Restoration of burned and post-fire logged *Austrocedrus chilensis* stands in Patagonia: effects of competition and environmental conditions on seedling survival and growth

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**Abstract.** In Andean Patagonia, Argentina, severe wildfires produce high mortality in *Austrocedrus chilensis* forests. Owing to its high timber quality, *A. chilensis* trees are generally salvage logged right after wildfires. Post-logged areas result in open, denuded stands with low herbaceous or shrub cover, which precludes natural *A. chilensis* regeneration. In severely burned *A. chilensis* stands, we determined how different site conditions (salvage logging, SL, and without logging, WL), and the combination of different methods of control of above- and belowground competition and micro-environmental factors (incident radiation, soil temperature and soil moisture) affected survival, growth and hydric status of planted *A* seedlings. Two growing seasons after plantation, seedling survival was lower than 10% at SL sites, whereas it was near 90% at WL sites. Four seasons after establishment, and regardless of competition treatments, no seedlings survived at SL sites, whereas over 75% survived at WL sites. Radiation attenuation by canopy of WL sites benefited *A. chilensis* seedling survival, and no additional control of early successional herbaceous species should be necessary to attain high seedling survival 4 years after restoration. Selective logging, by leaving ~50% of burned snags, and active restoration practices, may help reconcile economic needs of society and ecological requirements of *A. chilensis* for recovering former structure and functions.

**Additional keywords:** Andean cypress, facilitation, seedling performance, temperate forests.

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**Introduction**

Wildfires have shaped the structure and functioning of most terrestrial ecosystems (Wright and Bailey 1982; Crisp et al. 2011). Every ecosystem, however, has its own fire history, specifically defined as its *fire regime*. The fire regime is composed of different elements, identified as (1) *fire type* and *intensity*, (2) *size* of typical significant fires and (3) frequency or *return intervals* for specific land units (Heinselman 1981). In forest ecosystems, long fire return intervals may produce high severity fires, whereas short fire return intervals may lead to low severity fires (Pyne 1984; Agee 1993; Beschta et al. 2004). High severity wildfires usually produce major changes in community and population structure, modifying resources availability and altering the physical environment (Agee 1993). These changes favour the establishment of early successional species, also creating suitable conditions for invasion by new species. High severity fires may also have long lasting effects on low fire-tolerant forests, especially in those whose most prevalent species reproduce strictly by seed, which are killed during, or because of, the fire disturbance. Moreover, if remaining burned snags are removed from stands after high severity fires, additional changes are created in the post-fire denuded environment, increasing solar radiation, and modifying soil temperature and wind speed (McIver and Starr 2000; Beschta et al. 2004). In other cases, in which seeds of forest species are slightly or not damaged by fire, the rapid growth of herbaceous species right after the fire disturbance may constrain their germination and establishment, by out-competing for water and nutrients (Rose and Rosner 2005). The combination of these factors generally delay the natural process of forest recovery and, in extreme situations, preclude burned forests from recovering to pre-disturbance conditions. Under these situations, the fire-disturbed stand needs active restoration practices to return to its normal successional path, by sowing or planting former species. However, in order to guarantee the establishment of the target species, restoration practices sometimes require additional microsite manipulation to avoid or diminish any deleterious biotic or micro-environmental effects on planted
seedlings (Whisenant 1999). These effects are mainly caused by competition for soil water or nutrients by early successional herbaceous vegetation, excessive soil heating by higher incoming radiation reaching denuded soils, or by the lack of nurse plants (generally shrubs or half-shrubs) that protect seedlings from excessive temperature fluctuations (Padilla and Pugnaire 2006; Urretavizcaya et al. 2017). Forest-productivity research has identified interactions between competing vegetation and logging debris left after forest harvesting. In general, retention of logging debris in temperate zones inhibits development of herbaceous and sometimes woody species. Retention of logging debris has also been associated with mulching effects. These effects imply conservation of soil water near the soil surface by reducing evaporation and topsoil temperatures (Harrington et al. 2013).

In restoration projects, controlling competing species may be sometimes necessary for diminishing their deleterious effects on planted seedlings (Jobidon et al. 1998; Löff et al. 1998; Dinger and Rose 2009; Maguire et al. 2009). This control may involve using herbicides, manual removal of competing vegetation or the application of mulch (Löff et al. 1998; Harper et al. 2005; Jylhä and Hytönen 2006). Removing competing vegetation by chemical or mechanical methods reduces soil cover and modifies some topsoil physical conditions, increasing incoming radiation and soil-surface temperature, and also influencing soil-water content (Defossé and Robberecht 1996; Harrington et al. 2013). For some forest species at seedling stages, high topsoil temperatures at stem-collar level may negatively affect their survival (Kolb and Robberecht 1996). However, the use of mulch around seedlings may increase survival by reducing temperature variations, conserving soil moisture and diminishing plant evapotranspiration (Flint and Childs 1987). Although mulching is widely used in horticulture, it has scarcely been used in restoration projects aimed at recovering forest species under field conditions (Blanco-García and Lindig-Cisneros 2005).

The forests of Austrocedrus chilensis (D. Don) Pic. Ser et Bizarri (commonly known as cypress), are part of the Andean–Patagonian temperate forests that run along the Cordillera de los Andes in southern Chile and Argentina (Cabrera 1976). In Argentina, A. chilensis is located in a narrow belt that goes from 37°08′ to 43°44′S in the eastern slope of the Andes (Pastorino et al. 2006). Austrocedrus chilensis forms either pure or mixed stands. Pure stands occupy ~95 500 ha, whereas those in which A. chilensis grows intermixed with Nothofagus dombeyi Mirb. Oerst. (locally named as coihue) cover ~32 800 ha. National fire statistics show that during the last 20 years, an average of 3000 ha of pure or mixed A. chilensis stands have been annually affected by wildfires (Secretaria de Ambiente y Desarrollo Sustentable de la Nación 2013; Mohr Bell 2015).

After wildfires, the main factors that influence recovery of affected forests are intensity and severity of each particular fire, size or area covered, and environmental conditions prevailing during and after the fire event (Connell and Slatyer 1977; Veblen et al. 1999). For A. chilensis forests, the lack of a soil-seed bank after severe wildfires is the initial limiting factor for natural stand recovery (Urretavizcaya and Defossé 2004). Likewise, severe wildfires produce very high A. chilensis tree mortality because its very thin bark is rapidly affected by the fire and, unlike other Patagonian species, it does not re-sprout after fires (Gallo et al. 2004). These conditions mean that active-restoration practices, such as seedling planting or seed sowing, are often necessary for restoring A. chilensis stands affected by severe fires (Urretavizcaya et al. 2012).

Most burned A. chilensis forests are logged right after fire events because of their high timber quality and market demands (Perdomo et al. 2007). Once burned-log extraction is finished, logged areas result in open, denuded stands with very low herbaceous or shrub cover, similar to what happens in other postfire-logged conifer forests elsewhere (McIver and Starr 2000; Beschta et al. 2004). However, for A. chilensis forests, there are no studies that analyse these conditions. The new macro- and micro-environmental conditions created in these post-burning logged areas may preclude survival and establishment of planted A. chilensis seedlings, sometimes making any restoration plan unsuccessful. Similar to what was stated in Castro et al. (2011) and Harrington et al. (2013, 2018), we hypothesise that, for A. chilensis forests, the presence of burned logs act as structural nurses that, without competing with established vegetation, help reduce incoming radiation and maintain topsoil moisture, crucial for seedling survival during the summer drought. In the case of salvage logging, mulch attenuates the desiccant effects of high levels of direct solar radiation and help reduce evaporative loss by diminishing high topsoil temperatures that are usual in this region during the summer. We also postulate that mulch, as well as a reduction in nearby vegetation competition, has the potential to benefit A. chilensis seedling performance by increasing resource availability and modifying microclimate conditions.

The objective of this study was to determine how different site conditions (salvage logging, SL, and without logging, WL), microsite interventions (different methods of control of above- and belowground competition), and micro-environmental factors (incident radiation, soil temperature and moisture) affect survival, growth and hydric status of A. chilensis seedlings planted in severely burned stands. The outcomes of this study not only cast light on the effects of these factors on A. chilensis seedling establishment, but also help to design appropriate protocols for successfully restoring fire-damaged A. chilensis forests. Additionally, this study provides some guidelines that should be followed for sustainable managing post-fire A. chilensis stands.

Materials and methods

Study site

The study area was located near the town of Corcovado (43°32′S, 71°33′W), in Chubut province, Patagonia, Argentina (Fig. 1). Its climate is “mediterranean-temperate”, with cold and rainy winters and dry and mild-to-hot summers and early autumn. Mean annual precipitation is 800 mm, 75% of which falls during the winter months. Average annual temperature is 9.3°C. Soils of the region derive from volcanic ashes (Etcheverre 1972) and correspond to Andisols in the study area. These are rich in nutrients (except phosphorous), and have high soil-water retention (Colmet Dáage et al. 1993). This area represents the southernmost location of pure stands of A. chilensis in the Patagonian region of Argentina (Dezzotti and Sancholuz 1991). At the end of February 2008 (summer in the
southern hemisphere), a wildfire categorised as extreme behaviour and high severity ravaged 80 ha within this area. The categorisation of this fire was based on regional observations and by applying the classification proposed by Mutch and Swetnam (1995). Under this classification, the studied fire falls in the category 5: very high severity; most sub-canopy trees killed and >50% of canopy trees killed (Fig. 1).

**Experimental design and treatments**

In an area representative of the burned forest, we selected two adjacent and similarly burned *A. chilensis* stands (Fig. 1). One of them (~4 ha) was salvage cut in 2010 (i.e. all remaining *A. chilensis* snags removed), whereas the other (2 ha) conserved all the standing burned trees. Both stands presented similar aspect and 0 to 5% slope. In August 2012 (winter in the southern hemisphere), a split-split plot experiment was installed. Site condition or logging was the main-effect plot and had two levels (SL and undisturbed or WL). Aboveground competition was the second-effect plot and had three levels (aboveground vegetation manually removed (RE), mulch application (MU) and no aboveground removal, or control (CO)). Root competition was the third-effect plot and had two levels (root exclusion (EX), and root control or without exclusion (RC)). The experiment was installed in eight plantation blocks of 8 × 24 m (4 in SL and 4 in WL conditions) (Fig. 1). In every block, 48 *A. chilensis* seedlings were planted at a distance of 2 × 2 m each. At the time of planting, seedlings were 3 years old and were produced at Centro de Investigación y Extensión Forestal Andino Patagónico nursery (Esquel, Chubut, Argentina) from seeds of the same genetic zone of the study site (Pastorino 2015). Each seedling was manually planted in similarly shovel-made holes. Before planting, a sample composed of 20 randomly chosen seedlings were tested to determine their morphological attributes. These attributes (mean ± standard error) were: seedling height (21 ± 0.5 cm); collar diameter (4.2 ± 0.1 mm), aerial
assigned to one of these plots (Fig. 1). In total, 456 aerial competition treatments (RE, MU and CO) were randomly three plots (sample units) containing 16 seedlings each. The three of neighbouring herbaceous species. Each block was divided into of the planted seedlings to isolate their root system from those dry weight (3.1 ± 0.2 g), root dry weight (2.5 ± 0.2 g), aerial to root dry weight ratio (1.3) and sturdiness index (50) (Hasse 2007). These values are within the range of customary morphological attributes used for commercial *A. chilensis* seedlings. At planting, plastic root-exclusion tubes 25 cm in diameter and 25 cm long (Cook and Ratcliff 1984; Defosse 1995) were installed in half of the planted seedlings to isolate their root system from those of neighbouring herbaceous species. Each block was divided into three plots (sample units) containing 16 seedlings each. The three aerial competition treatments (RE, MU and CO) were randomly assigned to one of these plots (Fig. 1). In total, 456 *A. chilensis* seedlings were used in the experiment (228 per stand and 48 per block). Right after plantation, all seedlings were identified and their specific height and collar diameter measured for determining further survival and growth. Each seedling was protected by a metallic mesh for avoiding foraging by the exotic European hare (*Lepus europaeus* Pallas).

The mulch came from chipped *A. chilensis* branches and slash residues remaining after the wildfire. Mulch cover was 3–5-cm high in a 40-cm radius around treated seedlings. Aerial vegetation was manually removed with a hoe and a rake from each removal treatment in a 40-cm radius around each seedling, trying not to disturb the topsoil around it (Fig. 2).

Characterisation of the vegetation of the study site
At 3 and 6 months after setting plantation (November 2012, spring in the southern hemisphere, and February 2013, summer in the southern hemisphere), early successional species grown in both SL and WL stands were cut, taken to the laboratory, identified and their dry weight determined. The sampling procedure involved collection of all biomass located within eight 0.5-m² plots randomly distributed within Control plots of SL (*n* = 4) and WL blocks (*n* = 4).

Environmental variables measured
Photosynthetically active radiation (PAR), and soil temperature (ST) and moisture (SM) were measured three times (mid-spring, 11 November 2012; mid-summer, 6 February 2013; and late summer, 22 March 2013) during the first growing season after the start of the experiment. These measurements were performed to determine if there were significant differences among sites (SL and WL) and aerial competition (RE, MU and CO) treatments. For measuring PAR, a Cavadevices Ceptometer (Cavadevices.com, Buenos Aires, Argentina) with Photosynthetic Photon Flux method was used. Topsoil temperature was measured with an infrared laser thermometer Cole–Parmer Model 39650-02 (Instrument Company, Chicago, IL, USA) and at 10-cm soil depth with a Digi-Sense DualLogR thermometer and thermocouples Model 91100-50 (Cole-Parmer Instrument Company Vernon Hills, IL, USA). Soil samples from 0 to 10 cm (2 per plot in each block, 48 per date) and 20 to 30 cm (1 per plot in each block, 24 per date) were taken and their water content determined gravimetrically. At the beginning of the second growing season (October 2013), automatic data loggers Decagon Em5b (Decagon Devices, Pullman, USA) with moisture sensors ECH2O (Decagon Devices, Pullman, USA) at 10-cm soil depth were installed in randomly selected SL and WL blocks and each aerial competition treatment. These measurements allowed continuous recording of soil moisture data to the end of the growing season (March 2014).

Seedling survival, growth and hydric conditions measurements
We recorded seedling survival and height growth at the end of the growing season in 2013, 2014 and 2016. In April 2014 and 2016, seedling-collar diameter was also measured. Survival, height and collar-diameter measurements were performed in all seedlings. In mid-summer (February 2013), the hydric status of seedlings was determined both at predawn and midday at both SL and WL sites, and also in aerial (RE, MU and CO) and belowground (EX and RC) competition treatments. For these measurements, we used a Scholander pressure chamber (BioControl Model 4, Buenos Aires, Argentina). The measuring technique involved cutting of a small portion of a top *A. chilensis* seedling branch, using 24 seedlings at predawn and 24 at midday. Two blocks were randomly selected for each WL and SL site treatment and 4 seedlings were sampled per each aerial (RE, MU and CO) treatment, 2 in EX and 2 in the RC treatments.

Analyses
Vegetation biomass was analysed by analysis of variance (ANOVA) of one factor: site (WL and SL), and independently for each measuring date. PAR, ST and SM determined during the first growing season, were analysed by ANOVA (ANOVA) of one factor: site (WL and SL), and independently for each measuring date. Numerical data were processed using the R 3.1.1 software.
Soil moisture, which started to be continuously measured from the beginning of the second growing season, was analysed through mixed lineal models. Soil-moisture values were averaged at monthly intervals for each treatment, and significant differences among treatments determined for each month during the growing season.

_Austrocedrus chilensis_ seedling survival was evaluated as a function of the first (SL and WL), second (RE, MU and CO) and third (EX and RC) effect treatments. The analyses involved repeated measures through lineal mixed models by considering the three effects as fixed, and blocks and time as random factors. These analyses were performed after the first summer, second winter and second summer (April and October 2013 and April 2014 respectively). Survival at the fourth summer (April 2016) was only measured in aerial (RE, MU and CO) and root (EX and RC) competition in WL treatment, because no survival was registered at SL plots and blocks. In these analyses, _P_-values with Bonferroni correction and a _P_ = 0.05 were used. When significant differences were found, mean comparison was made by using Fisher’s least significant difference. Height growth after the first growing season (2012–2013) was analysed with the same models used for survival. After that date, the same models were used but only for one site (WL), because most seedlings at SL plots had already died. Height and collar-diameter growth was analysed for the periods 2012–2014 and 2014–2016 and then for the entire study period (2012–2016). Seedling hydric status was analysed by ANOVA with three factors: site (SL and WL), aerial (RE, MU and CO) and root (EX and RC) competition treatments. For all analyses, we used INFOSTAT Statistical Package (Facultad de Ciencias Agropecuarias, Universidad Nacional de Córdoba, Córdoba, Argentina, www.infostat.com.ar, accessed 7 February 2019) (Balzarini et al. 2008; Di Rienzo et al. 2017) and their interface with the R Foundation for Statistical Computing Platform(R version 3.4.0 The R-CoreTeam 2017, https://www.r-project.org/, accessed 7 February 2019). Unless stated otherwise, the probability level was set at _P_ < 0.05.

### Results

#### Vegetation

At the beginning of the experiment, herbaceous plant cover was similar at all treatment plots and blocks, and the predominant species were _Holcus lanatus_, _Acaena ovalifolia_, _Carduus thoermeri_, _Rumex acetosella_ and other grasses. Total herbaceous (grasses and herbs) aboveground biomass was not significantly different at both SL and WL plots. Three and six months after plantation (4.5 and 5 years after the fire), the biomass averaged 272 ± 38 and 414 ± 25 g m⁻² at SL plots and 395 ± 44 and 431 ± 34 g m⁻² at WL plots.

#### Environmental variables

**Photosynthetically active radiation**

Photosynthetically active radiation was significantly higher in SL treatments as compared with WL treatments, but was not significantly different among aerial (RE, MU and CO) treatments. Within sites (SL and WL), the higher PAR differences were registered during mid-spring 2012. Salvage logging showed PAR values above 1000 µmol m⁻² s⁻¹, almost four times the PAR registered (290 µmol m⁻² s⁻¹) in WL treatments. Photosynthetically active radiation differences started to decline towards mid- and late summer (Fig. 3).

**Soil temperature**

Topsoil temperature in mid-spring and mid-summer of the first growing season was not significantly different between sites (SL and WL), but it was significantly different among competition treatments (Table 1a). In both SL and WL plots, topsoil temperature varied between 30 and 40°C in mid-spring, increasing to 40 to 50°C by mid-summer. The MU treatment significantly reduced topsoil temperature from mid-spring to mid-summer as compared with the other aerial competition treatments. During the summer, SL plots showed slightly higher topsoil temperatures as compared with WL sites, although they were significantly different only at the end of
Significant (SL and WL) and aerial competition treatments were marginally compared with the other aerial competition treatments measurement, although it was significantly lower in CO as not significantly different at SL and WL sites in the mid-spring located at WL sites. At 20- to 30-cm depth, soil moisture was significantly lower in all competition treatments at SL sites as compared with those MU treatments (Table 2). The CO treatment also showed a significantly lower 10-cm soil moisture in the first growing season, soil moisture was significantly lower in all treatments at mid- and late summer respectively.

### Soil moisture

During the first growing season after seedling plantation, soil moisture in the first 10-cm soil depth was not significantly different between SL and WL plots, but it was significantly different among aerial competition treatments (Table 1a). The MU treatment also showed a significantly lower 10-cm soil depth temperature as compared with the other aerial competition treatments. Temperature at 10 cm belowground was not significantly different between sites or among aerial competition treatments. Temperature at 10 cm soil depth was, in average, 18, 24 and 8°C lower than topsoil temperature in mid-spring, and mid- and late summer respectively.

#### Dynamics of soil moisture at 10-cm soil depth

Soil moisture in the CO and RE aerial competition treatments at WL site was highest during mid-spring, with 25.6 and 23.5% respectively. Aerial competition treatments at the same date but in SL showed intermediate soil moisture values, ~20%, while MU treatment showed the lowest soil moistures values 4.7 and 14.4% in SL and WL, respectively. Soil-moisture dynamics showed a constant decline from the beginning of the spring towards the summer, with some upward pulses that represent mid-summer precipitation events (Fig. 4). Application of the MU treatments appears to have a deleterious effect in both SL and WL site treatments, although the WL treatment seems to rapidly recover moisture values after two rainfall events. After three consecutive rainfall events in mid-summer (February), all treatment effects (i.e. sites and aerial competition) disappeared. Soil moisture steadily diminished in all treatments towards the end of the summer (Fig. 4).

#### Austrocedrus chilensis performance

### Seedling survival

Seedling survival was significantly affected by sites (SL and WL) during the whole study period, and by aerial competition treatments (RE, MU and CO) during the second winter and the second summer analysed (Fig. 5). However, during the first 2 years after starting the experiment, survival was not significantly different in both root (EX and RC) competition treatments.

Survival slowly diminished at SL sites as compared with WL sites after the first summer, and did not present significant differences related to aerial (RE, MU and CO) or root (EX and RC) competition treatments (Fig. 5). After the second

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**Table 1. Temperature per logging site and aerial competition treatments:**

<table>
<thead>
<tr>
<th>Date</th>
<th>SL Control</th>
<th>SL Mulch Removal</th>
<th>WL Control</th>
<th>WL Mulch Removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-spring (Nov-12)</td>
<td>35.7b</td>
<td>30.6a</td>
<td>37.8b</td>
<td>36.5b</td>
</tr>
<tr>
<td>Mid-summer (Feb-13)</td>
<td>44.3ab</td>
<td>42.3a</td>
<td>49.7bc</td>
<td>42.4a</td>
</tr>
<tr>
<td>Late-summer (Mar-13)</td>
<td>23.2b</td>
<td>23.1b</td>
<td>24.5b</td>
<td>18.0a</td>
</tr>
</tbody>
</table>

**Table 2. Soil moisture per logging site and aerial competition treatments:**

<table>
<thead>
<tr>
<th>Date</th>
<th>SL Control</th>
<th>SL Mulch Removal</th>
<th>WL Control</th>
<th>WL Mulch Removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-spring (Nov-12)</td>
<td>7.3a</td>
<td>13.4b</td>
<td>10.7a</td>
<td>11.9ab</td>
</tr>
<tr>
<td>Mid-summer (Feb-13)</td>
<td>4.3a</td>
<td>5.8b</td>
<td>4.6ab</td>
<td>9.6a</td>
</tr>
<tr>
<td>Late-summer (Mar-13)</td>
<td>12.8a</td>
<td>12.6a</td>
<td>12.6a</td>
<td>19.2b</td>
</tr>
</tbody>
</table>

The summer (18.0 and 23.4°C for WL and SL respectively, Table 1a).
winter, however; seedling survival steadily diminished at SL sites for all aerial competition treatments. At the beginning of the second summer, most seedlings at SL treatment had died, and only 20% of seedlings in the aerial competition control (CO) remained alive. No seedlings survived at SL sites by the fourth summer after plantation, whereas for all competition treatments of WL, survival remained above 85%, with the only exception of the combination of EX and MU treatments, which was 75%
The combination of WL with RE and EX treatments showed significantly higher growth as compared to SL sites, although they varied from 1.25 to 2.56 cm at WL sites. The combination of WL with RE and EX treatments showed significantly higher growth as compared to WL sites.

Hydric status of A. chilensis seedlings
In mid-summer of the first growing season, predawn and midday hydric status of cypress seedling branches was associated with site (SL and WL) but not to aerial or root competition treatments. At predawn, seedlings grown at SL sites showed significantly lower (more negative) water potential average values (−1.6 MPa) than at WL sites (−1.2 MPa) (Fig. 7a). At midday, when the higher evapotranspiration rates, average values were −2.2 MPa for SL plots and −1.6 MPa for WL plots (Fig. 7b).

Discussion
This study showed that, in A. chilensis post-burned stands, salvage logging significantly affected survival and growth of restored A. chilensis seedlings. Results showed that two growing seasons after plantation, seedling survival was <10% at SL sites, whereas it was near 90% at WL sites. Four seasons after establishment, and regardless of sub-treatments, no seedlings survived at SL sites, whereas over 75% survived at WL plots.
These findings are useful to help alleviate the lack of reforestation typically observed at burned and logged sites.

Tree seedlings, in general, experience transplantation shock during the first growing season after planting, while they try to acclimate to their new site conditions (Hobbs 1984; Ouzts et al. 2015). During their acclimation period, *A. chilensis* seedlings showed a little shoot growth, while post-fire herbaceous vegetation established vigorously. Competition by neighbouring species is one of the main interactions affecting vegetation growth in early post-fire successional stages (Jobidon et al. 1998; Löf et al. 1998; Dinger and Rose 2009; Maguire et al. 2009). However, competition might affect growth in a great variety of ways, and it may differ among communities (Reader et al. 1994). In a mesic area similar to our study site, *A. chilensis* seedling survival and growth were positively associated with post-fire cover during the 2 years after plantation (Urretavizcaya et al. 2017). In our study, however, this nursing effect appeared two seasons after acclimation. Neighbouring plants seemed to exert a nursing rather than a competing effect, as *A. chilensis* seedlings grew vigorously from the third season onwards. These results agree with the study of Holmgren and Scheffer (2010), who reported results for a similar mesic site, and with other authors (Veblen et al. 1995; Gobbi 1999) dealing with other pure and fire-undisturbed *A. chilensis* stands. These studies showed that facilitation overcomes other interaction effects, and occurs in small to intermediate canopy gaps (Veblen et al. 1995) associated with high herbaceous and shrub cover (Gobbi 1999). Our study also showed that the use of root-exclusion tubes neither benefited nor harmed seedling performance.

At the beginning of the first summer after seedling plantation, PAR at SL sites was almost 4 times higher than at WL plots, significantly increasing topsoil temperature at the end of that period. These factors created a more stressful microenvironment at SL sites, perhaps causing its high seedling mortality as compared with WL plots by the end of the second summer. By contrast, at WL sites, standing snags reduced PAR reaching the soil surface, attenuating topsoil heating and improving overall seedling micro-environmental conditions. This situation is similar to what Castro et al. (2011) reported for plants grown in other Mediterranean environments. Although PAR and topsoil temperature at the end of the summer were significantly higher at SL sites as compared with WL plots, soil moisture was similar in both sites. Why, then, when having similar soil moisture levels, was higher mortality produced at SL sites as...
compared with WL plots? The answer could be found by considering the way *A. chilensis* developed physiological mechanisms to avoid hydric stress. Gyenge *et al.* (2005) demonstrated that *A. chilensis* avoids water stress by strong stomatal control. In our study, *A. chilensis* seedlings were able to tolerate soil-water deficit in both SL and WL treatments, but were unable to cope with high radiation levels at SL sites as compared with WL plots. This inference is supported by other studies with the same species, which showed that, even with adequate soil moisture, high levels of incident radiation increase the risk of photo inhibition and high cell damage because of overheating, causing high *A. chilensis* seedling mortality (Gyenge *et al.* 2007; Caselli *et al.* 2019). Additionally at WL sites, shade protection exerted by standing snags not only reduced PAR and diurnal topsoil temperatures, but also attenuated low night temperatures especially during the winter, thus creating a better microclimate for planted seedlings. This inference is supported by Castro *et al.* (2011), whose study showed that stands with snags maintain higher topsoil temperatures in winter as compared with nearby bare-tree areas.

After the first dry summer season, soil water returned to field capacity in all treatments except in those covered by mulch. This MU treatment may have obstructed normal water infiltration into the soil profile, being more noticeable at SL sites than in WL plots, because of its higher levels of radiation and the desiccant effect of winds. This may help explain the poor performance of seedlings grown in this MU treatment as compared with other treatments. Mulch is a mechanical barrier that may protect the seedling collar from high PAR and high topsoil temperatures, but also intercepting precipitation and dew condensation. This may be the reason why, in other study dealing with *A. chilensis* seedlings (Lallement *et al.* 2010), no differences in survival and growth were detected by using 2-cm shredded wood as mulch, as compared with unprotected control.

**Conclusions**

Severe fires kill *A. chilensis* trees and seeds, producing drastic modifications in stand structure. Post-fire macro- and micro-environmental conditions, particularly high PAR levels, may limit seedling survival and growth if salvage logging is applied to remaining *A. chilensis* snags, making any restoration efforts futile. In burned sites without logging, however; PAR attenuation by a canopy of remaining snags benefitted *A. chilensis* seedling survival, and no additional control of early successional herbaceous species should be necessary to attain high seedling survival. Also at SL sites, diminution of aerial or belowground competition did not ameliorate the deleterious effects of PAR. The addition of mulch as it was applied and, contrary to what was expected, did not improve seedling survival or growth. Our results agree with other studies (i.e. Melver and Starr 2000; Castro *et al.* 2011), which showed that post-fire standing snags provide significant shade, diminishing soil-surface heating and modifying surface micro-environmental conditions as compared with nearby salvage logging areas. However, standing post-fire snags are often logged for economic reasons (Castro *et al.* 2011), posing socio-ecological conflicts should the stand need to be actively restored, because seedling survival under this situation is almost nil (Keyser *et al.* 2009).

Taking into account these results, and to achieve high seedling establishment rates, limited logging is suggested in burned forest, such as the one presented in this case. One generally accepted recommendation is to leave ~50% of the snags of each diameter class, and all snags whose diameter at breast height is >50 cm or were >150 years old (Beschta *et al.* 2004). This recommendation may also be compatible with objectives related to the many other ecological roles of snags in the recovering landscape, including the provision of habitat for a variety of species and playing a key role in biological and physical processes (Thomas 1979; Donato *et al.* 2013). In burned *A. chilensis* stands, if salvage logging is the only alternative, a restoration plan should consider waiting for seedling plantation till the lower strata of plant cover and some shrubs has recovered. These shrubs may act as nurse plants, facilitating seedling survival and assuring further establishment (Kitzberger *et al.* 2000; Urretavizcaya *et al.* 2012). Aerial devices could also be used to attenuate direct radiation during early stages of seedling establishment (Urretavizcaya and Defossé 2013). Finally, post-fire management of burned forests should consider a variety of options to help reconcile and better balance the competing society’s economical needs with the benefits of sustainable management through ecological restoration.

**Conflicts of interest**

The authors declare that they have no conflicts of interest.

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