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Enrichment-planting with pines alters fuel amount and structure in endangered *Araucaria araucana* forests in northwestern Patagonia, Argentina

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ABSTRACT

The introduction of non-native tree species for large-scale afforestation may alter the fire regime of native ecosystems by modifying fuel proprieties. We quantified changes in fuel abundance and structure resulting from the establishment of commercial *Pinus* spp. plantations in *Araucaria araucana* ecosystems in northwestern Patagonia, Argentina. Specifically, we assessed the amount, distribution, and condition (live/dead) of surface and standing fine fuel in *A. araucana* stands with mature pine plantations (*i.e.* > 20 cm dbh) and in stands dominated only by *A. araucana* (control). Our study shows that both types of stands are prone to wildfires, but pine plantations have fuel characteristics that imply greater flammability due to higher fuel load and vertical continuity in the understory and in the overstory canopy. In the absence of fuel mitigation practices, *A. araucana* stands with plantations exhibit greater flammability than the control *A. araucana* stands, potentially promoting the occurrence and spread of fires of greater severity. This study contributes to understanding the effects of enrichment planting of pines, and possibly pine invasions, on *A. araucaria* ecosystem flammability and their potential consequences on fire behavior.

1. Introduction

Planting of fast-growing tree species for large-scale afforestation has facilitated invasions by non-native tree species in many regions worldwide, often with unintended consequences (Richardson, 1998). Although plantations of introduced tree species contribute significantly to the economies of many countries, there often are also important ecological drawbacks associated with their widespread use in forestry. In afforested areas, planted non-native species can impact negatively on soil (*e.g.* Céspedes-Payret et al., 2012), hydrology (*e.g.* Scott and Prinsloo, 2008; Milkovic, et al., 2019),wildlife habitat and native biodiversity, food resources from native flora and fauna, and numerous other ecological processes (*e.g.* Lara and Veblen, 1993; Armstrong and Van Hensbergen, 1996; Araujo and Austin, 2015; Zaloumis and Bond, 2011; Principe et al., 2015). In addition, the traits that make some tree species highly suitable for forestry, like rapid growth rates and early sexual maturation, also allow them as escaped plants to spread quickly and to rapidly modify native environments (Richardson, 1998; Williams and Wardle, 2005). A widely recognized threat is the role of non-native trees (*e.g. Pinus* spp. and *Eucalyptus* spp.), both in plantations and as escaped invasive plants, in potentially increasing vegetation flammability and in turn contributing to altered fire regimes of surrounding native ecosystems, resulting in ecosystem-level transformations (Peña-Fernández and Valenzuela-Palma, 2008, Bowman et al., 2019, Hermoso et al., 2021, Castro-Díez et al., 2021).

Both planted and invading non-native plants may lead to changes in fuel load and vegetation structure, having crucial implications on fire activity, possibly altering the spread, severity and extent of fire events. A new species in the ecosystem may modify fuel through changes in the amount or spatial arrangement of the fuel load (Dibble and Rees, 2005)

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or it might strongly alter the flammability of the community by contributing allochthonous chemical compounds that may be flammable (Pausas and Keeley, 2014). This may lead to a change in the flammability of the ecosystem (Brooks et al., 2004; Mandle *et al.,* 2011) and endanger native plants that are adapted to a different fire regime (Keeley et al., 2011).

In northwestern (NW) Patagonia (Chile and Argentina), there is a particular type of forest vegetation dominated by the paleoendemic conifer *Araucaria araucana* (Mol.) C. Koch. (monkey puzzle tree, or pewén) which is globally recognized as an endangered species (Premoli et al., 2013, Sanguinetti et al., 2023) and is of central cultural significance for local (Pewenche) Indigenous Mapuche People (*e.g.*, Aagesen, 2004; dos Reis et al., 2014). *A. araucana* and some associated co-existing species (*e.g. Nothofagus antarctica*) have adaptations that make them relatively resistant and/or resilient to fire (Veblen et al., 1996, González and Veblen, 2007). Large (*e.g.* > 40 cm dbh) individuals of *A. araucana* have a thick, fire-resistant bark that develops into distinctive polygonal plates (Angli, 1918). Moreover, upon reaching an age of approximately 100 years, the basal branches begin to detach, generating umbrella-shaped crowns where the foliage is relatively distant from surface fires (Veblen et al., 2003). In some locations, mainly in the drier eastern slopes of the Andes, within a matrix of vast Patagonian steppe and shrublands, *A. araucana* stands consist of sparse individuals with limited or no canopy connections, thus reducing the chances of fires spreading among crowns. If fire reaches the crowns, individuals may still survive (under less extreme burning) and develop epicormic shoots on the branches and trunk (Schilling and Donoso, 1976). In addition, the terminal meristems of branches are protected by differentiated leaves that help trees survive and continue to grow (Montaldo, 1974). Altogether, these conditions favor the tolerance and resistance of this species to low and medium intensity fires (Alfonso, 1941; Tortorelli, 1942). Despite these adaptations and stand structure attributes, changes in the fire regime caused by introduced non-native tree species may endanger *A. araucana* ecosystems and surrounding natural and human-modified environments.

In the area encompassed by *A. araucana* distribution, pine plantations first appeared in the 1970s and continue to be established today (Schlichter and Laclau, 1998). The most widely planted species in *A. araucana*-dominated landscapes are *Pinus ponderosa* Dougl. (Laws) (ponderosa pine), *Pinus contorta* Dougl. (lodgepole pine) (Sarasola et al., 2006) and *Pinus radiata* D. Don. (radiata pine)(INFOR, 2020). The vast majority of these monoculture plantations are established on open ecosystems or where native forests had been cleared due to deforestation, fires or grazing (Lara and Veblen, 1993, Pauchard *et al.,* 2015). However, a small fraction of these plantations were established by planting pines under the canopy of adult *A. araucana* in relatively open stands, without felling individual trees of this native species. This method, currently known as "enrichment planting" (also named strip-, gap- and under-planting; Forest Restoration Research Unit, 2008), is defined as the introduction of valuable species to degraded forests without the elimination of valuable individuals already present (Weaver, 1987; Montagnini et al., 1997). The objectives of enrichment planting are diverse and site-specific, but generally this method is used to increase the economic value of the forest and curb forest degradation (Weaver, 1987). In the specific case of *A. araucana* vegetation, enrichment planting has been practiced as a way of producing a commercial timber source (*i.e.* pines) while simultaneously protecting a threatened tree species (*A. araucana*) from further logging. However, although this method can protect the native species from logging enrichment planting procedure raises concerns about the mid- to long-term persistence of *A. araucana* stands due to multiple factors, such as competition for resources (*e.g.* light, soil moisture) negatively impacting the recruitment of *A. araucana* saplings and potential changes in the fire regime.

It seems logical to expect that enrichment planting would increase stand flammability because of the known flammable traits of these pine species (Keeley, 2012; Cóbar-Carranza et al., 2014). Studies conducted in other Patagonian ecosystems (*e.g.* steppe and shrublands) have shown that pine plantations and non-forested areas invaded by pines have elevated fuel loads and altered potential fire behavior (Taylor et al., 2017, Paritsis et al., 2018). Widespread invasions of introduced pines into the natural and semi-natural systems contiguous to the plantations may contribute to greater potential for fire spread at a broad landscape scale (Higgins and Richardson, 1998; Sarasola et al., 2006). Nevertheless, the possible alteration of fire regimes in *A. araucana* ecosystems in response to non-native woody species is complex and deserves system-specific studies. Careful evaluation of native and non-native fuel attributes is needed to understand and predict potential changes in fire regimes in those ecosystems where non-native species are planted or invading. In our study, while we focus on the fuel characteristics of stands with enrichment planting those fuel characteristics are likely to become increasingly widespread as pines continue to invade the native vegetation.

The objective of this study was to evaluate changes in fuel amount and structure due to the enrichment planting of two *Pinus* species (*P. ponderosa* and *P. contorta*) in *A. araucana* ecosystems. Specifically, we studied and compared the amount, distribution, and condition (live/dead) of surface and standing fine fuel in mature (dbh > 20 cm) pine plantations established in *A. araucana* stands and in contiguous natural ecosystems dominated by *A. araucana*. We expect that stands of *A. araucana* with mature pine plantations will show more flammable attributes than their counterparts without plantations.

2. Methods

2.1. Study area

The study area is located in the eastern foothills of the Andes mountain range, between 38°50′ S and 38°58′ S in Aluminé county, Neuquén province, Argentina (Fig. 1A). The climatic conditions of this area are governed by Pacific Ocean air masses that bring rain to the Andes, generating a pronounced precipitation and moisture gradient that declines eastward (Barros et al., 1983). Annual rainfall varies from 2500 to 1200 mm/year at from 1600 to 800 m a.s.l., respectively, and decreases exponentially towards the east, reaching 200 mm/year in the steppe (Bianchi et al., 2016; Paruelo et al., 1998). Precipitation (rain and snow) occurs mainly during the cold season (April-September). Summers (December-February) are dry and warm with temperatures of up to 30 °C (de Fina, 1972; Heusser et al., 1988), making summer the season most prone to the occurrence of fires. The sites selected for this study are located at elevations ranging from 1600 to 1200 m.a.s.l., with a mean annual precipitation of *c.* 1600 mm/year, mainly as snow. *A. araucana* can form monospecific or mixed forests, establishing different associations with *Nothofagus* spp. depending on elevation, aspect and soil conditions (CIEFAP MAyDS, 2016). The monospecific *A. araucana* forests have little development of understory and can be found at forest edges on mountains or in isolated patches along the forest-steppe ecotone. In our study area, in mixed forests, *A. araucana* commonly associates with the shrubby subcanopy tree *Nothofagus antarctica* (ñire) at warmer and drier sites, or, at moister sites, with *Nothofagus pumilio* (lenga), reaching heights of *c.* 20 m but still typically much shorter than the tallest emergent *A. araucana* individuals. In these mixed forests, the native bamboo *Chusquea culeou* (caña colihue) forms dense thickets in the understory, often reaching heights > 4 m (Peña et al., 2008).

Extensive timber extraction in our study area began in 1945 with the exploitation of the native forest, mainly *N. pumilio* and *A. araucana*. At the beginning of the '70 s, a conservation policy that prohibited the extraction and commercialization of *A. araucana*'s timber products was implemented by the Argentinean and Chilean governments. This resulted in the beginning of commercial afforestation of the drier steppe with fast-growing non-native species of the genus *Pinus*, including enrichment planting in *Araucaria*-dominated stands (Schlichter and Laclau, 1998). The most widely planted species has been *P. ponderosa* Dougl.

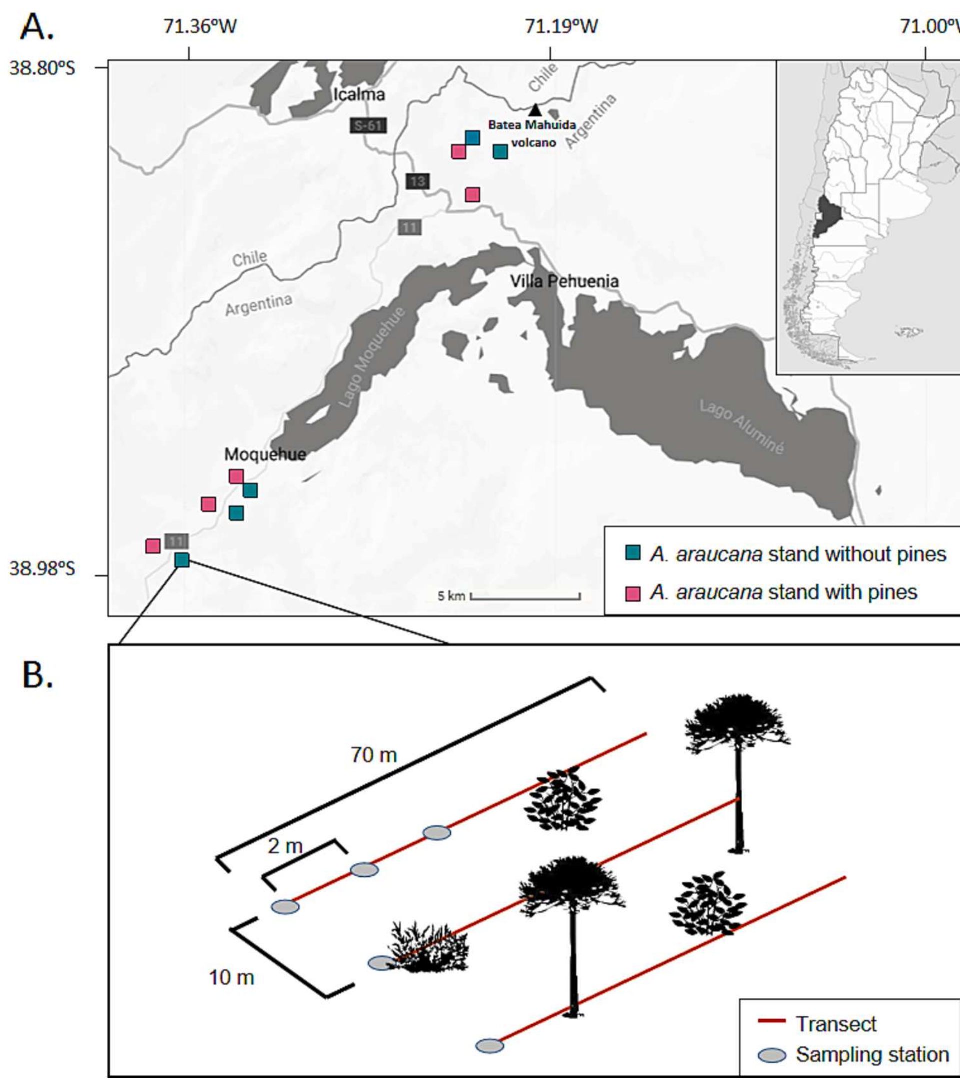

Fig. 1. A. Study Area. The sites selected for this study are located in the municipality of Villa Pehuenia-Moquehue, Neuquén, Argentina. Squares indicate the location of each site (10 sites), and the color shows the corresponding *A. araucana* stand type: without pines (green) or with pines (pink). Sites of different stand type separated by at least 200 m form pairs (5 pairs in total). **B.** Experimental design: scheme of a sampling site. Parallel transects placed in one site and sampling stations distributed along transects (distances are not to scale).

(Laws) (ponderosa pine), followed by *P. contorta* Dougl. (lodgepole pine) (Sarasola et al., 2006). By the year 2012, approximately 14,000 ha in Aluminé county had been planted with these species of *Pinus* (Stecher and Valverde, 2012). Due to the protected status of *A. araucana* against logging and high demand for pine timber and more recently carbon credits, some pine plantations were established under the canopy of adult *A. araucana* trees, avoiding the prior extraction of these protected native trees. Therefore, it is common to find pine plantations mixed with remnant *A. araucana* individuals. Today, the study area is within a jointly administered territory known as "Corporación Interestadual Pulmarí" (CIP, Pulmarí Interstate Corporation) –an administrative entity made up of the Argentine National Government, the Argentine Army, and the Province of Neuquén– where Native American Mapuche populations, livestock producers and private concessionaires are residents and resource users (Stecher and Valverde, 2012).

2.2. Experimental design

We established five pairs of measurement sites with the following contrasting forest structure (*i.e.*, stand types): 1. *A. araucana*-dominated forest with no pines (*i.e.*, stand without pines, or control) and 2. Pine plantations within originally *A. araucana*-dominated stands (*i.e.*, stand with pines; Fig. 2). Two pairs of sites were on the southern slope of Batea Mahuida volcano and three in the vicinity of Moquehue town (Fig. 1A). The stands with pines corresponded to mature (dbh > 20 cm, *c.* 50 years old) plantations of *P. contorta* and/or *P. ponderosa* (current mean density > 800 trees/ha), with a minimum of 15 % canopy cover of *A. araucana* (dbh > 20 cm). At a distance of at least 200 m from each stand with pines, the closest stand without pines with otherwise similar biophysical conditions was selected as a control, with a minimum of 15 % average canopy cover of *A. araucana*. In the stands with pines a very low proportion of pre-existing native vegetation was removed at the time of the enrichment planting. There were no signs of past or current management activities (*e.g.* thinning or pruning of basal branches) in any of the sampled areas, which had approximately 1400 m^2 (70 m × 20 m; Fig. 1B). The abundance of basal branches below 2 m in pine trees evidenced lack of thinning and pruning. Litter, coarse woody debris and branches of varying diameters were found accumulated on the ground surface of plantations, indicating that no understory clearing was done on the area. Also, some young pine individuals and native shade-tolerant plant species were growing under the pine canopy. In addition, there was a general pattern of an approximately 20 m wide swath of younger

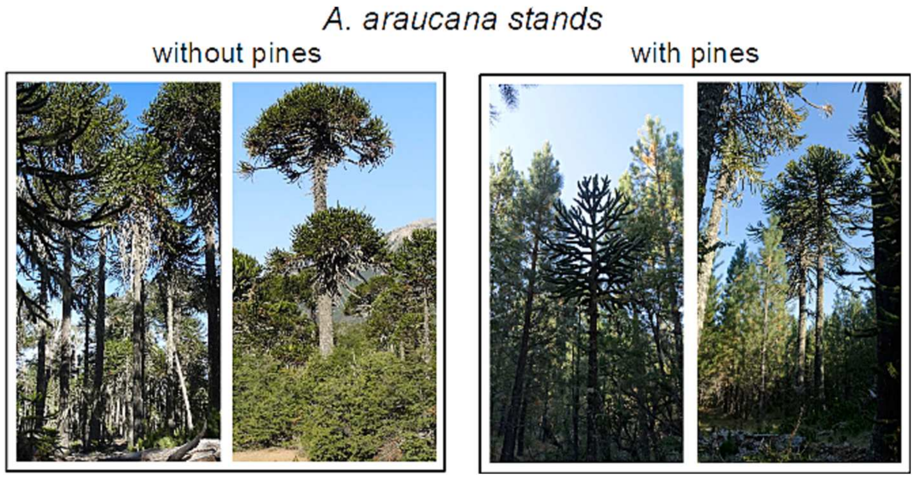

Fig. 2. Photographs of the analyzed stand types, without pines (left) and with pines (right), showing the differences in vertical structure due to the presence of non-native pine species in the *A. araucana* stands. The panels on the left show an *A. araucana* stand without pines, co-occurring with *Nothofagus pumilio* or *N. antarctica*. The panels on the right show *A. araucana* stands with mature pine plantations (*c.* 50 years old) with native vegetation in the understory.

pines invading from the edge of plantations into the native vegetation. None of the stands selected as "without pines" contained any pine individual. In both types of stands, there were small amounts of livestock feces and few overall signs of browsing, indicating a low livestock pressure. At each stand we established three 70 m parallel transects spaced 10 m from each other. Along each transect, fuel sampling stations were placed every 2 m (*i.e.*, 36 sampling stations per transect and 108 per stand; Fig. 1B). Sampling was carried out during the austral summer (February) of 2020, *i.e.*, dry season in the study area.

2.3. Fuel characterization

2.3.1. Vegetation structure and fine fuel measurements

At every sampling station, we characterized the vertical structure of fine fuels following the intercept methodology used by Paritsis et al. (2015) and Tiribelli et al. (2018) and detailed here. With a 25-mm diameter and 4-m height pole divided into 16 25-cm segments, we recorded intercepts between the pole and vegetation (twigs and leaves) of all fine fuels (<6 mm diameter; Anderson, 1990; Rothermel, 1972). For every fine fuel intercept, we recorded the species and its condition, which included both dead and live biomass. Recording the fine fuel condition is important in terms of flammability mainly due to their high surface-area-to-volume ratio and high rate of moisture loss in dead fuels, thus increasing the propensity of fires to ignite and spread (Rothermel, 1972). We quantified fuels from the ground up to 4 m because understory fuels are key for the start and spread of most fires (Pickard and Wraight, 1961). In addition, this procedure allows the characterization of potential ladder fuels, which are those that connect surface fuels with those of the tree crowns and, consequently, allow the spread of surface fires to the tree canopy (Merrill and Alexander, 1987; Dentoni and Muñoz, 2013). On the other hand, the accuracy of the field measurements decreases considerably above 4 m due to the reduced visibility of the pole apex. In each sampling station we also measured maximum shrub height to further characterize understory fuel structure.

To assess crown-level fuel continuity and structure we chose the nearest tree at every fifth sampling station (six trees per transect), with tree defined as an individual with one or more stems with diameter $\geq$ 4 cm at 1.3 m height (dbh) and height $\geq$ 4 m. For each individual (focal tree), we recorded its identity (species), distance from the transect (to estimate tree density), dbh, height, height of basal branches (*i.e.*, lowest height of branch tips), maximum diameter of the tree crown and horizontal distance from the crown to the four closest crowns (*i.e.*, distance between neighbours). This distance was measured at the height where the maximum width of the crown of the focal individual was located and only those trees taller than 4 m were considered, thus excluding individuals of *N. antarctica* which have a tall-shrub physiognomy. For each of the four neighbor trees we measured its height and dbh.

2.3.2. Flammability components

Flammability in ecosystems depends on the functional characteristics of the species present (*e.g.* proportion of fine fuels) and the spatial arrangement (*e.g.* horizontal and vertical structure) of the vegetation (White and Zipperer, 2010). Traditionally, four components define the concept of flammability, which was initially described by Anderson (1970) and Martin et al. (1994) and then modified by White and Zipperer (2010): 1) *ignitability* refers to how easy a fuel starts burning; 2) *sustainability* is the ability of a fuel to continue burning; 3) *consumability* refers to how much of the available fuel can burn in a fire event; and 4) *combustibility* refers to the rapidity of the combustion after ignition. These components are not directly quantifiable, but rather are indirectly measured by various estimators (Prior et al., 2018). To characterize the flammability of *A. araucana* stands, as described below, we used the fuel data as estimators of ignitability, consumability and sustainability. The estimators we used can be related to one or more flammability components.

We estimated ignitability using litter and understory vegetation variables. In each sampling station we measured litter cover and depth, and the near surface vegetation cover (cover below 50 cm height, Keane, 2015). These characteristics are related to the ignitability of a stand because greater litter depth and cover and/or more vegetation in the understory implies a greater accumulation of surface fuels, which are critical for increasing the probability of successful ignition and initial fire spread (Behm et al., 2004). Consumability was estimated with data from the intercept method. At each sampling station, we estimated the amount of dead, live and total fine fuels. We determined the amount of fine fuels as the number of segments with at least one fuel intercept divided by the total number of segments (16) at each sampling station (*i.e.*, proportion of fine fuels). These variables are related to consumability, given that they are good estimators of the amount of fuel readily available to burn (*i.e.*, fine fuel; Behm et al., 2004). Litter depth, litter cover and near surface vegetation cover can also be related to consumability. Vertical fine fuel structure was used to estimate sustainability. In each site we modeled a fine fuel profile of the understory up to 4 m: for every height (16 25-cm segments) we calculated the proportion of sampling stations with fuel presence, considering live and dead fuels together and separately (following Paritsis et al., 2015; Tiribelli et al., 2018). We used this fuel profile to determine the continuity of fine fuel in the vertical dimension, indicating how likely the fire is to spread from the understory to the canopy. Distance between crowns of the focal and neighboring trees served as an estimator of horizontal fine fuel continuity at crown height.

2.4. Data analysis

To compare vertical fine fuel profiles between types of stand we fitted a generalized additive model (GAM) with a binomial distribution (*logit* link function) for each fine fuel condition (dead, live, and total) using the *mgcv* R package (Wood, 2017). GAMs were fitted for the fine fuel proportion (%): the number of sampling stations with fine fuel over the number of sampling stations by height and site. Model predictors included the stand type (without pines and with pines) as a fixed effect, the site and the pair of sites as random effects to account for spatial autocorrelation and the height as a continuous predictor. The effect of the height on the fuel proportion was modelled as a smooth non-linear function using a thin plate spline for each type of stand separately, and the random effects of the site and pair of sites were modelled with thin plate splines allowing the intercept and the height effect to vary (base “fs”, for *factor smoothing*, in the *mgcv* package; Wood, 2017). To check model assumptions, we verified that the overdispersion parameter was less than 1.5 and graphically checked model fit. This analysis allows to estimate a continuous vertical profile of fuels, showing the probability of finding fine fuels at a given height. To compare the probability of fine fuel proportion between stand types, we calculated the predicted mean as a function of height and the corresponding 95 % confidence interval in each stand type.

To compare the remaining flammability proxies between stand types we fitted a Generalized Linear Mixed Model (GLMM) for each measured variable, using the *mgcv* R package (Wood, 2017). We included the stand type as a fixed effect, and the site and pair of sites as random effects. We assumed the following distributions and link functions for the response variables:

- fine fuel proportion: binomial (*logit* link),
- litter and vegetation cover < 50 cm in height: beta (*logit* link),
- distance from the focal tree to the transect and distance between crowns: gamma (*log* link).

All analyses were carried out in R version 3.4.1 (R Core Team, 2020).

3. Results

There were clear differences in the fine fuel amount and structure between the two types of *A. araucana* stands. The fine fuel vertical profiles showed that these differences become evident 2 m above ground level, where a greater load of total fine fuel was found in stands with pines and, therefore, greater fine fuel continuity (Fig. 3A). Between the ground and 2 m, the amount of total fine fuel (Fig. 3A.1) was similar between the two types of stands. Only within the first 0.25 m above ground level there was a tendency for a greater amount of total fine fuel in stands without pines (Fig. 3A.1). From 2 m and up to 4 m above ground level, the amount of total fine fuel was greater in stands with pines compared to stands without pines, with these differences increasing progressively with height (Fig. 3A.1). At heights close to 4 m, the total fine fuel in stands with pines reached between 5 % and 37 % greater than in stands without pines. Dead and live fine fuel profiles showed slightly different patterns with respect to the total fine fuel profile (Fig. 3A.2 and 3A.3). The dead fine fuel profile showed no differences between stand types up to 2 m above ground level (Fig. 3A.2). From there, up to 4 m, the amount of dead fine fuel increased progressively in stands with pines, reaching between 3 % and 22 % greater than in stands without pines near 4 m height. Live fine fuel showed no differences between stand types across most of the profile (Fig. 3A.3). However, from 3.1 m to 4 m it was higher in stands with pines.

Independently of height variability, a similar trend was found in the

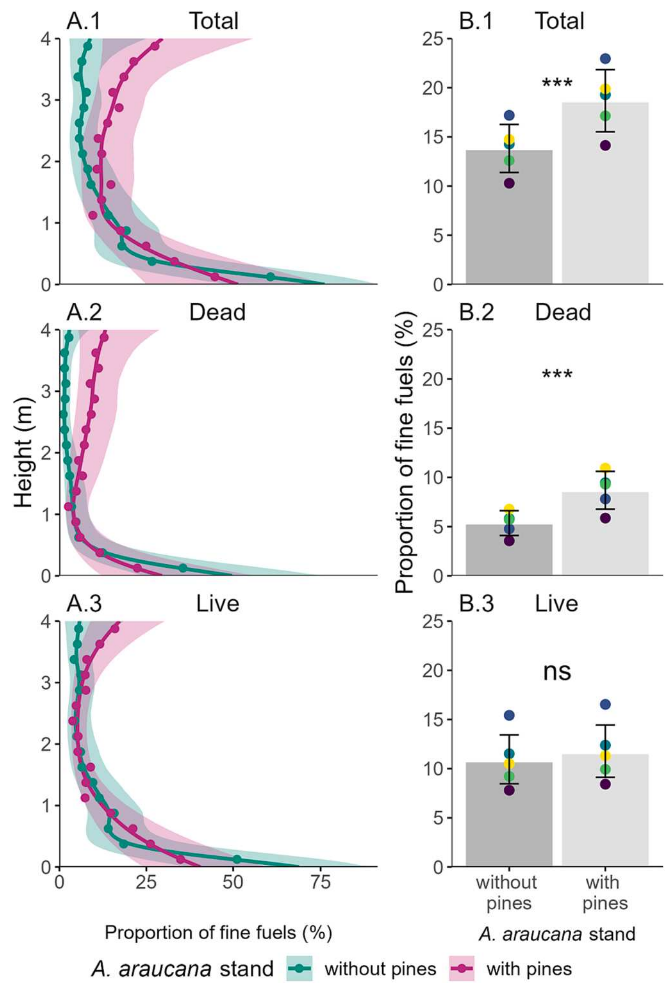

Fig. 3. Fuel amount quantification for each *A. araucana* stand type (without pines and with pines) and for each fuel condition (total, dead and live). **A.** Vertical distribution (meters) up to 4 m height of the mean fine fuel proportion (%). The color indicates the stand type. The lines and envelopes indicate the estimated mean fuel proportion with its corresponding 95 % CI, and the points show the observed mean proportion for each 25-cm segment. Note that the response variable is displayed in the y-axis. **B.** Proportion of fine fuels (%) in the understory (up to 4 m) by stand type. The columns indicate the estimated mean proportion of fine fuels, with bars showing the 95 % confidence interval. The points indicate the observed mean for each site and the same color links the paired sites. In all cases the statistical significance is indicated (*$p < 0.05$, ** $p < 0.01$, *** $p < 0.001$)..

proportion of fine fuels for stands with pines versus stands without pines (Fig. 3B). Stands with pines had an average of 5 % more vertical 25-cm segments with presence of fine fuel than stands without pines (Fig. 3B.1). The differences in the presence of total fine fuel between stand types is attributed mainly to differences in the proportion of dead fuel (Fig. 3B.2; 5 % [CI: 4 %; 6 %] in stands without pines and 8 % [CI: 6 %; 10 %] in stands with pines -hereafter, maximum likelihood estimate followed by the 95 % CI in brackets-), since the proportion of live fuel was similar in both stand types (Fig. 3B.3; 11 % [CI:8 %; 14 %] in stands without pines and 11 % [CI:9 %; 15 %] in stands with pines). The statistical method we employed is an effective approach for our analysis, but it is essential to recognize that there is autocorrelation between contacts at the same sampling station over the pole’s length, which could artificially reduce the uncertainty in our estimates. Considering the autocorrelation would have required a more complex model to be fitted, which was challenging with our data.

At ground level, differences were found in the litter properties, but not in the near surface vegetation cover (Fig. 4). Both values in litter

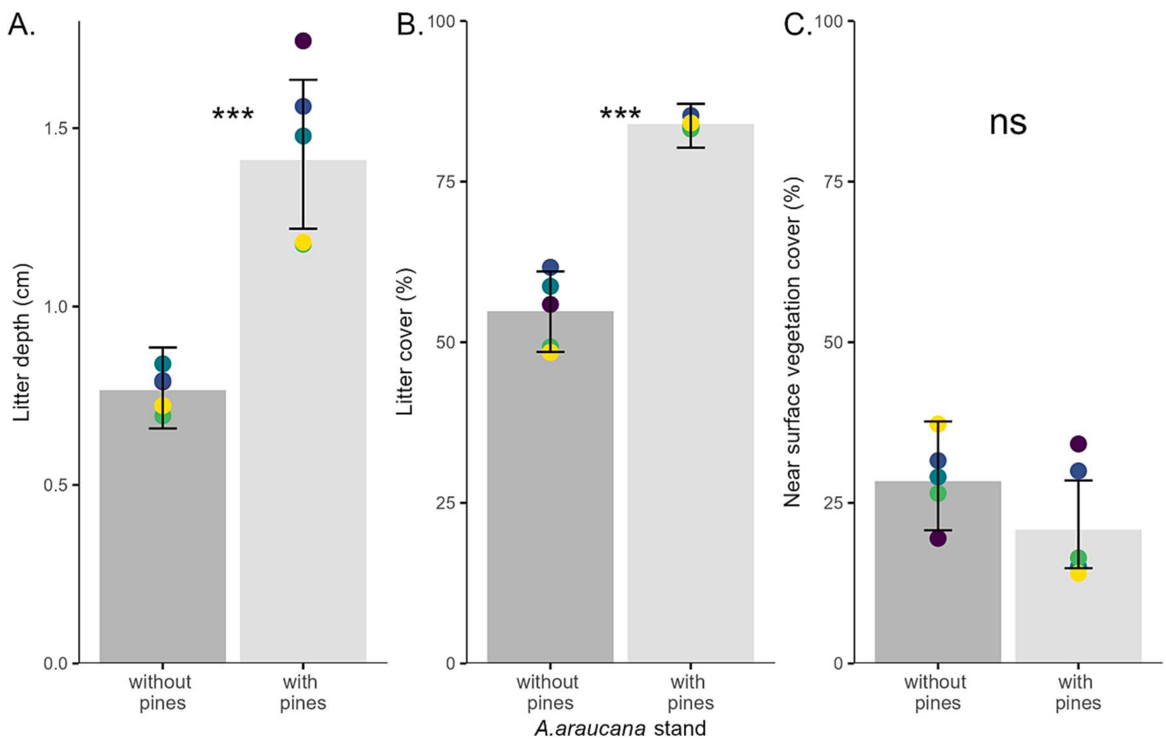

Fig. 4. Litter depth (cm; **A**), litter cover (%; **B**), and near surface vegetation cover (%; height < 50 cm; **C**) for each *A. araucana* stand type (without pines and with pines). Bars indicate overall means and colored dots indicates the mean at each site. Color indicates the pair of associated sites, consistent across panels. Whiskers denote a 95 % confidence interval. Asterisks indicate the level of significance for the statistical test (*p < 0.05, ** p < 0.01, *** p < 0.001).

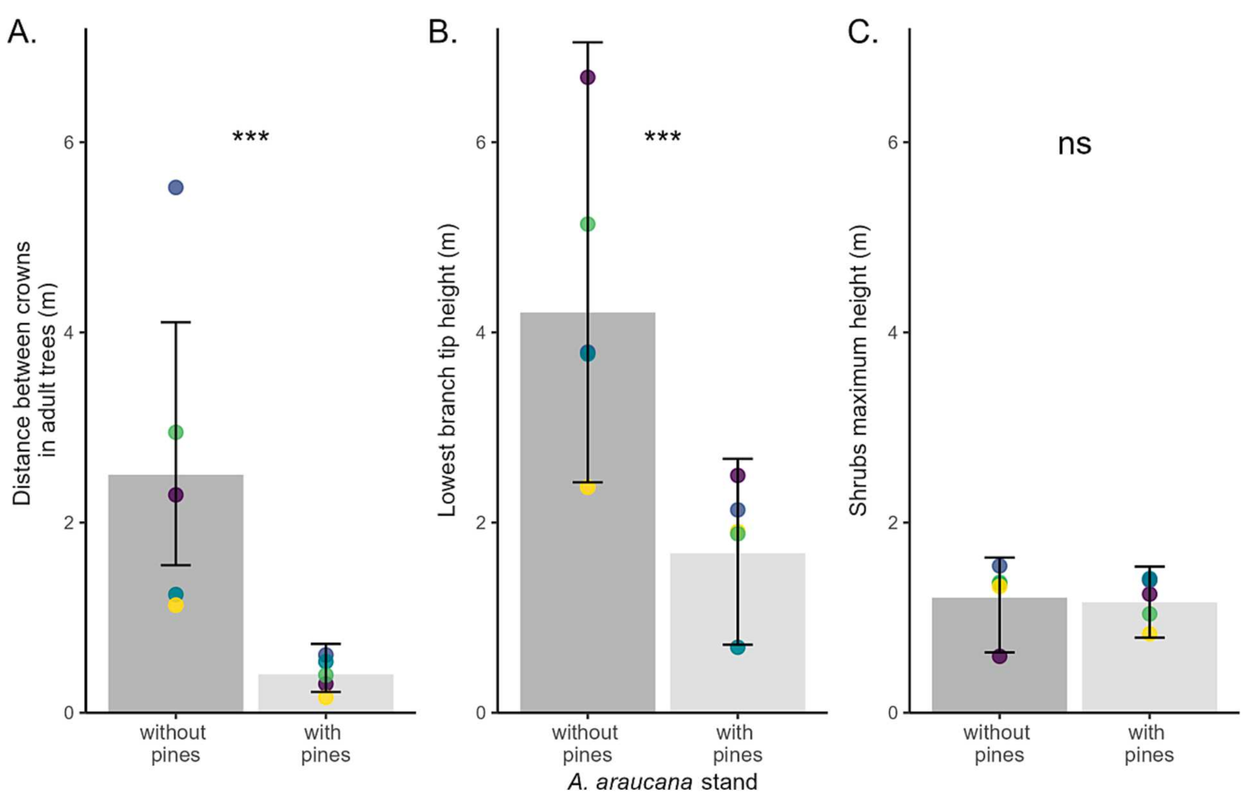

Fig. 5. Mean distance among the focal tree crown and four neighbouring trees (meters; **A**), height of lowest branch tips (meters; **B**), and shrub maximum height (meters; **C**) for each *A. araucana* stand type (without pines and with pines). Bars indicate overall means and colored dots indicate the mean at each site. Color indicates the pair of sites, consistent across panels. Whiskers denote a 95 % confidence interval. Asterisks indicate the level of significance for the statistical test (*p < 0.05, ** p < 0.01, *** p < 0.001).

coverage (Fig. 4A) and depth (Fig. 4B) were significantly lower in stands without pines than in stands with pines. Mean litter cover in stands without pines was 55 % [CI: 48 % – 62 %] with a mean depth of 0.74 cm [CI: 0.64 cm – 0.91 cm], whereas in stands with pines the mean cover was 83 % [CI: 79 % – 87 %] with a mean depth of 1.41 cm [CI: 1.19 cm – 1.67 cm]. In stands without pines, the litter consisted mainly of leaves of *A. araucana* and *Nothofagus* spp., whereas in stands with pines it consisted mainly of leaves of *A. araucana* and needles of *Pinus* spp. The bamboo *C. culeou* also contributed leaves to the litter in the sites where it was present. The near surface vegetation cover (height < 50 cm) was not significantly different between stand types (Fig. 4C).

Tree- and shrub-defined structural parameters were different between the two *A. araucana* stand types (Fig. 5). Our proxy of mean tree density was significantly higher in stands with pines (1290 trees/ha [CI: 880 trees/ha – 2044 trees/ha]) than in stands without pines (274 trees/ha [CI: 182 trees/ha – 412 trees/ha]). The method used to calculate the density could imply an underestimation of the true mean, but it is a systematic approximation that is appropriate for the scope of this study. The distance among tree crowns was significantly lower in stands with pines (0.41 m [CI: 0.22 m – 0.72 m]) compared to stands without pines (2.50 m [CI: 1.55 m – 4.10 m]), indicating higher fuel canopy continuity under pine presence (Fig. 5A). Height of the basal branches of trees was significantly lower in stands with pines (1.77 m [CI: 1.59 m – 2.01 m]) than in stands without pines (mean 4.35 m [CI: 3.82 m – 5.04 m]) (Fig. 5B). No significant differences were found for the maximum shrub height between both stand types (Fig. 5C). The allometric characteristics (tree height, dbh, crown diameter and height of basal branches) of each woody species present in both stand types were similar between stands (Fig. A1 and Fig. A2 in Appendix).

4. Discussion

A. araucana stands with pines exhibit fuel characteristics that imply greater flammability compared to similar stands lacking pines. This indicates that *A. araucana* stands with enrichment planting of pines (particularly in the absence of any fuel management) may have elevated flammability compared to native *A. araucana* ecosystems, thus promoting the occurrence and spread of fires of greater severity and higher tree mortality. These structural changes increased fuel connectivity due to pine planting in the native vegetation and are likely to become more widespread as pines continue to invade the native vegetation. These changes are significant not only because they involve increased flammability within a stand, but also because they can favor more extensive fire spread to the adjacent native ecosystems.

Fine fuel loads within the first 2 m vertical segment were similar between stands with and without planted pines, but at heights greater than 2 m stands with pines had a greater proportion of fine fuels. Whereas fine fuel amount and structure along the vertical profile in stands without pines is mainly explained by the *Nothofagus* species with which *A. araucana* is associated, in stands with pines it is mainly explained by the branches of *Pinus* spp., which also accumulate more fine fuels and have higher content of flammable oils than *Nothofagus antarctica* (*e.g.* Cóbar-Carranza et al., 2014). The height of the pines and the width of their crowns prevent the entry of light, which inhibits the growth and regeneration of the vegetation native in the understory (García et al., 2023; Paritsis and Aizen, 2008). The thick Araucaria-pine canopy decreases the native fuel load in the first few meters above the ground implying a proportion change in the dominant species contributing to the fuel load. In turn, the basal branches of the pines dry up due to the lack of light created by the increased canopy density of the plantation. The dry basal branches of pine trees, the presence of some native shade-tolerant understory species such as the flammable bamboo *C. culeou*, and, to a lesser extent, the dry branches of other woody species suppressed by the lack of light, provide additional dry fuel that favor rapid vertical fire spread. Even though we did not find evidence of change in the original native composition, it is expected that pines affect biodiversity by reducing native species richness (Taylor et al., 2016; Franzese and Raffaele, 2017; García et al., 2023). From 2 m to 4 m in height, the proportion of fine fuel in *A. araucana* stands with pines begins to progressively increase. From 2 m to 3 m in height, the dry branches of the pines are the ones that provide the greatest fuel load and from 3 m upwards the live branches also contribute to the fine fuel load. The high load and vertical continuity of fuels in stands with pines could favor the transition from surface fires to crown fires (Menning and Stephens, 2007; Paritsis et al., 2018), which could seriously damage large *A. araucana* individuals. Overall, our study not only reflects the fact that pines accentuate the difference in terms of fuel loads, but, more importantly, it highlights the difference in the vertical distribution of fine fuels, which may facilitate fire crowning.

Although in both stand types it is possible that fine fuels act as a fuel ladder, the transition to crown fires may be more likely in stands with pines, especially under non-extreme fire weather. First, the lower fuel load and the more discontinuous vertical distribution in *A. araucana* stands without pines imply lower probabilities of crown fire. Although in these stands there are *N. antarctica* individuals with a maximum height near to the lowest branches of *A. araucana* trees that may act as fuel ladders, these trees are mainly live fuel, whereas the basal branches of the pines that reach the same height as the shrubs are mainly dead fine fuel and are more flammable due to their lower moisture content (Bianchi et al., 2019). In addition, compared to stands with pines, the spread of a crown fire among individuals of *A. araucana* in stands without pines would be more difficult because, as we found, there is a lower density of trees and their crowns are further apart, possibly accumulating less crown area to allow propagation. Conversely, in stands with pines, although the load of fine fuel between the ground and 2 m is low on average, their dry basal branches could act as fuel ladders reaching the crowns (Paritsis et al., 2018). In this case, the remaining individuals of *A. araucana* in the plantation would be susceptible to severe damage because the spread of a crown fire would be facilitated by the continuity of the canopy of pines.

Enrichment planting of pines in *A. araucana* stands also changes the composition of dominant species that form the litter and produces an increase in the load of fine fuels on the ground. Several studies provide evidence that the degree of flammability of litter depends on the traits of the species making up the litter. For example, in ecosystems dominated by *Pinus radiata* litter has higher flammability than in temperate native *Nothofagus dombeyi* forests in Patagonia (Franzese et al., 2020). A similar change could be occurring in the flammability of our study system, since in stands without pines the litter is composed mainly of *A. araucana* leaves, which are broad and thick (*i.e.*, less surface-area-to-volume ratio), while in the presence of pines, it is mainly composed of thin pine needles (*i.e.*, more surface-area-to-volume ratio). The greater accumulation of litter in *A. araucana* stands with pines suggests that these have a higher probability of ignition, and that fire can spread more easily at the ground level (Varner et al., 2015). In addition, several studies found positive correlations between litter depth and fire-spread physical variables such as released heat (Ganteaume et al., 2011; Ormeño et al., 2009) and flame height (Ganteaume et al., 2011; Kane et al., 2008). On the other hand, shrub height and near surface vegetation cover are lower in stands with pines, but these differences are not statistically significant. This pattern could be explained by the lower light availability in stands with pines (García et al., 2023). Moreover, the difference may be small because shrub cover is also low in stands without pines, where thickets of *N. antarctica* alternate with open areas of bare ground or grasses. This pattern can be explained by the scarce precipitation in the study area. Thus, in stands with pines, the litter is responsible of the higher flammability at ground level, and it may facilitate fire to overcome the relative vertical discontinuity of the first 2 m, reaching dry branches and generating a crown fire.The results of the present study show that fine fuel loads within the first 4 m is higher in stands of *A. araucana* with mature pine plantations (*c.* 50 years old) than in *A. araucana* stands without pines. Contrary to our finding, Franzese et al. (2022) showed that mature plantations (purely of pines) and advanced invasions of *Pinus radiata* (both approximately 30 years old) have lower total fuel load within the first 4 m of height compared to the native *Nothofagus dombeyi* forest in more mesic habitats further south in Patagonia (*i.e.*, *c.* 42 °S). These differences may be due to the fact that native ecosystems dominated by *A. araucana* within our study area are drier and tend to be more open and with a lower density of understory vegetation than mesic forests dominated by *N. dombeyi* (Veblen et al., 2006). Additionally, in the enriched plantations evaluated in our study, a large portion of the original native vegetation was not removed when pines were planted and therefore large *A. araucana* trees (both canopy and subcanopy) and other vegetation can be found within a matrix of pines. Finally, the enriched plantations we studied have not been actively managed (*e.g.* no thinning nor pruning of basal branches); thus, dead fuels might be higher than in managed plantations (either pure or enriched). Our findings of fuel load and continuity in the *A. araucana* stands with pine plantations are similar to what Cóbar-Carranza et al. (2014) suggest about mature pine invasions in *A. araucaria* forests: they propose that the infilling of the pine-*A. araucana* stands is achieved by increasing canopy fuel load and connectivity compared with *A. araucana* dominated forests. Even though caution is advised when interpreting the flammability of pine plantations and pine invasions as equivalent (Franzese et al., 2022), our study of the effects of enrichment planting of pines on flammability suggests that pine invasion into open stands of *A. araucana* will similarly increase stand-level flammability (Cóbar-Carranza et al., 2014), thus providing a justification for preventive measures to be taken.

For the period 2000 to 2017, a total of 50,858 ha of plantations in Argentinean Patagonia were affected by fires, which represents approximately 3 % of the total burned area by year in this region (SAyDS Reports, 2018). Although they do not occupy an extensive area of the territory yet, plantations are foci where high severity fires can start and easily spread into surrounding native ecosystems (Raffaele et al., 2015). When proper management is not applied, plantations tend to increase their total fuel load as they age (Cruz et al., 2008), and thus increase flammability at both stand and landscape levels (Defossé et al., 2011; Raffaele et al., 2015). Because the area occupied by plantations in Argentinean Patagonia is relatively small, there is still time to take preventive and corrective management actions. As new areas are planted each year, they add to the existing mature plantations and increase density of individuals that can invade adjacent ecosystems (Godoy et al., 2013; Paritsis et al., 2018; Raffaele et al., 2015). Over time, as more of the landscape becomes dominated by pines, whether from planting or invasion, the occurrence of large and severe fires is likely to become more frequent (Godoy et al., 2013). Furthermore, the ongoing and predicted increase in temperature and decrease in precipitation for NW Patagonia is and will promote a decrease in fuel moisture, favoring extreme fire danger conditions (Ellis et al. 2021; Kitzberger et al., 2022). In addition, an increase in convective storm activity in Patagonia is predicted, which would increase the frequency of lightning ignitions (Veblen et al., 2008; Garreaud et al., 2014; Kitzberger et al., 2016). All of these conditions add to the urgency of understanding how pine establishment in NW Patagonia alters the flammability of the landscape to inform active management to reduce fuel loads and fire risk.

Although a local law for the classification of priority areas for conservation which prohibits installation of new enrichment planting in *A. araucana*-dominated stands has been implemented (Neuquén provincial legislation, 2011), such planting strategy has already affected hundreds of hectares. Furthermore, the ongoing expansion of pine invasions is expected to result in similar flammability outcomes as observed in the enrichment plantings of this study. Our study, along with previous researchs, shows that the impacts are not solely confined to fuel properties and fire risk; they extend to the loss of native biodiversity and other alterations in the characteristics of these ecosystems (Taylor et al., 2016; Franzese and Raffaele, 2017; García et al., 2023). Given the significant and growing impact of pines in the study area, it is important to take measurements that eliminate these pines from *A. araucana* ecosystems. The time lag and step-like pattern of conifer invasion suggest that the early stages of invasion are the most amenable to cost-effective management (Wittenberg and Cock, 2001), and focusing on isolated individuals or small groups of them can be crucial in preventing or at least greatly delaying widespread invasion from plantations to neighbouring native ecosystems (Simberloff et al., 2010). Early prevention and control of plantations are the best ways to reduce its economic, social and environmental costs of the widespread of invasions (Pauchard and Alaback, 2002).

Despite practical experience with fuels management in *A. araucana*-dominated stands is limited, as is any long-term monitoring of the outcomes of fuel treatments, we suggest some common sense practices to reduce the impacts of pine planting on fire potential in this ecosystem type (also see Paritsis et al., 2018 for pure pine plantations). Silvicultural practices can be applied to generate breaks in the vertical and horizontal continuity of fuels. For example, branch pruning can be applied to raise crown base height but must be followed by immediate removal or pile burning (outside the fire season) of flammable fine- and coarse-fuel residues that accumulate on the surface when pruning is conducted. Additionally, pre-commercial thinning could be performed to decrease overall fuel loads and potential of crown fire spread. Pruning and thinning must be implemented with caution and appropriate management of the generated residues, because otherwise, residues may increase, rather than reduce, fire hazard (Paritsis et al., 2018). Plantations near high fire risk areas (*e.g.* near settlements) should be prioritized for fire management (Mundo et al., 2013, Lindenmayer et al., 2023), as should incipient pine invasions before they change the flammability of native ecosystems (Taylor et al., 2016). Also, it would be critically important to monitor the medium-term effects of these silvicultural treatments on fuel loads of the non-target native species, such as *C. culeou* bamboos and understory shrubs and small trees (*e.g. N. antarctica*), which in the absence of pines contribute significantly to the flammability of Patagonian ecosystems (Kitzberger et al., 2016).

In conclusion, the enrichment of *A. araucana* stands by pine plantations leads to changes in the structure of the native plant community and increases fuel loads, contributing to increase the flammability of these ecosystems. Despite the current relatively limited area of enrichment planting of pines under open canopies of *A. araucana*, pine invasion into open stands of the native forest is likely to originate a similar increase in flammability over the larger landscape. This study contributes to the understanding of the effects of pine planting, and possibly invasions, on the flammability of *A. araucana* ecosystem. Detailed flammability studies on a larger scale and the adoption of appropriate fuel management procedures in areas of pine planting should be considered to help reduce the risk of fires in the region.

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CRediT authorship contribution statement

Sofía Cingolani: Investigation, Writing – original draft, Formal analysis, Visualization. **Ignacio A. Mundo:** Conceptualization, Methodology, Writing – review & editing, Supervision, Project administration. **Iván Barberá:** Formal analysis, Writing – review & editing. **Andrés Holz:** Conceptualization, Methodology, Writing – review & editing, Supervision, Funding acquisition. **Thomas T. Veblen:** Conceptualization, Methodology, Writing – review & editing, Supervision, Funding acquisition. **Juan Paritsis:** Conceptualization, Methodology, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix

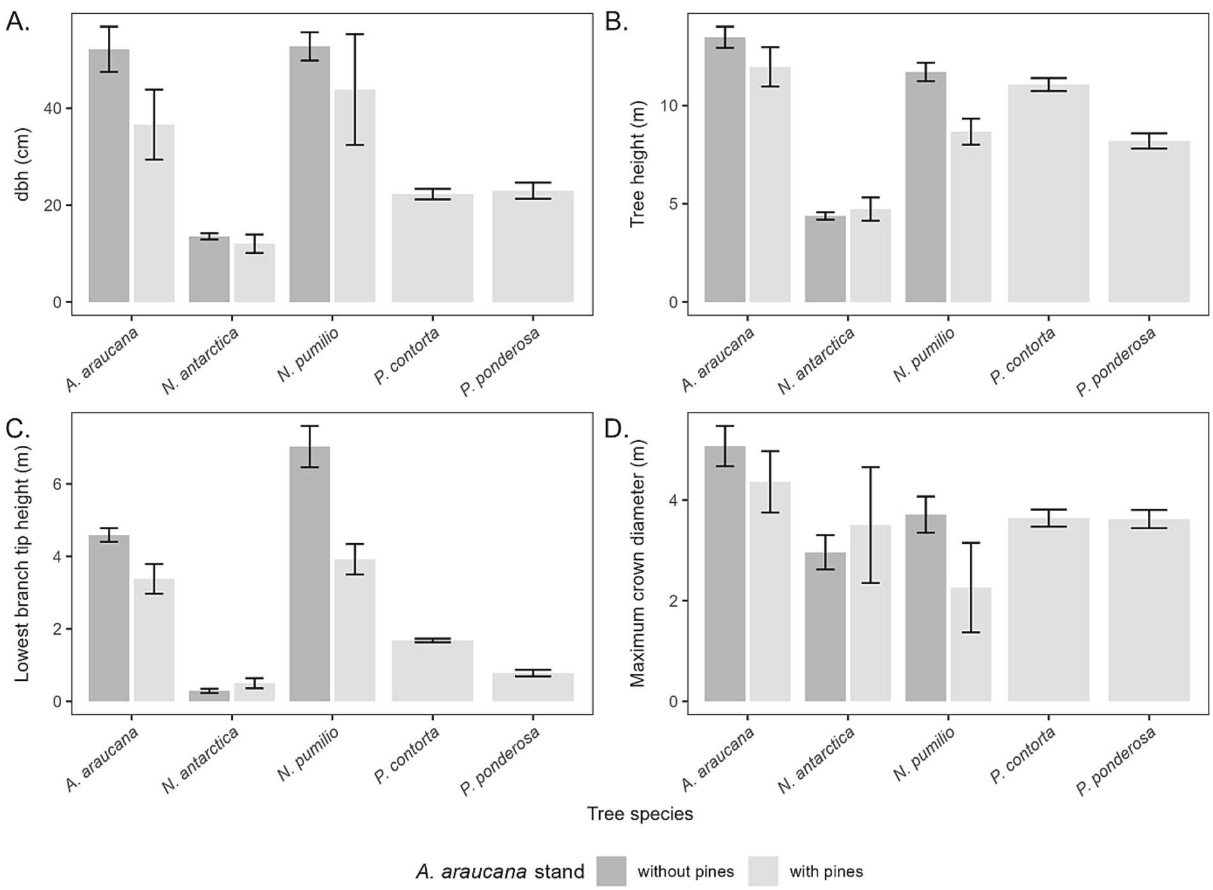

Fig. A1. Stand structure values (mean ± standard error) of the dominant species in A. araucana stand type (without pines and with pines). A. Diameter at breast height (cm) B. Tree height (m) C. height of lowest tree tip branches (m) D. Maximum crown diameter (m). For each variable (diameter at breast height, tree height, height of lowest tree tip branches, maximum crown diameter).

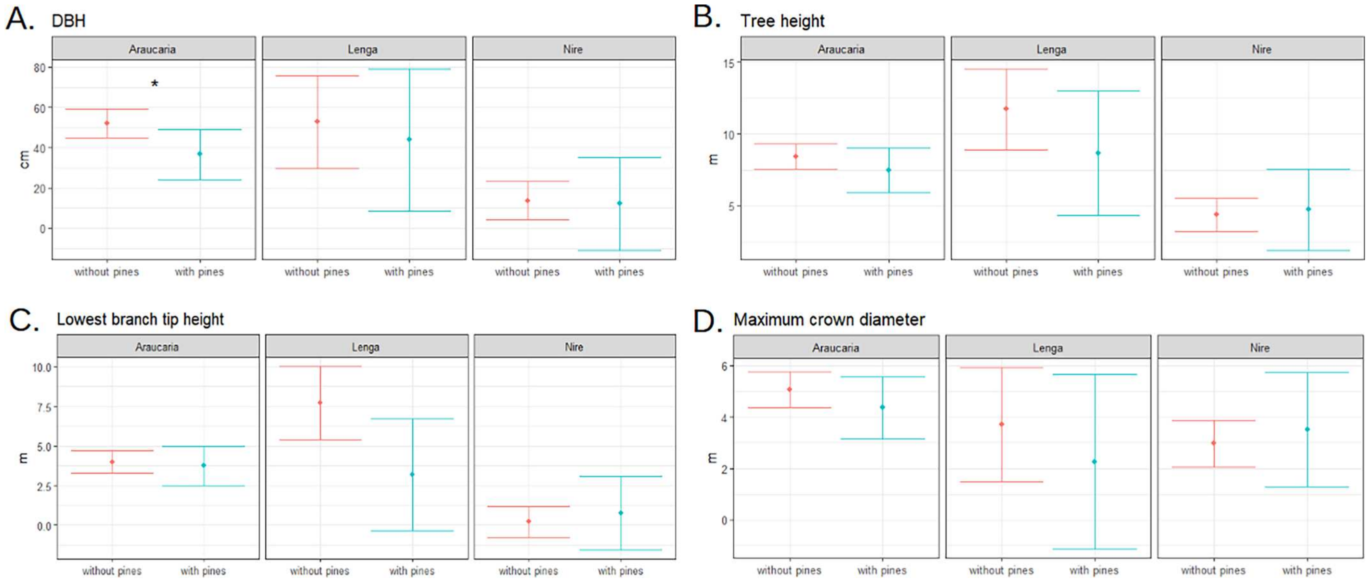

Fig. A2. We fitted a Generalized Linear Model (GLM), using the *mgcv* R package (Wood, 2017). We included the interaction between stand type and the species as fixed effect. We assumed a normal distribution for every variable analyzed. There was no evidence of differences between same species in different stand types. Stand structure mean values of the dominant tree species in *A. araucana* stand type (without pines and with pines). Dots indicate the mean value. Color indicates the stand type, consistent across panels. Whiskers denote a 95 % confidence interval. Asterisks indicate the level of significance for the statistical test (*$p < 0.05$, ** $p < 0.01$, *** $p < 0.001$). A. Diameter at breast height (cm) B. Tree height (m) C. height of lowest tree tip branches (m) D. Maximum crown diameter (m).

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