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Conversion of *Pinus radiata* plantations to native forest after harvest operations: a north Iberian Peninsula case study

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Abstract There is broad consensus on restoration of native woodlands in places where intensive forestry is nowadays not profitable. However, this consensus is lost when stakeholders need to implement forest management practices as restoration tool, especially because there is a substantial lack of empirical evidence about its feasibility. In this context, we assess the impact of two different harvest treatments on understorey plant species composition of *Pinus radiata* plantations as tools to recover native woodland vegetation in northern Iberian Peninsula. Here, common clear-cut treatment and restoration-clear-cut where only pine trees were removed (i.e. reducing the disturbance effect over understorey vegetation) were compared against understorey plant species composition of young and old plantations and restored tracks. The aim was to identify which treatment is more suitable to recover native woodland vegetation. The results reveal that both clear-cuts maintained species composition plus important understorey native species, some of them being restoration targets. However, both clear-cuts showed diversity reductions compared with old plantations, although there were not apparent retention effects on compositional change towards native communities at least two years after harvest. It seems that the remaining vegetation established by natural succession after both clear-cut treatments could be used to achieve initial restoration objectives for some native tree and understorey plant species at relatively low costs. In any case, it would be interesting to implement supplementary management measures to accelerate this conversion, such as invasive species elimination or target species seeding, to maintain local biodiversity and introduce native woodland species not present in the area.

Keywords Restoration · Native woodland · Clear-cut · Forest management · Oaks

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Introduction

Plantations represent 7% of the world's forests comprising around 75 million hectares worldwide (FAO 2015). Though native forest area is declining throughout the world, plantations are increasing significantly (Hartley 2002), mainly because tree plantations are becoming fundamental to satisfy the increasing demand for wood products worldwide (FAO 2015), especially those plantations of fast-growing exotic tree species (Pryde et al. 2015). Nowadays, most new plantations are composed of exotic coniferous or eucalypt species, which can be easily found in Europe and southern hemisphere (Simberloff et al. 2010; Calviño-Cancela et al. 2012; Calviño-Cancela 2013). However, the negative effect of plantations, i.e. simplifying tree structure and composition, on the maintenance of regional biodiversity are well known (Calviño-Cancela 2013). Thus, the current increase in intensive plantation area has raised concerns among forest managers and the general public over their implications for sustainable production and native species conservation (Carnus et al. 2006). As there is now increasing societal demand to obtain multiple outputs from silvicultural systems (Alday et al.

2010), where timber production and the conservation of native species are integrated (Kimmins 2004), a greater emphasis is now being placed on the restoration and enhancement of native woodlands (Brockerhoff et al. 2008; Brown et al. 2015), especially in places where intensive forestry is not highly profitable.

Disturbance plays an important role in structuring natural communities (Vandvik et al. 2005; González-Alday et al. 2009). For example, silvicultural treatments, especially harvesting, produce large-scale disturbances that could be expected to impact on remaining plant species composition (Bengtsson et al. 2000). Hence, disturbance-sensitive species may be unable to recover, or even become locally extirpated, while disturbance-insensitive species may use the new stand environmental conditions to thrive and grow (González-Alday et al. 2009). In this regard, works in old pine stands in Spain have demonstrated that spontaneously regenerated vegetation enhance their growth after harvest treatments (Alday et al. 2010). Usually, the created gaps are colonized by late successional native shrubs and broadleaved trees (e.g. *Quercus* spp. and *Fraxinus* spp.; Onaindia et al. 2013). This opens a new path to test whether harvest treatments in pine plantations can be used as a restoration tool to promote understorey spontaneously regenerated vegetation and henceforth the trajectory to native woodlands. Nevertheless, some studies have reported an increase of invasive species presence after harvest treatments (Dix et al. 2010), which can reduce the effectiveness of harvesting as a restoration tool, since some invasive species negatively affect forest ecosystems dynamics (Holmes et al. 2009). In this context, knowledge of understorey native plant layer response to different harvest treatments such as clear-cutting or shelter-wood, therefore, is a first requirement for developing sustainable native forest restoration practices in fast-growing pine plantations.

In the northern Iberian Peninsula, native forests were nearly eliminated over the past century due to demand for charcoal and timber. As a result, at the beginning of the twentieth century, native mixed-oak forests were scarce and highly fragmented. Simultaneously, during the twentieth century farm abandonment favoured the spread of rapid growth and fast turnover plantations (35–40 years rotations), mainly composed of *Pinus radiata* D. Don. However, recently the timber prices have decreased reducing the plantation's profitability (Onaindia et al. 2013). Thus, regional governments are enacting silvicultural policies and practices towards increasing native forest restoration and biodiversity conservation, especially in areas where tree plantations are not profitable or produce negative impacts on soil structure (Marcos et al. 2010). The inclusion of native forest restoration objectives in regional management strategies requires information about the understorey vegetation performance in response to forest management practices (D'Amato et al. 2009; Cristan et al. 2016). However, understorey species response to forest management practices, and tree harvesting in particular, has not been well-studied in the northern Iberian Peninsula (Burke et al. 2008; Torras and Saura 2008; Tárrega et al. 2011).

In this paper we assess the impact of two different harvesting practices on understorey plant species composition of *P. radiata* plantations: (1) clear cut (CC) and (2) clear cuts where only pine trees were removed reducing the disturbance effect over understorey vegetation (RCC). Here, *P. radiata* plantations at different successional stages (young and old), two harvest treatments (CC and RCC) and reclaimed mountain tracks (tracks) were selected to compare their understorey plant species composition and diversity. The aim is to identify which harvest practices (CC vs. RCC) are more suitable to recover native woodland vegetation, while reducing the occurrence of invasive species, in areas were recovering traditional woodland has great ecological interest (Onaindia et al. 2013). Specifically, we tested the following questions: (1) Are there differences in understorey plant species composition between harvested habitats, plantations and reclaimed tracks? Here, we hypothesize that CC will be compositionally more similar to early-successional stages (i.e. young plantations and tracks), while RCC will be more similar to old plantations. (2) Are there differences in native tree species composition between harvested (CC and RCC) and plantation habitats (young and old plantations)? Here, we hypothesize that native tree species composition of RCC will be more similar to old plantations, while CC will be more similar to young plantations considering the potential negative impact of harvesting on understorey vegetation (Newmaster et al. 2007). (3) Are there differences in tree and understorey vegetation diversity between habitats? We hypothesize that harvest treatments (CC and RCC) will show diversity values located in-between early and late successional sites (i.e. between young and old plantations) as a consequence of the feedback between harvest sensitive-insensitive species (Alday et al. 2010), and (4) does any treatment support the colonization of invasive plant species?

Materials and methods

Study area

The study was carried out within the UNESCO Biosphere Reserve of Urdaibai, Biscay in northern Spain. Precisely, in the area known as Undabaso watershed, close to Gernika (43.233770N, −2.673826E). The climate is temperate

Atlantic with a mean annual rainfall of 1200 mm and a mean annual temperature of 12 °C. The elevation of the plots range from 175 to 256 m.a.s.l. and the native vegetation comprised a mixed forest of *Quercus robur*, *Fraxinus excelsior* and *Crataegus monogyna,* with remnants of *Ulmus glabra* and *Ulmus minor*. However, during the past century, most native forests were replaced by fast-growing exotic conifer plantations; thus, nowadays native forests only cover 3% of their potential area in Urdaibai (Onaindia et al. 2013), as a result, landscape resembles a matrix of pine plantations with small patches of native mixed forests. Plantations are dominated by *P. radiata*, with approximately a 40-year rotation. Pine seedlings are planted in a density of 1000 trees/ha. Pruning and thinning treatments are common during the first half of the rotation (<20 years), thereafter (>20 years) tree density is approximately 400 trees/ha, and at this stage, silvicultural activities are uncommon (Onaindia et al. 2013).

Sampling design

The understorey plant species composition and diversity were monitored in ten sampling sites of 2–3 ha within Undabaso watershed, representing five types of habitats (two sites each): (a) reclaimed haul roads; here the soil removed was dumped back to the site path with reclamation purposes two years before sampling (tracks); (b) clear-cut stands; where pines and all shrub and tree species were removed two years before sampling (CC), here, harvest operations were done using mainly pine harvesters followed by a handsaw to eliminate shrubs and tree species present; (c) clear-cut stands where only pine trees were removed two years before sampling, preserving understorey vegetation, shrubs and tree species (RCC), here, harvesting was carried out manually with handsaw once all *P. radiata* trees selected for cutting were marked. In both harvested habitats, the harvested timber was removed from the plots using poles or directly by a timber lorry. The use of chainsaws to harvest the timber followed by low-intensity removal is aimed to minimize site disturbance (Alday et al. 2010). (d) 10–12 years old pine plantations with 500 trees/ha and 8 m^2 of basal area (Young_P); and finally (e) 30 years old pine plantations with 214 trees/ha and 9.5 m^2 of basal area (Old_P). In this study, reclaimed tracks were included because after their use to collect harvested logs they were reclaimed dumping the soil back, as a consequence they are useful sites to identify early-successional species in the area and compare them with the two clear-cut types analysed (CC and RCC). To record the variation caused by harvest treatments rather than to site variability, the sampling sites for each habitat shared the same abiotic conditions (harvested sites and plantations: south-east aspect, slope $\sim 10°$), similar structure of surrounding stands with native species at effective dispersal distance, forest structure (same age of planted trees and for harvested sites the same machinery and harvest month), and similar initial vegetation composition (Etxebarria 2014).

The sampling was carried out at the end of June and beginning of July of 2014. In each site, three plots (10×10 m^2) at least 100 m apart from each other were randomly located ($n = 30$ plots in total). In each plot, the number and height from all trees with diameter at breast height (DBH) greater than 7.5 cm were recorded. Thereafter, in each sampling plot, three quadrats (2×2 m^2) at least 10 m apart from each other were located randomly ($n = 90$ quadrats in total). Here, the bare soil and cover (%) of all vascular plant species was estimated visually by the same observer. Moreover, the Leaf Area Index (LAI) under the understory vegetation was measured by means of a LAI-2200 Plant Canopy Analyser (LI-COR) as a measure of productivity.

Data analysis

Statistical analyses were performed in the R software environment (v.3.3.2; R Development Core Team 2016), using the vegan package for multivariate and diversity analyses (Oksanen et al. 2017), and nlme and lme4 packages for linear and generalized mixed models (Pinheiro et al. 2013).

Understorey vegetation data and tree species data were analysed using both multivariate and univariate methods. First, nonmetric multidimensional scaling (NMDS, 'metaMDS' function with Bray–Curtis distance) was used to identify the understory compositional differences between the five habitats (tracks, CC, RCC, Young_P and Old_P). Previous to the analysis, understorey vegetation data was log-transformed ($\log(x + 1)$) to reduce the influence of rare species, while tree species count data was Hellinger-transformed. In understorey vegetation analysis, the outputs from the centroids for each habitat were overlaid in the ordination space ('envfit' function), with their standard deviational ellipses ('ordiellipse' function). The significance of understorey and tree compositional differences between habitats was tested using Permutational Multivariate Analysis of Variance (PMAV, 'adonis' function using Bray–Curtis distance). Second, to identify whether compositional differences were also related with bare soil and LAI, both variables were fitted onto the NMDS ordination plot using the 'envfit' function and 999 permutations. Also, bare soil and LAI response surface models were fitted over NMDS ordination results by general additive models (GAM) using 'ordisurf' function. Third, ANOVAs were used to test the differences in native tree number and tree height per plot between habitats.

Linear mixed models were used for Pielou's evenness (Magurran 2004) and generalized linear mixed models for richness with a Poisson family error structure. In both models, habitats were fixed factors and quadrats nested within plots were included as random factors to account for spatial autocorrelation of adjacent locations (Pinheiro and Bates 2000).

Results

In total 96 plant species were found in the samplings: 42 in young pine plantations, 27 in old plantations, 43 in each clear-cut site (CC and RCC) and 75 in tracks. The eight most frequent species were *Rubus ulmifolius* (30/30 plots), *Pteridium aquilinum* (29/30) and *Festuca arundinacea* (27/30), *Quercus robur* (24/30), *Arrhenatherum elatius* (23/30), *Potentilla erecta* (22/30), *Ulex europaeus* (21/30) and *Salix atrocinerea* (20/30).

Understorey compositional differences between habitats

There are significant plant compositional differences in understory vegetation between the five habitats (PMAV; F-model$_{[4,25]} = 3.65$; $P < 0.001$; $R^2 = 0.37$), accounting for 37% of the variance in the species data. As expected, the main compositional differences were between tracks ($P < 0.001$; $R^2 = 0.20$) and the rest of habitats, followed by compositional differences of young plantations ($P < 0.004$; $R^2 = 0.10$). Surprisingly, there was not understorey species compositional difference between both harvested habitats (CC and RCC) and old pine plantations ($P > 0.05$; $R^2 = 0.04$).

NMDS ordination (stress: 0.18; Fig. 1a, b) and standard deviational ellipses clearly showed that tracks and young plantations occupied different regions of the ordination space compared with CC, RCC and old-plantation habitats that were grouped. Tracks are located in the upper left corner, while young plantations are located in the lower central area. A great number of early-successional species were found in tracks such as *Ranunculus repens*, *Taraxacum officinale*, *Centaurea* spp. and *Carduus nutans*. Young plantations were related with early-successional species (*Sonchus oleraceus*, *Bellis perennis*) and medium-successional species such as *Crataegus monogyna, Rosa canina, Hypericum androsaeum*, *Dryopteris pseudomas*, *Hypericum pulchrum*. In contrast, both harvested habitats (CC and RCC) and old plantations were overlapped in the right side of the ordination space. Old plantations were related with *Lonicera periclymenum*, *Daboecia cantabrica*, *Angelica sylvestris* and *Hedera helix*, while RCC and CC habitats were related with some native species such as

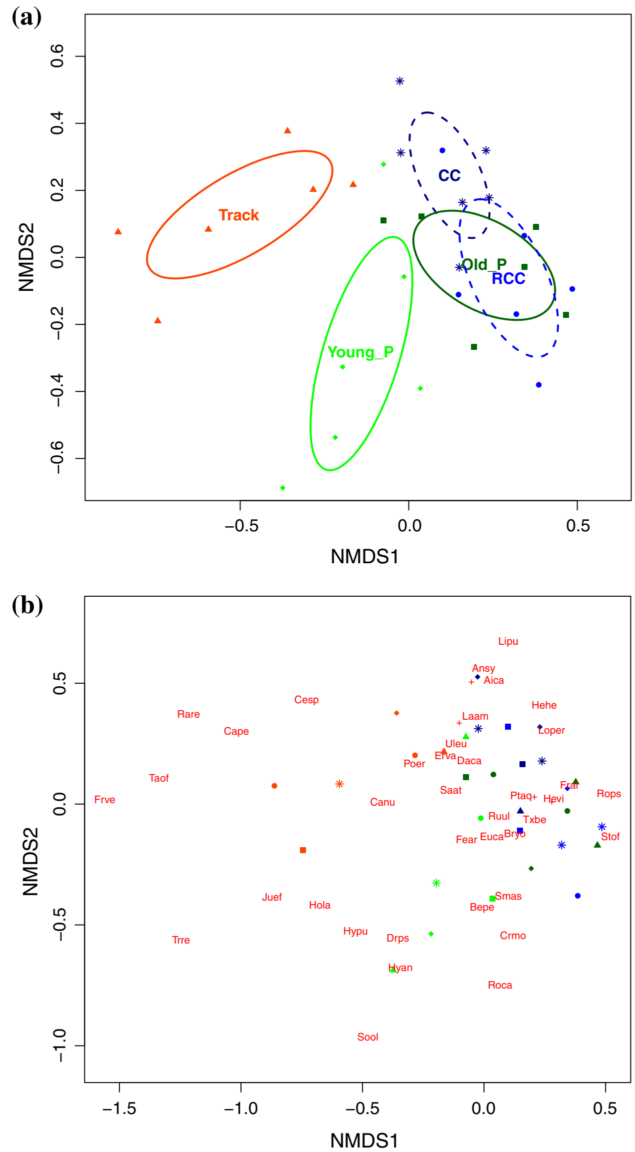

Fig. 1 NMDS ordination for the first two axes of understorey data from the five habitats in Biscay (northern Iberian Peninsula), illustrating: **a** Ordination biplot with deviational ellipses for each habitat: track, tracks; Young_P, young pine plantations; Old_P, old pine plantations; CC, clear-cut stands; and RCC, restoration-clear-cut stands; also sampling quadrats are included: *red triangles* tracks; *green diamond* young pine plantations; *green squares* old pine plantations; *asterisk* clear-cut stands and *blue dots* restoration-clear-cut stands; **b** species biplot where only the most frequent species are shown. Less important species are included as "+". Species codes are: Ansy, *Angelica sylvestris*; Bepe, *Bellis perennis*; Bryo, *Bryophites*; Canu, *Carduus nutans*; Cape, *Carex pendula*; Cesp, *Centaurea* spp.; Crmo, *Crataegus monogyna*; Daca, *Daboecia cantabrica*; Drps, *Dryopteris pseudomas*; Fear, *Festuca arundinacea*; Fral, *Frangula alnus*; Frve, *Fragaria vesca*; Hehe, *Hedera helix*; Hola, *Holcus lanatus*; Hyan, *Hypericum androsaeum*; Hypu, *Hypericum pulchrum*; Juef, *Juncus effusus*; Laam, *Lamium amplexicaule*; Lipu, *Lithospermum purpurocaeruleum*; Loper, *Lonicera periclymenum*; Poer, *Potentilla erecta*; Quro, *Quercus robur*; Rare, *Ranunculus repens*; Roca, *Rosa canina*; Rops, *Robinia pseudoacacia*; Ruul, *Rubus ulmifolius*; Saat, *Salix atrocinerea*; Sool, *Sonchus oleraceus*; Stof, *Stachys officinalis*; Taof, *Taraxacum officinale*; Trre, *Trifolium repens*; Uleu, *Ulex europaeus*. (Color figure online)

Festuca arundinacea, Stachys officinalis, *Quercus robur*, *Rubus ulmifolius* and Bryophytes.

Two environmental parameters were fitted significantly over the ordination space: bare soil ($R^2 = 0.44$; $P < 0.001$; Fig. 2a) and LAI ($R^2 = 0.36$; $P < 0.004$; Fig. 2b). Bare soil is clearly related with track habitats where the greatest vales are found (around 25–35% isolines). Young plantations showed lower bare soil values, around 5%, while there was no bare soil in harvested and old plantations (0% isolines). In contrast, LAI showed the opposite trend increasing from left side of the ordination, with tracks and young plantations showing the lower values (4–5 isolines), to right side where both harvested habitats (CC and RCC, 6–6.5 isolines) and old plantations (7 isoline) showed greater values.

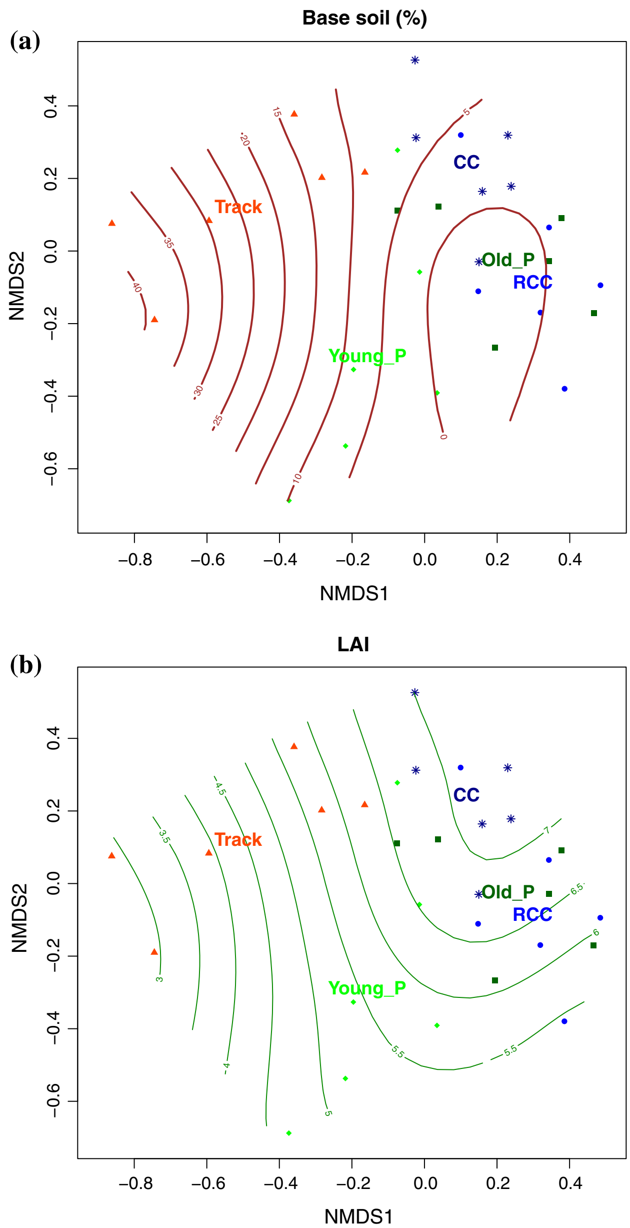

Fig. 2 Isolines representing **a** the predicted bare soil (%) and **b** LAI fitted values by GAM on the first two axis of NMDS ordination of understorey species composition. Symbols represent the sampling quadrats: *red triangles* tracks; *green diamond* young pine plantations; *green squares* old pine plantations; *asterisk* clear-cut stands and *blue dots* restoration-clear-cut stands. (Color figure online)

Understorey tree species composition between harvested and non-harvested habitats

There were significant differences in understorey tree species composition between planted and harvested habitats (PMAV; F-model$_{[3,20]} = 9.91$; $P < 0.001$; $R^2 = 0.60$), accounting for 60% of the variance in tree individuals. These differences were produced because harvested habitats (CC and RCC) were related with native tree species such as *Quercus robur*, *Fraxinus excelsior*, *Corylus avellana* or *Frangula alnus* compared to plantations that were dominated by *Pinus* saplings and by the invasive *Robinia pseudoacacia*. Surprisingly, CC had significantly greater number of native tree species per plot than the rest of habitats (Fig. 3a; 13 ± 3.30 individuals per plot, F-value$_{[3,20]} = 4.84$; P value = 0.011), although tree height was significantly greater in old plantations (Fig. 3b; 6.5 ± 1.06 m; F-value$_{[3,20]} = 11.35$; P value < 0.001).

Understorey vegetation diversity differences between habitats

There were significant differences in understorey plant richness between habitats ($\chi^2_{[4,5]} = 40.75$; P value < 0.001). Tracks showed significantly greater richness than the rest of habitats (32 ± 3.97; Fig. 3c). Young plantations and CC showed intermediate richness (19 ± 0.91), although they were not significantly different from old plantations and RCC (13 ± 1.50). At the same time, there were no significant differences in evenness between habitats (F-value$_{[4,5]} = 1.06$; P value = 0.463; Fig. 3d), with high mean evenness values around 0.78 ± 0.013.

Invasive species presence

Only two invasive species were found, mainly in pine plantations and harvested sites: *Robinia pseudoacacia* and *Cortaderia selloana*. *R. pseudoacacia* appeared in 8/30 quadrats with a mean height of 2–3 m, and being more frequent in old plantations and CC habitats. In contrast, *C. selloana* appeared only in two subplots (tracks and young plantation), but with high cover values around 10 ± 1.11%. At the same time, it is interesting to mention that *Pteridium aquilinum*, being a common native species, dominates in harvested sites (CC and RCC) with extremely

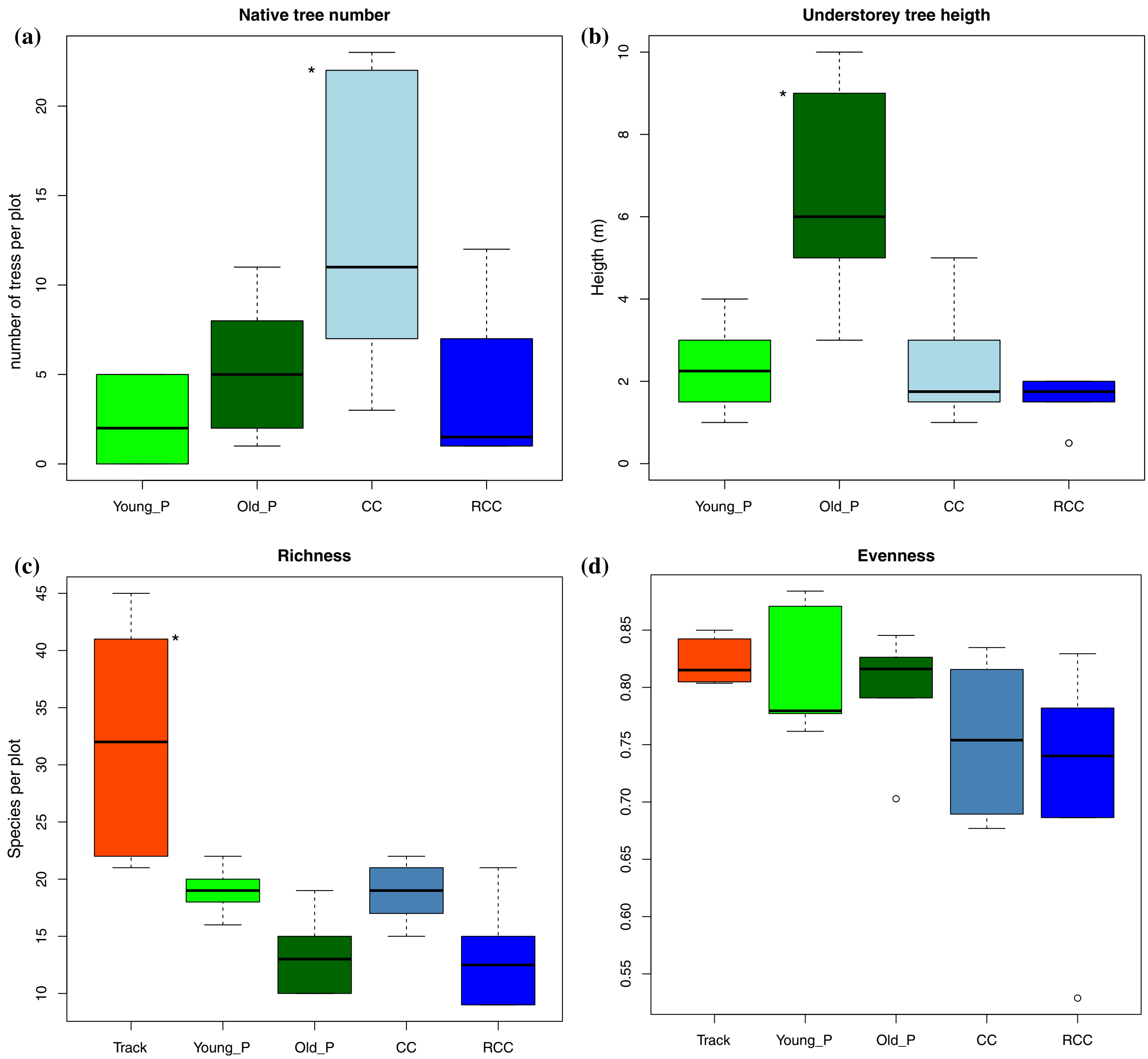

Fig. 3 Comparison of native tree number, height, understorey plant richness and evenness between the five habitats selected. Track, track plots; Young_P, young pine plantations; Old_P, old pine plantations; CC, clear cuts; RCC, clear cuts preserving native tree species. *Asterisk* next to the boxes indicates significant differences (*P* value <0.050) by Tukey's test

high cover and height values (cover 62 ± 8.4%, height 1.10 ± 0.05 m).

Discussion

Our findings suggest that both harvest treatments (CC and RCC) maintained important understorey native species: *Quercus robur*, *Fraxinus excelsior, Festuca arundinacea, Lonicera periclymenum, Stachys officinalis*. Also, both harvest treatments showed no reductions in diversity compared with old plantations; thus, there was not apparent retention effects on species compositional change towards native communities, at least two years after harvest. In contrast, the most disturbed habitat (tracks) showed clear differences in composition and richness driven by the colonization of early-successional species, while young plantations showed a mixture of early-successional and forest interior species. In general, our results indicate that after both clear-cut methods the remaining vegetation established by natural succession could be used to achieve initial native oak forest establishment at relatively low costs, almost for some native tree (*Quercus robur*, *Fraxinus excelsior* or *Corylus* *avellana*) and understorey plant species (*Lonicera periclymenum, Stachys officinalis*).

Understorey compositional differences between habitats

Our results showed clearly that main plant compositional differences were found in tracks ($R^2 = 0.20$). It seems that the soil disturbance produced by tracks and later restoration works (i.e. pushing the soil back onto the tracks) changed the plant species composition compared to plantation habitats (i.e. related with abiotic changes; Alday et al. 2011). Tracks plant community was dominated by early-successional species such as *Ranunculus repens*, *Taraxacum officinale*, *Centaurea* spp. and was environmentally characterized by high bare soil and low LAI compared with plantation habitats. These results were expected since it has been demonstrated elsewhere that tracks create edge-effect conditions (Hill and Curran 2001) and soil disturbances (e.g. increase of bare soil) creates conditions similar to agricultural and degraded lands where bare soil dominates, being susceptible to colonization by early-successional and disturbance-adapted species (Alday et al. 2011). It is worth mentioning that track geomorphology was restored putting back the removed soil (Bhuju and Ohsawa 1998), and as a consequence the natural succession happened over unstructured soil. At the same time, plant species composition of young plantations was also significantly different from harvested and old plantations, but proportionally more similar to tracks. The compositional differences of young plantation habitats and harvest or old-plantation habitats were mainly produced by the abundance of early-successional species (*Sonchus oleraceus*, *Bellis perennis*) mixed with mid-successional species (*Crataegus monogyna, Rosa canina*). Here, in young plantations, still enough light reaches the soil surface since the tree crown closure is not full developed (Heithecker and Halpern 2006) and planting soil disturbances somewhat remain (D'Amato et al. 2009). In any case, the description of vegetation changes in these areas can give us insights for forest management, as it lets us to identify which species can colonize and how. This should be the first step to identify the local availability of some characteristic understorey species of these ecosystems for planning future passive restoration plans.

A particularly noteworthy result was the lack of understorey plant compositional differences between two harvest treatments (CC and RCC) and old plantation habitats. It seems that from a compositional perspective, the drastic effects of clear-cutting on environmental conditions and associated disturbance produced by harvest operations (González-Alday et al. 2009) were not able to produce significant changes in the understorey plant species composition or LAI at least at short term (2 years). Surprisingly, similar results are not commonly described in literature (see Decocq et al. 2004; González-Alday et al. 2009; Kern et al. 2014), since most studies reported understorey compositional changes. In any case, this is an important result for the implementation of restoration programmes based on natural succession in pine plantations of northern Spain. Here, CC and RCC, when implemented sensitively, will maintain similar plant compositional structure of old plantations and native surrounding communities (Onaindia et al. 2013), being a positive ecological outcome in our case for the restoration of oak-mixed forest characteristic species.

Understorey tree species composition

Considering only native tree species composition, clear differences between both harvested (CC and RCC) and old-plantation habitat appear. Surprisingly, both harvested sites showed more abundance of native tree species with restoration interest than old plantations (e.g. *Quercus robur*, *Fraxinus excelsior* or *Corylus avellana*). Similar results have been found in temperate zone plantations where the planted species promote habitat conditions for understorey establishment of native tree species such as oaks and ashes (Truax et al. 2000; Cogliastro and Paquette 2012). Surprisingly, the number of native tree species present was significantly greater in CC than in other habitats, although the understorey tree height was similar to RCC. It seems that this outcome is produced by the colonization of CC sites by tree species present in the surroundings which increased tree number per plot but not their size (Piiroinen et al. 2015). Anyhow, these results suggest that both harvest treatments CC an RCC are adequate to maintain the restoration potential of native tree species complement in this area. Nevertheless, the reduction in native tree height in harvested habitats compared to old plantations may be produced by: (1) a negative effects such as physical destruction and damage of native tree component are produced by harvest operations (Newmaster et al. 2007; González-Alday et al. 2009) or (2) because new young seedlings or germinants that were released by the removal of the overstory reduced the mean habitat height. In any case, our results indicate that natural replacement of planted pines by native tree species using harvest is possible. These results agree with Brockerhoff et al. (2008) that advises the need to study the gradual replacement of plantations by native trees and especially the use when possible of low-cost harvest methods to recover native forest species. When harvest is properly done over old plantations maintaining tree and shrub components it will be easier to restore native forest (Alday et al. 2010). Thus, natural succession after harvest can be seen as a low-cost tool to restore native forests in this area, if there is no excessive grazing by domestic or wild ungulates, and there are available seed sources.

Vegetation diversity differences between habitats

The understorey compositional changes between habitats were only followed by differences in understorey species richness when tracks were considered. In general, the disturbance produced by tracks increases the plant species richness in comparison with other habitats. It seems that natural colonization of tracks by early-successional species, which were favoured by the modification of soil-conditions (Alday et al. 2010), were the cause of this richness increase.

A trend observed in coniferous plantations worldwide is that understorey species richness increases as pine plantations mature (Nagai and Yoshida 2006). Also, it is well known that harvest treatments and especially clear-cutting produce important increases in understorey species richness (González-Alday et al. 2009). However, we did not find significant differences in richness and evenness between both harvested habitats (CC and RCC), young and old plantations. It seems that in this area there are other filters driving the high richness found in all forest habitats. For example, the origin of the planted species, the site land-use history, and the landscape or forest management have been described as drivers of understorey species richness in plantations (Carnus et al. 2006; Macdonald and Fenniak 2007). Therefore, further research should be needed here to identify the relative importance of these factors to implement effective restoration plans.

Invasive species presence

It is well known that exotic invasive species are a global change consequence that threatens ecosystems and native species conservation (Mack et al. 2000; Levine et al. 2003), producing important economic loss (Pimentel et al. 2005). Environmental disturbance has been described as the main factor favouring ecosystems invasiveness (Burke and Grime 1996; Alpert et al. 2000), since disturbance creates 'windows' of opportunity for invasive species colonization and growth (Hobbs and Huenneke 1992; Davis et al. 2000). In this regard, human disturbed ecosystems usually have more invasive species than natural habitats (Hobbs and Huenneke 1992). Here, two invasive species were present in the studied habitats; i.e. *R. pseudoacacia* and *Cortaderia selloana*, even though all sites can be considered human perturbed. Surprisingly, both invasive species clearly prefer different successional sites, i.e. *R. Pseudoacacia,* being a tree species, is more abundant in old plantations and harvested sites (CC), while *Cortaderia selloana,* being a grass, prefers early-successional sites such as tracks and young plantations. It seems that invasive species functional traits have some influence on the selection of invaded habitat (Lee 2002). In any case, appropriate forest management and restoration plans will be needed in the future in order to eliminate invasive species from plots under restoration. For example, the negative impact of *Robinia* on understorey natural vegetation, diversity and soil properties seems remarkable worldwide (Medina-Villar et al. 2015; Crosti et al. 2016). In some cases, it may become the dominant tree species that can lead to the formation of monospecific *Robinia* stands. Thus, reducing or eliminating invasive species can have strong influence in the net outcome of native vegetation recovery and in the recruitment dynamics of native tree species in plantations under restoration.

It is interesting to highlight that *Pteridium aquilinum* was the dominant species in harvested sites (CC and RCC) and old plantations, having the highest cover and frequency (>50%). Although it is not an invasive species in Spain, *P. aquilinum* is one of the most aggressively spreading species in the world, being an invasive species in some regions (Marrs et al. 2013). It tends to colonize sites like grasslands, shrubland, harvested areas and pine plantations (Xavier et al. 2016). Nevertheless, if its presence is very high, such as in the studied plots, it can make the native forest restoration difficult by reducing understorey species establishment (Marrs et al. 2013; Alday et al. 2013). However, in our study, we have found no regeneration problems over the short term. Here, we expect that the growth of native vegetation (trees and shrubs) would reduce the dominance of *Pteridium* over the long-term by reducing available light and outcompeting it.

Conclusions

Both clear-cut methods (CC and RCC) applied over old pine plantations in the studied area preserved to some extent the native understorey tree and plant species composition, being most of the species preserved targets in the restoration programme carried out in the area (Onaindia and Mitxelena 2009). It is known that plantations catalyse the regeneration of most characteristic native woodlands tree, fern and some herb species in this area (Onaindia et al. 2013). Thus, forest management methods such as CC and RCC in these temperate ecosystems might enable the reorientation of pine plantations towards species compositional states that are more similar to natural oak habitats in a low-cost way (Rescia et al. 2010). Therefore, short-term outcomes using appropriate forest management practices (e.g. CC and RCC) and natural replacement of species to change plantations to native forest communities is possible. In any case, it would be interesting to implement supplementary management measures, such as invasive species elimination or target species seeding (e.g. oaks), to maintain local biodiversity and introduce native woodland species not present in the area (Onaindia et al. 2013).

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