

Survival vs. growth trade-off in early recruitment challenges global warming impacts on Mediterranean mountain trees

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Abstract

Climate change is expected to alter the geographic distribution of many plant species worldwide. However, there is still no clear evidence showing a generalized direction and magnitude of these distribution shifts. Here, we have surveyed, in nine mountainous regions in Spain, an array of tree species along entire elevational ranges, as surrogates of their global climatic ranges, to test for elevational shifts towards cooler locations. We analysed the distribution recruitment patterns of five dominant tree species, recording the abundance and measuring the primary growth of juveniles in 306 plots. Three of the species have a temperate-boreal distribution with populations at their southern edge in the Mediterranean mountain ranges: *Pinus sylvestris*, *P. uncinata* and *Fagus sylvatica*; and the other two species have a Mediterranean distribution: *Quercus ilex* and *P. nigra*. Despite the contrasting phylogenies and biogeographies, we identified a similar pattern in recruitment abundance across species, with an asymmetric distribution of juveniles (more recruits in the middle-upper elevation of their range), but higher annual growths at lower elevations. This survival-growth trade-off at the early recruitment stage may potentially counterbalance at population level the negative effect of global warming on recruit survival at the lower edge of species ranges. These findings suggest a demographic stabilisation process at the early recruitment stage of these tree species, and highlight the importance of considering the different demographic stages across the whole climatic range to understand the effects that climate change may exert on species distributions and population dynamics.

47 **Key words:** climate change; climatic gradient; demographic stabilising processes;
48 elevational shift; forest regeneration; Mediterranean forests; species distribution.
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Abbreviations: Sampled transects' position along elevational gradients: U: upper, MU: mid-up, M: middle, ML: mid-low, L: lower transects. Se: seedlings emerged during the current growing season; S1: saplings from one to five years; S2: saplings over five years.

Introduction

An increasing number of studies are showing latitudinal and elevational shifts of many species in response to climate change (Parmesan and Yohe, 2003; Rabasa et al., 2013). Although changes in traditional land uses contribute to some of these species distribution shifts (Peñuelas and Boada, 2003; Batllori and Gutiérrez, 2008; Ameztegui et al., 2010), most of the studies have suggested that global warming and the increased frequency of extreme climatic events are the main causes of gradual upward and poleward movements and consequent changes in the species composition of ecosystems (Wardle and Coleman, 1992; Peñuelas and Boada, 2003; Thuiller et al., 2005; Beckage et al., 2008; Bertrand et al., 2011b; Benavides et al., 2013; Matías and Jump, 2014). However, this evidence runs in parallel with other studies showing a lack of species distribution changes (Lenoir et al., 2009, 2010), species that have increased their cover and/or abundance with rising temperature or aridity (Wang et al., 2006; Gimeno et al., 2012) or even that have experienced a downslope range displacement (Lenoir et al., 2010; Crimmins et al. 2011; Rabasa et al., 2013). Lloret et al. (2012) shed some light on these apparent contradictions suggesting that some stabilising demographic processes may mitigate or compensate the negative effects in plant communities induced by extreme climate events (Table 1). Within the mitigation category, local factors (both biotic, like facilitation, and abiotic, like physiography or site quality) may provide microrefugia and attenuate the direct negative effects of a changing climate on individuals (Pearson and Dawson, 2003; Araújo and Luoto, 2007; Dobrowski, 2011; McLaughlin and Zavaleta, 2012). Moreover, species-specific characteristics related to their adaptation or acclimation ability may also buffer the expected climatic impacts on populations (Aitken et al., 2008; Nicotra et al., 2010; Richter et al., 2012). Accordingly, recent studies have suggested that the direct impacts of global warming may be smaller

than predicted, and have shown a high degree of homeostasis in different processes of some species in response to increasing temperatures (Gunderson et al., 2011; Baldi et al., 2012). Within a second category of compensatory processes, Lloret et al. (2012) included the beneficial effect that warmer temperatures may have over some life cycle stages and ecological processes. For instance, an increase in growth with temperature may compensate a lower seedling survival (Doak and Morris, 2010); or reduce current competitive or antagonistic interactions (McDowell et al., 2006; Carnicer et al., 2011), and increase mutualistic relationships (Le Conte and Navajas, 2008; Giménez-Benavides et al., 2011).

The mismatch between both types of observational outcomes (shift *vs.* no-shift) may not only reflect the species-specific characteristics, but also differences in the spatial and temporal study scales. Undoubtedly, some species are more vulnerable to environmental changes and will respond earlier than others. Nevertheless, the studies showing shifts have been frequently conducted considering a single species or population, or have been focused on changes occurring at the edge of species ranges, being then unable to reveal the whole picture by neglecting the overall distribution of the species (but see these studies with waterbird species -Lehikoinen et al., 2013- and stream fish species -Comte and Grenouillet, 2013-). Thus, it is a priority to conduct more comprehensive studies in which a set of species are simultaneously surveyed at several spatial scales, covering their whole distribution range (Lenoir and Svenning, 2015), and including different life stages (*i.e.* including the regeneration niche, see Grubb, 1977). This will shed light on the demographic mechanisms and trade-offs involved in the shift *vs.* no-shift responses, and will help to identify global responses to current warming and design strategies to mitigate its impact over plant communities.

In this study, we contribute to this goal and we analyse the distribution of the recruitment (seedlings and saplings), in terms of abundance and primary growth, of four dominant tree species, surveying their entire elevational range in Mediterranean mountains, as a surrogate of the climatic range that each species experiences along their latitudinal distributions. Using elevation as a surrogate for climate range is a useful tool to study responses under different climatic scenarios within a single population and, therefore, minimising differences due to genetic variability, compared to surveys carried out in large areas, and therefore, including different populations (Ruíz-Benito et al., 2012; Vayreda et al., 2013; Carnicer et al., 2014). Three of the target species have a temperate-boreal distribution with populations at their global southernmost limit in the Iberian mountain ranges: Scots pine (*Pinus sylvestris* L.), mountain pine (*Pinus uncinata* Ramond ex DC.) and European beech (*Fagus sylvatica* L.); and the other one has a Mediterranean distribution: Iberian black pine (*Pinus nigra* Arn. ssp. *salzmannii* (Dunal) Franco). Another Mediterranean dominant tree species was included in the target species pool of this study due to its great relevance in the Iberian Peninsula: holm oak (*Quercus ilex* L.). However, this species was not surveyed along its entire elevational range and its trailing edge was left out, because this species has no low elevational limit in the Mediterranean mountains. Our main aim is to test in the study populations whether there is an overall, pervasive elevational shift of the young cohorts towards cooler locations within the entire elevational range of the adult, which would be transferable to poleward shifts in response to the ongoing climate change, or if the altitudinal patterns are species-specific (Lenoir et al., 2009; Woodall et al., 2009). We expect that the recruits of the Mediterranean species, better adapted to summer drought, would be less affected by the ongoing climate change and be more abundant near their central part of the elevational range, where the density of conspecific adults and, therefore, seed rain are greater than at

the edges of their range. By contrast, we expect that the species with populations at their trailing edge of their distribution, therefore marginal populations, would show shifts of their optimum juvenile abundance towards higher elevations (Matías and Jump, 2014). This outcome would show the different climatic conditions imposed by the global warming between the conditions when the current adult stand established, and the present situation. Hence, we hypothesise an idiosyncratic response of the different species with no general upward trend, and where a potential compensatory process enhancing the performance of the juveniles under warmer conditions may be relevant, especially for those species whose populations are at their southern limit.

Material and Methods

Study sites

The study was conducted in nine mountain ranges across Spain during the early summer in 2010 and 2011 (Fig. 1). The entire sampling required two years due to the short lapse of time between the emergence of most of the seedlings and the onset of the summer drought. For every species we sampled two natural pure stands located in different mountain ranges (three in the case of *P. sylvestris*), making a total of 11 study sites (see Supporting Information, Appendix Table A.1). These stands showed neither visible signs of being planted, nor any evidence of management that could affect the current recruitment. Within each site, five transects were established at five elevational levels (Fig. 1). The upper (U) and the lower (L) transects were placed in the ecotones of each target species with the adjacent communities, and considered front and rear edges of their range respectively. In the majority of the cases, U transects were located at the treeline (*i.e.* transition from the uppermost closed forests to the treeless mountainous vegetation). The three intermediate levels: mid-low, middle and mid-up levels (henceforth ML, M,

and MU respectively) were evenly located within the elevational range of each species, with at least 100 m difference in elevation among them, being ML and MU located near the boundary with the ecotones (Fig. 1). In the case of *Q. ilex*, we only considered the three superior levels since this species has no low elevational limit in the Iberian Peninsula. Therefore, there was a total of 51 transects sampled ($[3 \text{ species} * 2 \text{ sites} * 5 \text{ transects}] + [1 \text{ species} * 3 \text{ sites} * 5 \text{ transects}] + [1 \text{ species} * 2 \text{ sites} * 3 \text{ transects}]$).

Recruitment sampling and environmental variable recording

We established six plots (4 x 4 m) within each transect, that is 306 plots in total (51 transects * 6 plots), separated by at least 100 m (Fig. 1). Three plots per transect were haphazardly located under fully canopied stands, and the other three in small forest gaps to include contrasting conditions within each forest and elevation, avoiding ravines, boulders, cliffs or other geomorphologic elements which could bias our measurements ameliorating the effect of climate by local physiography (Dobrowski, 2011). In every plot we recorded the number of recruits by counting the seedlings recently emerged and saplings (up to 1.50 m tall). Seedlings and saplings of resprouting species (*Q. ilex* and *F. sylvatica*) were correctly identified from vegetative offsprings following Espelta et al. (1995). We estimated the age of the recruits by counting the terminal bud scars (internodes) along the main stem and we sorted them into three different age classes (Espelta et al., 1995), assuming a different sensitivity to the environment (Collins and Carson, 2004): Se: seedlings emerged during the current growing season and without bud scars; S1: saplings from one to five years, that had thus experienced at least one summer drought and one winter; and S2: saplings over five years. The age of the individuals included in the last category was not fully dated due to the difficulty with distinguishing scars in the field as individuals grow and become woodier, but it was verified that they

exceeded 5 years. We also measured the primary shoot growth in previous years (growths occurred in 2007, 2008 and 2009) and averaged them, as a proxy of their individual performance. We identified the current shoot (not considered for the study because the growing season was not finished when the sampling took place) and measured the length of the previous internodes along the main stem not damaged by herbivores. The existing literature demonstrated that shoot growth is a good indicator of environmental favourability (Willms et al., 1998), as well as the impact of drought conditions on plant growth (Mutke et al., 2003; Peñuelas et al., 2004; Thabeet et al., 2009).

Some biotic and abiotic environmental variables were assessed in each plot. All adult trees within a 10 m radius around the regeneration plot were mapped and their diameter at breast height -dbh- was measured. Afterwards we calculated the basal area (both considering every adult tree and only the conspecific individuals) and tree density of adult trees of the target species in each plot. Finally, we translated the elevation and latitude of every plot (through the GPS coordinates) into climatic variables using the Climatic Digital Atlas of the Iberian Peninsula (Ninyerola et al., 2005) (see Appendices Table A.1. and Figure A.1.).

Data filtering and statistical analysis

The abundance of the individuals of each species and age class along the elevation and temperature ranges was compared using the Kruskal-Wallis test. However, as the amount of seedlings (Se) varies greatly among years, we considered only the saplings (S1 and S2) for further analyses, *i.e.* survivors after the first summer drought, which is considered the main bottleneck for recruitment in Mediterranean areas (Jordano and Herrera, 1995; Castro et al., 2004; Giménez-Benavides et al., 2007; Linares and Tíscar, 2010).

We analysed the variation of the regeneration pattern along the elevational gradient. Hence, we compared our response variables along the five elevational levels (fixed variable) using Generalized Linear Models (GLM; McCullagh and Nelder, 1989) and Generalized Linear Mixed Models (GLMM; Breslow and Clayton, 1993). For the response variable *abundance of saplings* per species and at plot level (age class was neglected due to the low figures in some transects of these count data), we run GLMs and assumed a negative binomial distribution and a log link function. We selected this distribution after noticing the presence of over-dispersion (McCullagh and Nelder, 1989; Hoef and Boveng, 2007). In addition to *elevational levels*, two covariates describing the structure of the adults within a plot, therefore related to the seed rain, were considered to fit the models: *tree density* and *mean dbh* of conspecific adult individuals (Vayreda et al., 2013; Matías and Jump, 2014). Finally, the variable *site* was treated as a fixed factor due to the low number of levels (two/three for each species) to be considered a random effect (Crawley, 2002; Bolker et al., 2008).

The other response variable, the *individual mean growth* per species and age class along elevational levels was log-transformed and analysed assuming a normal distribution and an identity link function, using a mixed approach (GLMM). This time the sample size (data at individual level) allowed us to consider the age class, and a model for each age class was run separately. Light availability is one of the best predictors for saplings' growth (Gómez-Aparicio et al., 2008; Benavides et al., 2013). Therefore, this time the total *basal area* of the plot was included as a covariate, considered as an index of competition for light with adult trees (Urbieta et al., 2011). Concerning the random variables, we considered the hierarchical structure of the data, with likely different level of correlation among observations from the same *site* and *plot* (nested design).

To analyse the overall trend in the regeneration pattern across the different elevational levels, and make the data from different species comparable, we standardised the abundance and growth values per species, *i.e.* we subtracted from each value the species averaged abundance or growth, respectively, and divided it by the standard deviation. Then, we pooled the data and compared them along the elevation gradient (levels) using a GLMM. Similar to the analyses conducted per species, for abundance data we included the covariates *density* and *mean dbh*. This time *site* was considered a random variable (with 11 levels), and we assumed a gamma distribution and a log link function (after adding two units to the standardised data to avoid negative values). For growth data we assumed a normal distribution, and we considered *basal area* as a covariate, and *plot* within *site*, as random factors.

We tested for significance of fixed effects comparing models with likelihood ratio tests, and we removed the covariates if they did not significantly improve the fits. The estimation of the parameters was done through Maximum Likelihood analyses for the GLMs, and for comparison between models with the same random structure (GLMMs), and through the Restricted Maximum Likelihood approach for assessing the parameters of the final selected GLMMs.

Finally, we implemented the Tukey's post-hoc test (Games and Howell, 1976) to examine significant differences in abundance and growth along the elevational levels for every single species and globally (*i.e.* for pooled standardised data). The GLMs and GLMMs were performed using packages MASS (Venables and Ripley, 2002), nlme (Pinheiro et al., 2013) and stats (R Team 2014) in R (R Core Team, version 3.0.2, 2013).

Results

Distribution of juveniles along the elevation

We recorded a total amount of 9310 juveniles in the 306 plots, sorted into 5283 juveniles of *P. sylvestris*, 547 of *Q. ilex*, 1143 of *P. nigra*, 740 of *P. uncinata*, and 1597 of *F. sylvatica*. The distribution of juveniles by age class (Fig. 2) along the elevation and temperature gradients showed a shift towards higher elevations within the elevation range of the adult individuals (therefore, towards cooler temperatures) of the saplings (particularly S1) of *Q. ilex* and pines, compared to the elevation at which most of the seedlings emerged. By contrast, *F. sylvatica* recruits occurred mainly in the lower areas (L), although there was no significant difference among the distributions of the different age classes.

Abundance and growth of saplings along elevational levels

Removing the seedlings (Se), the total amount of saplings found in the 306 plots were 3191, sorted into 1367 saplings of *P. sylvestris* (surveyed within 90 plots), 543 of *Q. ilex* (within 36 plots), 564 of *P. nigra* (within 60 plots), 608 of *P. uncinata* (within 60 plots) and 109 of *F. sylvatica* (within 60 plots).

At tree species level (analyses per species), the GLM showed a similar trend for abundance data of saplings of all species along the elevational levels, with a significant effect but for *P. nigra* (Table 2; Fig. 3). The U and L levels had the lowest saplings abundances. Focusing on the three intermediate elevational levels, all of them showed higher figures in MU than in the ML, with a significant difference for *P. sylvestris* and *F. sylvatica*. *Pinus uncinata* and *Q. ilex* had its maximum abundance of saplings at the MU level, showing an abrupt decrease between MU and U levels. Considering pooled data (global analysis), the estimates of the standardised mean abundance of saplings showed a

significant difference among elevational levels as well (Table 2; Fig. 3), with the highest number of saplings appearing in the M level, and followed by MU. These numbers were significantly higher than those found at the ecotones (L and U), which accounted for the lowest figures.

Concerning the covariates included in the models for abundance estimation (Table 2), significant differences among sites hosting the same species appeared, but for *F. sylvatica*. *Tree density* did not appear as an important factor affecting saplings' abundance (except for *P. nigra* with a negative effect), while the mean *dbh* of the adults affected positively the abundance of species with resprouting ability, *i.e.* *F. sylvatica* and *Q. ilex*.

Analyses run with mean annual growth data showed a clear decrease in annual growth for younger saplings (S1: from 1 to 5 years) with elevation, particularly when comparing L and ML levels with higher ones, both considering data per species and pooled standardised data. For older juveniles (S2: over 5 years), there was a global slight increase upslope (Table 3; Fig. 4), but there were large differences in the trends of the different species. In addition, a generalized negative effect of the *basal area* appeared affecting annual growth (Table 3), although not significant for every species, and mainly affecting the old saplings (older than 5 years).

Discussion

Opposite to our hypothesis, and despite the contrasted phylogeny and biogeography of the study species, we identified common patterns in recruitment abundance and growth along climatic gradients for these species. This result is especially relevant when it is considered that recruitment is very dependent on local processes and hence, highly site-dependent (Gómez-Aparicio, 2008; Matías et al., 2011; Ameztegui and Coll, 2013; Benavides et al., 2013). Therefore, this outcome shows the importance that climate exerts as one of the main drivers for seedlings' survival and ingrowth rate (Urbieto et al., 2011; Vayreda et al., 2013; Carnicer et al., 2014).

Sapling abundance exhibited a common asymmetric pattern in pine saplings independently of their biogeographical origin, with more saplings in the middle-upper than in middle-lower elevations. As expected, density and basal area of conspecific adults were higher in the three intermediate transects than in the ecotones (Fig. 2; Appendix Figure A.1.), in agreement with the number of saplings. Within these three elevational levels, it is remarkable that, despite the higher abundance of recently emerged seedlings near or even below the middle range transects, sapling abundance was skewed to higher elevations (lower temperatures). Lacking historical records, we cannot completely dismiss that this ontogenetic shift simply renders different requirements for emergence vs. seedling survival (Werner and Gilliam, 1984), regardless of the ongoing climate changes. However, we assumed that this pattern suggests that seedling survival is greater at higher elevations where heat- and drought-stresses are typically lower compared to seedling emergence, supporting the current expectations in response to climate warming (IPCC, 2007; Hughes, 2000). Nevertheless, we recorded the lowest densities at the uppermost transects suggesting that recruitment up there may be hampered, likely, either by climatic variables (not exclusively temperature and precipitation, but also radiation,

freezing events, wind, etc.) and other local conditions. Consequently we cannot prove an expansion of the current range limits of the study species beyond their current boundary (upward migration) as other studies did (Peñuelas and Boada, 2003; Peñuelas et al., 2007; Batllori and Gutiérrez, 2008; Jump et al., 2012; Matías and Jump, 2014). In other words, our results agree with an upslope movement of central tendencies of the study species distributions but we had no evidence for an extension of the species ranges upslope (Breshears et al., 2008; Kelly and Goulden, 2008; Lenoir et al., 2008, 2009). Such a statement would have required an additional study focused at the ecotones comparing the sapling abundance of the target species with those from competing species.

The results from both *Fagaceae* species were a bit less clear. The elevation range of *Q. ilex* sampled in this study only covered the upper edge of its distribution, exhibiting again a decrease of the overall sapling abundance in the U transect. However, looking at the elevational distribution of young saplings (under five years), they were more abundant near the upper ecotone compared to the older saplings, similar to the distribution of pines. This outcome agrees with studies carried out in Spain using data from the National Forest Inventories, showing an upwards shift of *Q. ilex* recruits towards cooler areas, especially relevant in stands where this species adjoins pine forests upslope, a situation that seems to facilitate oak seedlings success (Urbieto et al., 2011; Vayreda et al., 2013; Carnicer et al., 2014). On the other hand, *F. sylvatica* showed an unexpected high density of juveniles at the lower ecotone in our study. This pattern did not match with previous studies conducted in north-eastern Spain, which evidenced an upslope shift of this species during the last century with a consequent species replacement downslope (Peñuelas and Boada, 2003; Peñuelas et al., 2007). We posit that other reasons not linked with the species response to climate change may underlie this unexpected trend. In fact, the difference in the stand structure among transects may be interfering with the likely effect of the global

warming over the recruitment pattern in the short term. Indeed, the lower transects of these forests showed low densities of bigger adult trees (Appendix Figure A.1.), which means stands with less multi-stemmed individuals. This result matches with the positive effect that *dbh* exerted over the abundance of saplings, likely related to the higher vigour on seed production of one-stemmed individuals from the lower levels compared to the multi-stemmed individuals resulting from old coppicing (Rodríguez-Calcerrada et al., 2011).

Moreover, our study reveals opposite trends between recruitment abundance and performance across the elevational range suggesting a potential stabilising process at the population level. Unlike the sapling abundance pattern, mean growth of young saplings (S1) of every target species decreased with increasing elevation. This cohort is known to be more susceptible, not only to the climate and especially to extreme climatic events (Collins and Carson, 2004; Castro et al., 2005), but also to the effect of mammalian herbivores, which can impede regeneration in Mediterranean mountains (Zamora et al., 2001; Herrero et al., 2012). The pattern we found revealed more favourable conditions in warmer places for early recruitment performance, showing the positive role that warm temperatures may exert over growth even in relatively dry Mediterranean areas, when appropriately distributed along the year regarding the annual rainfall distribution, (see Herrero et al., 2013 for secondary growth in adult trees). Similar findings were reported by Doak and Morris (2010) studying the performance of two tundra species along their latitudinal range. They found higher growth rates at the southern edge, despite a lower survival rate of juveniles, suggesting as well a demographic compensatory effect in response to the climate shifts with latitude. Several studies have already reported evidence on the positive correlation between sapling size and their survival, including studies carried out in Mediterranean environments (Cuesta et al., 2010; García-Camacho

et al., 2012; Villar-Salvador et al., 2012). Therefore, these higher individual growths of young saplings (more vulnerable) provide more chances to survive because they can overcome quicker this critical stage in terms of climate sensitivity (Pfister 1998; Doak and Morris, 2010) and damage from browsers (Zamora et al., 2001), counterbalancing at the population level the negative effect of global warming over early survival at the trailing edges.

Nevertheless, this pattern (higher growths downslope) disappeared as saplings grew, showing again an ontogenetic shift between both cohorts (Werner and Gilliam, 1984), rendering into an idiosyncratic response of juveniles over five years among species and transects. Indeed, several studies have already proved that the niche may change over life cycle due to the different requirements and tolerance to environmental conditions (Miriti, 2006; Quero et al., 2008; Soliveres et al., 2010; Bertrand et al., 2011a). This outcome firstly warns about drawing general conclusions about the impact of global warming on plant populations without considering each particular life stage (Grubb, 1977). Moreover, the results hint that once the early seedlings are recruited, the climatic impact on the demography may be somehow attenuated by other factors that become more relevant at a local scale, like the basal area that is considered a proxy for available light. Accordingly, previous studies have supported the non-random effect that local environmental conditions (biotic and abiotic factors) exert over the juveniles' growth, including light availability (Gómez-Aparicio et al., 2008; Matías et al., 2011; Benavides et al., 2013) and plant-plant interactions (Davis et al., 1999; Ameztegui and Coll, 2013; Benavides et al., 2013). In other words, fine scale factors acting hierarchically may exacerbate (for instance, at southern, dry exposures) or mitigate (for instance, by facilitation) the impacts of the changing climate (Lloret et al., 2012).

We conclude that our results across entire elevation ranges of different species suggest the existence of a demographic compensation during the early recruitment stage in relation to the global warming impacts. Although in lower (warmer) areas fewer seedlings of different species overcome the first summer drought (*i.e.* main filter in Mediterranean areas), those that survive grow more than those at higher altitudes and would likely move out faster from this highly vulnerable life stage. So, we might expect that the net result turns out in certain stability at the population level, as long as climate change does not exceed certain thresholds. Thus, challenging opportunities arise to explore the persistence and relative importance of these stabilising processes along larger temporal scales, which would allow us to infer how these patterns will evolve as juveniles become adults (*i.e.* paying special attention on the temporal variation of ontogenetic niches shifts), and the role that local factors and management can play on population dynamics under climate change.

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Figure captions

Figure 1. Sample design: a) site locations, b) diagram of the transects' layout within a site, and c) display of the plots layout within a transect. Two sites for every species were sampled. Within each location, five transects were established at five elevations: upper (U), mid-up (MU), middle (M), mid-low (ML) and lower (L). MU, M and ML are transects located within the elevational range of a target species in a pure stand, while U and L are located at the ecotones with other communities. Within each transect six plots were established, three of them under fully canopy and the other three in gaps.

Figure 2. Distribution of juveniles along the elevational and climatic gradients (mean annual temperature) of each species by age class, and mean values (standard error) of the plot basal area assessed considering the conspecific adult individuals at the different elevational transects in each type of forest. Age classes were: Se: recently emerged seedlings; not shown for *Q. ilex* due to the small sample size (n=4); S1: saplings from 1 to 5 years; S2: saplings over 5 years. Different letters indicate significant differences among the medians of different age classes along the elevational gradient (Kruskal-Wallis test). Dotted lines show the surveyed elevational (climatic) range and the continuous black line the mean elevation (temperature) of the elevational range. Mean annual temperatures were obtained introducing the coordinates of each plot in the Climatic Atlas of the Iberian Peninsula (Ninyerola et al. 2005). U, MU, M, ML, L mean upper, mid-up, middle, mid-low and lower transects respectively

Figure 3. Estimates of the mean abundance of juveniles according to the elevational level. It was used Generalized Linear Models for the individual species analyses and Generalized Linear Mixed Models for the global analysis (in the latter case the standardised abundance -Std- was used). Thin gray lines show the 95% confidence

interval of the estimated means. Elevational transects within every site are (L) lower, (ML) mid-low, (M) middle, (MU) mid-up and (U) upper. Different letters (at the right bottom part of each panel) show the significant differences and *ns* means non-significant differences among elevational levels.

Fig. 4. Estimates of the mean growth of saplings across the elevational range of pooled data and per species. Analyses run using Generalized Linear Mixed Models, and analysis of pooled data used standardised data -Std-. Thin gray lines show the 95% confidence interval of the estimated means. Elevational transects within every site are (L) lower, (ML) mid-low, (M) middle, (MU) mid-up and (U) upper. Saplings' age class: S1: 1-5 years; S2: >5 years. Different letters (legend) show the significant differences, and *ns* means non-significant differences, among estimated means among elevational levels.

Figure1

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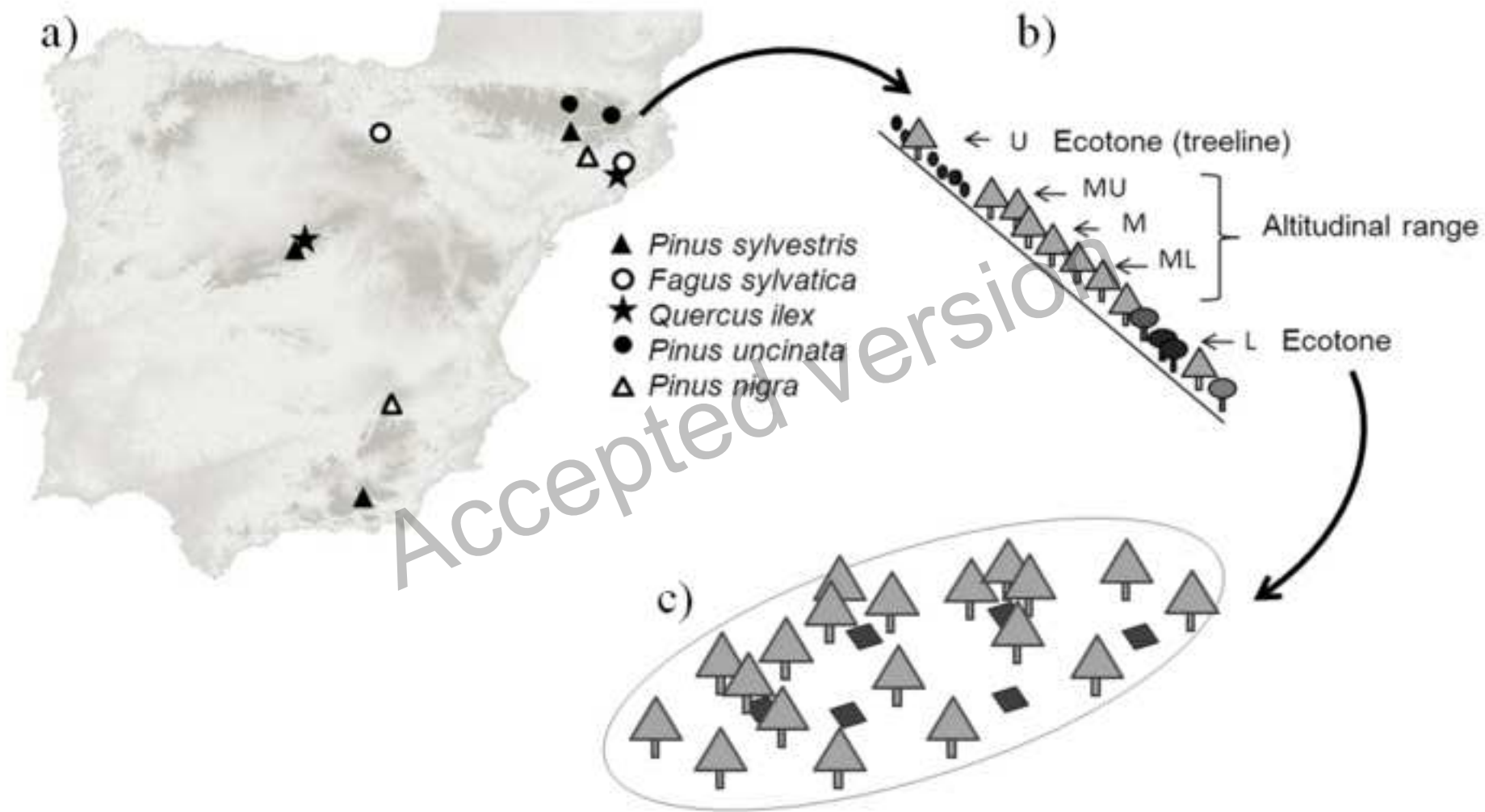


Figure2
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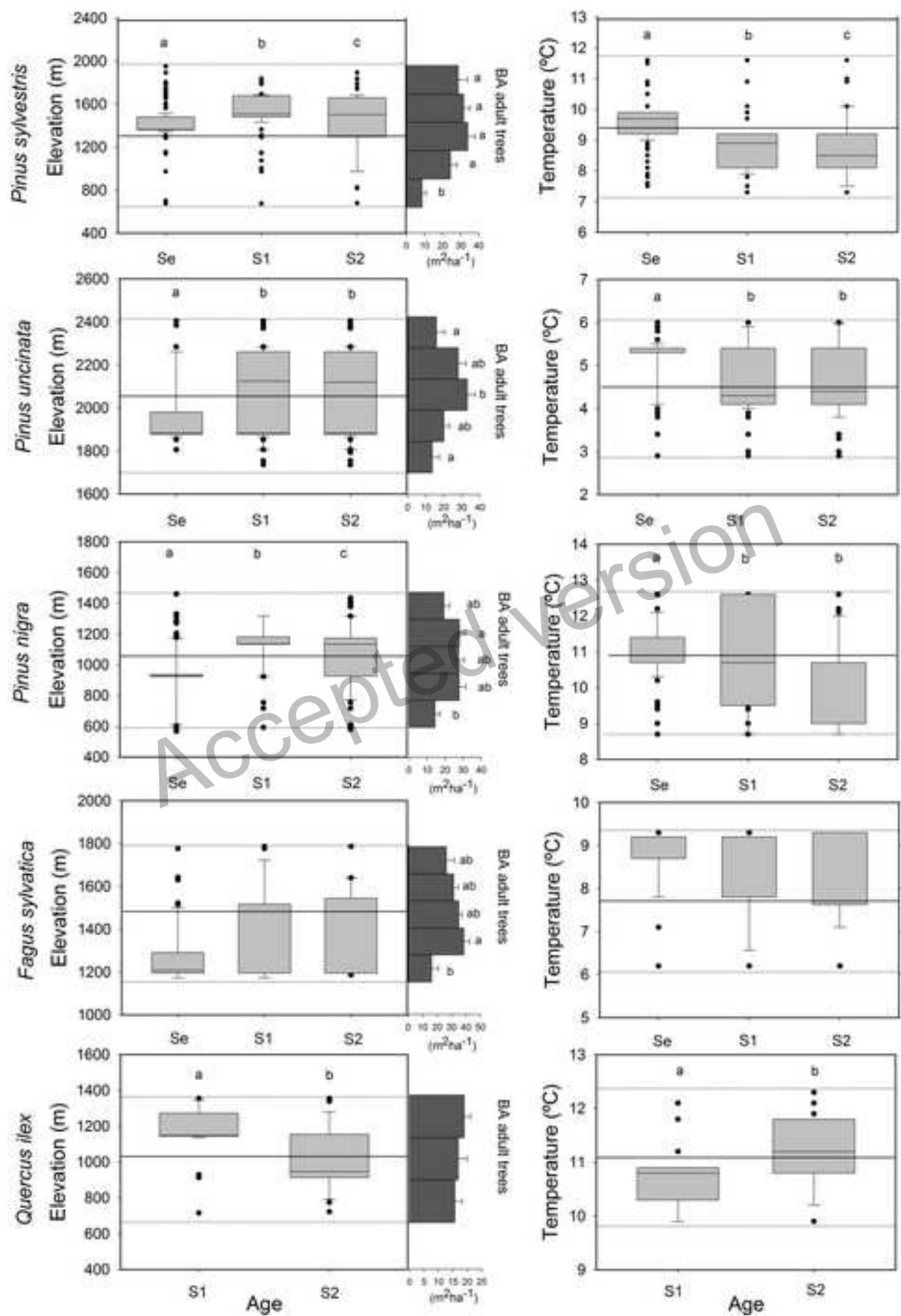


Figure 3

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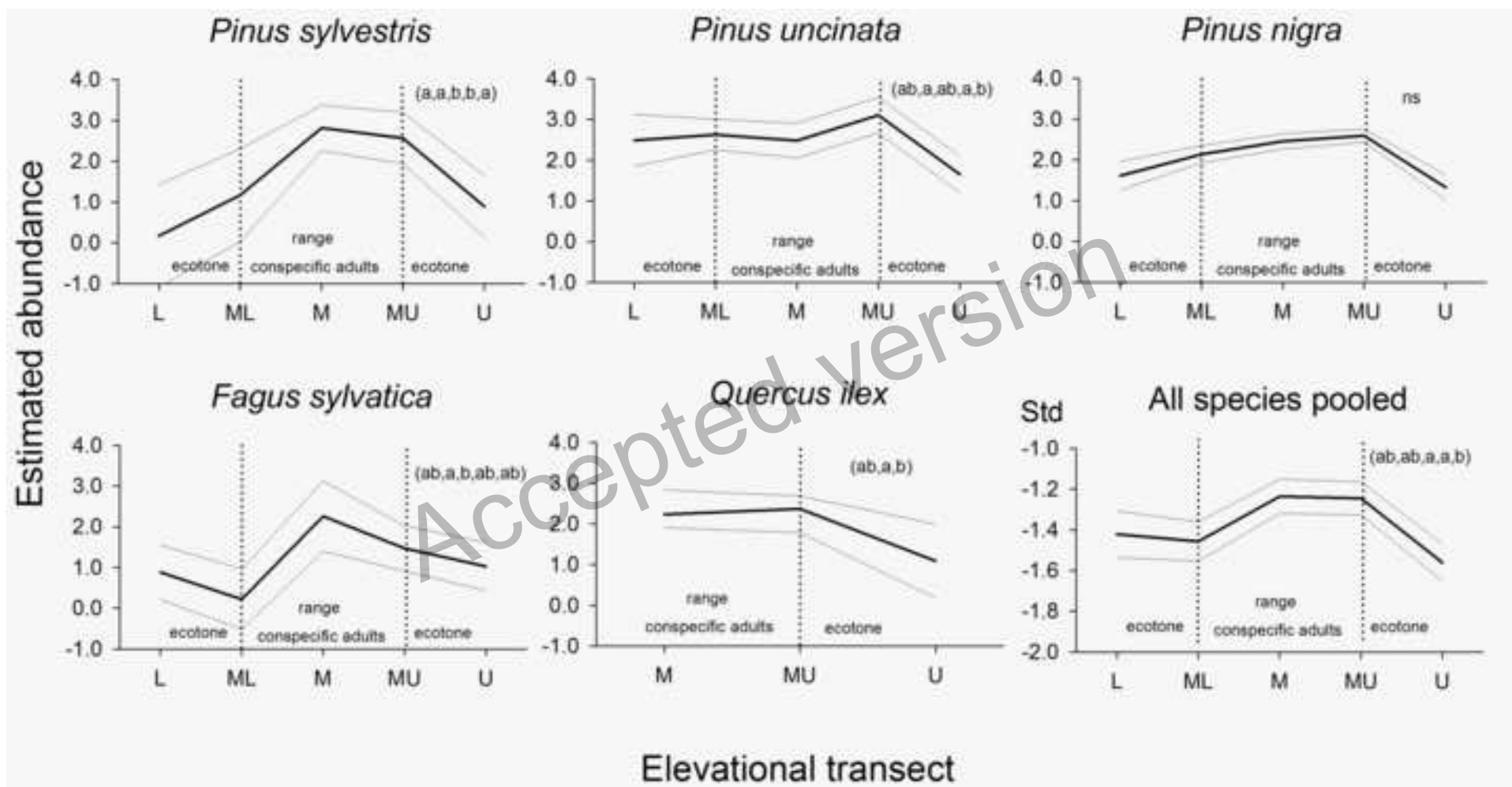


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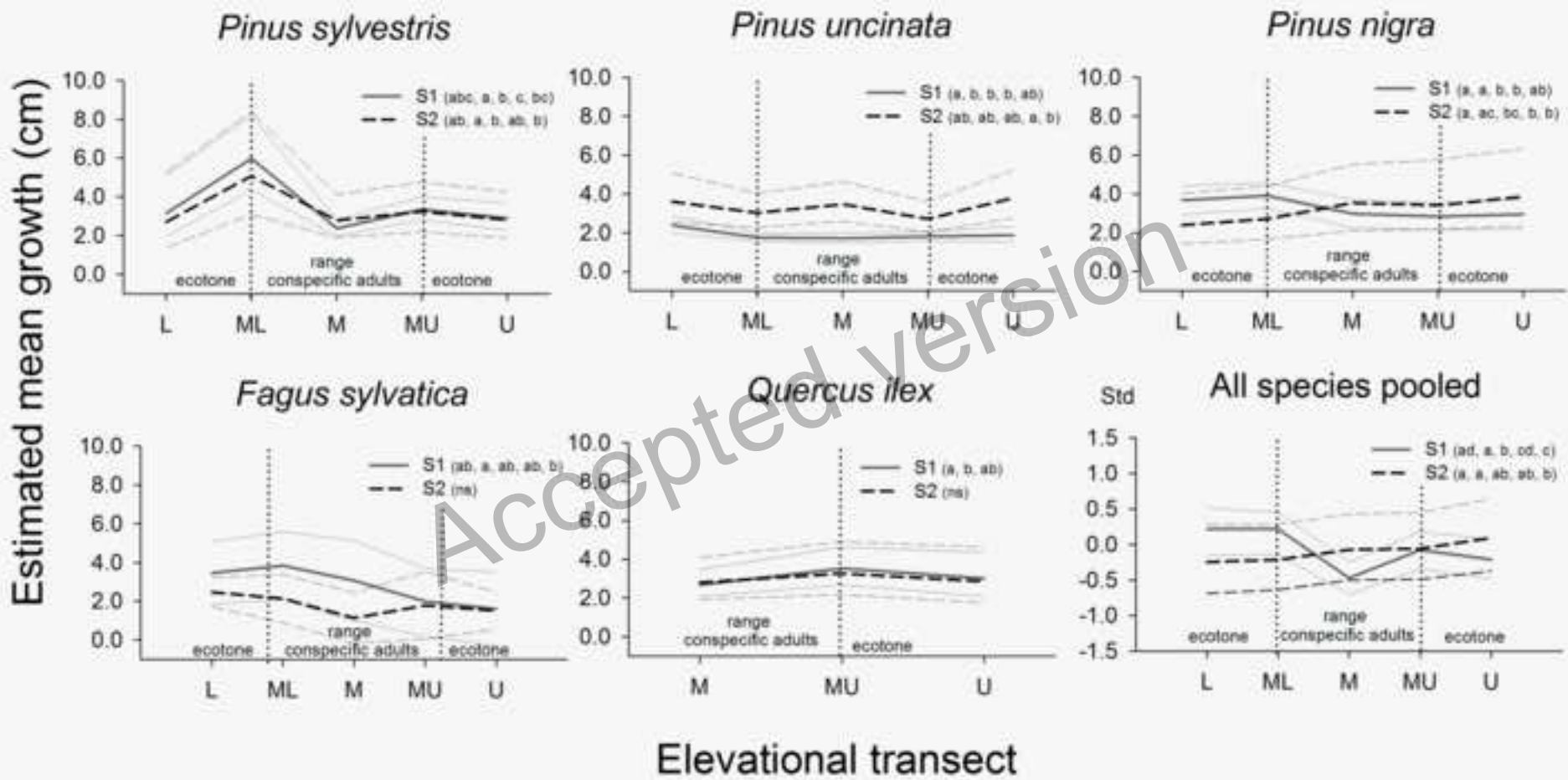


Table 1. Summary of the stabilising processes proposed by Lloret et al. (2012).

Stabilising processes	Basis	Underlying predisposing factors/ processes
Mortality mitigation	Existence of microrefugia	Site quality
		Facilitation
	Adaptation, acclimation ability	Tolerance
		Plasticity
		Phenotypic variability
Mortality compensation	Enhanced future survival or recruitment ¹	Beneficial effects of climate change
		Release competition
		Release antagonism
		Increase mutualism

¹ By improvement of the conditions for recruits or adult reproductive performance.

Table 2. Summary of the best Generalized Linear Models (species-specific data) and Generalized Linear Mixed Model (pooled data) analysing the effect of the elevation level on the abundance of juveniles (number of individuals per plot). It is specified the sample size -number of plots surveyed- (n), the degrees of freedom (d.f.), the explanatory variable deviance (D), the probability of the chi-square tests of the variable effects ($P(\chi^2)$), AIC_{model} for the fitted model; the variation of AIC_{model} referred to the null models ($\Delta AIC= AIC_{null}-AIC_{model}$) and the explained deviance of the model (i.e. the reduction in deviance from the null to the fitted model, as a percentage of the deviance for the null model, D_{model} , expressed as a decimal). Outcomes shaded are those variables with a significant effect over the response variable in terms of deviance at $P(\chi^2) < 0.05$. For the covariates *Tree density* and *dbh* (diameter at breast height) the sign of the effect is specified in brackets after the $P(\chi^2)$, when significant.

	<i>Pinus sylvestris</i> (n=90)			<i>Pinus uncinata</i> (n=60)			<i>Pinus nigra</i> (n=60)			<i>Fagus sylvatica</i> (n=60)			<i>Quercus ilex</i> (n=36)			All species pooled (n=306)		
	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$
Site	2	40.238	<0.0001	1	10.439	0.0012	1	23.430	0.0000				1	2.0446	0.0248	-	-	-
Elevation	4	28.217	<0.0001	4	21.221	0.0003	4	4.567	0.3347	4	10.001	0.0404	2	3.3223	0.0411	4	20.990	0.0003
Tree density	1	3.280	0.0701(-)				1	4.932	0.0264 (-)									
dbh										1	8.377	0.0038 (+)	1	4.0933	0.0430 (+)	1	3.840	0.0498 (+)
AIC_{model}		321.00			296.21			289.39			88.46			180.60			365.02	
ΔAIC		29.61			14.24			12.85			4.74			0.14			14.83	
D_{model}		0.64			0.43			0.43			0.53			0.26			0.07	

Table 3. Summary of the best Generalized Linear Mixed Model analysing the effect of the elevation level on the individual mean growth for each species and globally (pooled standardized data), for individuals from 1 to 5 years (S1) or individuals older than 5 years (S2). It is specified the degrees of freedom (d.f.), the explanatory variable deviance (D), the probability of the chi-square tests of the variable effects ($P(\chi^2)$), the sample size -number of individuals measured- (N), AIC_{model} for the fitted model; the variation of AIC_{model} referred to the null models ($\Delta AIC = AIC_{null} - AIC_{model}$) and the explained deviance of the model (*i.e.* reduction in deviance from the null to the fitted model, as a percentage of the deviance for the null model, D_{model} , expressed as a decimal). Outcomes shaded presented a significant effect over the response variable in terms of deviance at $P(\chi^2) < 0.05$. The sign of the effect of the covariate *BA* (basal area) appears in brackets after the $P(\chi^2)$, when significant.

		<i>Pinus sylvestris</i>			<i>Pinus uncinata</i>			<i>Pinus nigra</i>			<i>Fagus sylvatica</i>			<i>Quercus ilex</i>			All species pooled		
		d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$	d.f	D	$P(\chi^2)$
S1	Elevation	4	113.518	<0.0001	4	11.877	0.0183	4	14.401	0.0061	4	6.726	0.1511	4	6.203	0.0450	4	112.455	<0.0001
	BA	1	12.304	0.0005(-)				1	1.545	0.2139				1	5.389	0.0203 (-)			
	N	838			252			195			64			109			1458		
	AIC_{model}	1014.88			205.92			522.21			224.77			156.27			-169.2678		
	ΔAIC	115.82			3.88			14.71			-1.73			2.20			107.84		
	D_{model}	0.11			0.06			0.06			0.04			0.04			1.7		
S2	Elevation	4	17.425	0.0016	4	17.695	0.0014	4	11.532	<0.0001	4	6.238	0.1821	4	0.713	0.7001	4	9.758	0.0447
	BA	1	8.586	0.0034(-)	1	8.825	0.0030 (-)				1	0.808	0.3687	1	3.264	0.0708(-)	1	18.137	<0.0001(-)
	N	211			332			292			26			124			985		
	AIC_{model}	258.79			594.17			295.21			73.46			162.75			303.9562		
	ΔAIC	16.01			16.52			17.37			-2.95			-2.02			17.90		
	D_{model}	0.1			0.04			0.08			0.11			0.03			0.09		

Accepted version