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Using inventory variables for practical biodiversity assessment in plantation stands

Marija Cosovic

Forest Research Centre, School of Agriculture, University of Lisbon, Tapada da Ajuda, 1349-017 Lisbon, Portugal

Abstract

Aim of study: Practically and simply assessing biodiversity by using inventory variables in four types of forest plantation stands (mixed and pure) including species such are chestnut, blue gum and maritime pine.

Area of study: Northwest Portugal in Vale do Sousa (14,840 ha), which is 97% covered with plantation forests.

Materials and methods: Simulated data, from 90-year stand-level forest management planning, were considered using three indicators: tree species (number of different species and species origin—native or exotic), mean diameter at breast height (DBH), and shrub biomass. Two shrub regeneration types (fully regenerated by seed and fully regenerated by resprouting), and three site quality conditions were also considered.

Main results: Mean biodiversity scores varied between very low (10.13) in pure blue gum stands on lowest-quality sites with shrub regeneration by seed, and low (29.85) in mixed stands with a dominance of pine, on best-quality sites with shrub regeneration by resprouting. Site quality and shrub regeneration type significantly affected all biodiversity scores in mixed stands dominated by pine and pure chestnut stands, while less affected pure blue gum stands and mixed stands dominated by blue gum.

Research highlights: The considered biodiversity indicators cover the major biodiversity aspects and allow biodiversity assessment over time. The findings are relevant for biodiversity conservation and fire protection management.

Additional key words: biodiversity indicators; forest function; forest structure; tree species composition; inventory variables; site index; shrub regeneration

Abbreviations used: DBH (diameter at breast height); FMM (forest management model); SI (site index).

Authors' contributions: Conceptualization, methodology, analysis and interpretation of data, statistical analysis and writing the manuscript: MC.

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Correspondence should be addressed to Marija Cosovic: marijacosovic@isa.ulisboa.pt; cosovicmaja@gmail.com

Introduction

Forest plantations are often criticised due to a low compositional and structural biodiversity (Newbold *et al.,* 2015; Yamaura *et al.,* 2019), particularly mono-specific stands with exotic tree species (Hunter, 1990; Hartley, 2002; Carnus *et al.,* 2006). Forest biodiversity is the diversity of all forms of life and its organisation within the forest area (Hunter, 1990; Winter *et al.,* 2011). Forestry plantations are formed by planting or seeding for purposes that could be economical (*e.g.,* timber and fibre production) or protection (*e.g.,* soil conservation and carbon sequestration) (Carnus *et al.,* 2006; Stephens & Wagner, 2007). Indeed, natural forests typically host higher biodiversity than plantations, although the latter may support higher biodiversity than other intensive land uses, *e.g.* agriculture (Stephens & Wagner, 2007). The concern is that plantation forests with low biodiversity are considered more susceptible to disturbances and environmental changes than natural forests (Lugo, 1992; Carnus *et al.,* 2006; Bassi *et al.,* 2008; Proença *et al.,* 2010). Biodiversity contributes to the resilience of forest ecosystems and the delivery of different ecosystem services (https://millenniumassessment.org/en/Frameworkhtml; Proença *et al.,* 2010). It is necessary, therefore, to include biodiversity conservation aims in forest management plans (Ezquerro *et al.,* 2016). As most of the management operations in plantations are performed at a stand level, it is critical to ensure that biodiversity conservation is addressed at this scale (Similä *et al.,* 2006; Smith *et al.,* 2007). Moreover, assessing biodiversity at larger scales (*e.g.,* Botequim *et al.,* 2021) implies building from a smaller-scale approach such as the stand level.

Thus, forest managers need to include indicators for biodiversity optimization in forest management planning (Biber *et al.,* 2020) that may contribute to the design of resilient and sustainable landscape mosaics (Marto *et al.,* 2018; Botequim *et al.,* 2021). The definition and proper applications of biodiversity indicators are topics under permanent discussion. However, many scholars agree that biodiversity indicators need to be practical (*e.g.,* Ferris & Humphrey, 1999; Angelstam & Donz-Bruss, 2004; Smith *et al.,* 2007). Practical indicators are easy to apply, repeatable, cost-efficient and ecologically meaningful (Ferris & Humphrey, 1999; Smith *et al.,* 2007). Therefore, biodiversity indicators should be based on variables readily available in forest inventory datasets such as tree species composition (number of tree species per unit area) or diameter at breast height (DBH, *i.e,* diameter over bark measured at a height of 1.3 m above ground level) and tree height, for structural biodiversity (Ćosović *et al.,* 2020).

Trees are the dominating elements of forest ecosystems, and thus tree species composition is the most significant indicator of forest biodiversity which also contributes to structure definition (Stapanian *et al.,* 1997) and affects the composition of other forest communities (Martín-Queller *et al.,* 2011). For example, studies from North-Western Iberia have shown that plant and bird species composition are greater in native oak (*Quercus* spp.), maritime pine (*Pinus pinaster* A.), chestnut (*Castanea sativa* M.), and birch stands (*Betula alba* L.), than in non-native blue gum stands (*Eucalyptus globulus* L.) (Proença *et al.,* 2010; Goded *et al.,* 2019). Regarding structure variables that may influence biodiversity, such as tree height, biomass, diameter heterogeneity, and shrub volume, most are available in forest inventories and thus well known to forest managers, and also frequently available for public use (Ćosović *et al.,* 2020). In plantation forests, the understory layer is particularly important component of habitat structure and provides cover and food for wildlife (Smith *et al.,* 2007). Additionally, the understory is an indicator of ecological processes such as carbon storage, nutrient cycling and fire hazard risks (Botequim *et al.,* 2015). However, the structural properties of shrubs are species-dependent. Namely, the shrub species that regenerate by seeds typically develop lower bulk density than the shrubs that regenerate by resprouting, which is relevant for wildlife and the risk of wildfire predicting (Botequim *et al.,* 2015). Other variables, such as DBH, may also indicate stand structure as it relates to tree height, biomass growth or crown development. A particularly important aspect of forest biodiversity structure is trees with large diameters, as these are of great significance for the survival of numerous insects and birds (Badalamenti *et al.,* 2017). Additionally, forests that host large trees from diverse species are more resistant to disturbances than those with low tree species richness (Musavi *et al.,* 2017; Lutz *et al.,* 2018). Mature temperate forests, which are usually high in biodiversity, have higher large tree densities, mean diameters and total living biomass, in comparison to young stands (Burrascano *et al.,* 2013; Badalamenti *et al.,* 2017).

The present study aimed to demonstrate a practical and simple way to estimate stand-level biodiversity in plantation forests of Northwest Portugal. More precisely, for biodiversity assessment, the focus is on structural indicators: mean diameter (DBH) (cm) and shrub biomass (Mg ha^{-1}), but also compositional aspects such as tree species composition and species origin (native or exotic), and functional aspects such as shrub regeneration type and site index. To my knowledge, this research is the first to consider site index and shrub regeneration type in forest biodiversity assessment. Moreover, despite the large body of literature, the estimation of practical and quantitative indicators for application in the framework of managed forest management is still scarce.

Material and methods

Case study and collected data

The case study area was Vale do Sousa in Northwest Portugal, an area that extends over 14, 840 ha, out of which 97% is forest cover (for more information on the study area see Rodrigues *et al.,* 2020). Vale do Sousa can be considered representative of the forest landscape and forest management practices of this part of the country. The topography is very irregular, with a maximum elevation of 700 m (Marto *et al.,* 2018). The mean annual temperature is between 10 °C and 15 °C and the mean annual precipitation is quite high (1240 mm), though summer is typically dry, while autumn is very wet. Forest stands are managed according to four forest management models (FMMs), where each FMM has a different field management regime (silvicultural management). Two forest models (FMM1 and FMM2) assemble mixed stands of blue gum *(E. globulus)* and maritime pine *(P. pinaster)* where in FMM1, maritime pine is dominant (73%), while blue gum dominates in FMM2 (67%). FMM3 harbours pure chestnut (C. sativa) stands and FMM4 harbours pure blue gum stands. FMM1 and FMM2 are even-aged only at the beginning of the rotation period, while after the first eucalypt harvest, the stands became uneven-aged. Eucalypt is harvested three or more times before the first pine harvest (Table 1). FMM3 and FMM4 are even-aged stands.

There is a difference in tree species growth on various sites in Vale de Sousa, therefore, we considered tree site indexes (SI): SI1-low, SI3-medium and SI5-high. Site indexes were derived using data from the forest inventory such as the height and age of dominant trees for each stand (Rodrigues *et al.,* 2020). Blue gum and maritime pine stands are relatively evenly distributed across areas with all three site indexes, while chestnut covers mainly the high-quality sites (SI5).

The forest inventory data used here was collected within the ALTERFOR project (https://alterfor-project.eu/). These data were simulated along a 90-year forest planning horizon. The growth of maritime pine was simulated by the model PINASTER (Nunes *et al.,* 2011), and blue gum stands (FMM1, FMM2, and FMM4) were simulated by model GLOBULUS (Tomé *et al.,* 2006), where both models are implemented into the StandsSIM-MD module (Barreiro *et al.,* 2016). Chestnut stands growth (FMM3) was simulated by CASTANEA yield tables (Patrício, 2006). All growth models are empirical and such models "seek principally to describe the statistical relationships among data with limited regard to an object's internal structure, rules, or behaviour" (Korzukhin *et al.,* 1996). Shrub biomass accumulation (Mg ha^{-1}), was simulated according to Botequim *et al.* (2015) and considers the following management-related biometric variables: (i) stand basal area (m^2 ha^{-1}, with values obtained from growth and yield models described above); (ii) resprouter cover percentage (which considers fully seed and fully resprouting regenerative type strategies); (iii) shrub age (elapsed time since the last shrub clearing); and (iv) mean annual temperature (T = 14.5ºC).

Biodiversity indicators

The following variables were considered biodiversity indicators in this study: (1) tree species composition (tree species richness + species origin), (2) mean diameter (DBH, cm), and (3) shrub biomass (Mg ha^{-1}) with two levels of regeneration (by seed or by resprouting). A tree species composition indicator was created by combining tree species richness and the integer reflecting the number of native/non-native tree species in the stand. Thus, a value of '1' was assigned to one exotic species present in the stand and a value of '2' to one native species present in the stand. Correspondingly, a value of '3' was assigned to mixed pine/blue gum stands since maritime pine is a native species and blue gum exotic, '2' to pure chestnut (native species) stands and '1' to pure blue gum stands. The next step was to define the reference value of each indicator. This was based on the literature review, consulting peers and based on my own experience as a forest ecologist. In a report by Forest Europe (2020), tree species biodiversity is estimated across Europe in four categories: 1, 2-3, 4-5 and 6+ tree species. Forests with 6+ tree species are the rarest, and cover only 4.6% of European forests, while the most dominant are forests

Table 1. Four forest management models (FMMs) inventory variables and management practices.

FMMs	Tree density (trees ha^{-1})	% of study area	Thinning operation frequency	Fuel treatments	Harvesting
FMM1. Mixed maritime pine and blue gum forest system (*P. pinaster* + *E. globulus*) dominance of maritime pine 73%	Maritime pine 2200 Blue gum 1400	16.0	For pine—thinning every five years between 20 and 45 years	Every 5 years	Clear cutting systems for pine (45 years) / Coppice systems for blue gum (11 years)
FMM2. Mixed maritime pine and blue gum forest system (*E. globulus* + *P. pinaster*) dominance of blue gum 66%	Maritime pine 2200 Blue gum 1400	17.0	For blue gum —leaving two shoots at every stool on the 3rd year after the harvest		
FMM3. Chestnut *(C. sativa)* forest systems for the production of chestnut sawlogs	1250	1.0	Thinning every 5 or 10 years starting at age 15		Clear cutting systems (50 years)
FMM4. Blue gum *(E. globulus)* forest system for pulpwood production	1400	66.0	Leaving two shoots at every stool on the 3rd year after the harvest		Coppice systems (11 years)

with 2-3 species that cover half of all the European forests (Forest Europe, 2020). Regarding mainland Portugal, five native *Quercus* species comprise 93% of potential zonal native forests (Capelo *et al.,* 2007; Monteiro-Henriques & Fernandes, 2018). Therefore, the reference value of the tree species composition variable was considered to be 10. This means that it could represent the stand with, *e.g.,* ten exotic tree species, or five native species, or three native and four exotic species, and similar.

Regarding shrub biomass, two sources for the reference value were considered: the data used in the study and the literature. The highest value of shrub volume encroachment simulated in Vale Sousa with no shrub clearings scenario was 26 Mg ha^{-1}. Similarly, a study from northern Portugal reported 28.88 Mg ha^{-1} as the greatest shrub encroachment during 15 years of the post-fire period (Enes *et al.,* 2020). Then, in this work, the reference value utilized was the mean value between the data used in this study and the example found in Enes *et al.* (2020), which is 27 Mg ha^{-1}.

Regarding a reference value for mean diameter, according to national, European and global levels, about 60 cm of diameter is a suitable reference value. Hence, the average diameter of mature trees in Portugal is about 55 cm for 83 years old maritime pine (Pinto, 2004) and about 54 cm for 73 years old chestnut (Patrício & Nunes 2017). European forests are dominated by trees whose diameters are 21-40 cm, but about 8% of trees reaching 60 cm of DBH are found in uneven-aged forests (Forest Europe, 2020). Also, Lutz *et al.* (2018) recommended 60 cm as the fixed diameter threshold for large-diameter trees 'reached by at least some trees in almost all plots' in their study related to global forests. Therefore, the standard reference value of 60 cm was considered an appropriate value for a tree that contributes to biodiversity significantly.

Data analysis

The data of each indicator were normalized as percentages using the indicator's actual and reference values and the following formula: $x = (a/b) \cdot 100$, where x is the indicator's normalized percentage value, a is the actual indicator value and b is the reference value.

The normalized value was calculated for all three indicators. The average of these three indicator values was calculated to estimate the biodiversity value (index), ranging from 0 to 100. Further, five biodiversity categories were created as quintiles, where values between 0 and 20 corresponded to a very low value of biodiversity, 20-40 to low biodiversity, 40-60 to medium biodiversity, 60-80 to high biodiversity, and 80-100 to very high biodiversity. Normalization calculations were performed and the results were visualized in Excel (MS Office 2016). Also, a statistical summary, coefficient of variance, was carried out in the R programming language (R Core Team, 2020). Further, statistics were compared and biodiversity data was first assayed for normality with Shapiro-Wilks Normality Test. Since the data did not follow the normal distribution, the Kruskal-Wallis test was performed to check if there were differences in biodiversity data between groups with different site indexes and shrub regeneration types. After, multiple pairwise-comparison was carried out with Wilcoxon signed-rank test to see which groups were different. All comparative statistics were performed in the R programming language (R Core Team, 2020).

Results

Biodiversity in different forest management models (FMMs), shrub regeneration types and site indexes

The variations of biodiversity values of the four stand types over 90 years of management horizon, on three site quality conditions and two shrub regeneration types, are presented in Figs. 1 and 2. The summary of biodiversity data is presented in box plots (Fig. 3), Table S1 [suppl.] and Fig. S1 [suppl.] where it is shown that the highest mean value (29.85, low biodiversity) was recorded in mixed blue gum and maritime pine stands, with maritime pine dominance (FMM1) on the site index 5 (SI5) with shrub resprouting regeneration. The lowest mean value (10.13, very low) was recorded in pure blue gum stands (FMM4) on the SI1 with shrub seed regeneration. However, the highest maximum value (45.7, medium biodiversity) was recorded in pure chestnut stands (FMM3) with shrub resprouting regeneration and SI5, in the 49th year, right before the clear cut (Fig. 2). Medium category biodiversity score maximum values (41.72) were also recorded in FMM3 stands on shrub resprouting regeneration type and SI3 in the last year before the clear cut (49 years), and in FMM1 (41.24) in the year 40, which is 5 years before pine's clear cut. Regarding FMM2, all maximum values were low category biodiversity, while in FMM4 these were very low and low. Generally, across all site indexes, the mean biodiversity value was lower in shrub seed regeneration stands than in the resprouting stands (Fig. 3, Fig. S1 [suppl.]). In all stands with shrubs regenerating by resprouting, shrubs recovered faster (Figs. 1 and 2) and biomass was larger on average (Fig. 3). However, only in FMM3, there were clear differences in variations of biodiversity data between seed and resprouting stands (Fig. 4). Although, the coefficient of variations of all biodiversity data was high in all FMMs (>17). The highest variations were recorded in FMM4 (34-36), followed by FMM3 with variations between 26 and 33, while FMM1 and FMM2 had the lowest variations (18-21).

Regarding the site index, lower values generally reflected lower biodiversity within the same regeneration type.

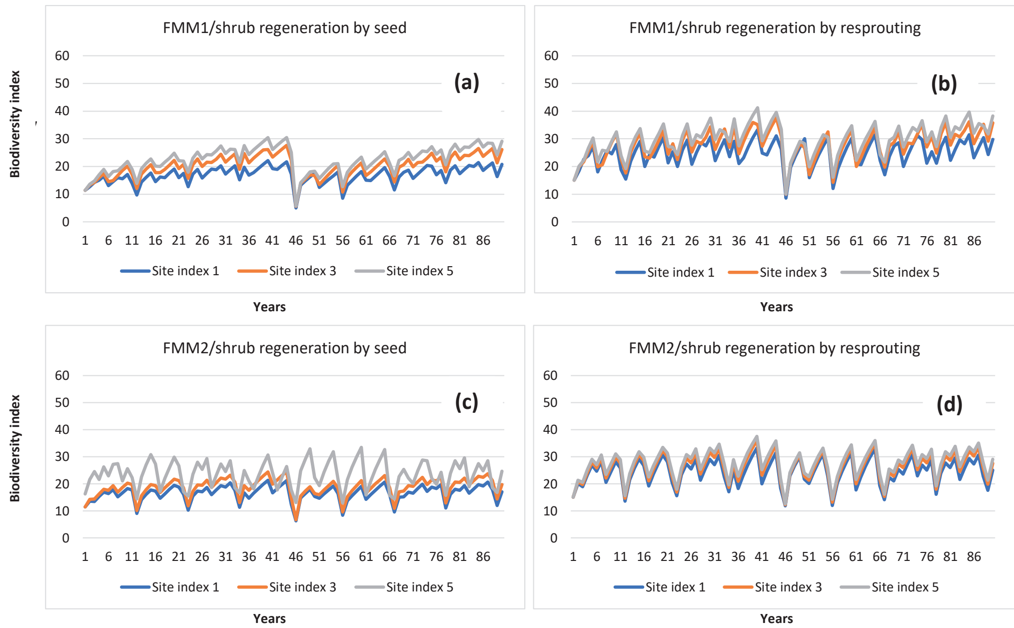

Figure 1. Biodiversity values for mixed blue gum and maritime pine stands, with a dominance of maritime pine (FMM1) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (a) and shrub resprouting regeneration (b); mixed blue gum and maritime pine stands, with a dominance of blue gum (FMM2) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (c) and shrub resprouting regeneration (d). Y axes show biodiversity values and X axes, 90 years of management horizon.

i.e., biodiversity values were always highest in SI5 and lowest in SI1 (Fig. 3). When biodiversity values between regeneration types were compared, SI1 with seed regeneration shrub type typically had the lowest biodiversity, and SI5 with resprouting regeneration type typically had the highest biodiversity (Fig. S2 [suppl.]). The majority of maximum values of all FMMs were higher in shrub resprouting regeneration type than in seed regeneration type, except in the case of FMM2 and FMM3, where SI1 shrub resprouting regeneration was lower than SI5 seed regeneration. There was a significant difference in biodiversity score between most of the site indexes within the same shrub

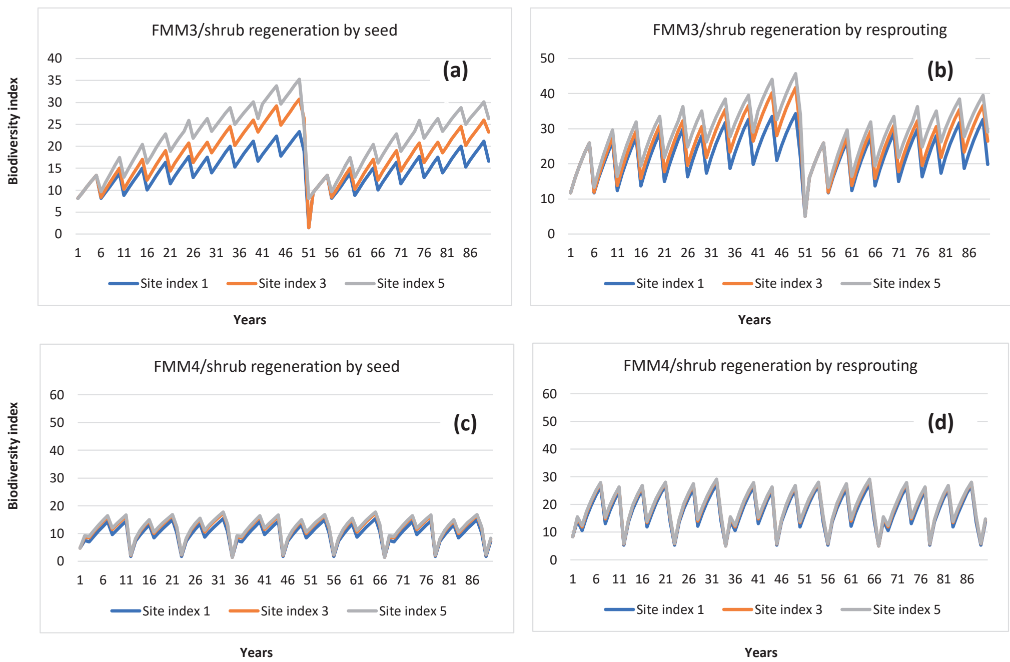

Figure 2. Biodiversity values for pure chestnut stands (FMM3) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (a) and shrub resprouting regeneration (b); pure blue gum stands (FMM4) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (c) and shrub resprouting regeneration (d). Y axes show biodiversity values and X axes, 90 years of management horizon.

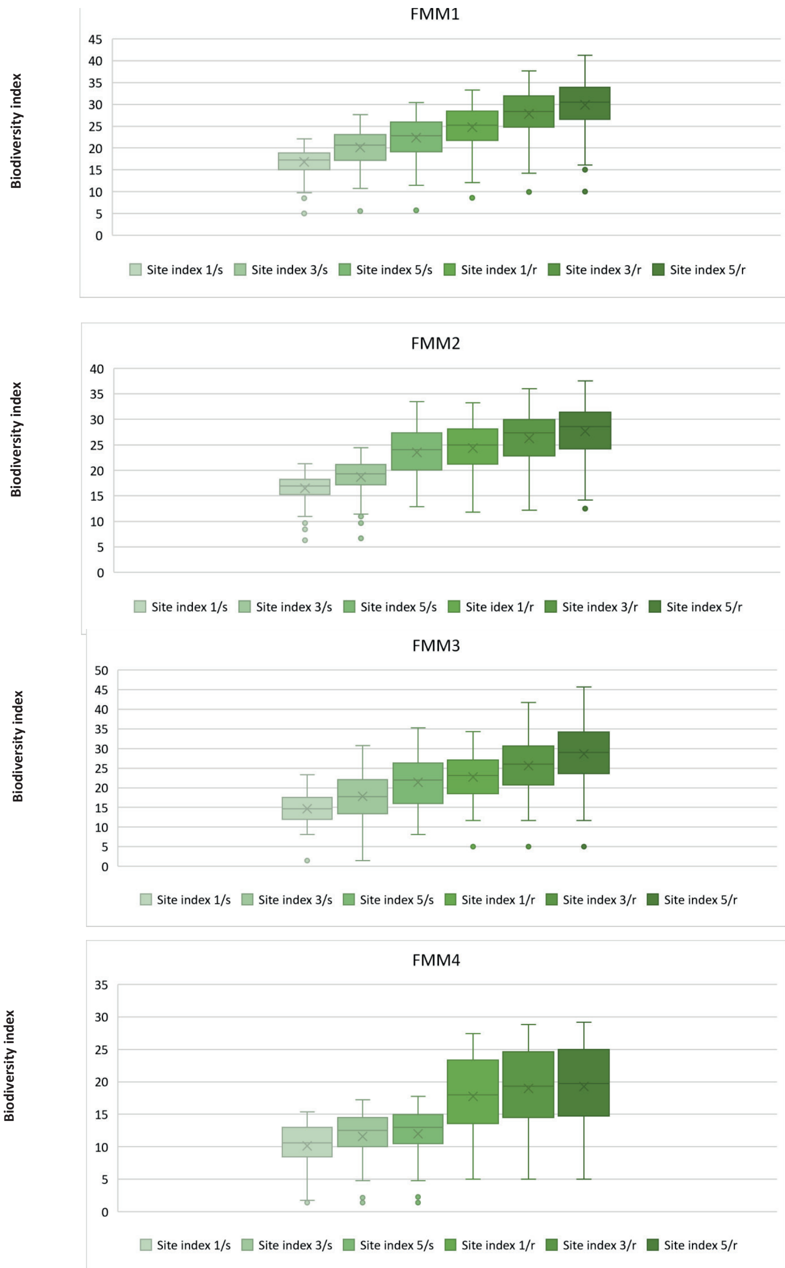

Figure 3. Boxplots of biodiversity values of FMM1, FMM2, FMM3 and FMM4 with three site indexes (SI1, SI3 and SI5), shrub seed regeneration (s) and shrub resprouting regeneration (r).

regeneration type, and between regeneration types (Table S2 [suppl.]). There was no significant difference (p>0.05) in FMM4 between SI3 and SI5 seed regeneration (p=0.30); between SI3 and SI5 resprouting regeneration (p=0.58); between SI1 and SI3 resprouting regeneration (p=0.13), and SI1 and SI5 resprouting regeneration (0.07) (95% of significance). Also, there was no significant difference in FMM2 between SI3 and SI5 resprouting (p=0.07), and between SI5 seed and SI1 resprouting (p=0.22).

Discussion

Biodiversity in plantation forests

In this study, biodiversity was assessed in four types of plantation stands (FMMs) in Northwest Portugal over the 90-year forest management planning horizon, using a method that combines biodiversity indicators and derives biodiversity scores varying from 0 to 100 (very low to very high). The

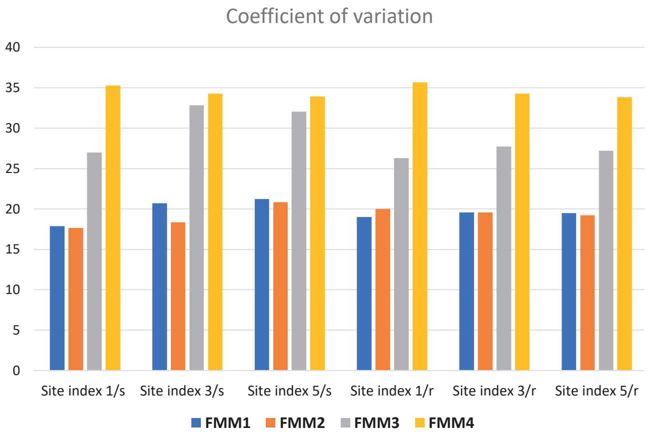

Figure 4. Coefficient of variation of biodiversity values in FMM1, FMM2, FMM3 and FMM4 with three site indexes (1, 3 and 5) and shrub seed regeneration (s) and shrub resprouting regeneration (r)

mean values of biodiversity, considering all FMMs, shrub regeneration category and site quality, varied between very low (10.13) and low (29.85), which confirms the statement that biodiversity in plantation forests, at a stand level, is typically low (Koh & Gardner, 2010; Newbold *et al.,* 2015; Yamaura *et al.,* 2019). However, maximum values were medium in the case of pure chestnut stands (45.71) and mixed stands with a dominance of maritime pine (41), and these values were reached only in mature years. Therefore, the state of biodiversity in plantations depends on the maturity stage. Pure exotic blue gum stands had the lowest mean biodiversity over the entire 90-year horizon and, in general, when compared to the other stands (mixed maritime pine and blue gum stands, and pure chestnut stands). This is in concordance with studies that argue that forests with pure stands of exotic species have the lowest biodiversity (Hunter, 1990; Hartley, 2002; Carnus *et al.,* 2006; Mikulová *et al.,* 2019). However, these low values in our case resulted from management practices such are frequent clear cuts (every 11 years). Plantation forests have very intense dynamics imposed by clear cuts (high coefficient of variation in the present study). Nevertheless, there is potential to develop higher biodiversity, if the time between clear cuts is extended. Therefore, in future research, it may be worth comparing the biodiversity value of exotic species plantations that have not had any silvicultural interventions, with unmanaged forests. Indeed, most plantations of exotic species have shorter rotation periods than native species plantations, and that is one of the main reasons why exotic species are introduced in the first place. For example, blue gum rotation in Portugal is typically 10-12 years (Deus *et al.,* 2019), while maritime pine is around 35 years (Oliveira, 1999; Dias & Arroja, 2012). The shorter the rotation period, the shorter the time necessary for biodiversity recovery and establishing of species interconnections and ecosystem stabilisation. These might be the main reasons for low levels of biodiversity in exotic mono-specific plantations.

Impact of shrub resprouting type and site index on biodiversity

There are differences in mean biodiversity values in the case of the shrub regeneration category in this study. Maximum values of biodiversity were also affected in the case of all FMMs except pure blue gum stands (FMM4). This can be explained by the short rotation of FMM4 (11 years), while other FMMs have nearly four times longer rotation periods such as maritime pine (45 years) in FMM1 and FMM2, and chestnut (50 years) in FMM3. There were also differences in the speed of shrub biomass development among stands between shrub regeneration by seed and by resprouting, where shrubs that regenerate by resprouting developed faster than shrubs that regenerated entirely by seed. Nevertheless, Botequim *et al.* (2015) reported opposite results, where shrubs that regenerate by resprouting developed lower biomass than shrubs that regenerate by seeds and particularly if the basal area was larger. Also, they reported shrubs that regenerated by seed recovered faster particularly if the basal area was lower. The reason for the difference might be that the study area was Mainland Portugal, covering managed, unmanaged forests and plantations, while the study area of this paper, had only plantations. Similarly, a study from central Argentina reported that, after a fire, shrub sprouting vigour was faster if wood density was low and the shrub was tall before the fire, while for small shrubs, wood density had no influence (Gurvich *et al.,* 2005). Also, a study that researched postfire shrub regeneration in heathlands from Australia (Pate *et al.,* 1990) reported slower growth of juvenile resprouters (<6 years), than non-resprouters. However, Pausas *et al.* (2004) found that resprouter traits can hardly be predicted on a global scale, but rather local, due to different responses of species in various areas. Therefore, there are indications that forest structure might influence the speed of shrub regeneration; however, more research is needed to examine locally regenerating traits of certain shrub species and their interaction with biodiversity.

Though numerous studies have reported better forest productivity in mixed stands than in monocultures (*e.g.,* Zhang *et al.,* 2012; Bielak *et al.,* 2014), such a case does not apply to clonal *Eucalyptus* plantations which are the world's fastest-growing plantations (Forrester & Bauhus, 2016). In this study, site index slightly affected mean and maximum biodiversity values in pure blue gum *(E. globulus)* while mixed stands with the dominance of maritime pine (FMM1), and pure chestnut stands (FMM3) were significantly affected by site quality. It can be concluded that the site index does not have a major effect on the biodiversity of short-rotation plantations. However, more research is needed on this topic.

Implications and suggestions for future management

Since the case study of this paper belongs to the Mediterranean geographic region, forest fires are widespread. In the past decade, forest fires became widespread all over the globe due to climate change. Frequently, maintaining a high conservation value habitat such as shrub formation may also imply a higher risk of wildfire (Silva *et al.,* 2020). Therefore, it is not advisable to increase shrub encroachment volume all over the area, but only where it is associated with high biodiversity importance (Botequim *et al.,* 2015). This will necessarily create trade-offs that need to be carefully considered. Additionally, knowledge about shrubs regenerating type may be used in fire prevention management, since shrubs that regenerate by resprouting develop much faster than shrubs that regenerate by seeds, in this case study area.

Extending the rotation period, increasing tree species composition with native species and leaving some big trees after clear-cutting may benefit biodiversity at the stand level. For example, Lafond *et al.* (2015) researched French Alps and found that retention measures of large trees, non-dominant species, and deadwood can compensate for the negative effect of intensive management practices. Initially, it might affect the income from timber production, but in the long run, it might decrease losses induced by pest and disease outbreaks. Additionally, payments for biodiversity conservation management may compensate for losses due to wood production. In Portugal and other Mediterranean countries, introducing species well adapted to forest fire, such as cork oak, may not only improve habitat quality for wildlife but, if well managed, even reduce fire risk.

Plantation forests can be useful in efforts for biodiversity conservation (Koh & Gardner, 2010), even plantations with exotic species can host native species. However, it would be more ecologically acceptable if native forests are restored and protected than to manage eucalypt plantations for biodiversity (Calviño-Cancela *et al.,* 2012).

Even though deadwood or old trees are fundamental indicators of forest biodiversity, those were not applied in this research due to the absence of such aspects in the case study area. However, since the results were mainly anticipated and demonstrated very low and low mean values of biodiversity, these indicators seem explicit and meaningful. The approach used in the present study is not intended for detailed scientific biodiversity assessments. It is rather suitable for initial estimation that will give the forest managers sense of the biodiversity state in their forests and help in management decisions. Also, it can serve as a base for detailed scientific estimations.

The present study demonstrates that forest inventory variables can be used as practical biodiversity indicators, and their combination can provide an overview of biodiversity at stand level over time. Site index and regeneration strategy are important aspects as they influenced the biodiversity of plantations with longer rotations such are those with maritime pine and chestnut, while less affected plantations with short rotations such are those with blue gum. Those findings are important for forest biodiversity conservation and fire prevention management.

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