and dynamics, but not for diversity-related attributes (see woody species richness and functional dispersion in **Table 1**).

Tree biomass, functional identity of maximum height and recruitment changed more rapidly in planted than in secondary forest in potential Mediterranean oak forests (**Fig. 2a, 2e, 2i**). The recovery of tree biomass to the reference values was attained after 59 years in planted forest and after > 80 years in secondary forests assuming a constant trend (**Fig. 2a**). In potential mesic oak forests, the recovery of CV of tree height in planted forest slightly decreased with age but it increased in secondary forests (**Fig. 2b**). The functional identity of maximum tree height of planted forest moved away from the reference values with age, while it was constant and similar to references for secondary forests (**Fig. 2e**). In potential Mediterranean oak forests, recruitment of reference species increased rapidly with forest age in planted forests and only slightly in secondary forests, attaining similar values in forests older than 71 years (**Fig. 2i**).

**Recovery of multifunctionality**

Multifunctionality was overall low. The maximum potential multifunctionality was 8; however, the mean multifunctionality was 1.32 (± 1.22 sd), 1.23 (± 1.13 sd) and 1.33 (± 0.96 sd) in planted, secondary and reference forests, respectively. Restoration strategy only influenced multifunctionality in potential Mediterranean oak forests, where planted forests had higher multifunctionality than secondary forests (see negative values of restoration strategy in **Fig. 3a**). Multifunctionality increased with forest age in all forest types except in potential mountain pine forests, with rainfall in potential Mediterranean pine forests, with slope (a proxy of soil erodibility) in potential mountain pine forests, and with forest cover in potential mesic oak forests (**Fig. 3a**). However, multifunctionality decreased with slope and forest cover in potential mesic and Mediterranean oak forests, respectively (**Fig. 3a**).

Multifunctionality increased faster with age in planted than in secondary forests in potential Mediterranean oak forests (see ΔAIC for the interaction *restoration strategy* × *forest age* in **Table 2**; **Fig. 3b**). In potential Mediterranean pine forests, secondary forests exceeded reference multifunctionality after 65 years and planted forest after > 80 years, assuming a constant trend (**Fig. 3b**). In potential mesic oak forests, secondary and planted forests exceeded reference multifunctionality after > 80 years, while in potential Mediterranean oak forests they exceeded reference multifunctionality after 41 and 43 years, respectively (**Fig. 3b**). We found no interaction between restoration strategy and forest age on multifunctionality in different contexts of abiotic or biotic conditions (ΔAIC ≤ 2 when three-way interactions were tested in **Table 2**).

Standardised biomass, woody species richness, functional dispersion and tree growth were higher, and recruitment of reference species was lower, in planted than in secondary potential Mediterranean oak forests (**Fig. 4**). The other multifunctionality components (coefficient of variation of tree height, cover of frugivore-dispersed shrubs and dead wood) were similar between planted and secondary forests.

## DISCUSSION

Restoration strategy determined forest restoration success, mainly in potential oak forests. Overall, active restoration did not accelerate forest recovery when compared to passive restoration. However, multifunctionality was higher and increased more rapidly in planted than in secondary forests in potential Mediterranean oak forests.

### Passive restoration is more successful than active restoration in potential oak forests

In accordance with our first hypothesis, secondary forests were more similar to references than planted forests in potential oak forests for all attributes except tree biomass. This could be partially explained because the dominant species in plantations is often different to the naturally dominant species. Wide use of oaks for forest restoration started in the 1990s in Spain, but the area of oak plantations is small compared to the pine plantations, widely established during the 20$^{th}$ century (Vadell et al., 2016). Moreover, the lack of thinning in pine plantations has resulted in high-density stands, which hinders the establishment of late successional plants (Gómez-Aparicio et al., 2009; Ruiz-Benito et al., 2012). Thus, most attributes were similar in reference and secondary forests where dominant species were the same. The exception was the tree biomass in planted forests that was more like references due to faster growth of pines compared to oaks.

The overall lack of differences between restoration strategies in potential pine forests, contrary to our first hypothesis, could also be explained by the wide use of pines in Mediterranean planted forests. Restoration strategy only affected woody species richness, functional identity of maximum height and tree growth. The higher richness in secondary forests may be related to the reduced plant colonisation in planted forests because some species are selected over others (Vadell et al., 2016), the site preparation techniques that might remove species (Maestre & Cortina, 2004), and high competition in dense forests (Gómez-Aparicio et al., 2009). The high functional identity of maximum height in planted forests in potential Mediterranean pine forests is probably due to the preferred use of *P. pinaster* (Vadell et al., 2016), which is taller than *P. pinea*, the other dominant species in these forests (Castroviejo et al., 1986-2012). Finally, tree growth was greater in planted than in secondary forests in potential mountain pine forests, where secular management in planted forest has maximised growth (Gil, Pardo, Velasco, & López, 2004).

**Speed of forest recovery does not depend on restoration strategy**

The speed of recovery of secondary and planted forests was generally similar. Forests planted by the end of the 19$^{th}$ century in the Mediterranean region and thinned later have shifted to mixed conifer-broadleaf forests (Carnus et al., 2006). However, in our study trait composition attributes remained constant along time in planted potential oak forests, suggesting slower forest recovery in central Spain. This can be due to: (1) our chronosequence is not long enough to observe significant changes (Vallauri et al. 2002); and (2) the lack of thinning in planted forests can constrain the facilitation of the canopy and the recovery of attributes such as plant richness (Battles, Shlisky, Barrett, Heald, & Allen-Diaz, 2001; Gómez-Aparicio et al., 2009).

Active restoration can speed recovery of some forest attributes in Mediterranean climate (second hypothesis). Specifically, ~~t~~he recovery of recruitment, biomass and functional identity of maximum height was accelerated in planted forests in potential Mediterranean oak forests. The increase of dispersal vectors and the nurse effect of tree canopy in planted forests (Gavinet et al., 2015; Sheffer, 2012) may cause recruitment acceleration (**Fig. 2i**). However, the harsh conditions under dense pine canopy are unsuitable for long-term oak development (Gavinet et al., 2015; Sheffer, 2012). The high growth of pines explains the faster biomass recovery in planted forests. Changes towards shorter trees would be the expected transition from planted pine to oak forests (Ruiz-Benito et al., 2017); however, it is not occurring

probably because younger planted forests that represent initial stages in the chronosequence had tree species with lower maximum height (i.e oaks) than older planted forests with pines (Vadell et al., 2016).

**Active restoration enhances Mediterranean oak forest multifunctionality**

Multifunctionality was higher and increased more rapidly in planted than in secondary forest only in potential Mediterranean oak forests. This difference between restoration strategies could be due to the relationships among the individual functions (Byrnes et al., 2014). For instance, tree biomass and growth are closely related attributes in early succession stages (e.g. Sheffer, Kigel, Canham, & Perevolotsky, 2014). Moreover, planted forests had greater tree biomass and growth than secondary forests. Thus, both attributes affected multifunctionality differently and as a joint set of variables according to the restoration strategy.

The influence of restoration strategy on multifunctionality did not change under different abiotic or biotic conditions. Thus, active restoration did not overcome the environmental limitations for multifunctionality recovery better than passive restoration (third hypothesis), even though multifunctionality was affected by restoration strategy, rainfall, soil erodibility and forest cover (see **Appendix S8. Supplementary discussion**). The selection of sites with relatively high initial degradation levels for active restoration projects could partially explain the similar multifunctionality outcomes of both restoration strategies and the fact that planted forests were less similar to references than secondary forests under a wide range of conditions (Reid, Fagan, & Zahawi, 2018). The magnitude of soil erosion was an important criterion in deciding where to allocate efforts for active restoration in Spain (Vadell et al., 2016). However, the simplification of the soil community in eroded sites may reduce plant diversity (Wagg, Bender, Widmer, & van der Heijden, 2014) and consequently the multifunctionality in planted forests (Hector & Bagchi, 2007; Meyer et al., 2018). Accordingly, we observed that maximum slope decreased multifunctionality in potential mesic oak forests, but not in other forest types (see **Appendix S8**).

### Multifunctionality vs. reference conditions

Reference forests showed an overall low multifunctionality. We propose three explanations: (1) the multifunctionality index may only capture a fraction of the entire forest multifunctionality (Manning et al., 2018), missing some functions typical of old forests, such as species habitat and resilience (e.g. Sutherland, Bennett, & Gergel, 2016); (2) reference forests may not maximise all functions because they are often monospecific in Mediterranean regions (Vilà et al., 2007), and there is evidence for a direct relation between multifunctionality and species diversity (e.g. Meyer et al., 2018); and (3) the Mediterranean basin lacks primary forests and the relatively old-growth forests have a legacy of millennia human-related perturbations (Scarascia-Mugnozza, Oswald, Piussi, & Radoglou, 2000). Thus, the use of relatively old-growth forests as references for restoring Mediterranean forests could be debated due to their low multifunctionality.

## SYNTHESIS AND APPLICATIONS

Our long-term assessment of forest recovery dynamics can guide future restoration projects in Mediterranean environments. The suitability of active vs. passive forest restoration depended on the restoration purposes (e.g. achieving similarity to reference forests, maximising certain functions of interest, or attaining target levels of multifunctionality), the potential forest type, and the desired time to achieve the restoration outcome. In potential pine forests, passive restoration allowed the recovery of forest attributes as fast as active restoration. In potential Mediterranean oak forests, however, active restoration promoted multifunctionality and a fast increment of biomass, but it did not recover the levels of some reference forest attributes.

Traditionally, native pines were planted in Spain, but since 1993 oaks and other native trees have been more commonly planted (Vadell et al., 2016; **Figures S1.2 and S1.3**). As a result, planted forests spread on 10% of the country (*c.* 5 million ha, Vadell et al., 2016). Extension of passive restoration is difficult to calculate, but we estimated it represents around 2/3 of the forest increase between 2000-2010 (from Vallejo, Torres-Quevedo, Robla, & Viejo, 2014). Our study captures both active and passive restoration strategies; however, 71% of analysed planted forests were monospecific (generally pine-dominated). Considering the dominance of pine plantations in active restoration in our study, we recommend that potential pine forests should be restored passively because it allowed the recovery of forest attributes at the same rate as active restoration, with no additional costs (active restoration costs are *c.* 2500 € $ha^{-1}$, Vadell et al., 2016). In potential oak forests, we recommend passive restoration to reach similar functioning as in reference forests, at the lowest cost. We suggest promoting active restoration where non-intervention may accelerate degradation (e.g. when soil erodibility is high or specific contaminants could be released), where there is no close natural forest remnants to ensure propagule supply (Andivia et al., 2017) or, in potential oak forests, if a rapid biomass accumulation or an increase in multifunctionality is needed. Our results suggest that planted forests have the long-term potential to reach references in potential Mediterranean oak forests, because the recruitment of reference species was accelerated, but costly stand thinning may be needed to shift to mixed conifer-broadleaf forests. Oak planting may be appropriate in planted potential mesic oak forests as we found that recruitment of reference species was not promoted. Future studies should target mixed plantations because mixing species promotes a broader range of functions (e.g. carbon storage; Ruiz-Benito et al., 2014) and species richness is a key driver of multifunctionality (Hector & Bagchi, 2007; Meyer et al., 2018).

**AUTHORS CONTRIBUTION**

All authors conceived the ideas and designed methodology; VCA and PBR collected and analysed the data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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**DATA ACCESIBILITY**

Data available via Figshare: https://dx.doi.org/10.6084/m9.figshare.7503860 (Cruz-Alonso et al., 2018a)

Tree functional traits available via Figshare: https://dx.doi.org/10.6084/m9.figshare.7503929 (Cruz-Alonso et al., 2018b)

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**Table 1.** Model summaries predicting the response ratio of 11 forest recovery attributes in each potential forest type. Response ratios were used to measure the extent of forest recovery. Model comparison was based on the Akaike Information Criterion (AIC).

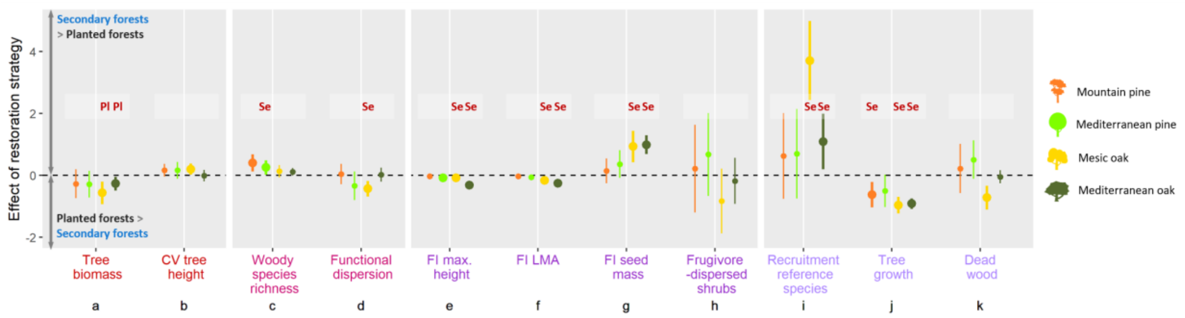
| Response ratio | Potential forest type | $\Delta AIC_{main-int}$ | Marginal $R^2$ | Conditional $R^2$ | n |
|---|---|---|---|---|---|
| Tree biomass | Mountain pine | -0.06 | 0.16 | 0.89 | 146 |
| | Mediterranean pine | 0.22 | 0.09 | 0.80 | 365 |
| | Mesic oak | -1.79 | 0.25 | 0.88 | 445 |
| | **Mediterranean oak** | **33.37** | 0.12 | 0.83 | 1507 |
| Coefficient of variation of tree height | Mountain pine | -1.96 | 0.11 | 0.47 | 146 |
| | Mediterranean pine | -1.90 | 0.02 | 0.68 | 303 |
| | **Mesic oak** | **3.24** | 0.05 | 0.57 | 398 |
| | Mediterranean oak | -1.86 | 0.02 | 0.45 | 1103 |
| Woody species richness | Mountain pine | -2.00 | 0.23 | 0.84 | 97 |
| | Mediterranean pine | 1.23 | 0.13 | 0.51 | 242 |
| | Mesic oak | -1.18 | 0.08 | 0.71 | 294 |
| | Mediterranean oak | -1.74 | 0.10 | 0.66 | 972 |
| Functional dispersion | Mountain pine | 1.37 | 0.05 | 0.68 | 146 |
| | Mediterranean pine | -1.53 | 0.04 | 0.81 | 364 |
| | Mesic oak | -1.87 | 0.09 | 0.85 | 444 |
| | Mediterranean oak | -0.70 | 0.02 | 0.78 | 1507 |
| Functional identity of maximum height | Mountain pine | -1.12 | 0.11 | 0.85 | 146 |
| | Mediterranean pine | -1.75 | 0.08 | 0.93 | 364 |
| | Mesic oak | -0.05 | 0.06 | 0.96 | 444 |
| | **Mediterranean oak** | **6.04** | 0.16 | 0.91 | 1507 |
| Functional identity of leaf mass per area | Mountain pine | -0.15 | 0.07 | 0.99 | 146 |
| | Mediterranean pine | -1.69 | 0.02 | 0.95 | 364 |
| | Mesic oak | -2.00 | 0.13 | 0.99 | 444 |
| | Mediterranean oak | -1.92 | 0.19 | 0.71 | 1507 |
| Functional identity of seed mass | Mountain pine | 1.42 | 0.06 | 0.99 | 146 |
| | Mediterranean pine | -1.62 | 0.02 | 0.90 | 364 |
| | Mesic oak | -0.43 | 0.24 | 0.98 | 444 |
| | Mediterranean oak | -0.62 | 0.16 | 0.88 | 1507 |
| Cover of frugivore-dispersed shrubs | Mountain pine | -1.57 | 0.06 | 0.67 | 86 |
| | Mediterranean pine | 1.31 | 0.10 | 0.72 | 209 |
| | Mesic oak | -2.00 | 0.08 | 0.80 | 253 |
| | Mediterranean oak | -1.41 | 0.07 | 0.64 | 931 |
| Recruitment of reference species | Mountain pine | -1.91 | 0.14 | 0.38 | 161 |
| | Mediterranean pine | -1.81 | 0.01 | 0.40 | 398 |
| | Mesic oak | -0.12 | 0.18 | 0.84 | 524 |
| | **Mediterranean oak** | **9.25** | 0.04 | 0.77 | 1795 |
| Tree growth | Mountain pine | -1.95 | 0.23 | 0.68 | 86 |
| | Mediterranean pine | -1.95 | 0.13 | 0.68 | 205 |
| | Mesic oak | -0.45 | 0.36 | 0.69 | 244 |
| | Mediterranean oak | -1.97 | 0.19 | 0.73 | 815 |
| Dead wood | Mountain pine | -0.90 | 0.20 | 0.44 | 89 |
| | Mediterranean pine | -1.78 | 0.05 | 0.06 | 231 |
| | Mesic oak | -1.66 | 0.28 | 0.47 | 262 |
| | Mediterranean oak | -1.65 | 0.03 | 0.41 | 894 |

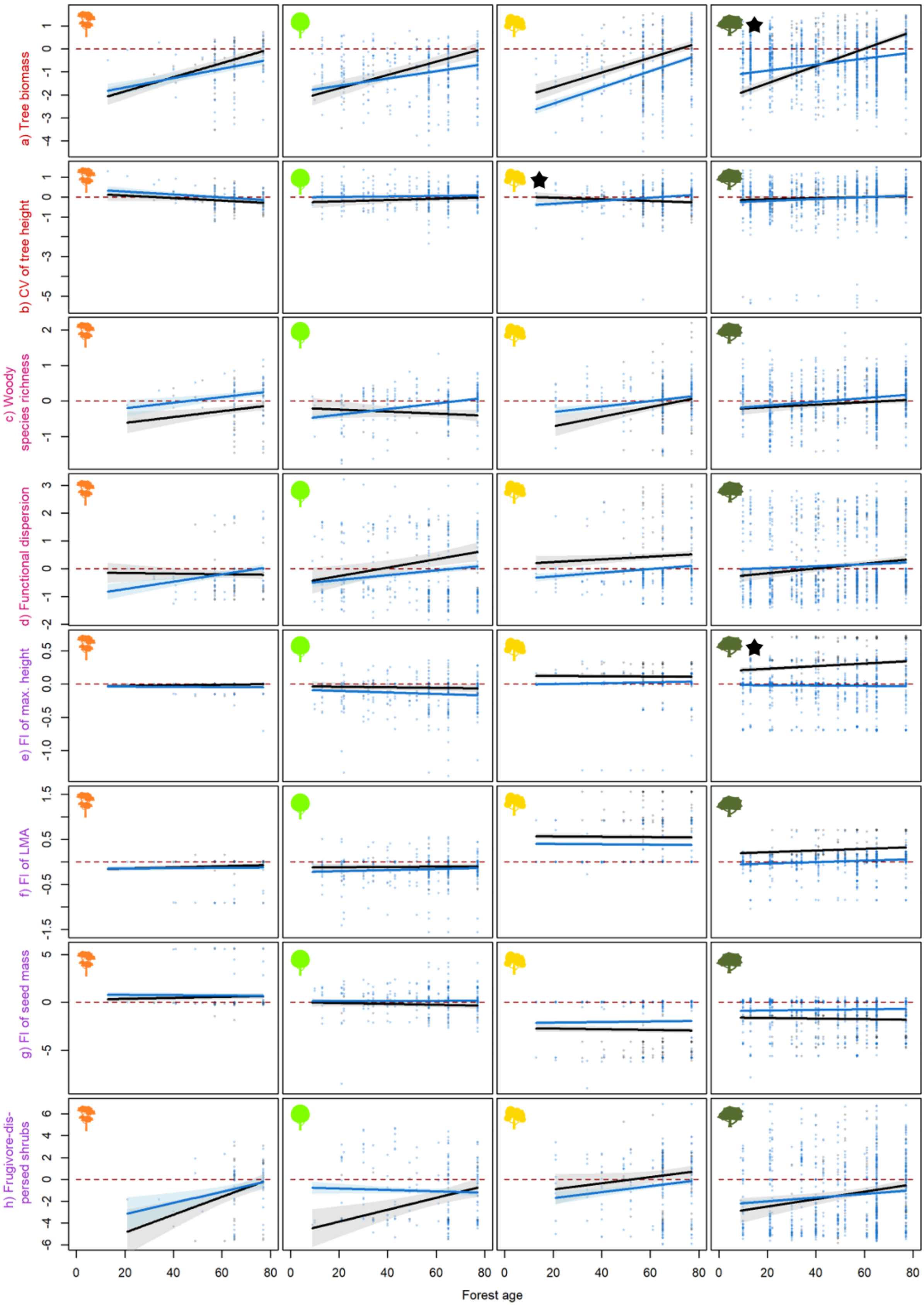
Main models include restoration strategy, forest age, rainfall, maximum slope, and forest cover. The interaction models (+ int) include the main effects and the pair-wise interaction between the restoration strategy and forest age. AIC comparisons ($\Delta AIC_{main-int}$) are the difference between each main model AIC and the alternate interaction model. When $\Delta AIC > 2$ (shown in bold), the interaction model is the best model and it supports the *restoration strategy x forest age* interaction, which suggests that the speed of recovery is different for each restoration strategy. For the best model we show the marginal and conditional $R^2$. n is the number of plots used in models that depend on each attribute and forest type.

**Table 2.** Comparison of alternative models of multifunctionality in four potential forest types, using the Akaike Information Criterion (AIC). Details are in the footnote.

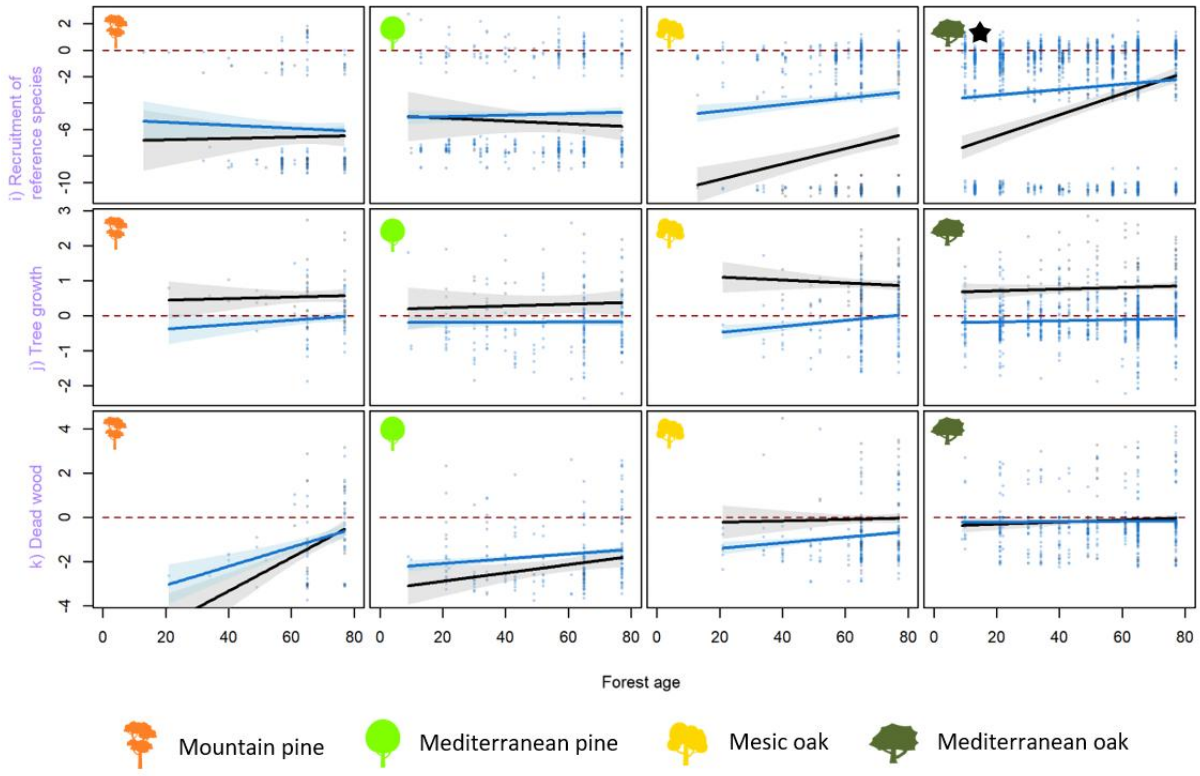
| **Potential forest type** | | Mountain pine | Med. Pine | Mesic oak | Med. oak |
|---|---|---|---|---|---|
| **n** | | 97 | 247 | 433 | 977 |
| **Main** | **Marginal $R^2$** | 0.07 | 0.11 | 0.04 | 0.06 |
| | **Conditional $R^2$** | 0.75 | 0.59 | 0.50 | 0.59 |
| **Interaction: restoration strategy × forest age** | **$\Delta AIC_{red\text{-}int}$** | -1.79 | -1.93 | -1.33 | **4.01** |
| | **Marginal $R^2$** | 0.07 | 0.13 | 0.05 | 0.07 |
| | **Conditional $R^2$** | 0.74 | 0.60 | 0.49 | 0.60 |
| **Interaction: restoration strategy × forest age × rainfall** | **$\Delta AIC_{red\text{-}int}$** | - | -3.68 | -6.37 | -5.23 |
| | **Marginal $R^2$** | - | 0.13 | 0.05 | 0.07 |
| | **Conditional $R^2$** | - | 0.60 | 0.49 | 0.60 |
| **Interaction: restoration strategy × forest age × max. slope** | **$\Delta AIC_{red\text{-}int}$** | - | -2.97 | -4.80 | -1.86 |
| | **Marginal $R^2$** | - | 0.13 | 0.05 | 0.07 |
| | **Conditional $R^2$** | - | 0.60 | 0.49 | 0.60 |
| **Interaction: restoration strategy × forest age × forest cover** | **$\Delta AIC_{red\text{-}int}$** | - | -5.52 | -5.33 | -1.47 |
| | **Marginal $R^2$** | - | 0.11 | 0.05 | 0.07 |
| | **Conditional $R^2$** | - | 0.59 | 0.50 | 0.60 |

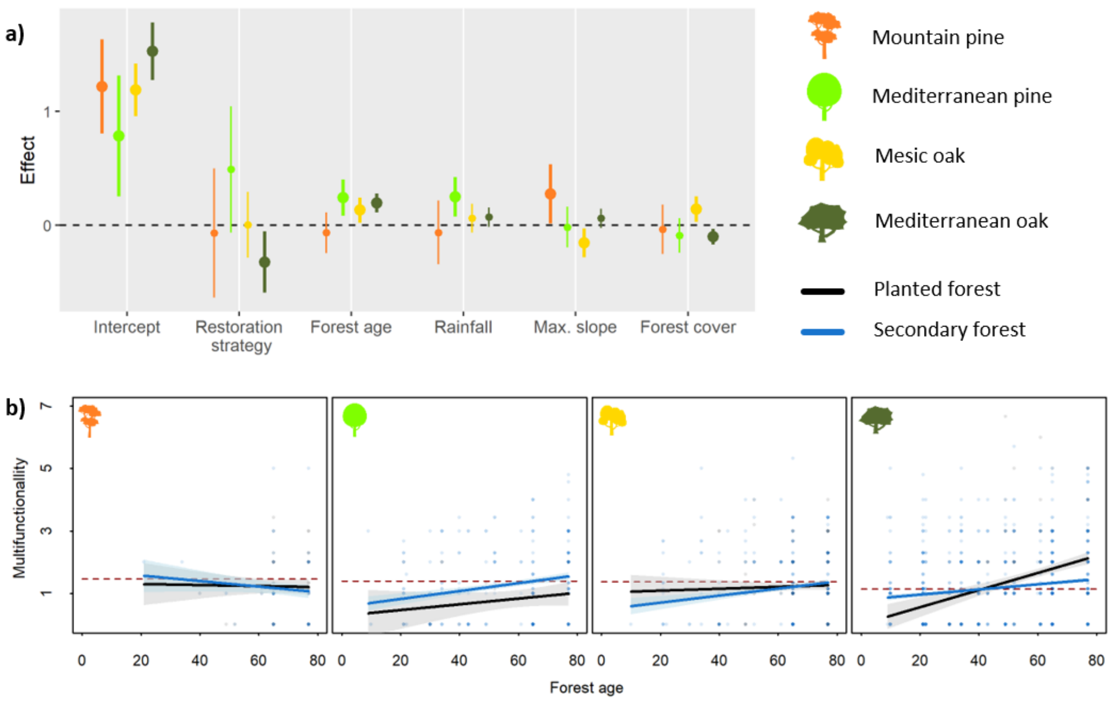
Main model includes the restoration strategy, forest age, rainfall, maximum slope, and forest cover. The interaction models include the main effects and the specified interaction in each case. Model comparison was based on $\Delta AIC$, and $\Delta AIC_{red\text{-}int}$ is the difference between the AIC of the best reduced model (without the specified interaction) and the full model (interaction model). When $\Delta AIC > 2$, the interaction model is the best model (shown in bold). For each model we show marginal and conditional $R^2$. n is the number of plots used in models that depend on each forest type.

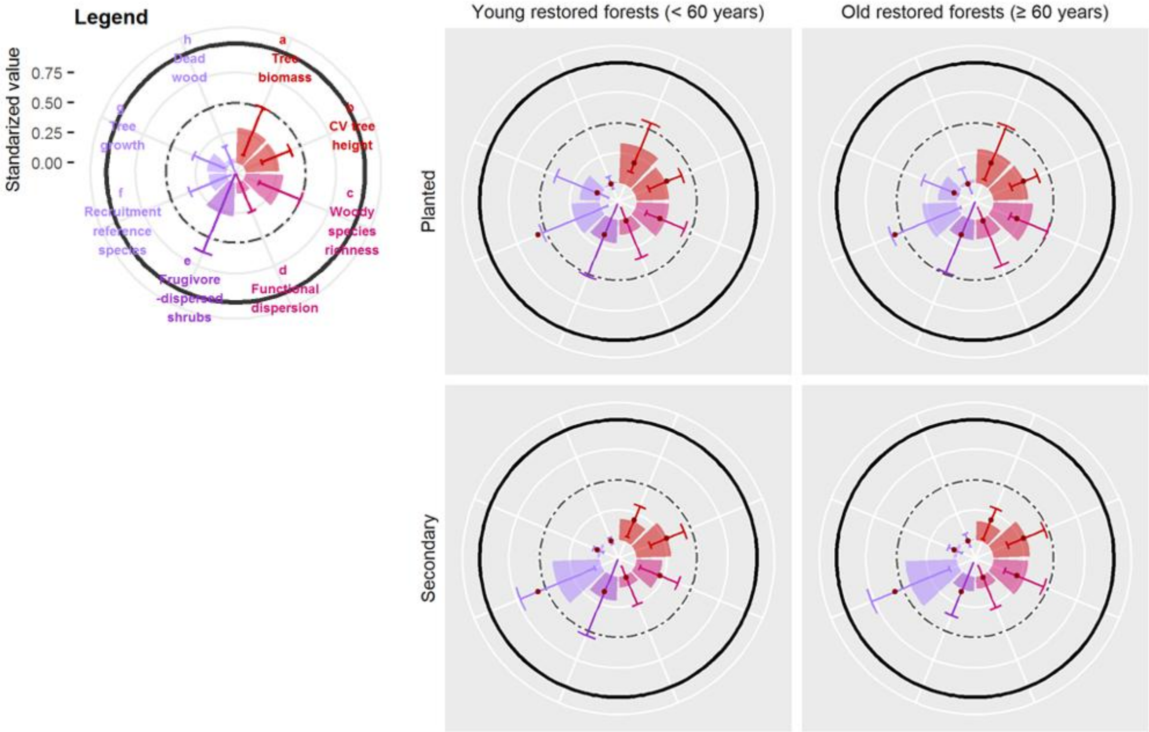




a) Tree biomass
b) CV of tree height
c) Woody species richness
d) Functional dispersion
e) FI of max. height
f) FI of LMA
g) FI of seed mass
h) Frugivore-dis-persed shrubs
0 20 40 60 80
Forest age


i) Recruitment of reference species
j) Tree growth
k) Dead wood
Forest age
Mountain pine
Mediterranean pine
Mesic oak
Mediterranean oak


a)
Effect
Intercept
Restoration strategy
Forest age
Rainfall
Max. slope
Forest cover
Mountain pine
Mediterranean pine
Mesic oak
Mediterranean oak
Planted forest
Secondary forest
b)
Multifunctionality
Forest age



Legend
Standarized value
0.75
0.50
0.25
0.00
a
Tree
biomass
b
CV tree
height
c
Woody
species
richness
d
Functional
dispersion
e
Frugivore
-dispersed
shrubs
f
Recruitment
reference
species
g
Tree
growth
h
Dead
wood
Young restored forests (< 60 years)
Old restored forests (≥ 60 years)
Planted
Secondary