

RESEARCH ARTICLE

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New Zealand Journal of Forestry Science

Silviculture of South American temperate native forests

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(Received for publication 13 July 2021; accepted in revised form 22 November 2021)

Abstract

Background: South America has the largest area of temperate forests in the Southern Hemisphere, which grow in diverse site conditions. The aim of this paper is to review the practices of silviculture applied and recommended for these temperate forests, and to discuss prospects to develop new silvicultural proposals to improve sustainability, adaptation and *in-situ* conservation of forest ecosystems.

Methods: We reviewed the silviculture knowledge in four major forest types: 1) The *Nothofagus*-dominated forests of south-central Chile; 2) the Angiosperm-dominated evergreen forests; 3) the *Nothofagus* and *Austrocedrus chilensis* (D. Don) Pic. Serm. & Bizzarri forests in the Argentinean Northern Patagonia; and 4) the Cool temperate *Nothofagus* forests and Magellanic rainforests.

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Abstract continued

Results: In Chile, both *Nothofagus*-dominated and Angiosperm-dominated evergreen forests are diverse in tree species, and mixed-species silviculture with commercially valuable species of variable shade tolerance is most promising. Some secondary forests can reach growth rates as high as $20 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$. After thinnings, stands with 35–60% of residual densities have shown the best responses in growth. Even-aged silvicultural methods have shown a rapid reorganization and development of new cohorts, although, where *Chusquea* species are conspicuous, regeneration establishment requires controlling competing vegetation. Preliminary results also show interesting prospects for single-tree selection cuts in uneven-aged forests. East of the Andes, in Argentina, mixed *Nothofagus* forests and pure and mixed *Nothofagus* and *A. chilensis* forests occur. The shelterwood method has been the most practised and successful in the mixed *Nothofagus* forests, with abundant regeneration, and good growth rates. For *A. chilensis* forests, an adaptive management approach is proposed, conditioned by the cypress disease attack (e.g., *Phytophthora austrocedri*). Conversion of pure *A. chilensis* to mixed *A. chilensis*–*N. dombeyi* forests could increase growth considerably. Finally, in the cool temperate *Nothofagus* forests and Magellanic rainforests, shared by Chile and Argentina, the regular shelterwood cuts have been the most common silvicultural method, with massive regeneration a decade after the regeneration felling. Since the two major *Nothofagus* species in these forests regenerate well in gaps, uneven-aged silviculture seems also promising.

Conclusions: There is abundant knowledge about the silviculture of these forest types. However, there are opportunities for several silvicultural systems to better contribute to sustainable forest management, reverse forest degradation, and cope with climate change challenges, primarily through developing mixed and single-species productive and carbon-rich forests, with greater adaptive capacity.

Keywords: *Nothofagus*-dominated forests; Angiosperm-dominated evergreen forests; pure and mixed *Austrocedrus chilensis* forests; cool temperate *Nothofagus* forests; Magellanic rainforests; Valdivian temperate forests; Magellanic subpolar forests

Introduction

Temperate forests in the Southern Hemisphere are scarce relative to those in the Northern Hemisphere (Dinerstein et al. 2017). The largest tract of temperate forests in the Southern Hemisphere is the one shared by Chile (14 million ha; CONAF 2021) and Argentina (3.6 million ha, Mohr Bell et al. 2019). These are included in two ecoregions: the Valdivian Temperate Forests and the Magellanic Subpolar Forests (Dinerstein et al. 2017).

In South America, temperate forests occur between 33°S and 55°S latitude, at the southernmost tip of the continent. Several forest types are found in the region. They represent a biogeographic forest island, surrounded by different physiognomic and taxonomic types of vegetation. Composition and distribution of these forests are regulated by environmental gradients (e.g., temperatures, precipitation) in latitude and longitude, the latter especially for the effect of the Coastal range in Chile, and the Andes range between Chile and Argentina. Both ranges act as obstacles for the humid winds coming from the Pacific Ocean, but especially the Andes (Loguerio et al. 2018b). The climate is characterised for its moderate to low temperatures that decrease with latitude. Precipitation increases progressively from north to south, being usually in the range of 3,000–5,000 mm yr⁻¹ on the western slopes of the Coastal and Andes Mountains south of 38°S, where humid air masses come from the Pacific Ocean. East of the Coastal Mountains, in the intermediate depression, precipitation declines to 2,000 mm yr⁻¹. On the rain shadow of the Andes, 50 km east from the highest peaks, precipitation drops dramatically to around 500 mm yr⁻¹, and an ecotone occurs between the forest and the steppe (Veblen et al. 1996; Kitzberger 2012; Soto et al. 2021). Most of this ecotone territory is located in Argentina, but farther south (from 43°S) it also occurs in Chilean territory (Pisano 1977).

These precipitation differences result in the occurrence of mixed and single-species forests. From north to south, and from west to east of the Andes, forest composition and structure become less complex (Veblen & Alaback 1995; Bannister et al. 2012; Donoso 2015; Loguerio et al. 2018b), i.e. from the mixed Valdivian Temperate Forests to the Magellanic Subpolar *Nothofagus*-dominated forests. In the middle, in northern Patagonia, conifers such as *Araucaria araucana* (Molina) K.Koch, *Fitzroya cupressoides* (Molina) I.M. Johnston and *Austrocedrus chilensis* (D.Don) Pic. Serm. & Bizzarri (this especially in Argentina) are common in more stressful site conditions, such as the shallower soils and lower temperatures at higher elevations (*Araucaria araucana* and *F. cupressoides*, both of which are more abundant in Chile), drier conditions and rocky sites in *Austrocedrus chilensis* forests (more abundant in Argentina), or in humid areas on nutrient-poor soils in *F. cupressoides* forests (La Manna 2005; Veblen et al. 2005). West of the Andes, in Chile, between 33°S and 37°S, there is the Mediterranean Sclerophyll Forest, and between 37° 25'S and 43° 20'S the Valdivian Temperate Rainforest. North Patagonian rain forest covers the region between 43° 20'S and 47° 30'S, and further south the Magellanic rainforest occurs (Veblen et al. 1983). At relatively high elevations along the whole region, pure *Nothofagus pumilio* (Poepp. & Endl.) Krasser subalpine forests are present, and short *Nothofagus antarctica* (G.Forst.) Oerst. forests or tall shrublands occur on a variety of sites, mainly on those poorly drained, cold valley bottoms, and in the ecotone. East of the Andes, both in Argentina and Chile, along the strong precipitation gradient from 37° 30'S to 55°S, the subantarctic *Nothofagus* forests are found. They are mainly stands of *Nothofagus* sp. (pure, or mixed in the ecotones, with the species *N. antarctica*, *N. pumilio* or *Nothofagus betuloides* (Mirb.) Oerst. (Promis et al.

2008, Toro Manríquez et al. 2019), or dry forests and woodlands, bordering grasslands and the Patagonian steppe (Veblen et al. 1996).

Different natural disturbances such as volcanism, earthquakes, landslides, snow avalanches, and wildfires, massive bamboo (*Chusquea* spp.) flowering and windstorms, have shaped these forest ecosystems (González et al. 2014; Veblen 1982). However, the distribution and current structure of the forest types, north of 43°S, have been strongly affected by anthropogenic fires since the middle 1800s, mainly during the European settlement, because fire was used as a tool to transform forests into agricultural and cattle raising lands (Willis 1914; Otero 2006). During the last century, part of the forest regenerated naturally as

secondary forests (Veblen et al. 2003). In the Coastal range of Chile windfall that create small- to medium-size gaps is the most common natural disturbance (Veblen 1985). In Magallanes and Tierra del Fuego (47°S–55°S) wind blowdown is the main disturbance (Rebertus et al. 1997).

In this article we present the state of the art in silviculture in native forests in four major regions of Chile and Argentina (Figures 1, 2 and 3). For each of these major forest types, we provide their distribution and site characteristics (climate, soils, topography), information about the silvics of their main tree species and forests dynamics, and finally a summary on silvicultural experiences, including tools developed, lessons learned, and approaches and prospects for the future.

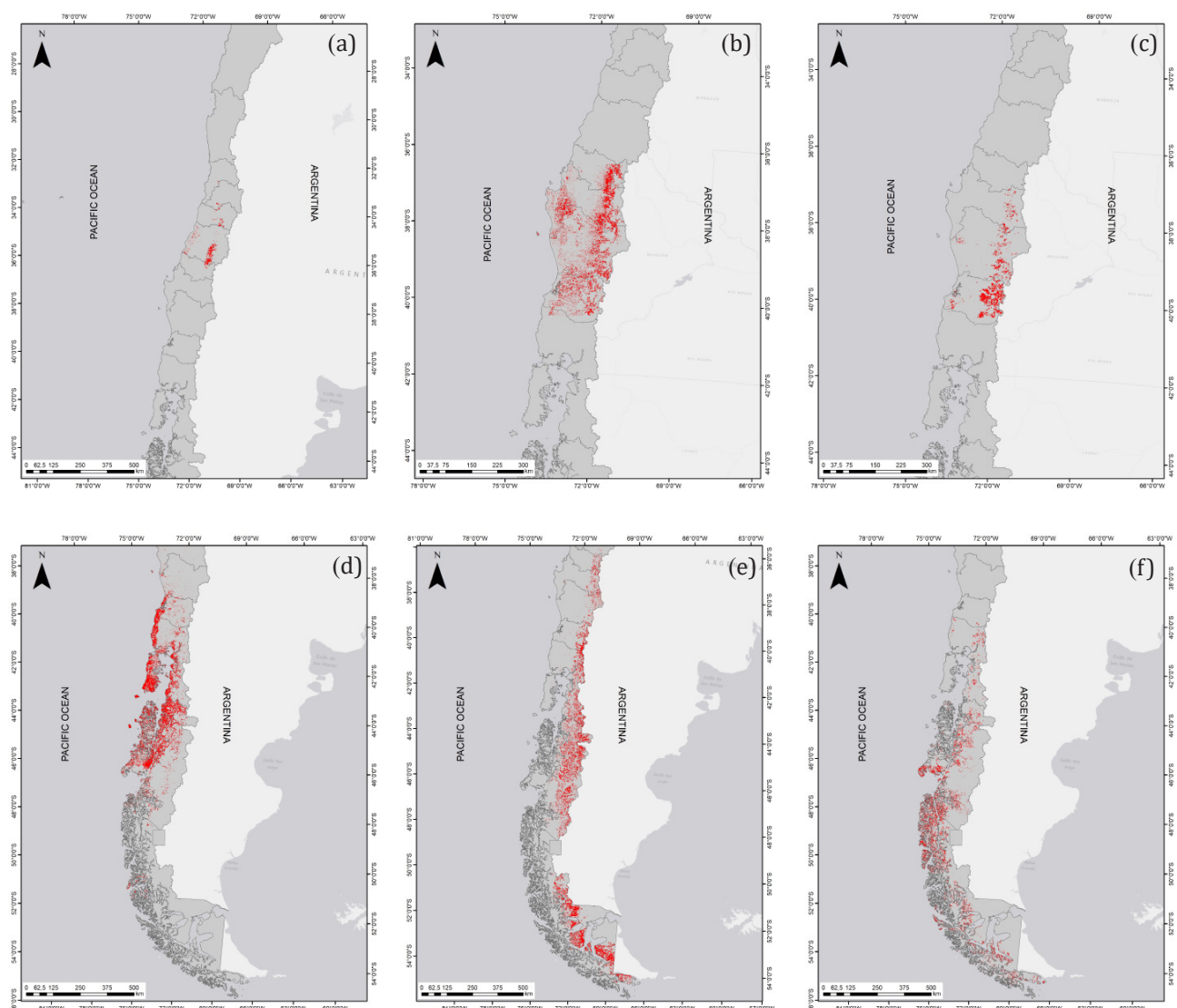


FIGURE 1: Distribution of the temperate Chilean forest types with greatest potential for management (area in parenthesis, x 1,000 ha). (a) *Nothofagus obliqua*–*Nothofagus glauca* (220); (b) *N. obliqua*–*Nothofagus alpina*–*Nothofagus dombeyi* (1,635); (c) *N. dombeyi*–*N. alpina*–*Laureliopsis philippiana* (846); (d) Hardwood-dominated evergreen forests (3,505); (e) *Nothofagus pumilio* (3,632); and (f) *Nothofagus betuloides* (1,999).

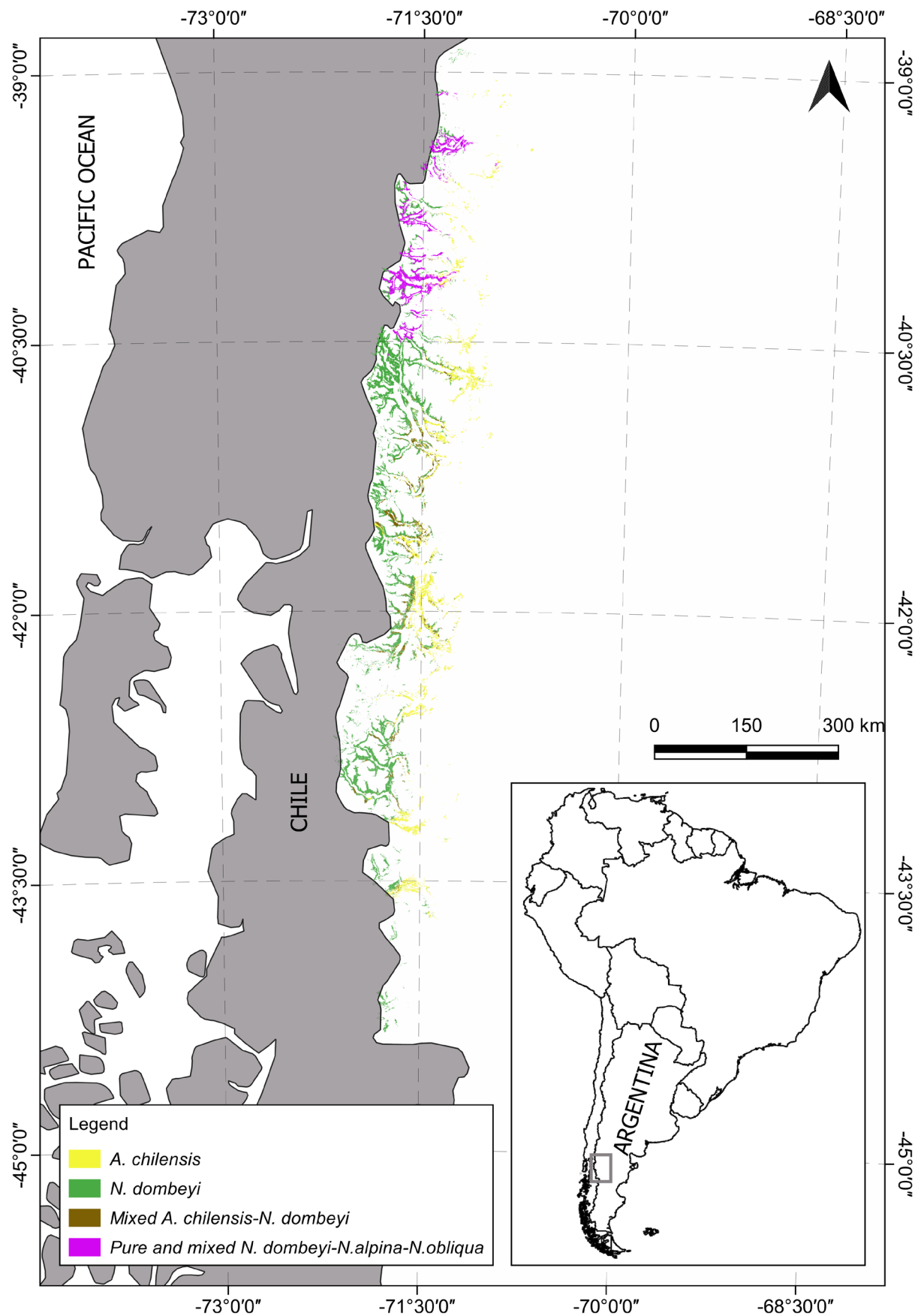


FIGURE 2: Distribution area of North Patagonian temperate forests in Argentina, including *Nothofagus alpina*, *Nothofagus obliqua* and *Nothofagus dombeyi* pure and mixed forests, and pure or mixed forests of *Austrocedrus chilensis* and *N. dombeyi* (modified from Mohr Bell et al. 2019).

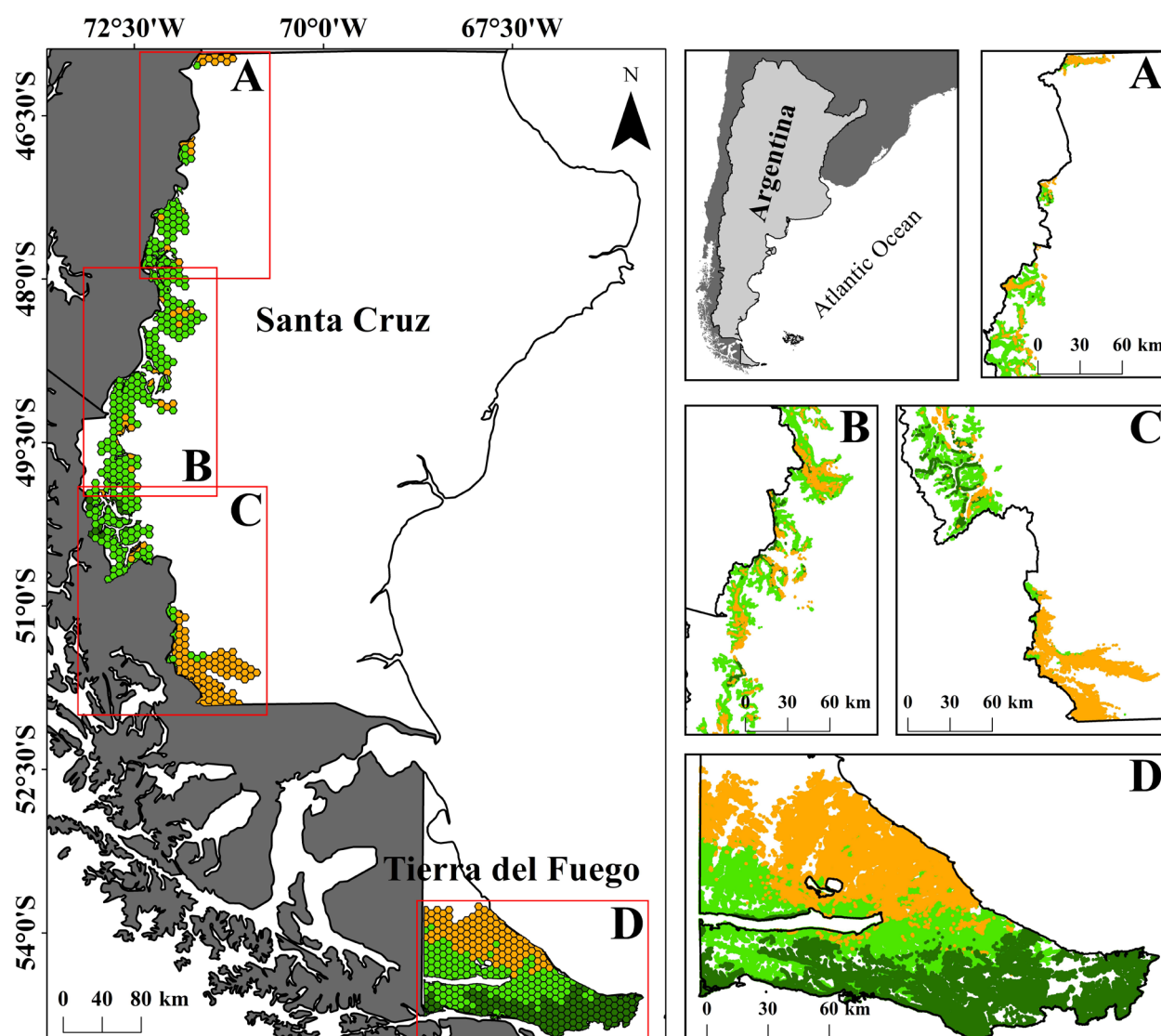


FIGURE 3: Distribution of Southern Patagonia temperate forest in Argentina, including *Nothofagus antarctica* forests (orange), *Nothofagus pumilio* forests (pale green) and mixed evergreen forests with *Nothofagus betuloides* (dark green) (based on Collado 2001; Peri & Ormaechea 2013; Peri et al. 2019).

***Nothofagus*-dominated forests of South-Central Chile (*Nothofagus obliqua* (Mirb.) Oerst., *N. alpina* (Poepp. & Endl.) Oerst., *Nothofagus dombeyi* (Mirb.) Oerst. and *Nothofagus glauca* (Phil.) Krasser).**

Distribution, composition and structure

Chile has 12 forest types (Donoso 1981; Donoso 2015), among which the three forest types we refer to in this section (Figure 1) are dominated by *Nothofagus* species. The *N. obliqua*–*N. glauca* (known as *Roble-Hualo*) and *N. obliqua*–*N. alpina*–*N. dombeyi* (known as *Roble-Raulí-Coihue*) forest types correspond to those that used to dominate the lowlands and medium elevations of Coastal and Andean cordilleras from central (35°S) to south-central (41°S) Chile, although the Sclerophyllous forest type extends through the central valley up to 37°S. The *N. dombeyi*–*N. alpina*–*Laureliopsis philippiana* (Looser) Schodde (known as *Coihue-Raulí-Tepa*) forest

type dominates especially in the Andes, at elevations between the upper limit of *N. obliqua* (500–600 m) and the lower limit of *N. pumilio* (1,100–1,200 m), a range within which *N. alpina* thrives (Donoso et al. 1986; Figure 1). These four species (*N. obliqua*, *N. alpina*, *N. dombeyi* and *N. glauca*) are shade-intolerant or mid-tolerant species, pioneers, and dominate the forests where they are present. Emergent *Nothofagus* trees in old-growth forests can reach heights close to 50 m and diameters greater than 2 m (Donoso et al. 1986; Salas et al. 2002; Parada et al. 2003). Past the stem exclusion phase of stand development (*sensu* Oliver & Larson 1996), these *Nothofagus*-dominated forests begin a process of vertical stratification due to the regeneration and increasing importance of tree species of greater shade tolerance (e.g., *Citronella mucronata* (Ruiz & Pav.) D.Don, *Cryptocarya alba* (Molina) Looser, *Persea lingue* (Ruiz & Pav.) Nees, *Laurelia sempervirens* (Ruiz & Pav.) Tul., *Podocarpus saligna* D.Don, *Drimys winteri* J.R.Forst. & G.Forst., *Aextoxicon punctatum* Ruiz & Pav., *Eucryphia*

cordifolia Cav., *L. philippiana*, *Archidasyphyllum diacanthoides* (Less.) Cabrera, *Saxegothaea conspicua* (Lindl.) that eventually dominate the canopy (22–35 m), the intermediate and the lower vertical strata in old-growth forests. Actually, once these forests have developed a vertical stratification, with dominant (secondary forests) or emergent (old-growth forests) *Nothofagus* trees above the other species, the basal area of both functional groups is additive (Lusk & Ortega 2003; Donoso & Lusk 2007; see also Parada et al. 2018 and Donoso & Soto 2016 in the Evergreen forest type).

It is important to mention that a great area of these *Nothofagus* forests has been burned in the process of conversion to agricultural lands, harvested for fuelwood over the last century or more, or completely replaced by plantations of *Pinus radiata* D.Don (Donoso 1996; Otero 2006; Lara et al. 2016). For these reasons, nowadays the structure of these forests corresponds mainly to secondary forests originating from seedling establishment or from vegetative resprouting, depending on the type of original human-caused disturbances. In the case of *N. glauca* forests, they usually correspond to resprouts (Donoso 1996, Promis et al. 2019).

Growth

Nothofagus species have rapid growth rates in secondary forests, where the highest growth rates in diameter and height occur during the second decade after establishment (Donoso et al. 1993a, 1999; Salas-Eljatib et al. 2018; Salas-Eljatib 2021). In 20–40 year-old secondary forests, and due to intraspecific competition and canopy stratification, most trees have periodic annual increment (PAI) in diameter between 0.3 and 0.4 cm yr⁻¹, but dominant trees reach PAIs in diameter of 0.8 cm yr⁻¹ (Figure 4; Puente et al. 1980). Time of maximum height growth-rates depend on the shade-

tolerance of the species, ranging between 10, 13 and 15 years for *N. dombeyi*, *N. obliqua*, and *N. alpina* (Salas-Eljatib 2020). *N. alpina* dominant trees reach a mean height growth of 0.56 m yr⁻¹ (Salas-Eljatib 2021). Diameter growth rates may be similar among dominant *Nothofagus* trees in secondary forests, but *N. dombeyi* has the capacity to sustain about 40% more trees in a stand with an equivalent quadratic stand diameter to a *N. obliqua*–*N. alpina* forest stand (Lara et al. 1998; Lusk & Ortega 2003), which means that it has greater basal area and greater volume growth rates. Donoso et al. (1999) report PAI in volume close to 20 m³ ha⁻¹ yr⁻¹ in both Coastal and Andean *N. dombeyi* secondary forests, while Donoso et al. (1993a) report PAI in volume of 12.6 (± 2.73) and 10.4 (± 2.05) m³ ha⁻¹ yr⁻¹ in the better sites for *N. obliqua* and *N. alpina*, respectively. Salas-Eljatib et al. (2018) studied 74 permanent sample plots in *Nothofagus* secondary forests in south-central Chile. They found a mean annual increment (MAI) in volume of 8.5 m³ ha⁻¹ yr⁻¹, but for a wide range of ages and geographic conditions.

Silvicultural experiences

Thinnings are an essential silvicultural treatment to improve growth and quality of secondary forests such as these *Nothofagus*-dominated forests. However, the information about effects of thinnings on *Nothofagus* secondary forests is very scarce. Gajardo-Caviedes et al. (2011) reported a crown area close to 40, 80 and 180 m² for dominant trees in a 48-year-old *N. dombeyi* secondary forests in plots without thinning, with one (age 22), and with two thinnings (ages 22 and 36), i.e. more than a four-fold increase after two thinnings. For 80-year-old *N. dombeyi* secondary forests, Ojeda et al. (2018) reported median PAI in volume of 12 to 16 m³ ha⁻¹ yr⁻¹ five years after crown thinnings. For secondary

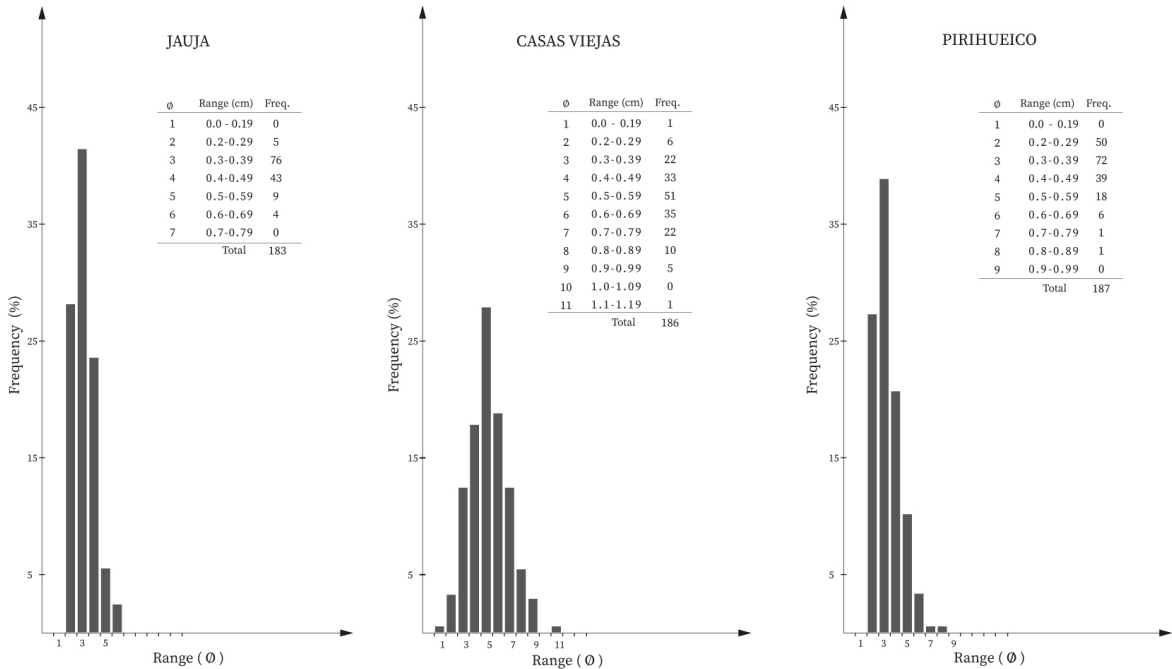


FIGURE 4: Diameter growth frequency distribution, by 2-mm categories, in secondary forests dominated by *Nothofagus alpina* in south-central Chile (38–40°S). (Puente et al. 1980).

N. alpina–*N. obliqua* forests, Donoso et al. (1993b) estimated MAI in volume for entire rotations in stands with, one, two or three thinnings, depending on their initial ages, and determined values between 15 and 20 m³ ha⁻¹ yr⁻¹ of gross growth (final harvest plus thinnings).

Navarro et al. (2016) evaluated 40 management plans in secondary forests of the Araucanía region (38–39°S), out of a base of 1,699 management plans approved between the years 2009–2015 by the National Forest Corporation (CONAF). The stands studied had medium to low cover, with an average of 833 trees ha⁻¹ (including saplings), 19.2 m² ha⁻¹ of basal area and 147 m³ ha⁻¹ of total volume. From these stands, 477 increment cores were collected, which showed that these thinned secondary forests were mostly between 41 and 69 years old (range 21 to 96 years). The post thinning PAI of these forests, which had an average quadratic stand diameter of 19.7 cm, were concentrated between 0.45 and 0.50 cm in diameter and 6 to 8 m³ ha⁻¹ in volume (Figure 5).

An interesting long-term thinning experiment in *N. alpina*–*N. obliqua* secondary forests (Figure 6) is the one initiated by Prof. Mario Puente and colleagues (Puente et al. 1981) in Jauja (38°S, 800 m a.s.l in the Andes). In a stand that was 40 years old and had 47 m² ha⁻¹ in basal area (70% *N. alpina*), four thinnings

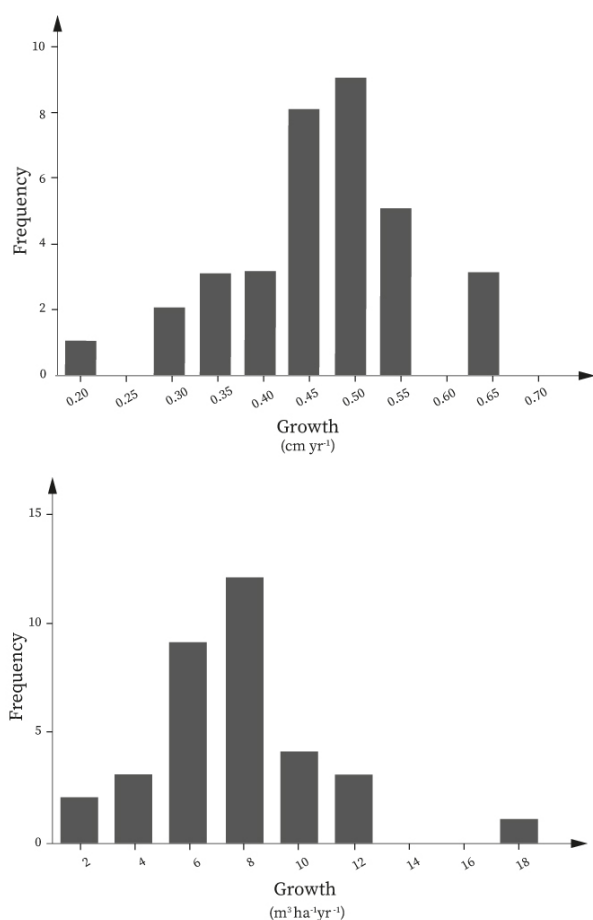


FIGURE 5: Diameter and volume growth frequency in thinned *Nothofagus*-dominated secondary forests in the Araucanía region (38–39°S) (Navarro et al. 2016).



FIGURE 6: Characteristic forests in south-central Chile. (top) A dense *N. alpina*–*N. obliqua* secondary forest without management at mid-elevations in the Andes (38°S), with some dead trees, partial *Chusquea* cover in the understorey, and regeneration of more shade-tolerant species. (bottom) Evergreen forest in Llancahue (40°S) 6 years after the implementation of a selection cut to a residual basal area of 40 m² ha⁻¹.

from below were implemented to different residual basal areas (40, 30, 20, and 10 m² ha⁻¹), with three 2,000 m² permanent sample plots for each treatment and for untreated forests. Although Prof. Puente envisioned application of more than a single thinning, funding restrictions as well as his untimely death did not allow further silvicultural interventions. Nonetheless, the data is available from a first measurement and four re-measurements (Pincheira 1993; Salas-Eljatib & Weiskittel 2018; Salas-Eljatib et al. 2018), including the last that is shown here for the first time. We computed stand variables for each plot. As expected, tree density decreased over time, but increased in the two more intense treatments (Figure 7) due to the re-sprouting of the cut *N. alpina* trees. Growth in basal area has tended to reach a plateau in the control and less intense thinning treatment, while basal area continues to accumulate in the more intense treatments (Figure 7). An overall assessment of these plots after 40 years since they were thinned indicates that the most convenient treatment has been the one with 30 m² ha⁻¹ in residual basal area, since basal area is close to the one in untreated plots and average individual tree volume of the residual trees is >1 m³.

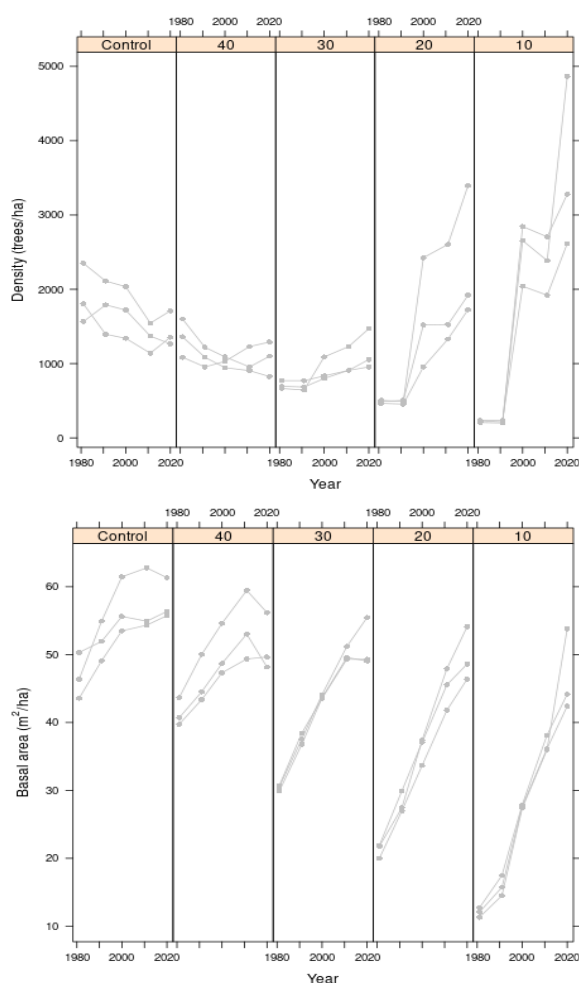


FIGURE 7. Tree density (a) and basal area (b) by thinning treatments (residual basal area in m² ha⁻¹) in a secondary forest stand of *Nothofagus alpina*-*N. obliqua* in southern Chile. Dots joined by lines correspond to remeasured plots.

In stands where owners decide to implement a silvicultural method to regenerate their forests, it is becoming common to use strip shelterwood (strips with a width equivalent to the height of dominant trees, i.e. near 30 m; *sensu* Nyland 2016) in poorly stocked stands, and also the single seed tree method (*sensu* Nyland 2016) with two 500 m² retention groups per ha. While preliminary results seem promising, long-term monitoring is necessary to evaluate whether bamboo (*Chusquea* spp.) competes with regeneration (Soto & Puettmann 2018, Patricio Toledo (forest consultant), personal communication). In fact, Soto et al. (2015, 2019) and Soto and Puettmann (2018) reported that numerous efforts to control bamboo competition following shelterwood cuts (*sensu* Nyland 2016) in *N. dombeyi*-*N. alpina* old-growth forests in the Andes failed due to lack of understanding of regeneration ecology of these *Nothofagus* species. These studies highlighted the fact that neither of these silvicultural methods applied in south-central Andes had incorporated the well-known research about the regeneration dynamics that most *Nothofagus* species follow. Most of the silviculture applied in *Nothofagus*-dominated stands is based on the premise that light is the main driver of succession. However, since landslides, fires or massive blowdowns are common large-scale disturbances in the Andes (*sensu* Veblen et al. 1981, 1996), forest regeneration is dominated by *Nothofagus* species adapted to the catastrophic regeneration mode, and forest succession occurs in a quite predictable way (stand recovery). This illustrates that light is not the sole factor controlling regeneration of these *Nothofagus* species.

Soto and Puettmann (2018) and Soto et al. (2019) showed that small-scale soil disturbance through topsoil scarification that eliminates the first layer of the soil provides the safe-site conditions for *Nothofagus* establishment, while effectively controlling the persistent dense bamboo (*Chusquea* spp.) thickets that are common in these forests (e.g., González et al. 2015). These studies showed that this management technique can overcome the arrested succession condition generated by bamboo thickets and re-allocate resources to encourage desired tree regeneration and understorey plant diversity. In ecological terms, these studies concluded that there is a potential divergence between the: (1) regeneration niche (establishment of seedlings); and (2) growth niche (established saplings) stages. Regeneration niche (i.e. resources and environmental factors that influence the seedling establishment) of both species (*N. alpina* and *N. dombeyi*) was mostly explained by soil resources, such as soil water content and exposed mineral soil (Soto et al. 2019), while the growth niche of the same species was mostly driven by the interaction of light availability and nitrogen at high light levels for *N. dombeyi*, and the proportional effects of nitrogen along the entire light gradient (ca. 3–77%) for *N. alpina* (Soto et al. 2017). The topsoil scarification should be practised carefully, since bamboos provide soil protection, functional diversity and complexity to the forest understoreys (Soto & Puettmann 2018, Seidel et al. 2021). Figure 8 summarises the main findings of these works in terms of the regeneration niche of *N. alpina* and *N. dombeyi*.

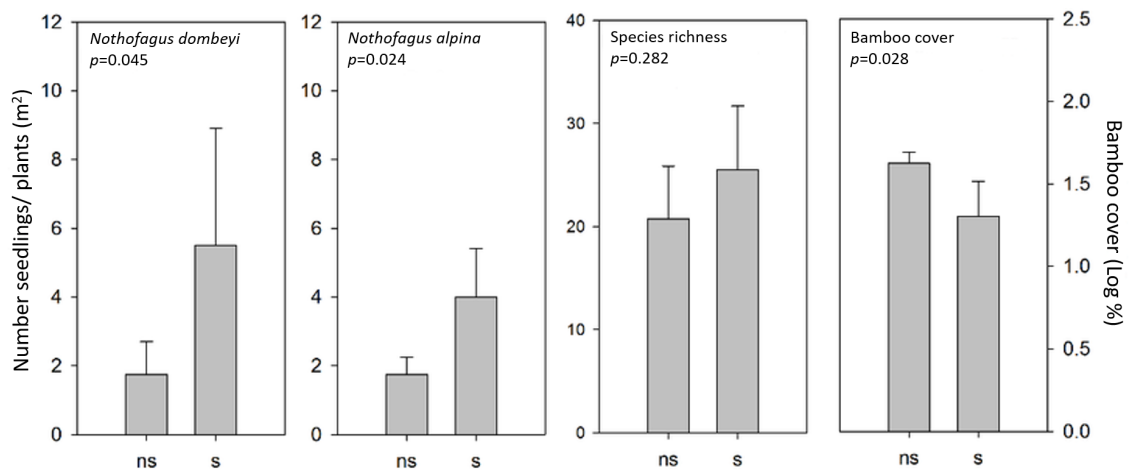


FIGURE 8: Bar plots (mean + standard deviation) showing results of regeneration, richness of vascular species, and bamboo (*Chusquea* spp.) cover under conditions of scarified (s) and non-scarified (ns) soils. P-values highlight the significant differences between scarified and non-scarified soils (t-test), which occurred for all four variables evaluated.

Angiosperm-dominated evergreen forests

Three main long-term silvicultural experiences in Chile have been developed in the Evergreen forest type (Figure 1), which include: (a) the implementation of even-aged silvicultural methods (or final harvests) in old-growth forests; (b) the implementation of uneven-aged silviculture in partially cut old-growth forests; and (c) thinnings in *D. winteri*-dominated secondary forests. Also, silviculture to rehabilitate degraded forests has been implemented.

Even-aged silvicultural methods in old-growth forests

In 1983, two experiments were established in the Coastal (40°S) and Andean (42°S) ranges in forests that had in common *L. philippiana* and *Eucryphia cordifolia* among the main canopy species (in addition to *Myrtaceae* species in the lower canopy), where in the Coastal forest also *A. diacanthoides* was an important

canopy species, and in the Andes *Nothofagus nitida* (Phil.) Krasser and *S. conspicua* were among the main canopy species (Donoso 1989). Three plots for each of three even-aged silvicultural methods (Table 1) were established in each experiment, with 100 x 100 m plots in the Coast and 120 x 120 m plots in the Andes. The experiment in the Coastal range is between 550 and 650 m a.s.l., with an annual precipitation close to 4,000 mm yr⁻¹, and a mean annual temperature of 11–12°C. The experiment in the Andes is between 350–550 m a.s.l., with an annual precipitation close to 5,000 mm yr⁻¹, and a mean annual temperature of 9–10°C. The major difference between these two areas is in their soils, since those in the Coastal range are of medium depth (40–60 cm) and with a mica-schist metamorphic bedrock, while those in the Andes are deep and correspond to recent volcanic ashes. Differing results of these two experiments were interpreted by Donoso (1989) for the

TABLE 1: Density of seedlings three years after the cuttings in both the Coastal and Andean large-plot experiments with silvicultural methods, and of trees 26 years after the implementation of the cuttings in the Andes.

Region	Silvicultural method*	Total **seedlings after 3 years (x 1,000)	Main tree species***	Total* trees after 26 years	Main tree species***
Coastal range	Clear Cut	74	<i>Ec, Wt, Dw</i>	-	-
	Irregular Shelterwood	58	<i>Ec, Sc, Pn</i>	-	-
	Seed Tree	327	<i>Ec, Wt, Ga</i>	-	-
Andean range	Clear Cut	826	<i>Wt, Ecm, Dw</i>	3,148	<i>Ecm, Dw, Ec</i>
	Irregular Shelterwood	898	<i>Wt, Ecm Ec</i>	2,055	<i>Ecm, Dw, Ec</i>
	Strip cut	1,146	<i>Ec, Wt, Ecm</i>	2,384	<i>Ecm, Ec, Lp</i>

*Definition of silvicultural methods follow Nyland (2016) except for the Irregular Shelterwood that follows Raymond and Bedard (2017).

**Totals do not include non-commercial (timber) species, especially those grouped in the *Myrtaceae* family.

***Ecm: *Embothrium coccineum*; Ec: *Eucryphia cordifolia*, Dw: *D. winteri*, Sc: *S. conspicua*, Pn: *P. nubigena*, Wt: *Weinmannia trichosperma* Cav.

first three years after implementation (Table 1). These illustrate a much greater seedling density in the Andean range, and in general, in both experiments, the dominance of shade-intolerant or mid-tolerant species, the only exception being the regeneration of the shade-tolerant Podocarpaceae species, *S. conspicua* and *Podocarpus nubigena* Lindl. in the irregular shelterwood (*sensu* Raymond & Bedard 2017) experiment in the Coastal range, which also had abundant regeneration. After 26 years in the Andes, in all cases trees were dominated by *Embothrium coccineum* J.R.Forst. & G.Forst., and both in the clear cut and the irregular shelterwood methods this species was accompanied mostly by *D. winteri* and *Eucryphia cordifolia*. In the case of the strip cut the second and third species of greater density were *E. cordifolia* and *L. philippiana* (Donoso et al. 2019). By age 26, basal area in these experiments was $43.6 (\pm 4.2, \text{standard deviation})$, $40.4 (\pm 2.6)$ and $25.9 (\pm 2.9) \text{ m}^2 \text{ ha}^{-1}$, for the strip cut, clear cut and irregular shelterwood, respectively (the latter significantly lower). While in the experiment in the Coastal forests there is no published recent data, the secondary forests formed after almost four decades in these experiments seem to have a more homogeneous tree distribution and less cover of *Chusquea* spp. in the plots with the irregular shelterwood method (PJ Donoso, personal observation).

Uneven-aged silviculture in partially cut old-growth forests

In 2012, two single-tree selection experiments were established at mid-elevation sites within the Evergreen forest type in the Coastal range (Llancahue and Los Riscos, at 40 and 42 °S). They had an average initial basal area of $70\text{--}80 \text{ m}^2 \text{ ha}^{-1}$, after being partially cut in the past with selective cuttings (*sensu* Nyland 2016). In each stand, we established plots with residual basal areas of $\sim 40 \text{ m}^2 \text{ ha}^{-1}$ and $\sim 60 \text{ m}^2 \text{ ha}^{-1}$. We planned for a maximum residual diameter of 80 cm but needed to leave 20–25% of the residual basal area in larger trees due to their great abundance in these old-growth forests. Plots were re-measured 5–6 years after the cuttings (Donoso et al. 2020a). At Llancahue, the PAI in diameter of individual trees was significantly greater in the treatment with lower residual densities, especially for mid-tolerant species in lower diameter classes (5–20 cm). At both sites, the PAI in volume was greater in the more heavily stocked treatment (median 7 and $5 \text{ m}^3 \text{ ha}^{-1} \text{ y}^{-1}$ in Llancahue and Los Riscos, respectively), but differences were significant only at Llancahue. Tree regeneration was more abundant and more diverse at Llancahue (Figure 6), but no differences were found in regeneration responses between the lower and higher levels of residual basal area. While both sites had many similar trends after implementing selection cuts (a greater individual growth in the treatment with lower basal areas but a higher stand-level growth in the treatment with a high basal area, more abundant regeneration of shade-tolerant species, etc.), they illustrate a differential potential for implementing uneven-aged silviculture, especially due to site-species interactions, i.e., similar to differences reported above for the even-aged experiments. These are the only experiments in Chile with single-tree selection

cuttings. They are only near the middle of the expected length of a cutting cycle for these forests, so there is still much to learn from these experiments, not only for a first complete cutting cycle but also for successive cutting cycles. There is no experiment nor any reported experience with selection cuttings in Andean forests, so that in general there is still much work to do with, and many lessons to learn from uneven-aged silviculture in these evergreen forests comprising diverse and valuable mid- and shade-tolerant tree species, mostly hardwoods (Donoso 2013).

Thinnings in *Drimys winteri*-dominated secondary forests

Drimys winteri forests occupy an area of 252,000 ha, ranging from 38.5° to 43°S, representing 7% of the Chilean secondary forests (Navarro & Cabello 2018). Navarro et al. (2017), on a basis of 360 sample units established between 39 and 43°S, determined that these secondary forests mostly ranged between 40 to $80 \text{ m}^2 \text{ ha}^{-1}$ in basal area, 1,000 to 4,000 trees ha^{-1} , 200 to $500 \text{ m}^3 \text{ ha}^{-1}$ in volume, 12 to 18 m in dominant height, and 10 to 25 cm in quadratic mean stand diameter (Figure 9).

In these forests, a series of studies have been carried out in relation to silvicultural experiences, highlighting the work on the size-density relationships in different sites and species composition (Donoso et al. 2007), the evaluation of thinning trials (Navarro et al. 1997, Reyes et al. 2009; Navarro et al. 2010; Navarro 2011), the development of density management diagrams (Navarro et al. 2011; Navarro et al. 2017) and the exploratory definition of potential areas for silvicultural management (UGS for Unidades de Gestión Silvícola in Spanish) (Navarro 2011; Navarro & Cabello 2018). The study developed to define UGS across the distribution area of these forests determined 96,000 ha that should be assigned mainly for the production of quality wood, and a similar area for sites of high fragility and very low productivity that should be allocated to non-timber purposes, such as the maintenance of ecosystem services (e.g., water production, conservation of biodiversity).

In one of the high-quality sites a thinning trial was established in 1985 in pure 30-year-old *D. winteri* forests, which had $6,174 \text{ trees ha}^{-1}$, $58 \text{ m}^2 \text{ ha}^{-1}$ in basal area and $355 \text{ m}^3 \text{ ha}^{-1}$ in volume (Navarro et al. 1999). The design included five treatments, with three mixed thinnings aimed to leave a distance between residual trees of 4 x 4 (4 m or R4), 3 x 3 (3 m or R3) and 2 x 2 (2 m or R2), one crop tree thinning (RL), plus the control. Harvesting in number of trees was 64, 81 and 89% for the treatments with spacing at 2, 3 and 4 meters, respectively, and the RL had a similar thinning intensity to that of the treatment at 4 m (residual densities were 2,494, 1,094, 611, 716 trees per ha for R2, R3, R4 and RL thinning, respectively). The growth trend in diameter after 21 years indicated that the treatments of greater intensity (R3 and R4) maintained high growth rates with respect to the control and treatment R2 (Figure 10). The RL treatment and 2–3 m treatment reached very similar basal areas after 21 years, while the 4 m treatment remained with lower figures due to the low density in

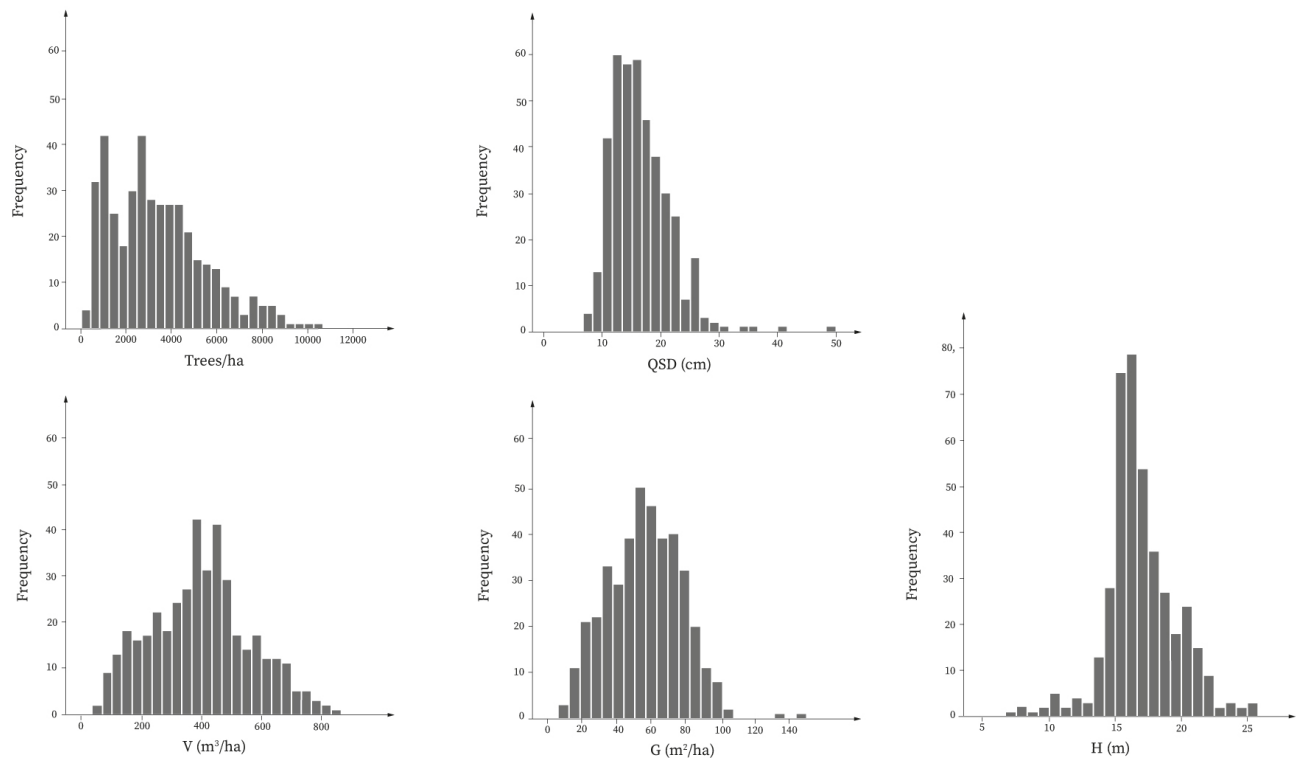


FIGURE 9: Frequency observed for different stand variables in *Drimys winteri* secondary forests. QSD (cm): Quadratic Stand Diameter; V (m³ ha⁻¹): volume; G (m² ha⁻¹): basal area; H (m): dominant height.

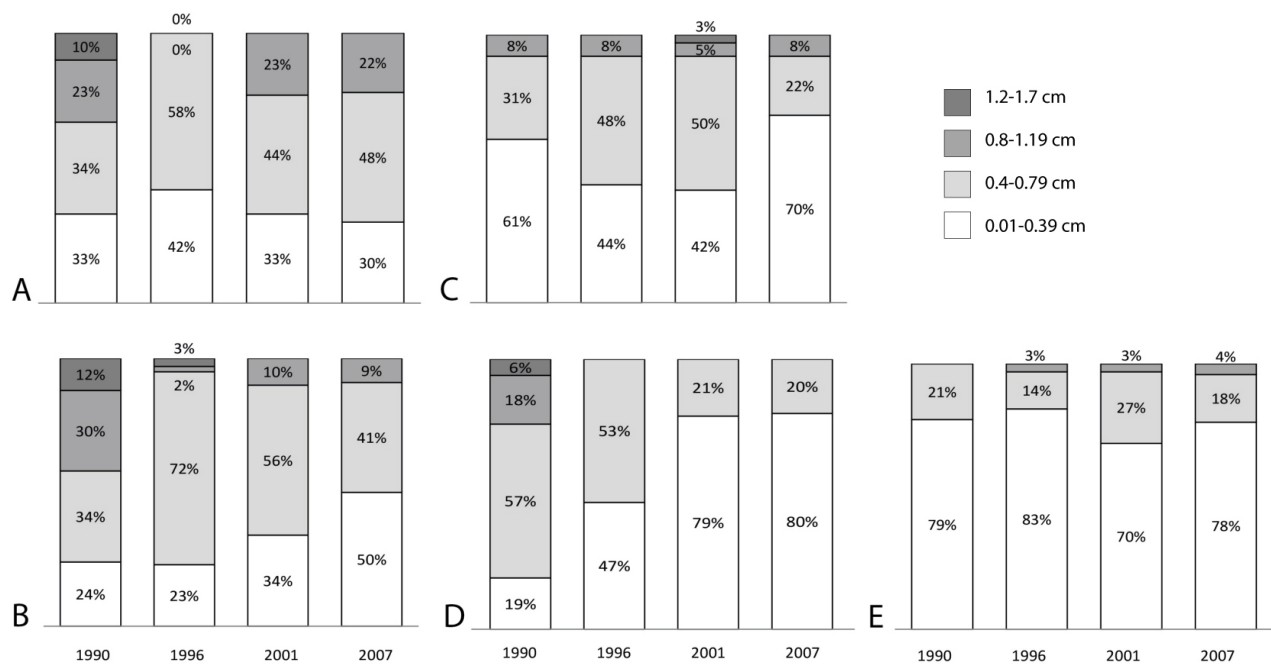


FIGURE 10: Distribution of annual diameter growth frequencies per thinning treatment: A (4 m R4); B (Crop tree RL); C (2 m R2); D (3 m R3); and E (control).

relative terms (Navarro 2011). In volume, treatments showed similar trends, highlighting a PAI in volume of 4 to 6 m³ ha⁻¹ yr⁻¹ in the control and 3-m treatment, respectively. In gross volume growth, the most intense treatments, i.e. R4, R3 and RL, had 16, 17 and 19 m³ ha⁻¹ yr⁻¹, while the R2 treatment had 13 m³ ha⁻¹ yr⁻¹, and the control close to 5 m³ ha⁻¹ yr⁻¹. These results reveal the silvicultural potential of these forests, and the need for their timely intervention (Navarro 2011).

Navarro et al. (2011) determined that at the age of harvesting, 52 years after the thinnings, the treatment with a highest net present value (NPV), at a rate of 6%, was thinning at 3 m, with US\$ 2,724 ha⁻¹, and a modified internal rate of return (MIRR) of 19%. When comparing thinning treatments with the control, the NPV at the rate of 6% is at least doubled, and at a rate of 12% this figure rose to at least 6 times. Navarro et al. (2011) suggest maintaining site occupancy levels between underutilisation areas (relative density index, RDI, of 30%) and that of imminent mortality due to competition (RDI of 45%).

There are several other recent novel silvicultural experiments in these forests, many of which fit within ecological silviculture (*sensu* Palik et al. 2021). These include variable-density thinnings in different types of secondary forests (Donoso et al. 2020b), and irregular shelterwood cuttings in mature secondary *N. dombeyi*-dominated forests (PJ Donoso unpublished data). These studies are providing promising results in growth and regeneration.

Silviculture to rehabilitate degraded forests

Forest high-grading in Chile has been the result of frequent harvesting of the most merchantable trees of commercially valuable species, especially in south-central Chile (the forests of this and the previous section; Vásquez-Grandón et al. 2018). In addition, especially in the forests closer to urban centers in south-central and in North Patagonia in Chile (Coyhaique), the unsustainable exploitation of forests to produce firewood has been an important cause of forest degradation. RedPE (2020) estimates that there is a minimum of 14 million m³ of firewood annually consumed in Chile, of which 40–50% corresponds to native species. If these types of exploitation are accompanied by cattle grazing within the forest, the situation becomes more severe (Zamorano-Elgueta et al. 2014), and these forests become degraded systems in terms of tree composition, structure, and regeneration (Vásquez-Grandón et al. 2018). In the worst cases, these systems are in a scenario of arrested succession, in which the ecological processes associated with the dynamics of the forest are reduced or severely limited (Ghazoul et al. 2015).

Without silviculture, these forests will unlikely recover their attributes, and therefore will continue in a state where they do not provide their potential goods and services to society. Human intervention is then required to restart essential processes, such as regeneration, through rehabilitation approaches aimed to trigger these processes (Ghazoul & Chazdon 2017). In all cases, the focus is to promote tree regeneration

of the main and dominant species according to the forest type and with the aim to restoring desired species composition, structure, or processes (Stanturf et al. 2014). Rehabilitation is complex, therefore, due to the range of species, sites, and levels of degradation (Clatterbuck 2006).

Rehabilitation is silviculture applied to restore desired characteristics of degraded stands (Kenefic et al. 2014). These efforts largely rely in the implementation of well-known silvicultural techniques for regeneration, tending or harvesting the forests, which will depend in their type and magnitude upon the degree of degradation of the forest (Vásquez-Grandón 2020). In this sense, Vásquez-Grandón (2020) studied several old-growth forests that have been high-graded in the past, analysed 26 variables of forest composition, structure, and regeneration, and found that the common patterns resulting from past unsustainable practices were: (i) low quadratic stand diameter; (ii) low to moderate basal area of commercial tree species; (iii) high basal area of non-commercial tree species; (iv) low to moderate basal area of large-diameter trees; (v) low to moderate total regeneration density; and (vi) low total regeneration density of mid-tolerant species. These forests were classified in a gradient of light to severe degradation (Figure 11). The variables that determined the categorization of forests according to their degree of degradation were, in order of importance, tree regeneration, basal area of non-commercial tree species, and tree regeneration of mid-tolerant species. Different techniques such as improvement cuts, understorey competition control and soil scarification have been implemented in these forests (Figure 11), but results are yet to be reported. These measures are usually in the list of many options that have been proposed to face the recovery of degraded forests across temperate regions, including southern Chile (Russell-Roy et al. 2014; Stanturf et al. 2014; Bannister et al. 2016; Nyland 2016; Soto & Puettmann 2018, 2020; Prévost & Charette 2019; Soto et al. 2019, 2020). Also supplementary plantation of tree species is regarded as an alternative technique to rehabilitate or to restore degraded forests (e.g. Soto et al. 2020; Bannister et al. 2021; Caselli et al. 2021), but so far most of these efforts have been conducted at small scales. Forest rehabilitation or restoration at large scales is currently limited by the scarcity of high quality seedlings (Bannister et al. 2018) and poorly developed tree seed systems (Atkinson et al. 2021).

Mixed *Nothofagus* (*N. obliqua*, *N. alpina* and *N. dombeyi*), *N. dombeyi* - *Austrocedrus chilensis*, and pure *A. chilensis* forests in the Argentinean Northern Patagonia

The Andean-Patagonian Forests – in Argentina also known as Subantarctic Forests (Cabrera 1994) – develop east of the Andes in northern Patagonia, with a strong precipitation decrease, between 2,500 and 500 mm yr⁻¹, in no more than 80 km from West to East (Veblen et al. 1996). These are pure forests with less complex



FIGURE 11: Forest stands of Valdivian rainforests (40–41°S in Chile) with different degrees of degradation and after the implementation of silvicultural treatments aimed to trigger regeneration (soil scarification and control of understorey competition) and to improve the quality of residual trees (improvement cuts) that play a role as potential valuable timber and as seed source for regeneration.

structures, dominated by broadleaf *Nothofagus* sp. (*N. pumilio*, *N. antarctica*, *N. dombeyi*, *N. obliqua* and *N. alpina*) and the conifers *Austrocedrus chilensis* and *Araucaria araucana*, among the main ones, and with mixed transitions between them (Loguerio et al. 2018b). Among mixed forests, the old-growth forests of *Nothofagus* spp. stand out in the province of Neuquén. They consist of *N. alpina*, *N. obliqua* and *N. dombeyi*. They extend between 39° 29' and 40° 22' S and 71° 14' and 71° 40' W, occupying 45,800 ha (Mohr Bell et al. 2019) (Figure 2). Approximately 90% of *Nothofagus* mixed forests are located in Lanín and Nahuel Huapi National Parks and are used for conservation, recreation and productive purposes (Sabatier et al. 2011). In Lanín

National Park, legislation allows forest management as a strategic conservation policy, strictly in areas categorized as Reserve (195,010 ha, 47.3% of the total protected area) but not in areas classified as intangible (the remaining 53.7%). In this area, the last 30 years there have been near 1,000 ha under forest management on public, private and indigenous community lands, to produce mainly wood, firewood and *Chusquea* cane. These forests grow in valleys and slopes up to 1,000 m above sea level, with annual rainfall between 1,000 and 2,000 mm and deep, well-drained soils, originated from volcanic ashes (Hoffmann 1982; Oyarzabal et al. 2018). Although their extension is greater in Chile, in Argentina they stand out for their greater genetic diversity (Marchelli & Gallo 2004, 2006; Azpilicueta et al. 2009).

The forests dominated by *A. chilensis* develop a little further east, extending to the south. They appear scarcely from 32° 39' S and are more common from 40° to 43° 44' S (Donoso 1981; Pastorino et al. 2006). They occupy 92,900 ha, 33% of which are in protected areas (Lanín, Nahuel Huapi and Los Alerces National Parks) (Mohr Bell et al. 2019) (Figure 2). Where precipitation varies between 2,200 and 900 mm yr⁻¹ *A. chilensis* is associated with *N. dombeyi*, forming mixed forests. They are distributed in 31,800 ha, 67% of which are in National Parks (Mohr Bell et al. 2019) (Figure 2). With less precipitation, *A. chilensis* forms pure stands, being the most common species in the ecotone with the steppe in open stands (Veblen et al. 2005; Kitzberger 2012). The soils derived from volcanic ashes (andisols), are also deep, with abundant organic matter, especially where *N. dombeyi* grows, while *A. chilensis* can be found in shallow soils with rocky outcrops (Veblen et al. 2005).

These forests play a key role in the provision of ecosystem goods and services in the region. They provide both timber and non-timber goods and services, regulating the climate and the water cycle, sustaining the soil, providing habitats for the maintenance of biodiversity, mitigating carbon emissions and being a scene of great beauty, which contributes to the comprehensive development of the region.

Structure and dynamics

North Patagonian forests east of the Andes are largely post-fire secondary forests too, originating after large scale anthropogenic fires that occurred between the late 1800s and early 1900s (Willis 1914; Veblen et al. 1999). Only in some more humid areas or at higher elevations do old-growth forests retain their natural dynamics. One case is the mixed forests of *Nothofagus* spp., whose structures corresponds generally to the transition between understorey reinitiation and old growth stages (Oliver & Larson 1996). At the present, these forests have simple and regular structures with an average diameter of 45 cm and average height of 30 m, generally accompanied by overmature trees (Figure 12). The basal area fluctuates between 40 and 60 m² ha⁻¹, with a maximum of 70 m² ha⁻¹. Stands exhibit mature trees with dominance of ages between 100 and 150 years and total volumes between 600 and 900 m³ ha⁻¹. Stands in the initiation and stem exclusion phases are less frequent.

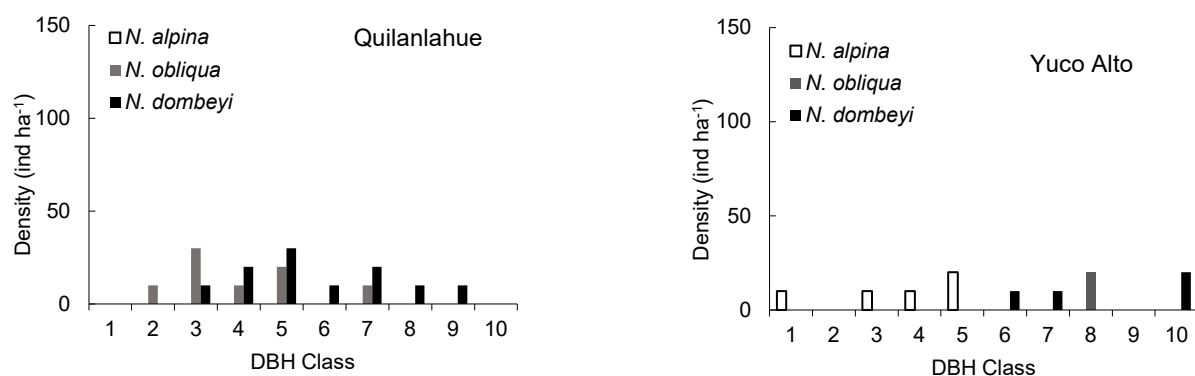


FIGURE 12: Frequency distribution of diameter at breast height (DBH) of *Nothofagus* in two virgin stands of Lanín National Park. DBH class 1: 10 - 19.9 cm, 2: 20 - 29.9 cm, ..., 10: > 100 cm (Dezzotti et al. 2016).

Unlike what happens in Chile, and due to the lower rainfall in Argentina, these forests are not replaced by shade-tolerant or mid-tolerant species during advanced stages of development (Veblen et al. 1996). Regeneration requires the protection of the canopy against the direct and indirect effects of extreme solar radiation in a climate with marked seasonality, which cause mortality due to drought and freezing (Dezzotti et al. 2004; Donoso et al. 2013). At age 10, saplings of *N. alpina* and *N. obliqua* reach an average of 4 m height, while *N. dombeyi* averages 3 m height (Sola et al. 2015). *N. alpina* shows similar growth patterns in height with *N. obliqua*, and in diameter with *N. dombeyi* (Attis Beltrán et al. 2015, 2016, 2018). Based on simulations, in the most productive sites, at the age of 100 years, *N. dombeyi* and *N. alpina* reach 40–50 m in height, while *N. obliqua* 35–40 m (Carrizo 2001; Attis Beltrán et al. 2015). On average, the mixed *Nothofagus* forests show volume increments of $7.4 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ (González Peñalba et al. 2010).

Within the post-fire secondary forests, those of *A. chilensis* stand out. From survivors in rock shelters, it recolonises large areas, forming even-aged structures in mesic sites and, to a lesser extent, uneven-aged structures in xeric sites (Dezzotti & Sancholuz 1991, Loguercio 1997, Veblen et al. 2005). The mesic forests, with 100–120 years of age, reach heights of up to 25–30 m, basal areas of $70\text{--}80 \text{ m}^2 \text{ ha}^{-1}$ and PAI values of $4\text{--}6 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$, while in the xeric sites these values are around 15–20 m in height, $35\text{--}40 \text{ m}^2 \text{ ha}^{-1}$ in basal area of and $2\text{--}4 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ in volume (Goya et al. 1998, Loguercio et al. 2018b). *Austrocedrus chilensis* presents a sanitary problem, known as “cypress disease”, produced by *Phytophthora austrocedrae* Gresl. & E.M. Hansen (Greslebin et al. 2007), prevailing in clayey soils (Vertisols; La Manna et al. 2008). It is evidenced by foliage loss, progressive growth loss and eventually by mortality (Loguercio & Rajchenberg 2005; Amoroso et al. 2015). Any tree can be affected, modifying the structure and dynamics of the stand. The PAI in mesic sites is reduced to $3\text{--}4 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$, a level similar to mortality (Loguercio et al. 2018a). The canopy opening due to the disease, triggers the natural regeneration of *A. chilensis* and also allows the entry of *N. dombeyi*, when there is a nearby seed source (Loguercio 1997; Amoroso et al. 2012). Mixed secondary *N. dombeyi*–*A. chilensis* forests, as a transition between the pure forest types, present two strata, with the first species in the upper

strata and the second one in the lower strata (Veblen & Lorenz 1987; Dezzotti 1996; Caselli et al. 2021). Despite the younger age of *N. dombeyi*, it contributed more to the stand growth, because of its greater growing space efficiency (relation between PAI in volume and leaf area index) (Caselli et al. 2021). Conversion from pure *A. chilensis* forest to mixed *N. dombeyi*–*A. chilensis* forest can increase the PAI to $10\text{--}15 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ (Loguercio et al. 2018a; Caselli et al. 2021). But in xeric sites, it has been observed that the regeneration in gaps of *N. dombeyi* can be affected by extreme drought events, particularly in rocky soils with steep slopes (Suárez & Kitzberger 2008).

Silvicultural experiences

Silvicultural studies have been carried out on these forest types in the region based on the interpretation of the natural dynamics in forests without interventions and their responses to cuttings, covering the different stages across the forest management cycle (Chauchard et al. 2012). Forest management based on silviculture supported by ecological processes of mixed *Nothofagus* forests began in the Lacar lake basin in the late 1980s within Protected Areas. The main silvicultural system applied in the *Nothofagus* forests of northern Patagonia (Reserve of Lanín National Park) includes the shelterwood method for the harvest and regeneration of even-aged stands (Chauchard 1988, 1989; Chauchard et al. 1995, 1998, 2003), through the gradual elimination of the mature cohort and the establishment of natural regeneration during approximately a period of 20–25 years (Chauchard & González Peñalba 2008) (Figure 13a). Dominant, healthy, well-formed, and stable trees of the older cohort, with the potential to grow larger, are retained for variable periods. This is carried out to preserve a seed source until immature plants become established and protect them from the cold and desiccation of the winter and summer seasons. Also, these trees are reserved for soil protection and improve drainage. The system design is completed with standing or dying trees and fallen trees that are kept in the forest site to preserve the structural complexity and promote the maintenance of the biological and functional diversity of the ecosystem. In this way, a variable and dispersed structural retention system is formed in a productive stand.



FIGURE 13: Mixed *Nothofagus* sp. forest in understorey reinitiation stage with first cutting of shelterwood system in Lanín National Park (above); even-aged thinned *Austrocedrus chilensis* forest in El Guadal Reserve (Rio Negro Province) (below right); and mixed *Nothofagus dombeyi* and *Austrocedrus chilensis* secondary forests in El Manso valley (Rio Negro Province) (below left).

In the shelterwood system, the preparatory cuttings for the beginning of the harvest and renewal of the stand are implemented at the end of the stem exclusion stage and onset of the understorey reinitiation stage (*sensu* Oliver and Larson 1996), and they are basically carried out to increase the diameter of the stem and expand the tree crowns that will supply seeds to the stand. This treatment is not intended to begin the renovation process. The objective of the dissemination cuttings is to open the site sufficiently to allow the establishment of regeneration under the protection of the remaining adult trees. After 2–3 years, secondary cuttings are carried out in order to homogenise the spatial distribution of open areas for regeneration. Final cuttings are applied to free regeneration from competition and promote the potential of the remaining mature trees to increase their value. The final cuttings, which could not be applied and conserve these trees for a biodiversity reservoir (retention), represent the last intervention within the regeneration period, and they must be carried out when a minimum recruitment is guaranteed (2,000–2,500

saplings ha^{-1} , minimum 2 m height and good sanitary) and with a homogeneous distribution in the stand. However, in those management units in which this period has concluded, the final cuttings have not yet been carried out.

In stands, in the stem exclusion phase, the remaining crown cover of the preparatory cut is between 70–80% of the original (total crown cover), trying to leave a minimum of $35 \text{ m}^2 \text{ ha}^{-1}$ of basal area. Then, with the reproduction cuttings, the crown cover remains between 30 to 40% and the basal area between 15 to $25 \text{ m}^2 \text{ ha}^{-1}$ (González Peñalba et al. 2008, 2010). Timber production from harvest cuttings in mature stands was estimated from 150 to $400 \text{ m}^3 \text{ ha}^{-1}$, depending on the site quality. The remaining trees in mature stands showed a PAI in diameter of 0.36 cm yr^{-1} and a PAI in volume of $5.8 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ ($1.4\% \text{ yr}^{-1}$), for periods of 15 to 26 years. However, the main objective of the treatment in mature stands is to favor the establishment of a new cohort in suitable physical conditions.

Since 1988, a forest management monitoring program has been carried out in Lanín National Park, in which the composition, abundance, size and growth of recruited trees and seedlings are evaluated every 5 years, as well as the abundance of stump sprouts of *N. obliqua* and *N. alpina* (González Peñalba et al. 2016). In addition, the abundance and dynamics of tree regeneration is a key indicator of forest management (Raison et al. 2001), and for that reason it has been intensively assessed in the management area (Dezzotti et al. 2003, 2004; Sola et al. 2015; Dezzotti & Ponce 2018; Sola et al. 2020). The mean regeneration density recorded included 4,963 seedlings ha^{-1} (height < 2 m) and 5,735 saplings ha^{-1} (≥ 2 m). The response of *Nothofagus* stands to the shelterwood system is very positive considering the quantity and quality of the natural recruitment, whose values exceed the pre-established prescriptions. The silvicultural system also generally maintained the original composition of species present in the upper canopy. However, there are particular situations in the environmental gradient in which less intense cuts are recommended to favor the establishment of *N. alpina* (Sola et al. 2015).

With regards to vegetative reproduction in *N. obliqua* and *N. alpina* stands under management, the number of healthy sprouts in the stumps was recorded after 5 and 10 years of felling the trees (González Peñalba et al. 2016). After cutting, 67% of the stumps presented sprouts, with an average of 23 sprouts per stump, of which only 28% were healthy. During the second measurement a few years later, there were down to 11 sprouts per stump, of which only 32% were healthy. These sprouts had a DBH of 5.3 cm, a height of 5.9 m, and height growth of 0.36 m yr^{-1} . The high proportion of stumps with live shoots, but in poor condition, suggest that sprout management should be applied to improve their performance.

On the other hand, results of studies regarding silviculture of pure and mixed *A. chilensis* and *N. dombeyi* post-fire secondary forests, suggest that adaptive management should be implemented. In pure *A. chilensis* forests, different silvicultural approaches are

recommended for healthy stands and for stands affected by the “cypress disease” (Loguercio et al. 2018a). For healthy stands, predominantly in the stem exclusion stage, the management proposal is aimed at improving the quality based on the best trees (healthy, vital and with good stem shape), according to the current structure and the site. In mesic sites, the application of low and free thinning in even aged stands of 60–70 years (Figure 13b), with cutting intensities between 15 and 30% and a harvested volume between 50 and $100 \text{ m}^3 \text{ ha}^{-1}$ (Figure 14a), presented a PAI in volume between 4 and $5.6 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$. On average, the PAI in diameter was between 1.7 and 2.8 mm yr^{-1} (in dominant trees between 3.3 and 4.5 mm yr^{-1}). In more xeric sites and stands with irregular structure, the single-tree selection method, combined with release of the best trees of small diameter classes, yielded $50\text{--}60 \text{ m}^3 \text{ ha}^{-1}$ and a PAI of 2 to $4.2 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ (Figure 14b) (Loguercio 1997).

In diseased stands of *A. chilensis*, the management possibilities with ambitious productive objectives are limited. There, improvement thinning and cutting of diseased trees are proposed, but only after their death. The goal is to take advantage of the long growth period that diseased trees can live by contributing to the production of the stand (Loguercio & Rajchenberg 2005, Loguercio et al. 2018a). Sanitation felling to control the disease, as removing live diseased trees, has not been effective, since soon new healthy individuals are affected. Natural regeneration must also be ensured when the density requires it, preserving or promoting a protective understorey, necessary for the establishment of the regeneration. As a result, the stand structure becomes stratified, and the stand gradually converts into an uneven-aged forest. Under these conditions, it has been registered that mortality is similar to PAI, which is around $3\text{--}4 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$, with felling cycles of 3–5 years (Loguercio et al. 2018a). In this way, extensive forest management should be carried out to ensure productive sustainability. The intensity and frequency of the cuttings are subject to the evolution of mortality.

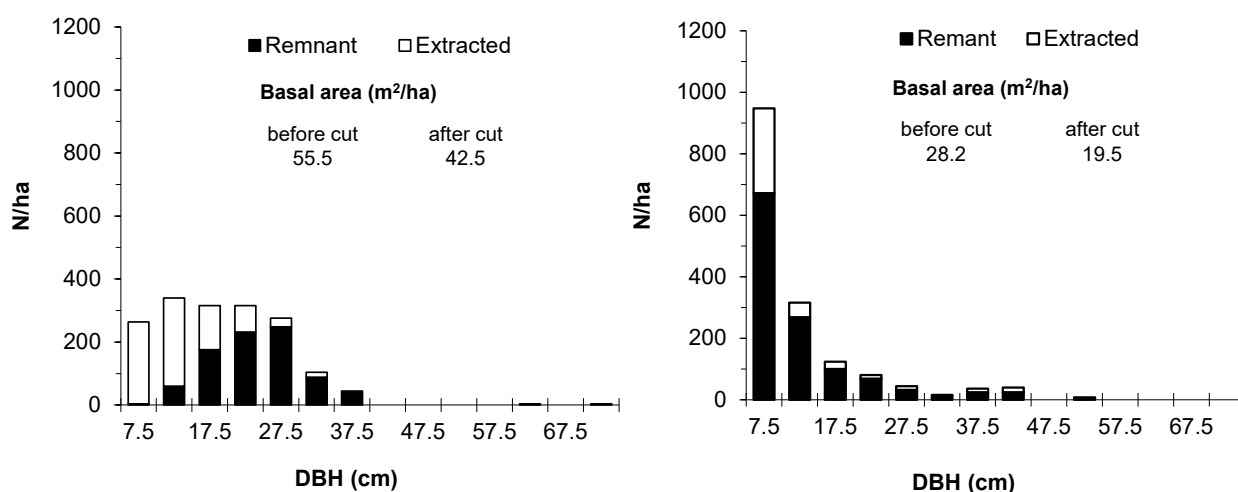


FIGURE 14: Diameter frequency distribution of an even-aged stand with low thinning (left); and an uneven-aged stand with a single-tree selection cutting, including release of best trees with small diameters (right), in pure *Austrocedrus chilensis* forests.

In areas with precipitation above 900 mm yr⁻¹, the regeneration entry of *N. dombeyi* observed in diseased *A. chilensis* stands (i.e., in the Reserve Loma del Medio-Rio Azul in El Bolson, Rio Negro Province) can generate the possibility of conversion to more productive mixed stands with stratified structures (Loguercio 1997, Amoroso et al. 2012, Caselli et al. 2021). There are still no silvicultural management experiences on these mixed forests. However, studies have begun from which first recommendations emerge based on the leaf area index (LAI) as a control variable. Stand structure could be manipulated through silviculture to provide more LAI to the most efficient components of the stand (species/strata), improving the productivity (O'Hara 2014). The growth of these mixed secondary forests is positively correlated to the LAI, but it is more related to the distribution by species and strata than to the total LAI of the stand (Figure 15a) (Caselli et al. 2021). In forests with 65% of the stand LAI occupied by *N. dombeyi* in the upper stratum, the PAI can reach 15 m³ ha⁻¹ yr⁻¹ (Figure

15a and 15b). Then a higher productivity of any mixed stand will be achieved by increasing the participation of *N. dombeyi* in that stand LAI, especially in the upper strata (Figure 13b). The growing space efficiency (GSE) (PAI in volume per unit of leaf area) of *N. dombeyi* doubles that of *A. chilensis* (Figure 16a) (Caselli et al. 2021). However, the GSE decreases as the LAI of the stand increases (Figure 16a), because the proportion of the shaded areas of the canopy increases (Waring & Schlesinger 1985).

Both species present their maximum GSE in the juvenile stage (Figure 16b). To take advantage of the rapid initial PAI of *N. dombeyi*, the best trees must be released from their competitors so that they quickly reach and dominate the upper stratum. However, considering the differences in tolerance of the two species, it would be appropriate to promote a greater participation of *A. chilensis* in the lower stratum, below *N. dombeyi* in the upper strata.

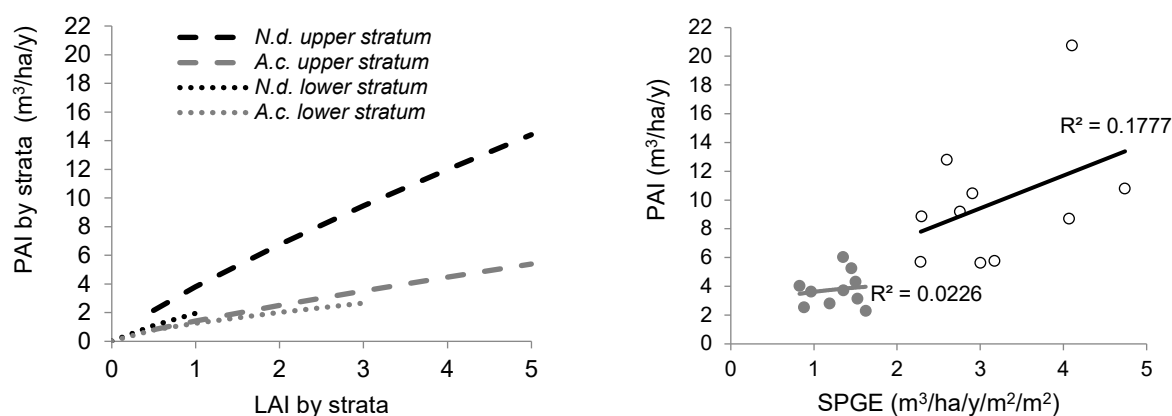


FIGURE 15: Periodic annual increment (PAI) in volume of *Austrocedrus chilensis* (grey) and *Nothofagus dombeyi* (black) in relation to their stand leaf area index by strata (LAI) (left); and PAI by species in relation to the specific growing space efficiency (SPGE) (right) from stands in El Manso valley and Loma del Medio-Río Azul Forest Reserve (El Bolsón), Rio Negro province and Los Alerces National Park (Caselli et al. 2021).

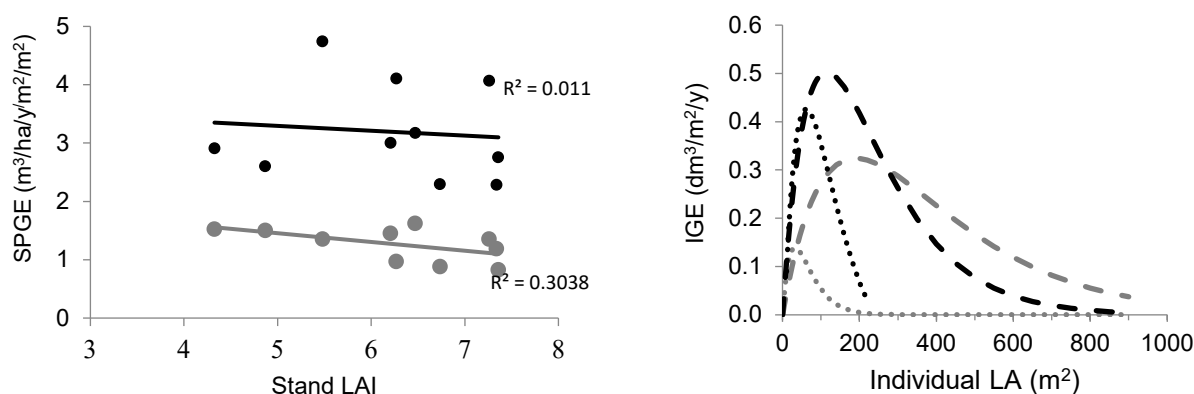


FIGURE 16: Specific growing space efficiency by species (SPGE) of *Nothofagus dombeyi* (black) and *Austrocedrus chilensis* (grey) in relation to stand leaf area index (LAI) (left); and individual growing space efficiency (IGE) in relation to individual leaf area (LA) by species (right). The dashed and the dotted black lines correspond to *N. dombeyi* in the upper and in the lower strata, and the same applies for *A. chilensis*, respectively, from stands in El Manso valley and Loma del Medio-Rio Azul Forest Reserve (El Bolson), Rio Negro province and Los Alerces National Park (Caselli et al. 2021).

Cool temperate *Nothofagus* forests and Magellanic rainforests

Distribution, structure and dynamics

The cool temperate *Nothofagus* forests and the Magellanic rainforests of South America (*sensu* Veblen et al. 1996) belong to the southernmost terrestrial forest ecosystem in the world. These forests occur from northern Patagonia (around 43°S) towards the south in the Cape Horn (Figures 2 and 3).

In the Chilean Patagonia and Tierra del Fuego (Chilean administrative regions of Aysén and Magallanes), the *N. betuloides* and *N. pumilio* (including *N. antarctica*) forest types occur in 1,869 and 2,714 thousand ha, respectively (CONAF 2021), of which around 55% and 14% are protected by the Chilean National System of Wild Protected Areas. Meanwhile, in Southern Patagonia in Argentina (Santa Cruz and Tierra del Fuego provinces), most of the forest corresponds to *N. pumilio* (617,000 ha) and *N. antarctica* (229,000 ha). However, evergreen pure and mixed forest also occupy a large portion in the rainy and temperate areas (105 and 33 thousand ha, respectively). The *N. pumilio* forests are protected in National Parks and Provincial Reserves (26%), while 21% are in state lands and 53% in private lands. *N. antarctica* forests are not well protected in formal reserves (2.1%), where most of them are in private lands (96%). Finally, other evergreen and mixed forests are well protected (10.8%), where most of the off-reserve belongs to the state (64%) (Rosas et al. 2021).

In general, the forest stand dynamics of *Nothofagus* forests are influenced by both coarse- and fine-scale disturbances (Veblen et al. 1996). Coarse-scale natural disturbances (e.g., disturbances associated to tectonic origin, glacial processes, snow avalanches, windthrow and fire) produce the whole-stand replacement, which creates even-aged forest stands (Veblen et al. 1996; González et al. 2014). In this case, the stand dynamics fits quite well with the model of Oliver and Larson (1996): stand initiation, stem exclusion, understorey reinitiation and the old growth stage. With the absence of coarse-scale disturbances, the forests reach the old growth stage, and fine-scale disturbances model the forest structure, with the fall of one or several (small groups of) trees, fostering the regeneration of trees in gaps (Figure 17a) (Veblen et al. 1996; Fajardo & de Graaf 2004; González et al. 2014; Promis et al. 2018). The small canopy openings promote irregular structures, which typically form uneven-aged structures through a patchy spatial distribution (Gea-Izquierdo et al. 2004). Also, two-aged or two-story stands are left after windstorms or other disturbances (Schmidt & Urzúa 1982; Martínez Pastur et al. 2020a). In old-growth forest stages, the seedling establishment in small canopy gaps has been documented for *N. pumilio* and *N. betuloides* (Veblen et al. 1996; Donoso & Donoso 2006; González et al. 2006; Promis et al. 2010; Promis 2018). Moreover, *N. betuloides* can persist as advanced regeneration in the understorey for a long time, whereas *N. pumilio* tends to be short-lived under shaded conditions in the understorey, providing important insights about the shade tolerance of both

species; the former has been considered more shade-tolerant than the latter (Veblen et al. 1996; Martínez Pastur et al. 2012). On the other hand, *N. antarctica* depends more on vegetative reproduction and it can vigorously resprout from roots and stumps after being burned or harvested (Veblen et al. 1996; Donoso et al. 2006; Soler et al. 2013; Promis et al. 2018).

Silvicultural experiences

These *Nothofagus* forests have been used historically for firewood and timber (Gea-Izquierdo et al. 2004; Promis et al. 2018). Since the end of the 19th century, a huge area of forests was burned, logged and converted to farmland and pasture (Martín 2005, 2006). However, at the present, *N. pumilio* is the most important native species for timber production in Chilean and Argentinean Patagonia and Tierra del Fuego (Martínez Pastur et al. 2000; González et al. 2006; Cruz et al. 2018). These forests were intensively harvested during the last century, mainly for sawnwood purposes, although firewood production was very intensive prior to the natural gas-based energy in Southern Patagonia. *N. betuloides* forests have not been managed extensively, although an ca. 280,000 ha has been estimated as suitable for timber production in southern Patagonia and Tierra del Fuego (Promis et al. 2008). Meanwhile, *N. antarctica* forests have been historically used for pastoral purposes (e.g. mainly cattle), however, recently silvopastoral systems have been proposed for managing these forests (Donoso et al. 2006; Peri et al. 2016; Sotomayor et al. 2016). In Argentina, its expansion is promoted considering a property planning in homogeneous areas (herbivorous steppe, forest and riparian grasslands), adjusting the livestock load according to the net primary productivity of the grasslands, and protecting tree regeneration from browsing (Peri et al. 2016). Furthermore, an ecosystem service-based framework for the sustainable management of the natural ecosystems of Patagonia is being promoted, and several planning proposals were developed for the region considering the synergies and trade-offs with biodiversity (Peri et al. 2021). For example, most of the forest planning at Southern Patagonia (Argentina) were based on provisioning ecosystem services despite the trade-offs with other activities (e.g., tourism or recreation) (Carrasco et al. 2021).

In Chile and Argentina, to harvest a forest a compulsory forest management plan must be accepted by the Forest Service. Following the Chilean forest law, *N. pumilio* and *N. betuloides* forest types can be managed with the shelterwood method (Figure 17b) and selection method (Figure 17f). In Argentina, different regeneration systems can be applied. Shelterwood cuts were the preferred one for *N. pumilio* forests, however, the variable retention system with aggregated retention and dispersed retention have gained attention during recent decades (Figure 17d) (Martínez Pastur et al. 2019). *Nothofagus betuloides* forests were harvested during the 1990s but currently they are not being harvested. Besides this, thinning was implemented in *N. antarctica* forests following silvopastoral prescriptions



FIGURE 17. Forest regeneration in a natural canopy gap in: (a) *Nothofagus pumilio* forests; (b) thinning in a secondary *N. pumilio* forest; (c) application of single-tree selection in a mixed *N. pumilio* - *Nothofagus betuloides* forest (d) regeneration established before final cutting in a shelterwood method in a *N. pumilio* forest; (e) regeneration cutting in a shelterwood method in a *N. pumilio* forest; (f) variable retention system with aggregated retention in a *N. pumilio* forest; .

(Peri et al. 2016), as well as in secondary *N. pumilio* forests (Figure 17e), but over small areas compared to the harvested ones (less than 5% each year) (Martínez Pastur et al. 2013).

The shelterwood method is the most frequent silviculture treatment applied in these native forests in Chile (Figure 17b) (Schmidt et al. 2003; Cruz & Schmidt 2007). It removes the overstorey in successive cuttings to promote the establishment of natural regeneration leading to an even-aged structure (Cruz & Schmidt 2007). Frequently, it has been applied through uniform shelterwood variant, which leaves the trees that provides shelter and seed production (between 40% and 60% of the initial basal area of the natural stands) (Schmidt et

al. 2003). After the cuts, the natural regeneration quickly reacts, and new seedlings establish, covering most of the forest floor (> 30 thousand plants per ha) (Schmidt et al. 2003; Martínez Pastur et al. 2017). Between 5 and 10 years after harvesting, when saplings reach heights of 50 to 100 cm (Figure 17c), the final cutting is proposed, removing the remnant overstorey trees (Cruz & Schmidt 2007). Clearing in the secondary forest should be initiated 20–40 years after the harvesting, while pre-commercial thinning in the pole development stages and commercial thinning from below or selective thinning should be applied in the following decades (Martínez Pastur et al. 2013; Cruz et al. 2018). The management rotation age ranges from 120 and 160 years, with stand

volumes up to 500 m³ ha⁻¹, where potentially near 20% corresponds to saw timber logs and 50% to small pole-sized logs (i.e., firewood or biofuel) in these forests that have not had previous management (Schmidt et al. 2003; Martínez Pastur et al. 2004; Cruz & Schmidt 2007).

The site quality of *N. pumilio* forests ranges between 15 and 30 m height in the mature stands. Harvesting is mostly conducted in forests growing in mid- to high-quality sites and when the dominant trees reach > 20 m in height (Martínez Pastur et al. 1997; Schmidt et al. 2003). The basal area of these stands can range from 60 to 80 m² ha⁻¹, but it can reach up to 100 m² ha⁻¹ in over-stocked stands, while total over bark volume can reach between 400 and 1,200 m³ ha⁻¹, which is affected by site quality, stocking density, stand stage and previous use (González et al. 2006; Martínez Pastur et al. 2000, 2004). The harvested basal area after the regeneration cutting ranges between 33% and 67% and the over bark volume between 33% and 46% (Martínez Pastur et al. 2000; Schmidt et al. 2003). The PAI in diameter before harvesting is between 1.4 and 2.1 mm yr⁻¹, and the residual trees improve to 2.5–4.2 mm yr⁻¹ after management. Thus, the residual canopy in the regeneration cutting showed a PAI in volume between 3.0 and 4.2 m³ ha⁻¹ yr⁻¹. Tree regeneration 10 years after harvesting has been successfully established in the managed stands, with more than 30,000 plants per ha, but it can reach to up 400,000 plants per ha. Usually in southern Patagonia, saplings reach an average of 100 cm tall after 10–15 years, evidencing low growth rates compared to the other *Nothofagus* forests described before. However, in some cases these seedlings may only reach an average of 20 cm in height at these ages due to the browsing damage caused by *Lama guanicoe* (e.g., Martínez Pastur et al. 2016). This undesirable effect of over-browsing in combination with an extreme climate (e.g., freezing and dryness) can delay the growth of regeneration, and the final cut must be postponed by 10–20 years (Martínez Pastur et al. 2017). Beside this, in the following years, successive thinnings must be applied to enhance the quality and growth of trees. With different intensities of thinning, the PAI in diameter increases from 1.5–2.2 mm yr⁻¹ (without thinning) to 1.9–3.9 mm yr⁻¹ in southern Patagonia (Schmidt et al. 2003; Cruz et al. 2018; Mundo et al. 2020). However, further north in Patagonia (45–48°S), with better site conditions, the PAI in diameter ranges from 2.3 to 6.0 mm yr⁻¹ after different thinning intensities and 26 growing seasons (Nuñez & Vera 1992).

On the other hand, since *N. betuloides* and *N. pumilio* forests follow the gap regeneration mode (Figure 17a), and can form irregular forest structure following small-scale disturbances (e.g., canopy gaps), it is possible to think in single- and group-selection silviculture for these forests (e.g. Promis 2013). This system presents unique advantages compared to even-aged systems, mainly related to biodiversity conservation and provision of ecosystem services such as aesthetic, supporting of biodiversity and soil protection (Promis 2013; Martínez Pastur et al. 2020a; Peri et al. 2021). However, it has

not been applied yet as such. Most of the selection harvestings are confused with selective cuttings, where the landowner cuts the best and leaves the worst in terms of wood for timber (*sensu* Donoso 2013; Nyland 2016). It is expected that single-tree selection (Figure 17e) could be applied in places where the mean annual precipitation is above 800 mm yr⁻¹ (DP Soto unpublished data). Below this precipitation value, *N. pumilio* forests are located in the ecotone with the steppe and much research is needed to potentially sustain management operations or leave these marginal forests for biodiversity conservation initiatives, especially within the scenario of climate change (e.g. regeneration process can fail due to dryness) (Aschero et al. 2021; Soto et al. 2021). Group-selection cuttings for *N. pumilio* have also been proposed, but the rainfall level must be taken into account (see López Bernal et al. 2012). Therefore, selection cuttings (single-tree or group-selection) may be implemented in some cases to improve the plant community (Soto et al. 2021), structure and wood quality of irregular stands dominated by *N. pumilio*.

There is an increasing concern regarding the provision of ecosystem services (e.g. those values without market value) and biodiversity conservation in the managed forests (e.g. Native Forest Law 26,331 of Argentina) (Martínez Pastur et al. 2020b). In this framework, variable retention harvesting offers an alternative that combines timber and *in-situ* conservation (Martínez Pastur et al. 2009). Martínez Pastur et al. (2019) reported that legacies maintained in managed stands varied according to the different forest management plans, from small patches (e.g., aggregates of 3,000 m² to a couple of hectares) to isolated trees (e.g., dispersed retention) with desirable characteristics for conservation (e.g. hollow trees). The implementation of this system generated stands with greater heterogeneity, allowing to regenerate the stands in the harvested areas, but maintaining 80% of the original biodiversity in the managed areas (Martínez Pastur et al. 2013).

Prospects for silviculture of South American temperate forests

The southern cone of South America hosts a great variety of temperate forests between 37 and 55°S. These forests are mixed and diverse in its northern part, and mostly pure in south and east Patagonia. The mixed forests of south-central Chile host a great diversity of tree species due to the existence of many shade-tolerant and mid-tolerant species, while towards Argentina they are less complex due to the drastic reduction in precipitation. For each, here we have provided the main results and findings about their silviculture.

South American temperate forests basically represent a relevant global biome not only in terms of their great diversity and endemisms, but also in terms of their potential to provide high-quality timber. A vast area of native forests is available to supply timber and non-timber forest products as well as many other ecosystem services (water, recreation, tourism, etc.) for local populations. These forests are dominated by *Nothofagus* species, which are pervasive throughout

the region, but they also have many other angiosperm species (especially in Chile), and also the conifer *A. chilensis* (especially in Argentina), within Valdivian and Magellanic Subpolar ecoregions. These forests are, in general, highly productive, greater in the north respect to the south, and in Chile compared to Argentina. Coastal Temperate Rainforests are more productive than other temperate forests (Pan et al. 2013), and in the case of Chile and Argentina the influence of soils of volcanic origin adds to their high productivity. The simple fact that basal area in these South American forests reach 80–100 m² ha⁻¹ in old-growth conditions (e.g., Donoso & Lusk 2007 (Andes of Chile); Gutiérrez et al. (2009) (Angiosperm-dominated evergreen forests), Donoso & Soto (2016) (Coastal and Andean forests of Chile)) compared to near 30–70 m² ha⁻¹ in most (except for the Pacific coastal forests of North America) forests of the Northern Hemisphere (e.g., Ziegler 2000 (New York); D'Amato & Orwig (2008) (Massachusetts), Motta et al. (2014) (Italian Alps) is a reflection of the differences in the capacity of site occupancy in these different biomes. Volume growth rates that can reach values near 20 m³ ha⁻¹ yr⁻¹, especially in secondary forests, are similar to values in plantations of some exotic species (Cubbage et al. 2007). These high growth rates also represent an opportunity for carbon sequestration for climate change mitigation. However, the diversity of management tools, including those that in addition to producing timber also promote the development of old-growth forest structures or maintenance of legacies, is a great opportunity within the management of South American temperate forests.

We have addressed the main experiences in silviculture of temperate forests in Chile and Argentina. This review reflects that there are still many aspects related to silviculture that have not been researched or informed yet, and for which we face many challenges. Of special importance and urgency is to develop climate-smart forestry, an emerging branch of sustainable forest management that aims to manage forests in response to climate change, with a special emphasis in enhancing the provision of ecosystem services (Bowdtich et al. 2020). Progress has been made in recent research on the sensitivity of tree species to climate change. For instance, it has been found that extreme drought events produced mortality and reduced growth of *A. chilensis*, *N. dombeyi* and *N. pumilio* in Northern Patagonia (Suarez & Kitzberger 2010; Amoroso et al. 2015; Rodríguez-Catón et al. 2016; Marcotti et al. 2021). Although a rise in temperature should favour height growth rates, the decrease in precipitation in critical periods of a growing season is predicted to decrease the height growth of *N. alpina* (Salas-Eljatib 2021). New silvicultural research and practice should consider these aspects into measures for adaptive management. Investigations have also been initiated on carbon storage estimation over regional scales for *N. pumilio* forests (Poulain 2009), on carbon dynamics due to the effect of fires (Bertolin et al. 2015; Defossé et al. 2020), carbon storage along the sequence of stand development after the application of the shelterwood method in *N. pumilio* forests (Schmidt 2009), and also studies on carbon reserves and sequestration

in successional forests are in process (G Loguercio, pers. communication). Similarly, ecological silviculture (Palik et al. 2021) provides a framework to sustain the provision of commodities and ecosystem services from forests by anticipating and accommodating social and biological changes (e.g., those triggered by global change).

While many scientists, politicians and stakeholders are willing to achieve greater levels of implementation of silviculture in the field to sustain goods and services of local relevance, and to reverse forest degradation, we must at the same time promote silviculture to mitigate climate change and increase the adaptive capacity of forests. In this sense, the experiences gained in both Chile and Argentina with irregular silviculture (e.g., continuous cover forestry, including selection cuttings and irregular shelterwood cuttings), in Argentina with adaptive management of damaged *A. chilensis* forests, and in Chile with variable-density thinnings to enhance old-growth attributes in secondary forests (Donoso et al. 2020; Biscarra et al. 2021), are a first step in the direction of developing climate-smart forestry, or ecological silviculture. There is still, however, a lot of research needed and work to do in regard to adaptive strategies (resistance, resilience, transition) of forests to climate change (Palik et al. 2021).

There are also interesting prospects in the management of plantations with native forests (Donoso et al. 2015). These plantations can aid the recovery of high-graded forests or can be established as pure or mixed forests to rebuild forests where they have been eliminated in the past (i.e. Promis 2020). In other words, these planted forests can be managed in even-aged systems that are more adequate for pioneer and valuable tree species, or can be a starting point to recover high-value mature or old-growth forests, a process that requires adaptive silviculture during the transition, and continuously in many cases (except those for preservation). Mixed-species plantations are an excellent and viable alternative (e.g. Ojeda et al. 2020), especially in Chile that has a greater diversity of tree species of different shade tolerances compared with Argentina. However, in Argentina mixed plantations including species with differential drought resistance are also proposed as an alternative for adaptation to climate change (Caselli et al. 2019, 2021). These plantations may not only be important at the stand level but can also aid build more resilient landscapes (e.g., in the degraded Coastal range of Chile with large-scale monocultures of *Pinus radiata* or *Eucalyptus* sp. plantations), which are also facing more frequent and severe fires (Veblen et al. 2008; González et al. 2011). Therefore, plantations with native species are a good option to build more resilient landscapes to face the current climate and ecological crises.

The silvicultural knowledge exposed here comes from experimental trials and technical management in specific areas. The expansion into long-term demonstration units for different silvicultural systems in different forest types, at an operational scale, accompanied with precise economic assessments, is a challenge to build economies based in these native forests for the benefits of people

and the environment. For this effort to succeed, it must include the urgent need to create forests of adaptive capacity to novel disturbances and with great capacity to sequester and store carbon to contribute to the mitigation of climate change.

Competing interests

The authors declare that they have no competing interests.

Acknowledgements

Pablo J. Donoso acknowledges research grant FONDECYT N°1210147. Daniel P. Soto acknowledges research grant FONDECYT N° 11181140.

Abbreviations

Diameter at breast height (DBH)
Crop tree thinning (RL)
Growing space efficiency (GSE)
Individual growing space efficiency (IGE)
Individual leaf area (LA)
Leaf area index (LAI)
Low thinnings to 2 x 2 (2 m or R2)
Low thinnings to 3 x 3 (3 m or R3)
Low thinnings to 4 x 4 (4 m or R4)
Mean annual increment (MAI)
Modified internal rate of return (MIRR)
Net present value (NPV)
Periodic annual increment (PAI)
Potential areas for silvicultural management (USG)
Relative density index (RDI)
Specific growing space efficiency (SPGE)

Authors' contributions

PD and AP participated in the conception and design of the review. PD, AP and GL were in charge of critically improving and editing the manuscript before and after receiving the peer review from three reviewers. PD, AP, CN and CS-E participated in the design and helped to draft the section of the *Nothofagus*-dominated forests of South-Central Chile. PD, CN, DS and AV-G participated in the design and helped to draft the section of the Angiosperm-dominated evergreen forests. GL, HB, MC, LC and MGP participated in the design and helped to draft the section of the *Nothofagus* and *Austrocedrus chilensis* forests in the Argentinean Northern Patagonia. AP, GC, GM-P, PN, DS participated in the design and helped to draft the section of the Cool temperate *Nothofagus* forests and Magellanic rainforests. All authors read and approved the final manuscript.

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