

## Bird assemblages in pine plantations replacing native ecosystems in NW Patagonia

María Victoria Lantschner · Verónica Rusch ·  
Celina Peyrou

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**Abstract** Forest plantations of exotic conifers represent an important economic activity in NW Patagonia, Argentina. However, there is a remarkable lack of information on the impact of forestry on native biodiversity. We analyzed the effect of *Pinus ponderosa* plantations on bird communities, considering different stand management practices (dense and sparse tree covers), and different landscape contexts where they are planted (*Austrocedrus chilensis* forest and steppe). Ultimately we wished to assess in which way plantations may be designed and managed to improve biodiversity conservation. Bird richness and abundance did not change significantly in the steppe, although community composition did, and was partially replaced by a new community, similar to that of ecotonal forests. In contrast, in the *A. chilensis* forest areas, species richness decreased in dense plantations, but bird community composition remained relatively constant when replacing the native forest with pine plantations. Also, in *A. chilensis* forest, stand management practices aiming at maintaining low tree densities permit the presence of many bird species from the original habitat. In the steppe area in turn, both dense and sparse plantations are unsuitable for most steppe species, thus it is necessary to manage them at higher scales, maintaining the connectivity of the native matrix to prevent the fragmentation of bird populations. We conclude that pine plantations can provide habitat for a substantial number of native bird species, and this feature varies both with management practices and with the landscape context of areas where afforestation occurs.

**Keywords** *Austrocedrus chilensis* · Bird diversity · Exotic tree plantations · Forest management · Landscape context · Steppe

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M. V. Lantschner (✉) · V. Rusch · C. Peyrou  
Grupo de Ecología Forestal, INTA EEA Bariloche, CC 277, Bariloche 8400, Argentina  
e-mail: mvlantschner@bariloche.inta.gov.ar

## Introduction

Biodiversity is an issue of increasing significance for the development and management of plantation forests and for their long-term sustainability (Carnus et al. 2006). Although the primary goal of plantation forestry is the efficient production of timber and pulp, it also offers important opportunities for biodiversity conservation if plantation design and management are appropriate (Lindenmayer 2002).

It is widely thought that exotic plantation forests are less favorable habitats than native forests (Hartley 2002; Carnus et al. 2006). Comparisons of unmanaged forests and plantations have found impoverished flora (Shankar et al. 1998; Humphrey et al. 2002) and fauna (Pomeroy and Dranzoa 1998; Lindenmayer and Munks 2000; Schnell et al. 2003) in the latter. Plantations may be unsuitable for many native species, because of the loss of some of the structural components of native habitats, such as understory vegetation (Yirdaw 2001; Brockerhoff et al. 2003), snags, and old or dead trees (Clout and Gaze 1984; Gjerde and Saetersdal 1997; Humphrey et al. 2002), which are critical for some wildlife. However, the effect of forest plantations on biodiversity depends on the type of plantation and the natural structure of surrounding native forests (Hartley 2002), and plantations can still contribute to biodiversity conservation if they are correctly designed and managed (Hartley 2002; Sayer et al. 2004; Carnus et al. 2006).

In Argentina, plantations of exotic fast growing conifers have been promoted by the state through subsidies in the last decades. In NW Patagonia, plantation forestry is a new activity that replaces traditional sheep production systems, representing an important economic alternative. Consequently, plantation rates show a rapidly increasing tendency (Schlichter and Laclau 1998). Plantations are established mainly in two types of ecosystems: *Austrocedrus chilensis* xeric forests and steppes. In steppes more than 2 million hectares are regarded as potentially useful for pine plantations (SAGPyA 1999).

*Austrocedrus chilensis* forests belong to the Valdivian Temperate Rainforest ecoregion, and steppe to the Patagonian Steppe ecoregion. Both have high conservation value; they were included in the WWF's "Global 200" conservation strategy (Olson and Dinerstein 1998) because they harbor some of the world's most outstanding and representative biodiversity. Both ecoregions have many endemic species. In the forest area they represent a high proportion (ca. 50% of species), whereas in the steppe area the proportion is smaller (ca. 20% of species) (Vuilleumier 1972). At the same time, they are threatened because of human intervention. The *A. chilensis* forest subregion is the most heavily altered and threatened within the Argentine portion of the Valdivian Temperate Rainforest ecoregion, as it has undergone forest fires, overgrazing and a high pressure for timber extraction (Laclau 1997). These forests currently have a low level of protection. Only 7% of the area is under the Protected Areas system, and mainly within the less restrictive status of national reserves (Vila 2002). In the Patagonian Steppe ecoregion, sheep overgrazing appears to have modified the vegetation and accelerated soil degradation processes (Soriano et al. 1983). About 4% of this ecoregion is under protected areas. Besides, most of these reserves are only nominally so, since little real protection is offered to wildlife (less than 1%) (Walker et al. 2005).

Potential negative environmental impacts of pine plantations have been the focus of great public concern in Patagonia (Rusch and Schlichter 2005). Pine plantations have been defined as "green deserts" and opposition to several afforestation projects has arisen, despite the fact that scientific information regarding their detrimental or beneficial effects is limited. In NW Patagonia there are some studies that suggest that changes in biodiversity in forest plantations depend on certain stand structural characteristics and on the taxa

considered (Paritsis 2002; Rusch et al. 2005b; Corley et al. 2006). However, there is scarce information about the impact of pine plantations on biodiversity, particularly with respect to the landscape context.

The aim of this study is to assess the effects of plantation forestry in NW Patagonia, by considering its effects on the avifauna in two different ecosystems (*Austrocedrus chilensis* forest and steppe). We also look at different stand management practices (dense and sparse plantations), in order to provide information aiming at enhancing biodiversity conservation.

## Methods

### Study area

The study was carried out in NW Patagonia (from 39° 55' S to 41° 51' S; and from 71° 03' W to 71° 33' W) (Fig. 1). The climate of this area is temperate, dominated by a marked west-to-east decrease in precipitation (from 3,000 to 600 mm in less than 100 km). Vegetation types reflect this climatic pattern, and they are distributed as north–south orientated belts, parallel to the Andes. The study region is located in the forest/steppe ecotone, with vegetation ranging from xeric forests of *A. chilensis* in the west, to a shrubby steppe in the east. In the *A. chilensis* forest area, mean annual rainfall ranges between 900 and 1,200 mm/year (Barros et al. 1983), and the prevailing vegetation is a pure *A. chilensis* arboreal stratum, and an understory of shrubs and trees such as *Aristotelia chilensis*, *Maytenus boaria* and *Lomatia hirsuta*. In steppe area where pines are planted, mean annual rainfall ranges between 700 and 1,000 (Barros et al. 1983). Vegetation corresponds to that of a cold semi-desert, dominated by bunchgrasses (*Stipa* spp. and *Festuca* spp.) and low shrubs (*Mulinum spinosum* and *Senecio filaginoides*) (Cabrera 1976).

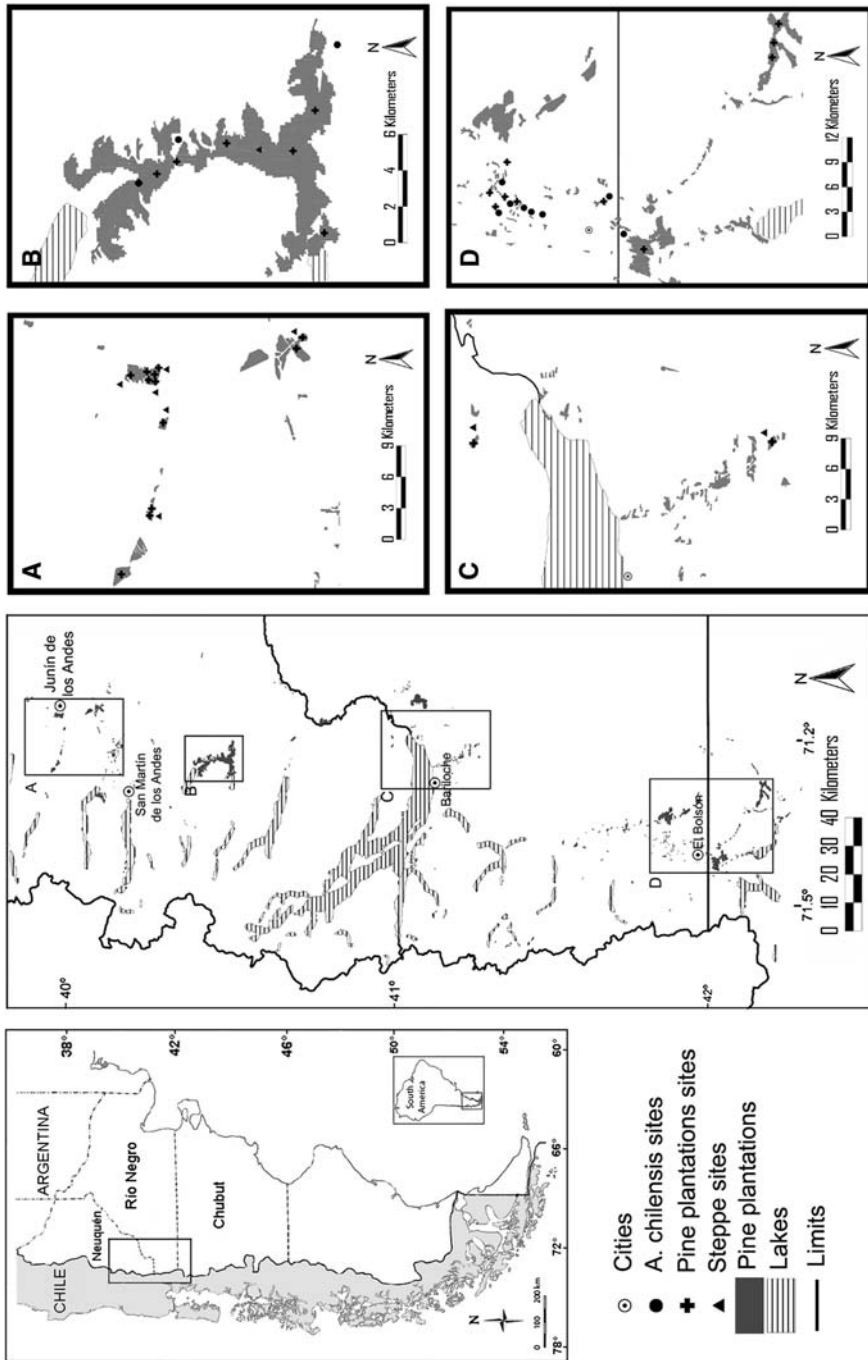
Plantations in the region comprise mainly of three species: ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*) and douglas fir (*Pseudotsuga menziesii*). We selected three situations (treatments) in both steppe and *A. chilensis* forest habitats (Fig. 1):

- Dense pine plantations: ponderosa pine plantations, where tree density was high, the herbaceous-shrubby cover was less than 15%, and the canopy cover was higher than 80% (N = 8 in steppe area, and N = 10 in *A. chilensis* forest area).
- Sparse pine plantations: ponderosa pine plantations where tree density was low, the herbaceous-shrubby cover was higher than 15% and the canopy cover was less than 80% (N = 9 in steppe area; and N = 6 in *A. chilensis* forest area).
- Native vegetation: continuous areas with the native vegetation, managed in the traditional way (cattle grazing of steppes and light selective logging of *A. chilensis* forests) which were sampled as control, and were located close to plantations (N = 9 in steppe area, and N = 11 in *A. chilensis* forest area).

Independent replicates were called “sites”. The size of plantation blocks was between 9 and 200 ha (see Table 1).

### Habitat characterization

Vegetation structure and composition were characterized on each site. Herbaceous (0–50 cm height) and shrub (50–200 cm height) cover were estimated through 10 random



**Fig. 1** Map of the study area, North Western Patagonia, Argentina

**Table 1** Habitat variable values (mean ± standard error), in the different vegetation types

	Steppe			<i>A. chilensis</i> forest		
	NV	SPP	DPP	NV	SPP	DPP
Herbaceous cover (%)	45.5 ± 4.4	24.56 ± 1.9	0.5 ± 0.3	27.2 ± 2.8	24.0 ± 7.0	2.5 ± 0.5
Shrub cover (%)	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	8.7 ± 2.3	8.2 ± 4.7	0.2 ± 0.2
Arboreal cover (%)	0.0 ± 0.0	50.5 ± 6.3	86.9 ± 1.6	62.5 ± 7.7	66.9 ± 6.6	88.7 ± 1.9
Canopy height (m)	0.0 ± 0.0	9.5 ± 0.6	13.0 ± 1.1	21.6 ± 1.9	11.6 ± 1.1	21.0 ± 1.9
Number of herbaceous-shrubby spp.	31.3 ± 2.45	25.1 ± 1.5	11.6 ± 2.1	44.0 ± 1.5	29.7 ± 3.6	15.4 ± 2.6
Number of arboreal spp.	0.0 ± 0.0	1.1 ± 0.1	1.3 ± 0.2	3.7 ± 0.4	3.7 ± 1.0	2.1 ± 0.4
Mean DBH (cm)	0.0 ± 0.0	18.1 ± 1.8	18.5 ± 1.2	22.8 ± 2.5	20.3 ± 1.6	22.1 ± 1.2
Basal area (m <sup>2</sup> /ha)	–	15.1 ± 3.0	29.9 ± 4.1	31.6 ± 4.1	28.8 ± 5.9	42.8 ± 5.5
Precipitation (mm/year)	911 ± 61	922 ± 691	900 ± 77	990 ± 9	1016 ± 40	1000 ± 0
Plantation age (years)	–	12–19	17–22	–	18–23	23–32
Plantation area (ha)	–	106.8 ± 56.5	100.7 ± 74.5	–	51.2 ± 46.7	55.1 ± 53.5

NV, Native vegetation; SPP, Sparse pine plantation; DPP, Dense pine plantation

quadrates of 45 cm × 45 cm per plot. All species in these strata present in the plot were recorded in order to estimate richness.

To characterize the arboreal structure, a 500 m<sup>2</sup> circular plot (or 1,000 m<sup>2</sup> in cases where tree density was less than 200 trees per ha) was established in each site. In each plot, diameter at breast height (DBH) of all the trees taller than 5 cm were measured, and canopy height was estimated with clinometer. These data were used to estimate basal area. Arboreal cover for each species was estimated using a densitometer (four readings per point at 10 random points per plot). Although it is very well known that structural elements such as snags and logs may influence birds, we did not include them in the analysis because these forests are not pristine and they almost lack these structural elements. Additionally, mean annual rainfall for each site was obtained from precipitation maps (Barros et al. 1983).

### Bird surveys

Bird counts were conducted using fixed 50-m-radius point-counts (Ralph et al. 1993), from December 2001 to March 2002, on clear days, from sunrise to around 10 AM. In each site, seven plots were established, separated by a minimum of 120 m from adjacent points and from vegetation edges. In the cases where plantations were too small to include seven point counts, we established only six or five plots. At each plot, all bird species heard and/or seen were recorded, during a 7 minute-period, once in each site. Birds flying over were not recorded unless they were somehow using the vegetation below them. Taxonomy of the birds follows Narosky and Babarskas (2000). Bird abundance was estimated as the mean number of individuals per point, in each site.

### Data analysis

Differences in the habitat variables across the treatments were assessed through one-way ANOVA in the cases where data met normality assumptions (Kolmogorov-Smirnov tests), followed by Waller-Duncan multiple comparisons (as we had an unequal sample size), to determine sources of differences. A Kruskal-Wallis test was performed in the cases where data were not normally distributed.

As sample efforts varied across sites, bird richness in each site was estimated with the Chao 1 estimate, using the program EstimateS 7.5 (Colwell 2005). The Chao 1 estimate is a non-parametric method for estimating total species richness. It is an abundance-based estimator, which uses the number of rare species to estimate the number of missing ones, and allows to standardize the survey effort (Chao 1984). Bird abundance was expressed as the mean number of birds per point, in each site. Differences between avian richness and abundance across vegetation types—both, in the steppe and the *A. chilensis* forest area—were assessed through one-way ANOVA, as data met normality (Kolmogorov-Smirnov tests) and homoscedasticity requirements. Waller-Duncan multiple comparisons for unequal sample sizes were used as post hoc tests. Comparisons of abundance of each species across vegetation types were done using the non-parametric Kruskal-Wallis test, as they were not normally distributed. Spearman rank correlations were performed in order to relate bird species abundance with vegetation variables (Sokal and Rohlf 1981).

As there is a wide variation in the sizes of plantation fragments, and this can affect the ability of birds to explore into the plantation from the surrounding native matrix, Pearson

correlations between plantation size and total bird abundance and richness were estimated, to assess if there is a significant relation between these variables. Additionally, Spearman correlations were estimated between plantation size and the abundance of each species in each site, to assess if there is a relation between plantation size and any of the species.

We also assessed whether bird species of conservation importance changed their abundance in sparse and dense pine plantations. The conservation status of each species was obtained from the IUCN Red List of Threatened Species (IUCN 2006). Additionally, we selected the species classed as having high conservation importance by the “SUMIN” index, for Nahuel Huapi National Park, located nearby the study area (Grigera et al. 1996). The SUMIN is an index comprising 12 survival-related variables, which are considered essential for the conservation of bird species. Variables with the greatest influence upon the index value are those related to distribution, space-use plasticity, reproductive potential and trophic amplitude. The index ranges between 2 and 18, and we arbitrarily selected all species with an index of 13 or higher. A Kruskal-Wallis test was performed on all those species with more than 3 detections, to compare the difference in abundance of each species between treatments.

Multivariate methods were used to analyze the distribution of bird species among the different vegetation types and to assess the effect of substitution of the native vegetation by exotic pine plantations on native bird communities. We performed an analysis of similarity (ANOSIM) with the Bray-Curtis similarity index, including all vegetation types together, to determine the existence of differences in the composition of bird communities. ANOSIM is a non-parametric test to establish differences between two or more groups, based on distance measures. Distances are converted to ranks, and then distances between groups are compared with distances within groups. The test uses the statistic *R*, which can take values between 0 and 1. Large positive *R* means dissimilarity between groups (Hammer et al. 2001). The significance of the test was determined by permutation of group membership, with 5,000 replicates. The test was performed with PAST 1.46 (Hammer et al. 2001).

The relationship between the composition of the bird species communities and habitat variables across all vegetation types, together, was examined by a Canonical Correlation Analysis (CCA, ter Braak 1986). Ordination axes represent the maximum variability that is attributable to the environmental parameter. Relative effects of individual environmental parameters were then visualized by the relative length of the respective vectors in the ordination space (Kent and Coker 1992). A Monte Carlo permutation test was performed, to test the significance of the relation between species and environmental variables, based on the first ordination axis, and on all canonical axes together (Kent and Coker 1992).

## Results

### Habitat characterization

In the steppe area, that native vegetation showed significant structural differences from pine plantations (Table 1), particularly to dense plantations, whereas differences from sparse plantations were not so marked. Herbaceous cover and species richness showed significant differences ( $F = 58.151$ ,  $P < 0.000$ ; and  $F = 23.475$ ,  $P < 0.000$ , respectively): they were highest in native vegetation plots; they decreased in sparse pine plantations, and practically no herbaceous vegetation was recorded in dense pine plantation plots. Mean canopy height, arboreal cover, and basal area were also significantly different across

treatments ( $F = 100.061$ ,  $P < 0.000$ ; and  $F = 123.951$ ,  $P < 0.000$ ;  $F = 30.088$ ,  $P < 0.000$ , respectively): they increased inversely, being higher in dense pine plantations and lower in native vegetation. The dbh values also showed significant differences ( $F = 78.294$ ,  $P < 0.017$ ), being similar in dense and sparse pine plantation, and null in native vegetation. Mean annual precipitation, and plantations areas were similar in the three vegetation types (Kruskal-Wallis  $P > 0.926$ ; and  $F = 0.017$ ,  $P > 0.983$ ).

In the *A. chilensis* forest area, in turn, differences in vegetation structure across the different vegetation types were less significant (Table 1), with dense pine plantation plots being the most different. Herbaceous, shrub, and arboreal cover showed significant differences across the treatments ( $F = 18.348$ ,  $P < 0.000$ ;  $F = 4.189$ ,  $P < 0.028$ ; and  $F = 5.746$ ,  $P < 0.009$ ). Herbaceous and shrub cover values were similar in native vegetation and sparse pine plantations, and higher than those of dense pine plantations (Waller-Duncan,  $P < 0.050$ ), whereas arboreal cover was higher in dense pine plantations than in native vegetation and sparse pine plantations (Waller-Duncan,  $P < 0.050$ ). Canopy height also showed significant differences across treatments ( $F = 7.207$ ,  $P < 0.004$ ), being similar in native vegetation and dense pine plantations, and less high in sparse pine plantations (Waller-Duncan,  $P < 0.050$ ). Herbaceous-shrubby species richness was also significantly different across treatments ( $F = 41.901$ ,  $P < 0.000$ ), being highest in native vegetation, intermediate in sparse pine plantations and lowest in dense pine plantations (Waller-Duncan,  $P < 0.050$ ). Arboreal species richness, basal area, and dbh, in turn, did not show differences across treatments ( $F = 3.028$ ,  $P > 0.067$ ;  $F = 2.106$ ,  $P > 0.144$ ; and  $F = 0.350$ ,  $P > 0.708$ ). As in the steppe area, mean annual precipitation and plantation areas were similar in the three vegetation types (Kruskal-Wallis,  $P > 0.794$ ; and  $F = 0.010$ ,  $P > 0.990$ ).

### Bird abundance and richness

A total of 41 bird species was recorded in the study area (Table 2). All except one (*Lophortyx californica*) were native to the region. Thirty three species were recorded in the steppe area and twenty-six in the *A. chilensis* forest area.

In the steppe area, neither species richness nor bird abundance was significantly different across native vegetation, sparse pine plantations and dense pine plantations (Chao's mean species/site estimate: 9.81, 10.09 and 5.79; and mean individuals/point: 2.61; 2.44 and 1.87, respectively) (ANOVA for richness  $F_{2,23} = 1.14$ ;  $P < 0.337$  and ANOVA for abundance  $F_{2,23} = 0.61$ ;  $P < 0.553$ ) (Fig. 2). Species richness and bird abundance did not show a significant correlation (Pearson) with the plantation area, neither in dense nor in sparse plantations (richness:  $P > 0.300$ ;  $P > 0.256$ , respectively; abundance:  $P > 0.099$ ;  $P > 0.565$ , respectively).

In *A. chilensis* forest area, species richness did not differ significantly across the different types of vegetations (ANOVA  $F_{2,24} = 0.631$ ;  $P < 0.541$ ) (Fig. 2). (Chao's mean species/site estimate: 8.77, 9.89 and 7.30). In turn, total abundance of birds was significantly different across the different types of vegetation (ANOVA  $F_{2,24} = 4.677$ ;  $P < 0.019$ ) (Fig. 2). Bird assemblages living in native vegetation and sparse plantations (4.51 and 4.19 individuals/point, respectively) had significantly higher abundance than those in dense pine plantations (2.35 individuals/point) (Waller-Duncan  $P < 0.050$ ). As in the steppe area, species richness and bird abundance did not show a significant correlation with the plantation area, neither in dense nor in sparse plantations (richness:  $P > 0.388$ ;  $P > 0.454$ , respectively; abundance:  $P > 0.526$ ;  $P > 0.900$ , respectively).



**Table 2** Mean number of bird species per point, in the different types of habitats

Species	Abbreviation	SUMIN <sup>a</sup>	Steppe		<i>A. chilensis</i> forest			
			NV	SPP	DPP	NV	SPP	DPP
<i>Agriornis</i> sp. (Shrike-Tyrant)	Agr sp	–	0.019	0.000	0.000	0.000	0.000	0.000
<i>Anairetes parulus</i> (Tufted Tit-Tyrant)	Ana par	10	0.082	0.138	0.099	0.183	0.433	0.031
<i>Aphrastura spinicauda</i> (Thorn-tailed Rayadito)	Aph spi	12	0.000	0.143	0.021	0.487	0.390	0.173
<i>Ashlhenes pyrrholeuca</i> (Lesser Canastero)	Asi pyr	8	0.206	0.019	0.025	0.013	0.298	0.014
<i>Buteo polyosoma</i> (Red-backed Hawk)	But pol	11	0.000	0.016	0.000	0.000	0.000	0.000
<i>Caprimulgus longirostris</i> (Band-winged Nighthjar)	Cap lon	9	0.000	0.016	0.000	0.000	0.000	0.000
<i>Carduelis barbata</i> (Black-chinned Siskin)	Car bar	9	0.175	0.368	0.265	0.247	0.157	0.017
<i>Cinclodes</i> sp. (Cinclodes)	Cin sp.	–	0.000	0.016	0.000	0.000	0.000	0.000
<i>Cistothorus platensis</i> (Grass Wren)	Cis pla	6	0.019	0.000	0.000	0.000	0.000	0.000
<i>Colaptes pitiuus</i> (Chilean Flicker)	Col pit	11	0.000	0.000	0.000	0.087	0.000	0.060
<i>Colorhamphus parvirostris</i> (Patagonian Tyrant)	Col par	13	0.000	0.000	0.000	0.013	0.000	0.000
<i>Columba araucana</i> (Chilean Pigeon)	Col ara	18	0.000	0.000	0.000	0.109	0.000	0.000
<i>Columba picazuro</i> (Picazuro Pigeon)	Col pic	–	0.016	0.251	0.154	0.013	0.000	0.020
<i>Coragyps atratus</i> (Black vulture)	Cor atr	9	0.000	0.016	0.000	0.000	0.000	0.000
<i>Curaeus curaeus</i> (Austral Blackbird)	Cur cur	8	0.016	0.032	0.208	0.009	0.067	0.000
<i>Diuca diuca</i> (Common Diuca-Finch)	Diu diu	7	0.474	0.019	0.057	0.000	0.462	0.000
<i>Elaenia albiceps</i> (White-crested Elaenia)	Ela alb	6	0.241	0.733	0.583	2.472	1.203	1.220
<i>Falco sparverius</i> (American Kestrel)	Fal spa	4	0.037	0.019	0.000	0.000	0.000	0.000
<i>Geositta cunicularia</i> (Common Miner)	Geo cun	6	0.056	0.000	0.000	0.000	0.000	0.000
<i>Leptasthenura aegithaloides</i> (Plain-mantled Tit-Spinetail)	Lep aeg	8	0.000	0.000	0.000	0.018	0.024	0.040
<i>Lophortyx californica</i> (Californian Quail)	Lop cal	–	0.111	0.000	0.000	0.000	0.000	0.000
<i>Milvago chimango</i> (Chimango Caracara)	Mil chi	6	0.016	0.000	0.000	0.031	0.000	0.000
<i>Muscivora sp.</i> (Ground-Tyrant)	Cus sp	–	0.016	0.000	0.000	0.000	0.000	0.000
<i>Phrygilus fruticeti</i> (Mourning Sierra-Finch)	Phr fru	7	0.079	0.000	0.000	0.000	0.000	0.000

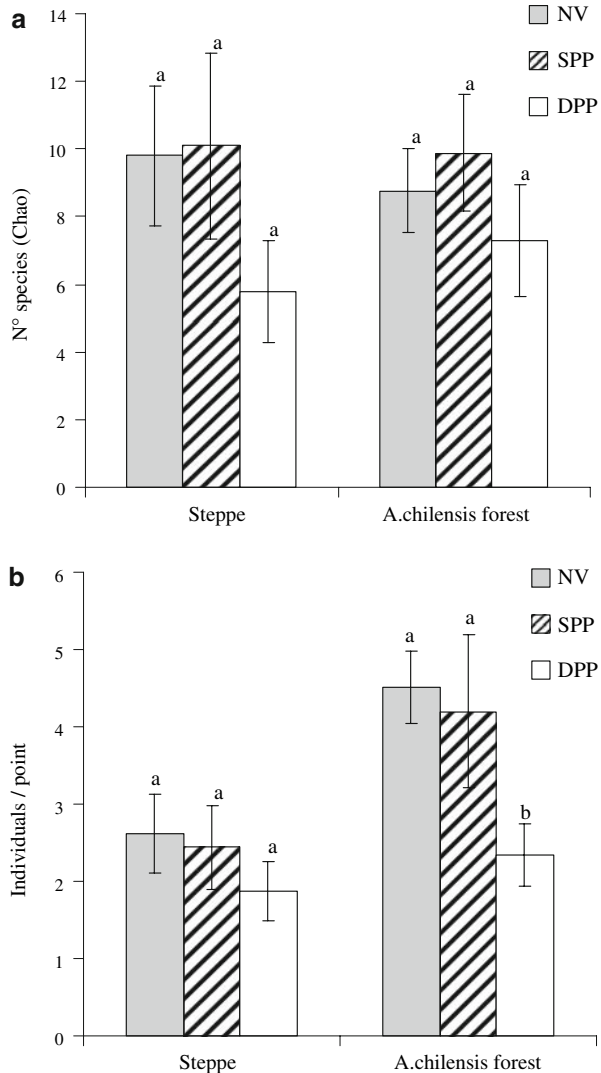
Table 2 continued

Species	Abbreviation	SUMIN <sup>a</sup>	Steppe		<i>A. chilensis</i> forest			
			NV	SPP	DPP	NV	SPP	DPP
<i>Phrygilus gayi</i> (Gray-hooded Sierra-Finch)	Phr gay	7	0.071	0.000	0.000	0.018	0.000	0.071
<i>Phrygilus patagonicus</i> (Patagonian Sierra-Finch)	Phr pat	13	0.000	0.053	0.158	0.009	0.117	0.396
<i>Polyborus plancus</i> (Crested Caracara)	Pol pla	6	0.000	0.016	0.036	0.000	0.000	0.000
<i>Pteroptochos tarnii</i> (Black-throated Huet-Huet)	Pte tar	11	0.000	0.000	0.000	0.000	0.033	0.000
<i>Pygarrhichas albularis</i> (White-throated Treerunner)	Pyg alb	13	0.000	0.000	0.000	0.028	0.000	0.000
<i>Scelorchilus rubecula</i> (Chucac Tapaculo)	See rub	10	0.019	0.000	0.000	0.279	0.105	0.063
<i>Scytalopus magellanicus</i> (Andean Tapaculo)	Sey mag	12	0.000	0.000	0.000	0.000	0.000	0.014
<i>Sicalis lebruni</i> (Patagonian Yellow-Finch)	Sic leb	–	0.032	0.000	0.000	0.000	0.000	0.000
<i>Strix rufipes</i> (Rufous-legged Owl)	Str ruf	13	0.000	0.000	0.061	0.015	0.000	0.014
<i>Sturnella loyca</i> (Long-tailed Meadowlark)	Stu loy	7	0.280	0.000	0.000	0.000	0.000	0.000
<i>Tachycineta leucopyga</i> (Chilean Swallow)	Tac leu	6	0.103	0.016	0.036	0.031	0.213	0.000
<i>Troglodytes aedon</i> (House Wren)	Tro aed	3	0.079	0.183	0.018	0.285	0.371	0.088
<i>Turdus falcklandii</i> (Austral Thrush)	Tur fal	6	0.257	0.272	0.122	0.096	0.156	0.105
<i>Tyto alba</i> (Barn Owl)	Tyt alb	6	0.000	0.000	0.000	0.000	0.000	0.000
<i>Vanellus chilensis</i> (Southern Lapwing)	Van chi	6	0.066	0.000	0.000	0.000	0.000	0.000
<i>Zenaidura macroura</i> (Eared Dove)	Zen aur	5	0.019	0.000	0.025	0.013	0.024	0.000
<i>Zonotrichia capensis</i> (Rufous-collared Sparrow)	Zon cap	2	0.124	0.114	0.000	0.052	0.142	0.020

NV, Native vegetation; SPP, Sparse pine plantation; DPP, Dense pine plantation in the steppe and the *A. chilensis* forest area

<sup>a</sup> SUMIN index (Grigera et al. 1996), it indicates the conservation importance of the bird species, based in survival-related variables (higher numbers indicate higher conservation importance)

**Fig. 2** Bird community parameters in the different types of vegetation (NV: native vegetation, SPP: sparse pine plantation, DPP: dense pine plantation) in steppe and *A. chilensis* forest habitats. **(a)** Richness (Chao estimate) and **(b)** abundance (individuals/point) of birds (means  $\pm$  standard error)



Community analysis

The ANOSIM showed that, in the steppe area (Table 3), bird communities of sparse and dense pine plantations were similar, whereas those of native vegetation differed. Although species richness in native vegetation and pine plantations was similar, differences between bird communities can mainly be accounted for by differences in species composition. Thirteen bird species present in the native vegetation were exclusive to this habitat (52% of the total number of species), and twelve (48%) were shared with pine plantations.

In *A. chilensis* forest areas, species composition in native vegetation sites, and sparse and dense pine plantations, was similar, most species (93%) being present in these vegetation types. The ANOSIM (Table 3) showed differences between native vegetation and dense plantations, but these were due to lower species richness and abundance (i.e., only

**Table 3** Analysis of similarity for bird communities (ANOSIM) in native vegetation (NV), sparse pine plantation (SPP) and dense pine plantation (DPP) for both steppe and *A. chilensis* forest areas

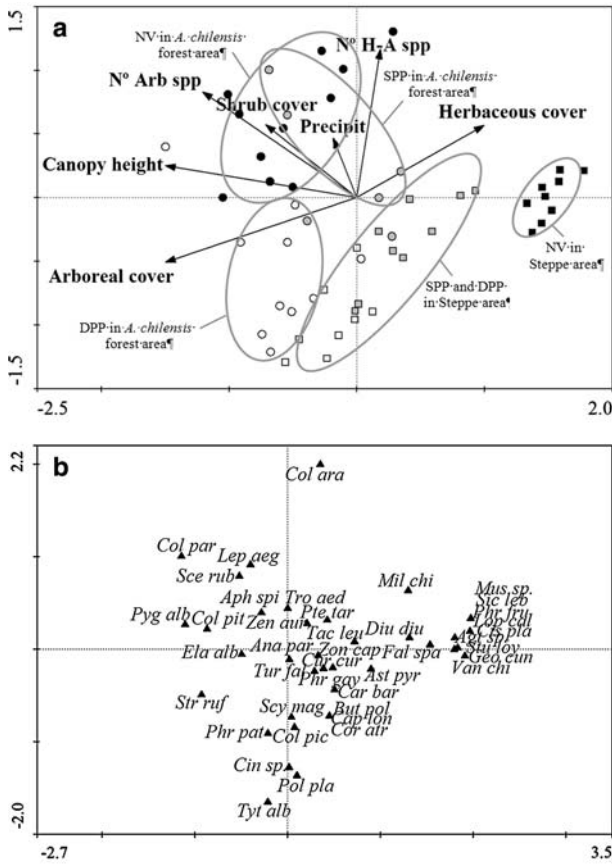
			Steppe			<i>A. chilensis</i> forest	
			NV	SPP	DPP	NV	SPP
Steppe	SPP	R	0.354				
		<i>P</i>	0.000				
	DPP	R	0.235	0.028			
		<i>P</i>	0.010	0.333			
<i>A. chilensis</i> forest	NV	R	0.688	0.343	0.510		
		<i>P</i>	0.000	0.003	0.000		
	SPP	R	0.223	0.040	0.221	0.207	
		<i>P</i>	0.033	0.297	0.054	0.055	
	DPP	R	0.519	0.267	0.322	0.261	0.289
		<i>P</i>	0.000	0.005	0.003	0.003	0.018

R (which can take values between 0 and 1) reflects the degree of separation of the assemblages based on their species composition; *P* is significant at the  $\alpha$ -level of 0.05

58% of the total species found in the *A. chilensis* forest area was also observed in dense plantations).

The CCA revealed that 16.3% of the total variance in species dispersion can be explained by the measured environmental variables. Canopy height ( $r = -0.792$ ), arboreal cover ( $r = -0.791$ ), and the number of arboreal species ( $r = -0.637$ ) were the strongest variables correlated with the first axis. On the other hand, the richness of herbaceous-shrubby species ( $r = 0.633$ ), arboreal species richness ( $r = 0.450$ ), shrub cover ( $r = 0.311$ ), and herbaceous cover ( $r = 0.307$ ) were the variables most correlated with the second axis. The relationship between species and the environmental variables was significantly correlated with the first ordination axis, and with all canonical axes together (Monte Carlo  $F = 4.164$ ;  $P < 0.002$  and  $F = 0.908$ ;  $P < 0.014$ , respectively).

Figure 3a and b displays the CCA diagram. When analyzing all the sites together, the diagram shows that there is a group, corresponding to steppe native vegetation sites, which is clearly separated from the other sites, with species such as *Vanellus chilensis*, *Milvago chimango*, *Sturnella loyca*, *Phrygilus gayi*, *Melanodera xanthogramma* and *Geositta cunicularia*. This group is associated with a high herbaceous cover, and presented a group of bird species characteristic of open areas. The rest of the sites were distributed close together, and presented a bird community composed mainly of species typical of native forests, such as *Phrygilus patagonicus*, *Aphrastura spinicauda* and *Elaenia albiceps*. Sparse and dense pine plantation sites in steppe areas tended to be grouped, and were associated with a low number of arboreal species, low shrub cover, and low canopy height. Dense pine plantation in *A. chilensis* forests areas were also distributed close together, and showed an association with high arboreal cover. *A. chilensis* forests were associated with a high number of arboreal species, shrub cover, and canopy height; whereas sites corresponding to sparse pine plantation in *A. chilensis* forest areas were distributed between pine plantations in steppe areas and native *A. chilensis* forest vegetation sites.



**Fig. 3** Ordination plots from canonical correspondence analysis of the species/environment data in the studied sites. **(a)** Environmental variables (rows) and Sites (■ Steppe native vegetation, □ Sparse pine plantation on steppes, □ Dense pine plantation on steppes, • *A. chilensis* forest native vegetation, ○ Sparse pine plantation on *A. chilensis* forest, ○ Dense pine plantation on *A. chilensis* forest). The arrows are plotted pointing in the direction of maximum change of the environmental variable across the diagram, and the length of the arrow is proportional to the magnitude of change in that direction. **(b)** Species, indicated with the three first letters of the genus and the species (see Table 2)

### Relationships between bird species and habitat structure

In steppe areas, four species were significantly more abundant in native vegetation: *Sturnella loyca*, *Asthenes pyrrholeuca*, *Phrygilus gayi* and *Diuca diuca* (Kruskal-Wallis  $X^2 = 8.519$ ,  $P < 0.014$ ;  $X^2 = 9.428$ ,  $P < 0.009$ ;  $X^2 = 6.132$ ,  $P < 0.047$  and  $X^2 = 6.713$ ;  $P < 0.035$ , respectively). The first three species were positively correlated with herbaceous cover ( $P < 0.032$ ;  $P < 0.014$  and  $P < 0.009$ , respectively) and negatively correlated with arboreal cover ( $P < 0.011$ ;  $P < 0.008$  and  $P < 0.034$ , respectively), and basal area ( $P < 0.011$ ;  $P < 0.024$  and  $P < 0.034$ , respectively). *Diuca diuca* was negatively correlated with arboreal cover ( $P < 0.034$ ) and basal area ( $P < 0.030$ ). On the other hand, *A. spinicauda* was significantly more abundant in sparse pine plantations (Kruskal-Wallis  $X^2 = 7.581$ ;  $P < 0.023$ ), *Columba picazuro* was significantly more abundant in both sparse

and dense pine plantations than in the other habitats (Kruskal-Wallis  $X^2 = 6.655$ ;  $P < 0.036$ ) and showed a positive association with arboreal cover ( $P < 0.039$ ); and *Tyto alba* was most abundant in dense pine plantations (Kruskal-Wallis  $X^2 = 7.313$ ;  $P < 0.026$ ), showing a negative association with herbaceous cover ( $P < 0.020$ ). Only one species showed a significant correlation with plantation size: *A. spinicauda*, which was negatively correlated ( $P < 0.047$ ).

In *A. chilensis* forest areas, the abundance of four species differed significantly between the different types of vegetation: *Elaenia albiceps* which was most abundant in native vegetation (Kruskal-Wallis  $X^2 = 9.254$ ;  $P < 0.010$ ); *Anairetes parulus*, which was most abundant in native vegetation and sparse pine plantations (Kruskal-Wallis  $X^2 = 6.315$ ;  $P < 0.043$ ), and showed a positive association with herbaceous cover ( $P < 0.021$ ); *Diuca diuca* which was most abundant in sparse pine plantations, and was negatively associated with canopy height (Kruskal-Wallis  $X^2 = 11.328$ ;  $P < 0.003$ ); and *Tachycineta leucopyga* which was most abundant in sparse pine plantations (Kruskal-Wallis  $X^2 = 6.374$ ;  $P < 0.041$ ), showing a positive association with herbaceous cover ( $P < 0.036$ ) and a negative association with arboreal cover ( $P < 0.002$ ) and canopy height ( $P < 0.026$ ). No species were correlated with plantation size.

None of the species recorded in the study area was listed as threatened in the IUCN Red List (IUCN 2006); they were all at low risk. Also, one species recorded in the steppe area and five in the *A. chilensis* forest habitat, presented a SUMIN index (Grigera et al. 1996) equal to or higher than 13 (Table 2). Among these, *Phrygilus patagonicus* appeared in both, steppe and *A. chilensis* forest areas. In the steppe area there were no significant differences in abundance between habitat types (Kruskal-Wallis  $X^2 = 3.130$ ;  $P < 0.209$ ), whereas in the *A. chilensis* forest area, it was not abundant enough to determine significant differences across treatments. *Columba araucana* was only recorded in the *A. chilensis* forest although differences between habitats were not significant, it did not present significant differences across treatments either. *Colorhamphus parvirostris*, *P. albogularis*, and *Strix rufipes* were also recorded in the *A. chilensis* forest, but because of their low abundance, we could not determine significant differences in their densities across the different vegetation types.

## Discussion

Changes in bird abundance and richness due to the replacement of native vegetation with exotic pine plantations differed depending on which type of vegetation was replaced. In the steppe areas, there were no changes in species richness and abundance, whereas in the *A. chilensis* forests, bird richness decreased in dense pine plantations.

Several studies in different regions of the world have found that conifer plantations support fewer bird species and lower total density of birds than native vegetation (Driscoll 1977; Carlson 1986; Leberton and Pont 1987; Mitra and Sheldon 1993; Estades 1994; Gjerde and Saetersdal 1997; Pomeroy and Dranzoa 1998; Marsden et al. 2001; Paritsis 2002; Lindenmayer et al. 2003; Zurita et al. 2006). However, others have noted that the avifauna in plantation forests, may be as diverse and abundant as in the natural vegetation they replace (Clout and Gaze 1984; Estades and Temple 1999; Vergara and Simonetti 2004; Lantschner 2005; Gonzalez-Gomez et al. 2006). Our results are in line with the view that changes in bird richness and abundance depend strongly on site and regional characteristics.

A recurring dilemma for land managers worldwide, however, is the trade-off between managing for maintenance of total species diversity and the need to pay special attention to the species belonging to the original native systems (Petit and Petit 2003). When native vegetation was replaced with pine plantations in the steppe areas we studied, species composition changed substantially although numbers of species and individuals changed little overall. In the *A. chilensis* forest area, in turn, some changes in species richness occurred but species composition remained quite similar. These findings suggest that replacing native vegetation with exotic conifer plantations does not always lead to changes in overall bird species richness or bird abundance, but that changes in the species composition of the bird community may occur and must also be considered.

Stand-management practices influence the presence of several bird species, and sparse plantations, particularly in the *A. chilensis* forest area, tend to have less impact on the native bird communities than dense pine plantations. Sparse plantations have different origins; some are sparse simply because trees are still small, while others have undergone thinning, or were planted at low densities. In all cases, however, the fact that they have low canopy cover, leads to increased light availability, and consequently, the development of a higher shrub and herbaceous cover (Miller 2001).

Native vegetation inside pine plantations may partly account for the composition of the bird communities observed in the study. Past studies in South American temperate forests have concluded that the presence of native vegetation is one of the most important factors determining the use of plantations by native birds (Estades 1994; Estades and Temple 1999; Vergara and Simonetti 2004; Lantschner and Rusch 2007). Our results confirm observations by Estades and Temple (1999) for Chilean temperate forests that the abundance of many bird species, (e.g., *L. aegithaloides*, *S. rubecula*, *C. parvirostris*) is positively associated with the amount of native vegetation in the understory, particularly in the *A. chilensis* area. Understorey vegetation may provide escape cover against predators, safe nesting sites, and food resources for birds. Its importance for some native bird species was also documented by several studies in the Chilean temperate forests, in sites associated with other disturbances, like agriculture, livestock grazing, and logging (Willson et al. 1994; Sieving et al. 1996, 2000; Reid et al. 2004; Díaz et al. 2004, 2005; Willson 2004; Castellón and Sieving 2006).

The landscape context also affects the composition of bird communities. Small changes resulted when plantations replaced native forest having a similar vegetation structure, whereas changes were more marked when plantations replaced steppe habitats. Consequently, when plantations were established in forest areas, the surrounding matrix affected the composition of the avifauna in plantations, and hence most forest birds were as likely to be found and as abundant in sparse plantations as they were in native vegetation. In dense plantations, richness decreased, but the assemblage found was a subset of the assemblage found in native vegetation. In contrast, when plantations were established in steppe areas, structural changes of the vegetation were so important that most steppe bird species did not find suitable habitat inside the plantation, particularly those that feed and nest in open grasslands (e.g. *S. loyca*, *P. gayi*, *A. pyrrholeuca*, *M. chimango*) (Christie et al. 2004). Therefore the steppe bird community was partially replaced with a new community made up of generalist birds and species typical of the forest ecotonal areas located near the study area, such as those that use trees to feed, nest, or take refuge (i.e., *A. spinicauda*, *T. alba*, *C. picazuro*) (Christie et al. 2004). The fact that steppe bird communities were more affected than *A. chilensis* communities can be also explained by the fact that bird species in Patagonian temperate forests have broad niches and wider distributions across habitats than steppe birds (Vuilleumier 1972, 1985) because these forests are isolated and have evolved

as ‘islands’ (Vuilleumier 1985). This ability of forest birds to adapt to different types of habitats might enable them to adopt pine plantations more easily as new forest habitats, which was suggested by Estades and Temple (1999) for pine plantations in Chile’s temperate regions.

According to these results, in the *A. chilensis* forest area, the habitat connectivity for most of the forest bird species may be maintained in sparse pine plantations but may be affected in dense pine plantation. Almost all bird species that were present in the native vegetation were also actively using sparse pine plantations, whereas dense plantations supported fewer species. In steppe areas, in turn, most steppe bird species were absent in both sparse and dense pine plantations. Large plantations could therefore act as barriers for many species and fragment their habitat (Fahrig 2001).

In NW Patagonia, plantation forests still occupy relatively small areas within native vegetation at present and exist as small scattered patches. Thus their impact on the connectivity of native bird populations may be less serious than in other parts of the world where plantations cover more extensive areas (Clout and Gaze 1984; Estades and Temple 1999; Lindenmayer et al. 2003). Nevertheless, strong subsidies on exotic forest plantations can increase planted areas substantially. Planning for biodiversity conservation in managed areas based on local information is, then, urgently needed in the region to anticipate severe negative impacts.

#### Changes at species level

In the steppe area, some species were clearly affected by the replacement of native vegetation with pine plantations where their abundance was greatly reduced. This was mainly the case with those species that require open areas, such as *Sturnella loyca* and *Phrygilus gayi*, which feed on the soil in open grasslands (Christie et al. 2004); *Asthenes pyrrholeuca*, which forages and hides within low bushes in open areas; and *Milvago chimango*, which usually looks for carrion in open lands, roads, and forest openings (Christie et al. 2004).

On the other hand, plantations benefited some other bird species, particularly those typical of ecotonal forest areas, which use trees to feed or take refuge. *Aphrastura spinicauda* was more abundant in sparse plantations, probably because it feeds on insects on the trees, but it was absent in dense plantations, which indicates that these are unsuitable for this species probably because they are too dense. On the other hand, *Tyto alba*, which is a nocturnal raptor owl that hides in holes in trees or caves during the day, appeared only in dense plantations. Finally *Columba picazuro* was more abundant in sparse and dense plantations than in native vegetation. This is a common species in other regions that has expanded its distribution to the south in the last decades (Narosky and Babarksas 2000), and has been reported as common in agricultural areas and artificial forests (Christie et al. 2004).

In the *A. chilensis* forest area only one species, *Elaenia albiceps*, was significantly more abundant in native vegetation than in pine plantations. Although this species was abundant in plantations, it was twice as abundant in native vegetation. These results are in line with other studies carried out in Chile (Estades and Temple 1999). The explanation for this pattern could be the greater heterogeneity of the native forest vegetation which provides a greater diversity and abundance of foraging resources for this species. In addition, three species were more abundant in sparse plantations than in native vegetation and dense plantations: *Diuca diuca*, *Anairetes parulus* and *Tachycineta leucopyga*. These species are



typical of ecotonal forests, and they may benefit from sparse pine plantations because they prefer open or disturbed forests areas where they look for insects in the soil, shrubs or air (Christie et al. 2004).

Five of the species recorded in the *A. chilensis* forest area (*E. albiceps*, *S. rubecula*, *P. tarnii*, *A. spinicauda* and *S. rufipes*), were considered focal species for the South American temperate rainforests, due to their habitat needs, range sizes, and/or importance in the food chain (Vila 2002; Rusch et al. 2005a). All these species were present in pine plantations, which implies that these are not unsuitable for those birds. However, the foliage insectivorous *E. albiceps*, as seen above, was negatively affected by the replacement of native forest with pine plantations. In addition, the understory insectivorous *S. rubecula* and the foliage insectivorous *A. spinicauda* also tend to decrease in pine plantations, particularly in dense plantations, although the differences were not significant. As pine plantations appear to be less suitable than native forests for some keystone species, detailed studies of population dynamics should be conducted to improve our understanding of the conservation value of these anthropogenic homogenous ecosystems. On the other hand, the nocturnal raptor owl *S. rufipes* and the understory bird *P. tarnii* did not show any negative tendency in pine plantations. Both species have been seen nesting in pine plantations (Vergara and Simonetti 2003).

As the bird surveys were carried out without taking into account differences in detectability of species between habitat types, species abundance values should be interpreted with caution. The abundance numbers of some species was probably underestimated. However, we do not think that there are important differences in abundance values across the different species and habitats as we only recorded the bird species heard or seen inside the first 50 m radius of each plot.

### Management and Conservation Implications

The traditional view of conservation reserves is of large, untouched areas. However, few landscapes provide the opportunity to preserve large tracts of land, and conserving biodiversity within the matrix of multiple-use lands becomes essential (Lindenmayer and Franklin 1997). Our results show that the type of management applied to pine plantations influences their suitability as habitat for birds, and so appropriate changes in design and management regimes of pine plantations can contribute to biodiversity conservation.

At the stand-scale, the maintenance of some forest structural elements is likely to permit the conservation of forest birds in planted forests. In this sense, the presence of native understory vegetation is of great importance, thus ideally the management of pine plantations should enhance the native understory vegetation to provide additional conservation benefits (Estades and Temple 1999). Based on our results, one of the most important ways to promote the presence of native vegetation in the understory is to plant at low densities or by early thinning (Zobrist and Hinckley 2005). Additionally, management practices such as herbicide application, removal of the stumps and roots of native trees, and other soil disturbances may reduce habitat quality for these birds (Vergara and Simonetti 2003). On the other hand, retention of biological legacies such as long-lived trees, snags and downed wood within plantation stands gives plantations a structure more similar to natural stands (Clout and Gaze 1984; Gjerde and Saetersdal 1997). In our study, different canopy heights and the number of tree species also proved to be important in bird community composition.

At the landscape scale, the most important factors to consider are plantation size, shape, and location (Díaz et al. 1998), extent to which a landscape has been and will be planted,

the similarity of plantation structure to natural vegetation (Gjerde and Saetersdal 1997), and what habitats are being converted into plantation (Hartley 2002). Our results show that pine plantations, particularly in the steppe area, are unsuitable for some local bird species. This implies that it is important to consider landscape design alternatives for maintaining or enhancing diversity in planted landscapes, and avoiding the fragmentation of bird populations. In this regard, special areas of high diversity and those areas that provide habitat for threatened, rare, or endangered species should be identified and specifically managed. In addition, it would be desirable to consider the connectivity of the remaining natural habitats. Unlike to other studies, which found that the number of bird species was higher in small patches of plantations (Curry 1991; Lindenmayer et al. 2002), our results showed that plantation size was not an important factor of the ability of birds to use the plantations. However, it would be necessary to carry out further studies covering a wider range of sizes to provide more certainty about this.

The impact of plantation forestry on biodiversity also depends on the degree to which the landscape is natural versus degraded. While the conversion of natural ecosystems to plantation forests will rarely be desirable from a biodiversity point of view, planted forests often replace other land uses (Carnus et al. 2006). Thus, an objective assessment of the potential or actual impacts of planted forests on biodiversity requires appropriate reference points. In our study area, the native vegetation sites were not pristine prior to the establishment of plantation forests (i.e., cattle grazing occurred in the steppe area and light selective logging in *A. chilensis* forests). Therefore it is necessary to consider that bird communities in the replaced habitats were already affected by other disturbances before the replacement with forest plantations, and pristine systems may have shown stronger effects.

However, our study covered a relatively short-term, and long-term records may be necessary to better understand the effects of land management on the biodiversity of these environments. Additionally, further studies should include other taxa, such as mammals or invertebrates, which have requirements that differ from those of birds, and which may perceive the impact of pine plantations in different ways.

## Conclusions

The results of this study suggest that pine plantations can provide habitat for a substantial number of native bird species, and that this varies with the landscape context. Plantations established in a forest matrix generate less impact on bird communities than those in a steppe matrix. Thus, in the *A. chilensis* forest areas, stand management practices aiming at maintaining low tree densities enhance the retention of many bird species, as they enable the persistence of some critical structural elements of native vegetation. In steppe areas, in turn, both dense and sparse plantations are unsuitable for many species. In those areas it is necessary to manage plantations with consideration of higher scales, maintaining the connectivity of the native vegetation remnants to minimize the fragmentation of bird populations.

Landscapes comprising mosaics of native vegetation and forest plantations are more desirable from a conservation perspective than other land uses that are more structurally simplified, such as agriculture (Moore and Allen 1999), or intensive livestock grazing (Lantschner 2005). Thus, when analyzing the impact of plantation forestry on biodiversity, the ecological context of planted forest development, as well as the social and economic context shaping land-use changes must be considered (Carnus et al. 2006). The definition of management objectives linked to sustainability, considering endangered and functional

keystone species, and the integrated analysis of different spatial scales are important to find a balance between intensive land use and biodiversity conservation.

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