Native plantations as an important element for biodiversity in vanishing forested landscapes: A study of the near threatened araucaria tit spinetail (*Leptasthenura setaria*, Furnariidae)

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Abstract   Although forest loss is still a problem worldwide, estimated rates of deforestation have declined in the last decade, primarily because of an increase in the area of tree plantations. This leads to the central question of how suitable plantations are for indigenous species. Native plantations are thought to have higher value for biodiversity than plantations of non-native trees; however, not all studies support this view. We assessed occupancy and density of the araucaria tit spinetail (*Leptasthenura setaria*, Furnariidae), a near threatened species, in the highly endangered araucaria forests of north-eastern Argentina and in araucaria plantations, which comprise 90% of the remaining habitat for this species. All natural forest remnants were occupied by araucaria tit spinetails. Only 85% of the plantations were occupied; however, density was almost threefold higher in plantations compared with natural forests. Our models indicated that stand age was the most important factor in determining occupancy and density of this bird species in plantations. Plantations <10 years old exhibited lower densities than older plantations. This species does not occur in plantations of non-native trees, but our results indicate that native plantations may provide important habitat for the araucaria tit spinetail, particularly given that most native forest has been removed. Restoration of natural remnants and conservation of old, connected plantations may assure the protection of significant populations of spinetails. The role of native araucaria plantations as habitat for other species merits further examination.

Key words: *Araucaria*, Argentina, biodiversity, *Leptasthenura setaria*, plantation.

INTRODUCTION

Most of the Earth's terrestrial surface (83%) has been transformed by human activities in the last century (Sanderson *et al.* 2002), leading to substantial population declines in species and sometimes to extinction. In particular, deforestation is a globally significant concern, as almost half of the terrestrial plant and animal species live in forests (Brockerhoff *et al.* 2008). Although deforestation is still high, rate of native forest loss declined from 2000 to 2005 worldwide (FAO 2007), mostly because of an increase in area of plantation forests. Planted forests are expected to be the major source of the world's wood supply by 2030 (FAO 2009). These observations lead to the question of how suitable are plantations as potential habitat for indigenous species.

Throughout South America, conversion of native forest and grasslands to introduced pine and *Eucalyptus* plantations is widespread and of great conservation concern because of negative impacts on biodiversity (Wilson *et al.* 2005; Barlow *et al.* 2007; Fonseca *et al.* 2009), soils (Jobbagy & Jackson 2003; Farley *et al.* 2008) and hydrology (Buytaert *et al.* 2007; Huber *et al.* 2008). From a biodiversity perspective, native plantations should be favoured for biodiversity over non-native plantations because they may be more structurally similar to native forest, host species with particular adaptations, and maintain mutualistic interactions with other indigenous organisms. Some studies have attempted to elucidate the value of native plantations in conserving biodiversity; yet, clear answers have not been obtained. For example, plantations of the native hoop pine (*Araucaria cuninghami*) in Australia have higher bird diversity than introduced pines and *Eucalyptus* (Kanowski *et al.* 2005). Similarly, natural forests and plantations of native trees species in western Kenya held more bird species than plantations of non-native species (Farwig *et al.* 2008). Fonseca *et al.* (2009) observed the same pattern in southern Brazil in groups as diverse as woody plants, epiphytic angiosperms, ferns and flatworms. In contrast, in araucaria forests of northern Argentina, Zurita *et al.* (2006)

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found that non-native and native plantations were similar in terms of bird diversity although still less diverse than the native forest.

In plantations, stand age and management practices such as pruning, thinning of trees and reduction of understorey influence structural complexity (Loyn *et al.* 2007, Brockerhoff *et al.* 2008 for a review, Luck & Korodaj 2008). Structural complexity of native forest influences the diversity of habitats, shelter and food availability, and has been cited recurrently as a key factor affecting bird diversity (Clout & Gaze 1984; Duran & Kattan 2005; Barlow *et al.* 2007). The value of plantations for birds and other species that rely on forests may vary widely depending on management practices that impact habitat quality linked to forest structure. The utility of plantations for forest species also may depend upon spatial configuration of the plantations. Commercial plantations are intrinsically fragmented and transient habitat. Studies on habitat loss and fragmentation in other ecological systems have shown that habitat variables at the patch and landscape scales influence distribution of species. Patch size and patch shape are important determinants of regional population persistence in fragmented landscapes (Laurance *et al.* 2002; Debinski & Holt 2003; Parker *et al.* 2005; Schooley & Branch 2007). Isolation of patches limits dispersal, influences population viability and is a critical factor in maintenance of metapopulations (Hanski & Gaggiotti 2004). Most studies of habitat fragmentation have been conducted in natural habitat, but understanding the importance of spatial configuration of plantations for forest species could lead to management practices to mitigate effects of habitat loss and fragmentation (Diaz *et al.* 1998). We examined effects of patch- and landscape-level factors on a near threatened bird (araucaria tit spinetail, *Leptasthenura setaria*, Furnariidae, IUCN 2009) in a landscape with natural forest remnants and plantations of native and non-native species in north-eastern Argentina.

The Atlantic Forest of South America, which originally covered around 1.5 million km^2 in Brazil, eastern Paraguay and north-eastern Argentina, is among the most diverse and threatened ecosystems of the world, with only 8% of the original forest area remaining. This forest has been replaced by urban areas, agriculture, pasture and plantations. One of the endangered forest types within the southern part of the Atlantic Forest is the araucaria forest. The araucaria tree (*A. angustifolia*), which dominates this forest, is a critically endangered species (IUCN 2009) that occurs in temperate areas from 600 to 1200 m in South-eastern Brazil and extreme North-eastern Argentina. In Argentina, plantations of *A. angustifolia*, which have been established since the early 1950s, cover an area at least 15 times larger than natural araucaria forests. Declines of some species that inhabit araucaria forest may have been buffered by araucaria plantations, but data are lacking to evaluate this conjecture. An inadequate timber market and slow growth rate of these trees are leading to replacement of araucaria plantations with loblolly pine (*Pinus taeda*). Araucaria plantations comprised around 40 000 ha in 2001. Recent estimates indicate this area has decreased by more than half, highlighting the need to determine the value of araucaria plantations for native species.

In this study, we evaluated use of araucaria plantations by the araucaria tit spinetail, an insectivorous passerine bird that exclusively inhabits the canopy of araucaria forests. This species has been observed in araucaria plantations in recent years, expanding its known distribution in Argentina (Krauczuk 2001; Cabanne *et al.* 2007). Araucaria tit spinetails have not been recorded in pine plantations (Zurita *et al.* 2006), which are common in the region (Antunes *et al.* 2007; Cabanne *et al.* 2007). Although araucaria plantations are highly fragmented, they constitute most of the remaining habitat for the species. At least three other species of birds are associated with the araucaria forest (*Amazona pretrei*, *A. vinacea*, *Cyanocorax caeruleus*) but these species already are so scarce that sightings are infrequent (Bodrati & Cockle 2006a,b). The goals of this study were to: (i) compare occupancy and density of araucaria tit spinetails in natural remnants and plantations and (ii) model density and occupancy of this species with respect to habitat variables at patch and landscape scales for plantations. By identifying habitat variables related to distribution and density of spinetails, this study will contribute to understanding characteristics of native plantations that influence their value for biodiversity and provide a scientific basis to assess the impacts of loss of these plantations and replacement by introduced pines.

METHODS

Study area

We conducted this study in an area of about 30 000 km^2 in the province of Misiones in north-eastern Argentina (Fig. 1). This area has an E–W altitudinal gradient ranging from the araucaria montane forest (900 m) to lowland broadleaf forests (150 m) and encompasses the range of forest and plantations of *A. angustifolia* in Argentina. Natural remnants comprise approximately 19 stands highly connected by an agroforestry matrix with isolated araucaria trees. The remnants include a few large stands in protected areas (maximum size – 600 ha) and numerous small stands outside protected areas (Rau 2005). Density of araucaria trees in forest remnants averages about 6 ind ha^{-1} (Rau 2005; Ríos 2006). Plantations are scattered among natural remnants and also occur outside the natural range of *Araucaria* in northwest Misiones. Density of trees in plantations ranges from 150 to 1500 ind ha^{-1} depending on management and age of
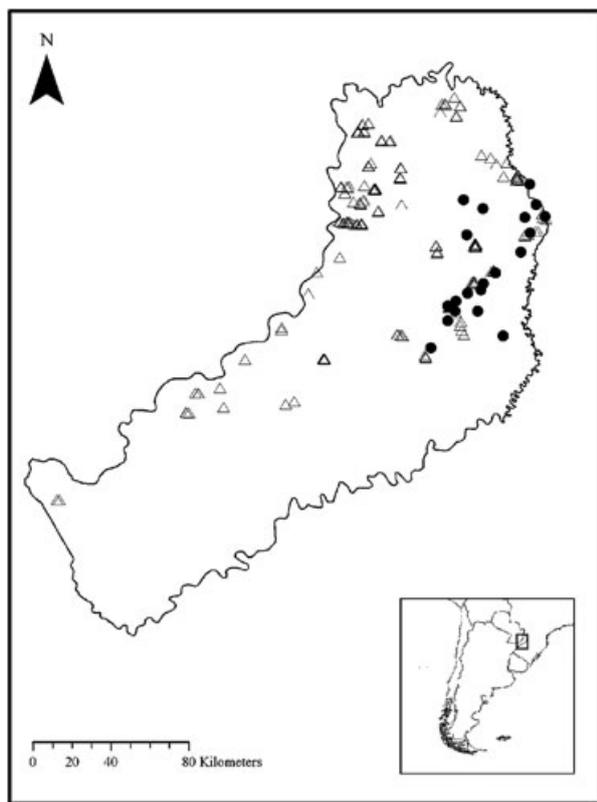
Fig. 1. Location of natural forests (●) and plantation (Δ) in Misiones Province, Argentina, where surveys were conducted for the araucaria tit spinetail.

the plantation. Most plantations have been managed to produce timber in rotations of 25–30 years. Silvicultural practices include pruning and thinning.

Site selection

Locations of natural remnants were obtained from a recent assessment conducted by Rau (2005) and all remnants were surveyed ($n = 19$, Fig. 1). Locations of araucaria plantations were mapped from Google Earth v4.3 (Keyhole Inc, Mountain View, California, USA, http://www.earth.google.com). Sixty plantations were selected randomly to encompass the wide array of sizes (1–435 ha), ages (4–60 years old) and isolation of plantations. Areas with planted araucaria trees were considered separate plantations if these areas had different age-classes of trees (see section on measurement of predictor variables for age-classes), or the distances between these areas were larger than 0.8 km, which is the median dispersal distance of several passerines similar in size to the araucaria tit spinetail (D'Eon *et al.* 2002) and more than eight times greater than the home range diameter of related furnarids (Díaz *et al.* 2006). Mean distance between plantations in the study was 1.9 km ± 2.2 (SD). Selection of plantations based on these criteria did not result in spatial autocorrelation in our dataset (see Results).

Density and occupancy data

We assessed occupancy and density of the araucaria tit spinetail in plantations and natural forests using point counts, separated by at least 250 m, and playbacks. If the area of the plantation was between 1 and 5 ha, one point was located at the approximate centre of the patch, and we randomly selected two additional points if the patch was large enough to meet the criteria for separation of points. If the plantation was larger than 5 ha, we conducted point counts along transects systematically placed at least 250 m apart along the length of the plantation while remaining at least 50 m from the edge of the plantation. We selected three random points for point counts and playbacks in each transect up to a maximum of nine points per plantation. A total of 150 point counts were taken in plantations. Because natural remnants were not always discrete patches and the density of araucaria trees was low, in remnants we followed the sampling criteria established for large plantations, but point counts ($n = 45$) were made at the closest araucaria tree to the randomly selected point.

Araucaria tit spinetails are highly mobile and easily detected by their characteristic vocalizations in the top of araucaria trees. Therefore, density of spinetails was assessed with point count surveys of 5-min duration using a snapshot approach in which the observer recorded locations of detected birds at a single moment, with time spent before this moment identifying and locating birds and afterwards confirming locations (Buckland 2006). Distances to birds from each point were recorded with a rangefinder. After point counts were obtained, occupancy was assessed at the same points with 5 min of playback. Given that detectability of the species using playback was 98.4% in a pilot study, once the species was detected, no further surveys of that site were conducted. Birds were considered to be absent if no birds were detected after three surveys of the point, each separated by at least 8 h. Surveys were conducted from September 2007 to January 2008 between 07.00–11.00 and 15.00–18.00 hours on days without rain and wind.

Measurement of predictor variables

We measured predictor variables for analysing factors affecting species distribution at patch and landscape scales in plantations. Patch scale represented the scale of individual plantations (1–435 ha). Our assessment at the landscape scale incorporated analyses of habitat within 5 km of each plantation and distances to neighbouring plantations that extended up to a maximum of 15 km. Because of the small number of remnants and high occupancy (100%), factors that affect occupancy and density of remnants were not assessed. At the patch scale we included measures of plantation area, stand age, and understorey height and density. Understorey density and height were estimated at a randomly placed point within a 10-m radius of the location of point counts by recording the number of vegetation contacts on a 20-mm diameter pole marked in 10 cm increments and by recording the height of the tallest understorey vegetation touching the pole. Understorey measurements were taken at one point in small plantations (1–5 ha) and three points in large plantations (>5 ha). Stand age, and tree density and tree

height for different aged stands were obtained from records of timber companies and landowners. We generated a categorical variable of four levels of plantation age: (i) 4–9 years old, (ii) 10–15 years old, (iii) 16–25 years old; and (iv) more than 25 years old, which corresponded to a gradient of decreasing tree density and increasing tree height (see Appendix SI).

For landscape-scale analyses, we used three measures of isolation for each plantation: (i) distance to the nearest neighbouring plantation (NN); (ii) mean distance to the three nearest plantations (THREE); and (iii) amount of available habitat (hectares of araucaria plantations) within a 5-km buffer around the plantation (BUFFER). Because birds were not surveyed in all plantations, we did not distinguish between occupied and unoccupied patches in isolation metrics. Plantation area and landscape variables were calculated using ARCGIS 9.2 (ESRI, Redlands, California, USA) (ESRI 2006).

Statistical analysis

We used a classification tree to model occupancy in plantations with the program DTREG (Sherrod 2003). Classification tree modelling involves partitioning of a dataset into increasingly homogeneous subsets (nodes), with each split defined by a simple rule based on the value of a single predictor variable (Breiman *et al*. 1984; De'ath & Fabricius 2000). Each variable entered in the model is assessed independently and the one generating the most homogeneous subsets determines the node splitting criteria. This method is especially appropriate for complex datasets that include imbalance and non-linear relationships. We included stand age and plantation area at the patch level in models and ran three simple classification trees each with a different isolation metric. Understorey height and density were positively correlated with stand age ($r = 0.59$, $r = 0.51$, respectively) and with each other ($r = 0.86$). Therefore, both understorey variables were excluded from all models. Because some plantations were spatially clustered, we also added the geographical location (x, y geographic coordinates) as another variable in the three models to assess potential effects of spatial location among samples.

We used Gini index of heterogeneity to determine optimal split and a minimum node size of 10 observations was required to perform a split and avoid model overfitting. Trees were constrained to the number of nodes allowed for one standard error from the minimum relative validation error. Model adequacy was assessed based on the percentage of data that were correctly classified, and classification accuracy of the trees also was evaluated using Cohen's Kappa (K) statistic (Cohen 1960; Fielding & Bell 1997). Kappa adjusts for bias associated with random model agreement by considering the difference of observed and expected agreement, given the frequency distribution within the dataset. Values of K can be used to classify model agreement as poor ($K \leq 0.4$), good ($0.4 \geq K \leq 0.75$) or excellent ($K \geq 0.75$, Landis & Koch 1977).

Estimates of absolute densities of spinetails in forest remnants and plantations from 10 to 60 years were obtained with distance sampling using the program DISTANCE 5.0 (Thomas *et al*. 2006). Plantations of this age were analysed in

order to match age structure of plantations and natural remnants. No natural remnants were comprised of young trees. Because araucaria tit spinetails were detected at a relatively small number of point counts in forest remnants (16 of 45), we modelled a global detectability function for both plantations and natural remnants and ranked the models following Akaike Information Criteria (Burnham & Anderson 2002). Encounter rate, rather than detection probability, was the main source of variation in both forest remnants and plantations, which supports the use of a global detectability function for different types of habitat.

Density of birds in plantations was modelled with respect to habitat variables using a Poisson generalized linear mixed model where two patch variables (stand age and plantation area) and one landscape variable (isolation) were fixed factors and identity of the plantation was a random effect. For modelling, we used PROC GLIMMIX in SAS v9.2 (SAS Institute, Inc., Cary, NC, USA) that produces true log likelihood estimates therefore enabling model comparison. We ran 15 models and then ranked models with the Akaike Information Criteria. Five models included one factor (stand age, plantation area or one of the three isolation metrics); seven models included two factors (stand age and plantation area, or one of these variables and one of the three isolation metrics); and three models included both stand age and plantation area and one of the three isolation metrics. To assess potential effects of spatial location among samples, we added geographical location (x, y geographic coordinates) to the best ranked model and compared this model with the same model without this variable.

RESULTS

All natural patches were occupied by araucaria tit spinetails, but only 85% of the plantations were occupied. The final classification tree for occupancy of plantations had two terminal nodes after pruning, and plantation age was the only predictor variable. Older plantations (≥ 10 years) had higher occupancy rates than younger plantations and a much lower misclassification rate (n, % occupancy, % misclassification; plantations ≥ 10 years old, 42, 97.6%, 2.4%; plantations <10 years old, 18, 55.6%, 55.6%). Overall the model correctly classified 49 of the 60 patches (81.7%) as occupied or unoccupied (Table 1). The Kappa statistic ($K = 0.49$, SE = 0.12) also indicated good model agreement with the data.

Table 1. Confusion matrix that indicates the agreement between observed occupancy of plantations and occupancy predicted by the classification tree model

Actual category	Predicted category		
	Occupied	Unoccupied	Misclassified
Occupied	41	10	19.60%
Unoccupied	1	8	11.10%

Table 2. Models of abundance of araucaria tit spinetails in plantations ranked with Akaike Information Criteria

Model[†]	k	AICc	ΔAICc	w_i
AGE + NN	3	297.07	0	0.344
AGE + BUFFER	3	298.04	0.97	0.211
AGE	2	298.78	1.71	0.146

[†]Models with $\Delta_i \leq 2$ are presented. k, no. explanatory variables plus 1; $\Delta_i = AIC_{ci} - $ minimum AIC_c; and w_i = Akaike weights. AGE, plantation age; BUFFER, amount of available habitat within a 5-km buffer around the plantation; NN, distance to the nearest neighbouring plantation.

Densities of spinetails were almost three times higher in plantations than in natural remnants (density, 95% CI, CV; plantations, 0.94 ind ha^{-1}, 071–1.23, 14%; natural forest, 0.36 ind ha^{-1}, 0.20–0.64, 29%). A hazard rate function with a polynomial series adjustment (Kolmogorov-Smirnov GOF $P = 0.37$) was the detection function that best fitted the data.

All highly ranked models (ΔAIC ≤ 2) for predicting abundance of spinetails in plantations based on patch and landscape variables included plantation age (Table 2). A model with age and the nearest neighbour measure of isolation ranked first, followed by a model with age and the amount of available habitat in a buffer. These two isolation metrics were correlated (see Appendix SI). Abundance of spinetails in plantations was best predicted by plantation age ($w_{sum} = 0.99$). Young plantations (4–9 years) showed lower abundances and were different from all the other categories of age (10 years and greater; Tukey's test for comparison of Least Squares Means, Table 3; Fig. 2). The sums of Akaike weights for other variables in the top models were much lower than for plantation age (NN, $w_{sum} = 0.45$; BUFFER, $w_{sum} = 0.28$), and models considering isolation as the only predictor variable ranked poorly (ΔAIC > 17). Plantation area was not included in competitive models, and incorporation of geographic location did not improve the best model.

DISCUSSION

Previous research has indicated that natural habitats generally are better for wildlife than human-generated landscapes (Lindenmayer & Hobbs 2004; Barlow et al. 2007; Du Bus de Warnaffe & Deconchat 2008). Our results highlight the importance of distinguishing between native and non-native plantations when plantations are evaluated for biodiversity value and when they are considered in mitigation of forest lost. Both natural remnants and plantations of native araucaria trees are important for araucaria tit spinetails. This species does not use pine plantations (Zurita et al. 2006) and continued replacement of araucaria forest and plantations with pine will have negative consequences for this species and other species associated with this forest (Fonseca et al. 2009).

The high occupancy (100%) of forest remnants by spinetails may be related to the high degree of connectivity of natural forests and possibly small area requirements of this species. Many forest remnants were located less than 5 km from the closest remnant, with an intervening matrix that included plantations and isolated araucaria trees. We observed spinetails in highly modified, natural forests with low densities of araucaria trees and no other trees. Spinetails can fly between araucaria trees separated by more than 80 m (Joenck 2005, A.G. Pietrek, 2007, pers. obs.), which suggests that the perceptual range of spinetails may permit the species to move among the canopies of relatively isolated individuals. No data are available on long-distance movements for this species. However, a closely related species (brown capped spinetail, *L. fuliginiceps*) displays seasonal movements in winter from the highlands to the lowlands of western Argentina (Narosky & Yzurieta 2003), indicating that at least this species can make long-distance movements when there is appropriate habitat. Similarly no data are available on the territory size of the araucaria tit spinetail, but the area used by a related species (des murs wiretail, *Sylviorthorhynchus desmursii*) in the southern temperate forest of Chile is estimated to be about 1 ha per pair (Díaz et al. 2006). If area requirements are similar for araucaria tit spinetail, this could promote persistence of this species in small remnants and plantations.

Although spinetails occupied 100% of the remnants, density was threefold higher in plantations that are greater than 9 years old. High densities do not necessarily indicate high habitat quality (Van Horne 1983) and habitat quality should be evaluated in terms of productivity of spinetails. However, the widespread occurrence and high densities of spinetails in plantations outside the natural range of *Araucaria* suggest that native plantations may be good habitat for this species (Krauczuk 2001, Antunes et al. 2007; Cabanne et al. 2007). The reason for higher densities in plantations as compared with natural remnants is unknown. Because these spinetails are closely associated with araucaria trees for foraging and, presumably nesting (only two nests have been found and both have been in araucaria trees, Bóçon 1993), tree density could be one factor in the low density of spinetails in natural remnants. In our study, spinetails rarely occurred in plantations with the highest density of trees (young plantations with >800 trees ha^{-1}), so clearly a simple linear relationship does not occur between bird density and tree density. However, degradation of araucaria forests through selective logging has decreased densities of trees in natural remnants so

Table 3.   Comparisons of abundance of araucaria tit spinetails in different age plantations with Tukey's test for comparison of Least Squares Means

Age comparison	Estimate	Standard error	d.f.	t value	Adjusted P
4–9 years vs. 10–15 years	−1.41	0.52	131.0	−2.72	0.048
4–9 years vs. 16–25 years	−2.04	0.51	131.0	−4.00	0.001
4–9 years vs. >25 years	−1.8	0.45	131.0	−3.97	0.002
10–15 years vs. 16–25 years	−0.62	0.42	52.8	−1.48	0.46
10–15 years vs. >25 years	−0.38	0.34	52.7	−1.10	0.69
16–25 years vs. >25 years	0.24	0.33	28.8	0.73	0.89
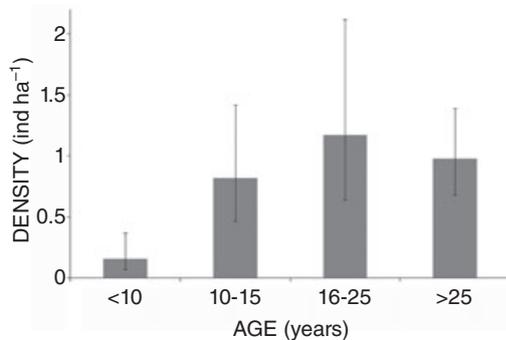
Fig. 2.   Density of araucaria tit spinetails (±95% CI) by age of plantation.

that araucaria densities in natural forest now are much lower than historical densities and much lower than in plantations of all ages. In the 1940s, densities of 48 trees ha^{-1} were recorded in some of the forest locations surveyed in this study (Ragonese & Castiglioni 1946). By the 1960s, average density was estimated at only 12 trees ha^{-1} (Rau 2005), but still far above the highest densities found in natural areas by the 1980s (7 trees ha^{-1}, Gartland 1984; Ríos 2006). Potentially, these very low numbers of trees in natural remnants could limit the density of spinetails.

Stand age influences occupancy of plantations by spinetails. All natural forest remnants in our study area contain old trees, so this relationship could not be evaluated for natural forests. Spinetails occupied most plantations greater than 9 years old, but often were absent from stands less than 10 years old, even when these plantations were less than 50 m from mature occupied plantations. This result is consistent with other studies that show that older plantations exhibit higher diversity and abundance of birds, a pattern generally linked to enhanced habitat structure in older plantations (Brockerhoff et al. 2008; Luck & Korodaj 2008). In southern Brazil, species richness and diversity of native plants increased dramatically with planting age up to 35 years in araucaria plantations and high canopy cover was an important factor explaining increased understorey diversity (Barbosa et al. 2009). Recent studies in other natural forests also indicate

that invertebrate fauna is more diverse and abundant in larger trees, resulting in higher abundance of insectivorous birds (Berg et al. 1994; Díaz et al. 2005, I. Díaz, 2008, in prep.).

Plantation area did not affect occupancy or density of spinetails in plantations, a result consistent with the hypothesis that this species can occupy small remnants because of low area requirements. Similarly, isolation was not a strong factor in predicting occupancy and models including only isolation ranked well below the first model. The large number and close proximity of plantations likely facilitated colonization of this species beyond its natural distribution. Surrounding habitat also has been shown to play an important role in patch isolation. Plantations and secondary forest are major components of the entire landscape that we surveyed in north-eastern Argentina. In contrast, in Brazil small araucaria patches are embedded in a hostile soy crop matrix and patches often are unoccupied even when potential source areas are as near as 700 m (A.G. Pietrek & M. Debarba, 2008, in prep.).

Occupancy of plantations was high in this study compared with previous estimates of 50% occupancy in 20 plantations in the same study area (Cabanne et al. 2007). We found all these 20 plantations to be occupied. These differences are unlikely to reflect changes in occupancy, but rather the previous study relied on passive observations, which may result in lower detectability than playbacks. Although occupancy estimates from our study were high, because of the rapid loss of araucaria plantations, distribution maps derived from our surveys may overestimate the area occupied by the species in the future. The two most isolated plantations in our study, which were at least 50 km from the nearest neighbouring plantation, were unoccupied. Furthermore as remnants and plantations become more isolated, demographic rescue and recolonization are likely to decline, resulting in an increase in unoccupied habitat.

Distribution of the araucaria tit spinetails in Argentina is strongly associated with araucaria plantations, as well as natural remnants. Plantations not only encompass nearly 90% of the remaining habitat but also exhibit high occupancy rates, although lower than natural forests, and high densities. Most of the remain-

ing natural stands are old and natural regeneration is low in many of these areas, threatening the viability of natural araucaria tree populations (Rau 2005). Restoration of these forests and conservation of old, connected plantations in Argentina may assure the protection of significant populations of spinetails.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

APPENDIX SI: (a) Characteristics of Araucaria plantations by age category. (b) Correlation between patch and landscape variables of plantations.