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The effect of native forest replacement by *Pinus radiata* plantations on riparian plant communities in Chile

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**Background:** As riparian habitats are legally protected, they have been maintained even in areas where extensive reforestation by exotic species occurred in areas surrounding riparian environments. However, the extent to which the riparian plant communities have been affected by the replacement of native forest on slopes has rarely been investigated.

**Aims:** In this study, we evaluated the effects of replacement of native forest by *Pinus radiata* plantations, on the diversity and structure of plant communities of remnant forests preserved in riparian habitats.

**Methods:** We selected five watersheds with native forest and five watersheds where the native forest had been replaced by pine plantations preserving riparian forests and compared composition, diversity and structure of riparian vegetation.

**Results:** In watersheds with pine plantation, riparian forests had lower adult tree density, tree cover, diversity and regeneration and higher shrub cover, diversity of herb species and diversity and richness of exotic species than riparian forests with abutting native forest.

**Conclusions:** The results suggest that the replacement of native forest by pine plantations negatively affects the diversity and structure of riparian forest. However, in other respects (e.g. shrub and climber richness), these habitats are not affected and they contribute significantly to the biodiversity conservation.

**Keywords:** diameter structure; forest structure; native riparian vegetation; *Pinus radiata* plantations; regeneration; species richness; species diversity

**Introduction**

The large-scale replacement of native forests by plantations of exotic tree species in recent decades has resulted in the loss of biodiversity, a rise in invasive species and changes in water resources (e.g. Parris and Lindenmayer 2004; Richardson and Rejmánek 2004; Huber et al. 2010; Altamirano and Lara 2010). In addition to these direct effects, plantations can impact the fragments of remaining native vegetation (Bustamante and Grez 1995; Bustamante et al. 2003; Kanowski et al. 2005; Langer et al. 2008).

While riparian habitats are preserved to protect water courses and biodiversity (Boothroyd and Langer 1999; Boothroyd et al. 2004; Langer et al. 2008), these riparian forests frequently remain as fragments, separated from riparian forests of other watersheds. Vegetation of riparian forests is generally more diverse and complex than vegetation of surrounding slopes (Naiman et al. 2000; Harper and Macdonald 2001; Granados-Sánchez et al. 2006), is usually more productive (Granados-Sánchez et al. 2006) and plays a fundamental role in hydrological processes (Tabacchi et al. 2000). Riparian forest can be adversely affected by factors related to the type and intensity of management on plantations that surround them as well as by the effects of fragmentation (e.g. edge effects) (Kanowski et al. 2003, 2005).

The establishment of forestry plantations can have different effects on riparian vegetation. For example, plantation management practices, such as thinning or pruning, can cause disturbances for riparian vegetation due to the fall of trees and branches, reducing tree cover (Graynoth 1979; Boothroyd et al. 2004; Langer et al. 2008). All these mechanical disturbances can affect regeneration and cause mortality of individual plants and with this alter the size structure of tree populations. At the same time, the clear-cutting of plantation areas implies a period of some years during which native riparian vegetation is bordered by open areas, which can have significant edge effects through macro- and micro-environmental changes (Chen et al. 1992; Burton 2002; Boothroyd et al. 2004; Harper et al. 2005; Langer et al. 2008; Becerra and Simonetti 2013). Newly cleared plantation areas are open to an invasion by exotic species (Frank and Finckh 1997; Boothroyd et al. 2004), thus serving as sources for exotic species from where they can invade adjacent riparian habitats. As a consequence of these biotic and abiotic changes in plantation areas, different vegetation attributes of remnant riparian forests can be affected. For example, changes can be expected in the composition of plant species, with the richness of exotic species increasing and native species richness decreasing (Thysell and Carey 2000, 2001; Boothroyd et al. 2004; Harper et al. 2005; Langer et al. 2008).
has been studied extensively in Chile, Spain, and Australia, New Zealand and South Africa (e.g. Richardson 1998; Baker and Murray 2012). Close to 64% of forest plantations in Chile are concentrated between 35° and 38° S ([CONAF] Corporación Nacional Forestal 2011), corresponding to the transition zone between Mediterranean type and Temperate forest zone in South America. This forest is characterised by a lower species richness than tropical and subtropical forests, but it has a high level of endemism (Armesto et al. 1998), and has been classified as a biological hotspot (Myers et al. 2000). Some of the main threats to the biodiversity of these forests are forestry plantations with exotic species, agriculture and grazing (Rozzi et al. 1994; Armesto et al. 2001; Lusk et al. 2001; Becerra 2006; Echeverría et al. 2006; Altamirano and Lara 2010).

The effect of replacing native forest with plantations of *P. radiata* has been studied extensively in Chile, Spain, Australia, New Zealand and South Africa (e.g. Richardson and Van Wilgen 1986; Slatter and Otero 1995; Merino et al. 2004; Parris and Lindemayer 2004; Williams and Wardle 2005; Echeverría et al. 2006; Gómez et al. 2009; Altamirano and Lara 2010; Huber et al. 2010; Meers et al. 2010). The main effects reported include biotic and abiotic changes in areas transformed to plantations, such as an increase in stream outflow rates, owing to less canopy interception, and a higher sediment load (Huber et al. 2010), changes in the chemical characteristics of the soil (Slatter and Otero 1995), soil loss and compaction (Oyarzun and Peña 1995; Merino et al. 2004), reduced richness of native species (Lindemayer et al. 2000; Meers et al. 2010; Onaindia et al. 2013) and increased presence of exotic species in plantations (Frank and Finckh 1997; Brockerhoff et al. 2003; Becerra and Simonetti 2013). At the same time, one of main collateral effects of replacement of native forests by plantations in remnant native forest areas is greater probability of invasion by exotic species (Bustamante and Castor 1998; Boothroyd et al. 2004; Langer et al. 2008; Rojas et al. 2011), among them, *P. radiata* (Lindemayer and McCarthy 2001; Bustamante et al. 2003; Bustamante and Simonetti 2005; Williams and Wardle 2005; Guerrero and Bustamante 2007; Baker and Murray 2012; Becerra and Montenegro 2013). Although there have been numerous studies of fragmentation and replacement of native forests, few studies have focused on the downslope effect of pine plantations on riparian vegetation. In addition, most studies have compared fragments of native forests and plantation areas in the same locality, while few studies have examined forest fragmentation and replacement effects at a larger geographic scale, in which different localities and watersheds are included. This study assessed the effects of replacing native forests on slopes by pine plantations on remnant riparian vegetation in watersheds, covering a major part of the geographic range in the distribution of plantations of *P. radiata* in Chile. In particular, we assessed the hypothesis that replacing native forest by plantations of *P. radiata* has negative downslope effects on the diversity and structure of riparian vegetation. Specifically, we predicted a (i) decreased richness and diversity of trees and climbers and increased species richness of shrubs and herbs, (ii) increased richness and diversity of exotic species and decreased richness and diversity of native species; (iii) lower density and richness of tree species for different size classes and (iv) increased understory cover, in watersheds with replacement as compared to watersheds without replacement of native forest.

**Materials and methods**

**Study area**

The study was conducted in Chile between 35° and 38° S. Ten watersheds were selected, five without replacement of native forest (hereafter “without replacement”) and five with replacement by pine tree plantations on both slopes (hereafter “with replacement”). In general, the watersheds were distributed in pairs, one with and one without replacement in the same locality to obtain the same climate range between the two types of watersheds (Figure 1). To ensure that changes observed in riparian vegetation be attributable to management of *P. radiata* plantations, all remnants of riparian vegetation were continuous with the plantation. So, there was no separation between riparian vegetation and plantation. According to Gajardo (1994), all the studied sites are classified in the Deciduous Forest Region, a temperate forest area dominated by deciduous species such as *Nothofagus obliqua*, *N. glauca* and *N. alpina* and evergreen species such as *N. dombeyi*, *Cryptocarya alba* and *Aextoxicon punctatum*, among others (Table 1).

All the watersheds with replacement, previous to planting *P. radiata* on the slopes, were covered with native vegetation (according to the information provided by the owners), so that any changes registered in the watersheds were attributable to the replacement of native forests by plantations of pine trees and the activities associated with their management. At the time of conducting the field work, the plantations were between 12 and 18 years old,
trees had a diameter at breast height (DBH) between 16 and 28 cm varied in height between 17 and 25 m, the canopy cover was between 23% and 46% and all had experienced only one rotation. The width of the riparian forest ranged from 12 (Los Potrerillos) to 20 m (El Duende). None of the watersheds with replacement had been cut recently, with the exception of El Duende, which, in one part of the watershed, had been clear-cut for the first time just 1 year before this study.

Sampling design and data collection
The sampling was carried out between August 2013 and January 2015. Within the same locality and month, a watershed with replacement and another without replacement was sampled, in all, five pairs of watersheds. In each watershed, five plots of 10 × 4 m were established on each side of the river (10 in total). The plots were 40 m apart along the river and were located closer than 5 m from the edge of the river.

Species richness and diversity. The composition of vascular plant species and the percent cover of every species were registered in all the plots. Percent cover was assessed by visual estimation in each plot (e.g., Lindenmayer and McCarthy 2001; Pauchard and Alaback 2004; Onaisiond et al. 2013). To do this, we started by recording all species in the plot, and then, inform standing in the centre of the plot, estimated the cover of each species (Mueller-Dombois and Ellenberg 1974). For species unidentified in the field, voucher specimens were deposited in the herbarium of the Natural History Museum of Santiago (Table 1S, supplemental data). For every plot, the Shannon–Wiener index (H) was calculated (using percent cover data) as a measure of species diversity for each life form (trees, shrubs, herbs and climbers) and origin (native and exotic). Species richness was analysed for each of life form and origin, for each plot. Unidentified species were included in the analysis of diversity and richness only when it was possible to determine their life form or origin. Species were classified following Marticorena and Quesada (1985). Species with potential height >5 m were considered trees; ferns were classified as forbs.

Size structure. DBH was recorded for all trees >2 m height and >5 cm in diameter to evaluate the size structure of tree species. Four diameter classes were established: C1 (5.1–15 cm), C2 (15.1–30 cm), C3 (30.1–50 cm) and C4 (>50 cm). Trees with DBH of <5 cm were classified as regeneration. We distinguished three regeneration classes: seedlings (<0.5 m in height), advanced regeneration (0.5–2 m in height) and saplings (>2 m in height and <5 cm DBH). Differences in tree species density and richness between the two watershed types were compared for each size class separately.

Stratification. Vegetation cover was recorded for ground vegetation (<1 m height), undergrowth (1–5 m) and canopy (>5 m), using the point intercept method, with 28 points located at each 1 m in the boundary of every plot, which yielded cover values for all the plots at every height class. No life forms or species were distinguished in this exercise, only the absence or presence of vegetation. Understorey was represented by the strata <5 m.

Data analysis
We compared watersheds with and without replacement for each variable using generalized linear models. The analyses were done in a nested design, where the watershed type (with and without replacement) were considered as the main factor and specific watersheds nested in each watershed type. We found a Gaussian distribution of data for Shannon diversity index and hence we used an “Identity” function, while for the data of species richness, density and cover, a Poisson distribution we observed and “Square Root” function were used. All analyses were made in R-3.1.2.

Results
Species richness and diversity
In the watersheds without replacement, 146 species were recorded, belonging to 119 genera and 80 families. Among
### Table 1. Location and characteristics of watersheds included in the study.

<table>
<thead>
<tr>
<th>Name</th>
<th>Region</th>
<th>Coordinates</th>
<th>Watershed type</th>
<th>Altitude (m a.s.l.)</th>
<th>Area (ha)</th>
<th>Climate</th>
<th>Vegetation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Los Piuquenes</td>
<td>Maule</td>
<td>35° 48' 49'' S</td>
<td>Without replacement</td>
<td>2087</td>
<td>660</td>
<td>Temperate mesothermal inferior stenothermal semi-arid Mediterranean</td>
<td>Deciduous mountain forest</td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 11' 6'' W</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Los Potrerillos</td>
<td>Maule</td>
<td>35° 31' 44'' S</td>
<td>With replacement</td>
<td>868</td>
<td>452</td>
<td>Temperate mesothermal inferior stenothermal sub-humid Mediterranean</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 11' 55'' W</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Las Arañas</td>
<td>Biobío</td>
<td>37° 33' 33'' S</td>
<td>Without replacement</td>
<td>982</td>
<td>560</td>
<td>Temperate infra-thermal stenothermal perhumid Mediterranean</td>
<td>Concepción Deciduous Forest</td>
</tr>
<tr>
<td></td>
<td></td>
<td>73° 13' 20'' W</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Los Cerezos</td>
<td>Biobío</td>
<td>37° 34' 05'' S</td>
<td>With replacement</td>
<td>730</td>
<td>224</td>
<td>254.30</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>73° 16' 58'' W</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Los Queñes</td>
<td></td>
<td>36° 39' 33'' S</td>
<td>Without replacement</td>
<td>1561</td>
<td>565</td>
<td>Temperate mesothermal inferior stenothermal semi-arid Mediterranean</td>
<td>Deciduous mountain forest</td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 34' 35'' W</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Las Cabras</td>
<td>Biobío</td>
<td>36° 40' 24'' S</td>
<td>With replacement</td>
<td>1168</td>
<td>576</td>
<td>1,122.00</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 37' 40'' W</td>
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</tr>
<tr>
<td>Manqui</td>
<td>Biobío</td>
<td>36° 22' 15'' S</td>
<td>Without replacement</td>
<td>571</td>
<td>78</td>
<td>492.30</td>
<td>Temperate mesothermal stenothermal Subhumid Mediterranean</td>
</tr>
<tr>
<td></td>
<td></td>
<td>72° 44' 27'' W</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Mela</td>
<td>Biobío</td>
<td>36° 20' 56'' S</td>
<td>With replacement</td>
<td>558</td>
<td>75</td>
<td>452.40</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>72° 46' 47'' W</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sin Puerta</td>
<td>Maule</td>
<td>36° 02' 46'' S</td>
<td>Without replacement</td>
<td>1153</td>
<td>520</td>
<td>235.00</td>
<td>Temperate mesothermal stenothermal sub-humid Mediterranean</td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 19' 21'' W</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>El Duende</td>
<td>Maule</td>
<td>36° 03' 30'' S</td>
<td>With replacement</td>
<td>1078</td>
<td>530</td>
<td>353.00</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>71° 20' 48'' W</td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

Note: Climate follows Santibañez and Uribe (1993) and vegetation follows Gajardo (1994).
them, 34.9% (51 species) were herbaceous, 26% (38 species) shrubs, 24.7% (36 species) trees and 14.3% (21 species) climbers. In the watersheds with replacement, an identical total number of species were recorded, belonging to 113 genera and 71 families, of which 43.8% (64 species) were herbaceous, 23.3% (34 species) shrubs, 21.9% (32 species) trees and 11% (16 species) climbers (Table 1S). In the watersheds without replacement, an identical total number of species were recorded, belonging to 113 genera and 71 families, of which 43.8% (64 species) were herbaceous, 23.3% (34 species) shrubs, 21.9% (32 species) trees and 11% (16 species) climbers (Table 1S). In the watersheds without replacement, of the 139 species of known origin, 5% (7 species) were exotics, while in the watersheds with replacement, of the 142 species of known origin, 16.2% (23 species) were exotics, mainly herbaceous (70%). Some of the native species present in most of the watersheds (at least six) were *Aextoxicon punctatum*, *Aristolotia chilensis*, *Citronella macroura*, *Cryptocarya alba*, *Lomatia dentata*, *Luma apiculata*, *Chusquea quila*, *Boquila trifoliata*, *Cissus striata*, *Hydrangea serratifolia*, *Lapageria rosea* and *Lardizabala biternata*. The most common exotic species were *Rubus ulmifolius*, *Rosa moschata*, *Prunella vulgaris* and *Rumex acetosella*.

There was a significantly higher diversity (Shannon–Wiener) of tree species in the watersheds without replacement (*P* = 0.002), while there was a greater diversity of herbaceous species in the watersheds with replacement (*P* = 0.005). No significant differences were found between the two types of watersheds in the diversity of shrub (*P* = 0.914) and climber species (*P* = 0.767) (Figure 2a). There was a greater richness of tree species in the watersheds without replacement (*P* = 0.028) and greater richness of herbaceous species in watersheds with replacement (*P* = 0.005). No significant differences were found in the richness of shrub (*P* = 0.219) and climber species (*P* = 0.783) between watersheds with and without replacement (Figure 2b).

There was greater diversity of native species (*P* = 0.004) in watersheds without replacement than in those with replacement, while there was greater diversity of exotic species (*P* < 0.0001) in watershed with replacement (Figure 3a). Similarly, there was significantly greater richness of exotic species in watersheds with replacement (*P* < 0.0001), but there was no significant difference in the richness of native species between watershed types (*P* = 0.273) (Figure 3b).

### Size structure of tree species

There was significantly greater density of individuals of all the considered DBH classes (*P* < 0.0001 for every class) in the watersheds without replacement (Figure 4a). Also, there was significantly greater richness in the class C4 (*P* < 0.0001) and C3 of tree species (*P* = 0.002) in watersheds without replacement, while there were no differences in the other classes (*P* = 0.58; *P* = 0.106 in C1 and C2, respectively) (Figure 4b).

There was a greater density of total regeneration and regeneration of seedlings and saplings in the watersheds without replacement (*P* < 0.0001), while in watersheds with replacement there was greater density of advanced regeneration (*P* < 0.0001) (Figure 5a). There was greater richness of species at the level of seedlings (*P* = 0.003), saplings (*P* = 0.022) and marginally for total regeneration (*P* = 0.051) in watersheds without replacement, while there was no difference between the two types of watersheds in the richness of advanced regeneration (*P* = 0.273) (Figure 5b).

### Stratification of vegetation cover

There were no significant differences between watersheds with and without replacement in plant cover at the stratum of <1 m (*P* = 0.234), while at the stratum 1–5 m, there was significantly more cover in watersheds with replacement (*P* < 0.0001). In contrast, cover in the stratum >5 m was significantly greater in watersheds without replacement (*P* < 0.0001) (Figure 6).

### Discussion

The structure of native riparian vegetation differed between watersheds with and without replacement of...
native forest by pine plantations. There was lower density of trees with DBH >5 cm and lower cover of the upper stratum in the forest of watersheds with replacement. This could have triggered the increased cover of undergrowth (stratum 1–5 m) observed in watersheds with replacement. Based on the observations of this study, it is possible to associate one or more causes with these structural differences between the two types of watersheds. A possible cause is the selective cutting of trees in the native forest remnants (Senbeta and Teketay 2001; Neira et al. 2002), which can be more common in watersheds with plantations because the remnant riparian forest are more accessible. For example, the wood of species like Gevuina avellana, Drimys winteri, N. dombeyi, Laurelia sempervirens, N. alpina and N. obliqua are widely used in Chile for diverse purposes, and none of these species, with the exception of N. obliqua, had diameters greater than 30 cm in the watersheds with replacement. It is also possible that plantation management activities such as thinning, pruning and harvesting could have caused disturbances in the canopy cover of remnant forests (Boothroyd et al. 2004), for example, by falling branches and even adult individuals of P. radiata (authors’ personal observation). It is also possible that the native trees in remnant riparian forest fall due to the effect of wind, which is generally more intense in fragments that are surrounded by an open matrix, as is the case after the plantations are cut (Burton 2002). Edge effects also generally raises the temperature and reduces the moisture levels of the soil within remnant forest (Didham and Lawton 1999), which may result in

Figure 3. Species diversity (Shannon index)/plot (a) and species richness/plot (b) according to biogeographic origin (native and exotic) in watersheds with and without replacement (mean ± 1 SE). N for each bar = 50. Different letters indicate statistical differences ($P < 0.05$) between watersheds with and without replacement for a single biogeographic origin.

Figure 4. Density (individuals/ha) (a) and species richness/plot (b) of trees species per DBH class (C1: 5.1–15 cm, C2: 15.1–30 cm, C3: 30.1–50 cm, C4: >50 cm) in watersheds with and without replacement (mean ± 1 SE). N for each bar = 50. Different letters indicate statistical differences ($P < 0.05$) between watersheds with and without replacement for a single DBH class.
higher mortality of senescent individuals and thus reduce canopy cover.

The creation of clear-cut spaces and reduced cover associated with fewer large-diameter trees, in addition to drastic change in the matrix that surrounds remnant riparian forest when plantations are harvested, may result in sharp micro-environmental changes in the remnant forests. These changes can include an increase in light (Burton 2002; Boothroyd et al. 2004; Langer et al. 2008), soil moisture reduction and temperature increases (Didham and Lawton 1999; Pearson et al. 2002; Pawson et al. 2006), among others. These structural and environmental changes may be related to the differences found in richness, diversity and density of the remnant riparian vegetation between the watersheds with and without replacement (Ramovs and Roberts 2003; Harper et al. 2005; Echeverría et al. 2007). As expected, we observed a reduction in the diversity of native species and increased richness and diversity of exotic species in watersheds with replacement. Plantation areas, especially after being cleared, are often invaded by exotic plants that can spread to native forest fragments (Frank and Finckh 1997; Kanowski et al. 2005; Langer et al. 2008; Becerra and Simonetti 2013), in this case, remnant riparian forests. Generally, exotic species are adapted to disturbed habitats (Thysell and Carey 2001; Boothroyd et al. 2004), as apparently are the riparian forests in watersheds with replacement. Also, the forest remnants surrounded by open areas, like riparian forests when plantations have been clear-cut, are more frequently visited by livestock that act as vectors for the dispersion of exotic species (Pauchard and Alaback 2004).

Despite the greater diversity of exotic species in watersheds with replacement, the regeneration or juvenile individuals of *P. radiata* were not registered in any of the assessed watersheds, although a few adult trees were present. The absence of invasions of *P. radiata* in riparian forests, including watersheds with pine plantations, may be due to their shade-intolerant character. It is possible that this species requires more canopy openings or much less cover of undergrowth strata to regenerate (Arévalo and Fernández-Palacios 2005; Williams and Wardle 2007). Consequently, the riparian forest remnants in watersheds with replacement show certain resistance to the invasion of *P. radiata*. However, the few individuals of *P. radiata* found in riparian forests suggests that pine invasions can occur, as occurs in other forests neighbouring pine plantations (Estades and Escobar 2005; Williams and Wardle 2007).
On the other hand, and also as expected, we observed an increase in herb species in watersheds with replacement, which could be triggered by an increase in light availability produced by a more open canopy. Similarly, the reduction in richness and density of tree species regeneration observed in replaced watersheds, mainly at the level of seedlings, could be related to disturbances and microenvironmental changes, apparently more frequent in watersheds with replacement. Various studies have reported how these types of changes, especially reductions in soil moisture and water stress, strongly affect the regeneration of woody species (Leishman and Westoby 1994; Burton 2002; Franklin et al. 2002; Pearson et al. 2002; Caccia et al. 2009). Bustamante et al. (2006) observed similar results in forests of the northern coastal range of our study region, where the regeneration of tree species was reduced in remnant forests (although not corresponding to riparian forests in this case) surrounded by *P. radiata* plantations.

Other mechanisms may also hinder forest regeneration in watersheds with replacement. For example, lower seed production is associated with lower density of adult and larger trees (Simonetti et al. 2001). Also, competition with exotic herbaceous species resulting from the invasions observed in remnant riparian forest could reduce regeneration (Hobbs 2001; Kanowski et al. 2003; Huth and Wagner 2006; Zhu et al. 2014). As well, higher rates of seed predation have been documented in forest remnants like those of watersheds with replacement (Donoso et al. 2003; Simonetti et al. 2006), reducing the density of regeneration. The presence of livestock, which can have negative effects on seedlings (Vazquez 2002), could also have reduced regeneration in riparian forests of watersheds with replacement, as cattle and horse faeces and hoof prints were found in 50% of watersheds without replacement compared to 80% of watersheds with replacement.

On the other hand, our results suggest that some attributes of riparian native forests (richness and diversity of shrub and climber species and total richness of native species) are not strongly affected by the replacement of native forests on slopes. It is possible that the presence of forestry plantations neighbouring native forest remnants instead of permanently open areas has avoided major changes in them. Plantations generate less drastic environmental changes than do open areas, as the tree cover is low only for a period of no more than 10 years (Brockerhoff et al. 2003). Ramos et al. (2008) even suggested that the presence of plantations helps conserve native forests in Chile when compared to areas where native forests border on pastures or open areas. Additionally, the plantations established in the studied watersheds have had only one rotation, that is, the riparian forest have only been bordered by open areas twice (once for the establishment of the plantation and the other after the first harvest). It is possible that after more rotations, variables of native vegetation composition or structure will be more affected.

Consequently, the preservation of remnant forests on the banks of waterways in watersheds with replacement of native forest by plantations on slopes (Boothroyd and Langer 1999) plays an important role in the conservation of biodiversity. Several studies have shown that remnant forests contribute to the conservation of biodiversity of temperate forests in Chile (e.g. Estades and Temple 1999; Grez 2005; Bustamante et al. 2006; Simonetti 2006; De La Vega and Grez 2008). Given the above, conservation should not be restricted to large-scale areas, especially when large areas are not available for some type of ecosystem, but rather should include small forest remnants (Grez 2005). However, more attention needs to be given to the increasing invasion of exotic species in small forest remnants (Pauchard and Alaback 2004; Becerra and Simonetti 2013). So, it is important to consider the breadth of the conserved riparian vegetation (Langer et al. 2008), as well as the management of neighbouring plantations (Pauchard and Alaback 2004), given that they can have a positive effect on invasion processes. Thus, with the aim of strengthening conservation of remnant riparian vegetation, we suggest (1) creating buffer zones between remnant vegetation and plantations (Baker and Murray 2012), for example selectively harvesting plantations in areas adjoining fragments, leaving some areas permanently with some pine trees on the outer border with native forest fragments (Thysell and Carey 2001; Roberts and Zhu 2002) to attenuate microenvironmental changes in the fragments; (2) reducing the internal degradation of remnant forests (Rojas et al. 2011) by restricting, for example, the entry of cattle and tree cutting; and (3) developing protocols prior to the use of machinery in managing plantations to avoid causing disturbances in the fragments (Lindenmayer and McCarthy 2001).

**Conclusions**

The results partially support the proposed hypotheses, with evidence of less diversity and richness of tree species, less regeneration of native tree species, more exotic species, and greater diversity of herbaceous species in riparian forest remnants of watersheds with replacement. These effects can be attributed to changes at the level of canopy coverage and the density of trees with diameters greater than 5 cm in watersheds with replacement and the resulting environmental changes. On the other hand, the absence of change in some variables, such as total richness of native species, and in particular of shrub and climbing species in riparian forests in watersheds with replacement, suggests that remnant vegetation contributes significantly to the conservation of biodiversity in forestry management systems, and consequently, actions should be taken to improve the conservation of these riparian forest remnants.

**Disclosure statement**

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Supplemental data
Supplemental data for this article can be accessed here.

Notes on contributor
Ivon R. Gutierrez Flores is a graduate student. She is interested in plant communities and the impact of economic activities on them.

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