Magellanic Woodpeckers in three national parks of central-southern Chile: habitat effects and population variation over the last two decades


ABSTRACT. A major challenge for protected areas is providing wildlife with enough suitable habitat to cope with stochastic environment and increased pressure from the surrounding landscapes. In this study, we addressed changes in local populations of Magellanic Woodpeckers (Campephilus magellanicus) occupying three national parks of central-southern Chile. We compared the breeding and postbreeding abundance of woodpeckers during the 1990s with the present (2016) abundance (n = 4 years), and assessed the extent to which abundance was explained by forest type and quality of foraging habitat (as quantified through the plant senescence reflectance index; PSRI). Results show a distinctive temporal variation in woodpecker abundance at each park, with local populations of Magellanic Woodpeckers declining by 42.2% in Conguillío National Park, but increasing by 34.3% in Nahuelbuta National Park. Woodpeckers responded to forest conditions within each park such that their abundance increased with high quality of foraging habitat, i.e., large PSRI values, and the presence of old-growth Monkey puzzle (Araucaria araucana) - Nothofagus pumilio mixed forest. Anecdotal evidence suggests that populations of woodpeckers in Conguillío National Park might have responded negatively to large-scale disturbances from recent forest fires affecting part of the forest area within park. Because stochastic events seemed to strongly mediate population changes, our findings suggest that regional conservation of Magellanic Woodpeckers requires expanding the current conservation area network in central-southern Chile.

Le Pic de Magellan dans trois parcs nationaux du centre-sud du Chili : effets de l'habitat et variation des populations durant les deux dernières décennies

RÉSUMÉ. Les aires protégées font face au défi important de fournir suffisamment d'habitat propice à la faune pour compenser le côté imprévisible de l'environnement et les pressions grandissantes provenant des paysages voisins. Dans le cadre de la présente étude, nous avons évalué les changements advenus dans les populations locales du Pic de Magellan (Campephilus magellanicus) qui fréquentent trois parcs nationaux du centre-sud du Chili. Nous avons comparé les effectifs de ce pic en nidification et en post-nidification dans les années 1990 avec ceux de 2016 (n = 4 années), et évalué dans quelle mesure les effectifs étaient reliés au type forestier et à la qualité des aires d'alimentation (quantifiée par l'indice de réflectance de la sénescence végétale; en anglais PSRI). Nos résultats montrent une variation temporelle des effectifs de pics distinctive dans chaque parc : la population locale du Pic de Magellan a diminué de 42,2 % dans le parc national Conguillío, mais a augmenté de 34,3 % dans le parc national Nahuelbuta. Les pics se sont adaptés aux conditions forestières dans chaque parc, de sorte que leur abondance a augmenté là où les aires d'alimentation étaient de grande qualité, c.-à-d. avec des valeurs de PSRI élevées, et en fonction de la présence de forêt mélangée âgée de Désespoir du singe (Araucaria araucana) - Nothofagus pumilio. Des observations anecdotiques laissent croire que la population de pics du parc national Conguillío pourrait avoir réagi négativement aux perturbations à grande échelle causées par les récents feux de forêt qui ont touché une partie de la superficie forestière du parc. Puisque les événements imprévisibles semblent avoir un effet important dans les changements de population, nos résultats indiquent que la conservation régionale du Pic de Magellan profiterait d'une expansion du réseau actuel d'aïres de conservation dans le centre-sud du Chili.

Key Words: Campephilus magellanicus; national parks; population changes

INTRODUCTION

Conserving biodiversity in an increasingly human-dominated world requires, in part, the establishment of protected areas that can support viable populations of species of conservation concern (Wilson 1989, Pauchard and Villaroe 2002, IUCN 2017). Unfortunately, there are many success stories around the world where national legislation and international agreements have facilitated the design and implementation of networks of protected areas, including national parks or reserves. However, the effectiveness of these protected areas in maintaining viable populations of plants and animals is usually difficult to...
demonstrate (Bücking 2003, Rodrigues et al. 2004). A major challenge for protected areas is providing wildlife with sufficient habitat of high quality, especially when faced with stochastic environments and increased pressure from the surrounding landscapes (Roux et al. 2008, Paudel and Heinen 2015).

The role of protected areas in biodiversity conservation is particularly critical for rare, endangered, or unique ecosystems (Noss 2000, Noss et al. 2015), yet seldom explicitly examined. The temperate rainforests of central-southern Chile are one exception because the current network of national protected areas (known as SNASPE; see Squeo et al. 2012) has been scientifically challenged regarding its capacity to support viable populations of forest-dependent wildlife (Tognelli et al. 2008, Soto et al. 2012). Since the 1940s Chilean conservation areas, including national parks and forest reserves, have been designated to protect natural scenic landscapes containing temperate rainforest and montane areas that receive relatively little pressure from humans (Otero 2001). Nevertheless, the amount of undisturbed temperate forest protected under Chilean law may be insufficient for meeting conservation goals (Tognelli et al. 2008). The potential insufficiency of forest habitat derives, in part, from high vulnerability to natural disasters, e.g., landslides or volcanic eruptions, that regularly occur within the Andes Mountains, as well as human-induced disturbances, such as large fires intensified by regional summer drought (González et al. 2005). The relatively modest area of forest currently being protected by the southern Chile national parks, in combination with the increased risk of large-scale disturbances, raises important questions about the ability of parks to conserve forest species (e.g., Araújo et al. 2002). One approach to evaluating the effectiveness of central-southern Chilean national parks for conserving wildlife populations is to quantitatively assess population status of species of conservation concern (Parks et al. 2013, Burns et al. 2016).

In this study, we address midterm changes (20 years) of local populations of Magellanic Woodpeckers (Campephilus magellanicus), occupying central-southern Chilean national parks. The Magellanic Woodpecker is the largest woodpecker species in South America and the main primary cavity excavator of the southern temperate rainforests (Short 1970, Vergara and Schlatter 2004, Ojeda and Chazarreta 2006). Like other large woodpecker species, Magellanic Woodpeckers have relatively long parental care duration (two to three years), large territories (0.2 to 1.3 km²) and low densities (0.1 to 1.8 individuals/km²), which make them highly sensitive to forest loss and degradation caused by logging, wildfires, and natural disasters (Vergara and Schlatter 2004, Chazarreta et al. 2011, Soto et al. 2012, Ojeda and Chazarreta 2014, Vergara et al. 2014; see also Lammertink et al. 2009). Specifically, the loss of large, dying, or dead trees reduces availability of foraging, roosting, and nesting sites and, thus, induces population declines across multiple species of woodpeckers (Lammertink 2004, 2014, Mikusinski 2006, Bull et al. 2007, Pasinelli 2007, Lammertink et al. 2009, Kumar et al. 2014, Nappi et al. 2015). Remote sensing-based methods have proven to be effective in identifying trees with advanced decay stages, thus providing information on the quality of foraging habitat of Magellanic woodpeckers (Vergara et al. 2016, Vergara et al. 2017, Soto et al. 2017). Accelerating loss and degradation of forests have constricted the distribution of several woodpecker species, and some viable populations of the woodpecker are now confined to protected areas surrounded by productive lands (Setterington et al. 2000, Melletti and Peteriani 2003, Roberge et al. 2008a).

The Magellanic Woodpecker lives in family groups (between two to five individuals), establishing territories that are relatively stable between seasons and consecutive years (Ojeda and Chazarreta 2014), thus making them an appropriate species to assess temporal changes in abundance (e.g., see Pasinelli 2006, Drever et al. 2008, Drever and Martin 2010, Vergara et al. 2016). Although the breeding biology and foraging behavior of Magellanic Woodpeckers have been described previously (Short 1970, Vergara and Schlatter 2004, Schlatter and Vergara 2005, Chazarreta et al. 2012, Espinosa et al. 2016, Soto et al. 2016, Vergara et al. 2016), to date, no study has evaluated their population dynamics. Temporal fluctuations in the population size of large-bodied woodpecker species remain poorly understood because of rarity and low demographic rates (but see Mattsson et al. 2008). Adding complexity, the level of threats facing Magellanic woodpeckers varies geographically (Ojeda et al. 2011). The most significant contributions to the conservation of global population of Magellanic Woodpeckers likely are the native forests currently protected under the SNASPE, which are concentrated in the Patagonian and Sub-Antarctic Ecoregions (40° 00′ S to 55° 30′ S) and comprise 40 conservation areas (~115,000 km²) facing relatively low human pressure. Conversely, woodpeckers living in the relatively small national parks located in central-southern Chile (north of 40° S) may be highly vulnerable to native forest replacement by exotic plantations and agricultural lands. In this study, we compared the abundance of Magellanic woodpeckers during the 1990s with that of the present (2016) in central-southern Chile's national parks surrounded by different matrix types and identified the habitat features that best explain variation in local abundance.

**METHODS**

**Study area**

We focused our study on three national parks in the Araucanía region of central-southern Chile (between 900 and 1450 m altitude), including Nahuelbuta National Park (63 km²; 37.81 S, 72.99 W), Tolhuaca National Park (64 km²; 38.22 S, 71.83 W), and Conguillo National Park (545 km²; 38.69 S, 71.68 W). These national parks differ in their amount of native forest, with Nahuelbuta, Tolhuaca, and Conguillo comprising, respectively, 61.91 km² (98.2%), 30.36 km² (47.4%), and 230.72 km² (42.3%). Nahuelbuta national park is the only protected area within the Coastal mountain range, while the other two parks are located along the Andes mountain range (Fig. 1). Soils in Conguillo and Tolhuaca parks are derived from recently deposited volcanic ashes and scoria, while Nahuelbuta park has clayey soils developed on metamorphic rocks. Nonforest vegetation in Conguillo and Tolhuaca parks includes perennial grasses, scrub, cushion plants, mosses, and lichens growing on volcanic rocks, while in Nahuelbuta park it includes shrub vegetation in degraded forest. All national parks were located adjacent (< 10 km) to highly fragmented landscapes dominated by exotic plantations and agricultural areas (Fig. 1). Because these parks have been protected for over 80 years, their forests have remained mostly intact and free from logging activities since then. Forests are
mainly made up of different species of southern beech trees (genus *Nothofagus*) mixed with the Monkey puzzle tree (*Araucaria araucana*), including the following forest types: (1) old-growth *N. pumilio* and Monkey puzzle forest; (2) old-growth *N. antarctica* and Monkey puzzle forest; (3) old-growth *N. dombeyi*, *N. pumilio*, and Monkey puzzle forest; and (4) second-growth *N. antarctica* forest (Espinosa et al. 2016). Conguillío National Park, mostly made up of pristine forests, experienced the eruption of Llaima Volcano in 2008, where lahar flows affected part of the protected forest and volcanic ash was deposited on the forest canopy. In addition, anthropogenic large-scale (> 1000 ha) fire events occurred recently in Conguillío National Park (summer 2015) and Tolhuaca National Park (summer 2002).

### Abundance surveys

Abundance of woodpeckers was monitored during the breeding season (austral spring) and post-breeding season (austral summer) in the late 1990s (i.e., 1995–1998) and during 2016. In each national park, we conducted 8 min (Vergara et al. 2010) point count surveys in 34 sites: 18 points in Conguillío National Park, 13 points in Nahuelbuta National Park, and 3 points in Tolhuaca National Park. Sampling points were randomly established in stands of different forest types (Table 1), at least 100 m from the stand edge. Because adult Magellanic Woodpeckers are conspicuous, noisy (e.g., Vergara et al. 2016), and thus easily detected, we recorded all woodpeckers seen and heard within 100-m-radius (e.g., Ralph et al. 1993). A single trained observer annually visited each point up to eight times between 1995 and 1998, and three to five times between November and March 2016.

### Table 1

<table>
<thead>
<tr>
<th>Variable</th>
<th>Tolhuaca</th>
<th>Conguillío</th>
<th>Nahuelbuta</th>
</tr>
</thead>
<tbody>
<tr>
<td>Monkey puzzle tree - <em>N. dombeyi</em> forest</td>
<td>0.0%</td>
<td>38.9%</td>
<td>46.2%</td>
</tr>
<tr>
<td>Monkey puzzle tree - <em>N. pumilio</em> forest</td>
<td>66.7%</td>
<td>33.3%</td>
<td>0.0%</td>
</tr>
<tr>
<td>Monkey puzzle tree - <em>N. antarctica</em> forest</td>
<td>0.0%</td>
<td>11.1%</td>
<td>15.4%</td>
</tr>
<tr>
<td>Plant Senescence Reflectance Index (PSRI)</td>
<td>-0.63 (0.032)</td>
<td>-0.65 (0.057)</td>
<td>-0.61 (0.074)</td>
</tr>
</tbody>
</table>

Points were spaced at least 500 m apart to reduce spatial autocorrelation arising from neighborhood effects. In order to increase detection probability, during 2016 the observer carried out an additional 8 min-long “active” survey by using a
“drumming device” (DD) once every minute. The DD is a wooden box with drum sticks to imitate the double-tap territorial drum that Magellanic Woodpeckers and other campophiline species make while patrolling their territories (Soto et al. 2016, Vergara et al. 2016). It produces a sound that is comparable to that broadcasted during playback surveys, but that is louder and more resonant than sound played from an electronic device (Saracco et al. 2011, Fogg et al. 2014). Abundance estimates derived from active surveys should be greater than those from passive surveys (without using the DD), being reflected by differences in detection probability between census techniques (see the Data analysis section).

**Habitat variables**

Forest habitat was characterized within 100 m around each point count using remote sensing (WorldView-2 and -3, and Landsat-8) and a GIS database of forest types (CONAF 2011). We estimated the plant senescence reflectance index (PSRI; Merzlyak et al. 1999), a measure of tree senescence, i.e., wood decay (Soto et al. 2017; Table 1). PSRI uses red and green bands to measure chlorophyll degradation and carotenoid content of the tree canopy, with higher values indicating more advanced decay stage of trees (e.g., see Soto et al. 2017). Previous studies have demonstrated that Magellanic Woodpeckers select individual trees based on their PRSI values (see details in Vergara et al. 2016). These tree preferences are consistent with the occurrence pattern of the long-horned beetle *Microplophorus magellanicus*, one of the main prey of woodpeckers, which respond positively to tree senescence (Vergara et al. 2017). In addition, tree senescence is a reliable indicator of the occurrence of nest-sites of woodpeckers because these cavities are easier to excavate in decayed trees (e.g., Blanc and Martin 2012, Zahner et al. 2012). We used WorldView-2 and -3 satellite imagery, which provide high spatial and spectral resolutions (0.5 m pixel size and 8 spectral bands), allowing a precise estimation of tree attributes. First, the satellite imagery was geocorrected and atmospherically corrected. Next, a digital supervised classification was carried out to discriminate between forest and nonforest land cover classes using ENVI v. 5.0, then supervised classification was carried out to discriminate between forest and nonforest land cover classes using ENVI v. 5.0, then

Data analysis

We used a N-mixture Bayesian model (Royle 2004) to estimate the abundance of woodpeckers at the *i = 1, 2, ... R* sites (point-count locations), at the *t* year, on each sampling replicate (*t = 1, 2, ... T*). We considered a multiyear dynamic model that assumes the population is demographically closed within each year (e.g., see dynamics models for woodpeckers in Fogg et al. 2014). The number of distinct woodpeckers counted at point *i* in year *j* and occasion *t* (*n* *j* *i* ) was modeled as an independent realization of a Binomial random variable dependent on the actual number of woodpeckers at point *i* in year *j* (*N* *j* *i* ) and the detection probability at *t* (*p* *j* *i* ), such that *n* *j* *i* ~ Binomial(*N* *j* *i* ; *p* *j* *i* ) with mean *p*N.

The actual number of woodpeckers at a point is a latent, unobserved Poisson distributed variable, *N* *j* *i* ~ Poisson(*λ* *j* *i* ), where *λ* *j* *i* represents the average abundance at site *i* in year *j* and is modeled as a function of independent covariates:

\[
\log(\lambda_{i,j}) = \beta_0 + \beta_1 H_i + \beta_2 Y + \beta_3 P S R I_i + \beta_4 Y \times P S R I_i \\
+ \beta_5 F_i \times Y + \beta_6 P S R I_i \times Y 
\]

(1)

where \( \beta_k = \delta_k \) is the \( k \)th fixed-effect coefficient (of \( k \) total coefficients, including an intercept). Covariates in Equation 1 were specified as follows: (1) Survey year (*Y*) is a factor with four levels, one for each survey year, i.e., 1996, 1997, 1998, and 2016; (2) Forest type (*F*) is a factor with four levels, one for each forest type, i.e., *N. dombeyi*- Monkey puzzle, *N. pumilio*, *N. antarctica*- *N. Antarctica*, and *N. antarctica*; (3) PSRI is a continuous variable ranging between -0.8 and -0.5; (4) Interactions between the above explained covariates, *Y* \( \times F \) and *Y* \( \times P S R I*.

The probability of detection *p* *j* *i* , at site *i* during the year *j* and sampling occasion *t* was estimated with the following logit function:

\[
\logit(p_{i,j,t}) = \alpha_0 + \alpha_1 F_i + \alpha_2 Y + \alpha_3 F_i \times Y \\
+ \alpha_4 P S R I_i \times Y + \alpha_5 S_i + \delta_i 
\]

(2)

where is the \( k \)th fixed-effect coefficient associated with the ability of observers to detect woodpeckers among different years (*Y*), seasons (*S*), and forest types (*F*). Annual turnover of individuals occupying territories, reproductive status and differences in conspicuousness between habitats may cause variation in the detection probability across years, seasons, and forest types, respectively. We controlled for spatial autocorrelation arising from counting the same woodpecker families at neighboring points by including a spatially structured term (\( \delta \)) from a conditional autoregressive model (e.g., Chandler and Royle 2013).

In a posterior analysis, we compared detection probabilities derived from passive and active surveys, both carried out during 2016 through a simplified function for the survey method *m*, at the site *i*, and during the time period *j* (*m = 1 for passive and *m = 2 for active):

\[
\logit(p_{i,j,m,t}) = Y_{i,j,m,t} 
\]

(3)

Parameter distributions and abundance estimates were based on three Markov Chain Monte Carlo (MCMC) samples, each with 20,000 iterations, discarding the first 10,000 iterations and thinning by 5. MCMC Convergence was visually examined and by using the potential scale reduction factor (Gelman et al. 2014). We evaluated 40 candidate models with different covariate combinations and ranked them by their deviance information criteria (DIC; Spiegelhalter et al. 2003). Models were run using OpenBUGS v. 3.2.3 (Lunn et al. 2009) via the R2OpenBUGS package of R v. 3.2.0 (Sturtz et al. 2005, R Development Core Team 2008). The regular and intensive sampling frequency in each survey point (~20 surveys per point), the seasonal stability in site occupancy, and the reduced number of model covariates contributed to improve model fitting. Indeed, model estimation did not exhibit convergence problems despite the number of sampling points (*n* = 34 points). The Bayesian credible intervals (BCI) estimated from posterior distribution of parameters was used to evaluate the importance of each. In addition, we estimated
Table 2. The five best-ranked Bayesian N-mixture CAR candidate models used to predict the abundance of Magellanic Woodpecker (*Campephilus magellanicus*). Covariates included in abundance model and probability functions are shown separately. Deviance’s information criterion (DIC), DIC difference with the lowest DIC model (ΔDIC) and model weights (ω) are shown.

<table>
<thead>
<tr>
<th>Abundance function</th>
<th>Probability function</th>
<th>DIC</th>
<th>ΔDIC</th>
<th>ω</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>Year + Season</td>
<td>5142.22</td>
<td>0.00</td>
<td>0.37</td>
</tr>
<tr>
<td>Forest + Year</td>
<td>Year + Season</td>
<td>5143.79</td>
<td>1.58</td>
<td>0.17</td>
</tr>
<tr>
<td>Forest + Year + PSRI</td>
<td>Year + Year + Season</td>
<td>5144.86</td>
<td>2.65</td>
<td>0.10</td>
</tr>
<tr>
<td>Forest + PSRI</td>
<td>Year + Season + PSRI</td>
<td>5145.69</td>
<td>3.48</td>
<td>0.07</td>
</tr>
<tr>
<td>Forest</td>
<td>Year + PSRI</td>
<td>5146.20</td>
<td>3.99</td>
<td>0.05</td>
</tr>
</tbody>
</table>

The coefficients of the best supported models (Table 3) showed a significant positive effect of the Monkey puzzle - *N. pumilio* forest on the woodpecker abundance, when compared to the other forest types (Table 3). The abundance of woodpeckers was also positively affected by the PSRI, as supported by the second-best model (Table 3; Fig. 2). Despite being included in the best supported models, neither year nor season had significant effect over the abundance or detection probability (Table 3). Bidecadal variation (1990s vs. 2016) in density of Magellanic Woodpeckers was significant (p < 0.05) in both Conguillío and Nahuelbuta National Parks (Fig. 3, Table 4). However, woodpecker density decreased by 42.2% ± 9.64 (SD) in Conguillío National Park, while in Nahuelbuta National Park it increased by 34.3% ± 20.65 (SD; Fig. 3, Table 4). The detection probability showed no significant annual variation (Table 4).

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**RESULTS**

A total of 218 individuals were detected during 649 different field surveys at the 34 sampling points, averaging 4.9 (SD = 2.1) visits per year and point (range: 4 to 7 visits). Detection probability (Mean ± SD) did not differ between passive (p = 0.43 ± 0.14) and active surveys (p = 0.45 ± 0.15).

We found support (ΔDIC ≤ 2) for two models explaining the abundance and detection probability of Magellanic Woodpeckers, with both models being better supported than the null model (ΔDIC = 15.46). The most parsimonious candidate model included only the effect of forest type on the abundance, and the effect of year and season on the detection probability (Table 2). The second best supported candidate model (ΔDIC = 1.58) retained the effect of forest type on the abundance, but also included the additive effect of year and PSRI (Table 2). The latter candidate model included the effect of year, season, and forest type on the detection probability (Table 2).
Table 3. Coefficients from the two best-supported Bayesian N-mixture CAR models explaining the abundance of Magellanic Woodpeckers (Campephilus magellanicus) in the studied national parks. The mean, standard deviations (SD), p-value, and 95% lower and upper Bayesian credible intervals (CI) are presented. For comparison reasons, coefficients for Year 2016, reproductive season and forest of N. dombeyi - Monkey puzzle tree were set to be zero.

<table>
<thead>
<tr>
<th>Function</th>
<th>Variable</th>
<th>Mean</th>
<th>SD</th>
<th>Lower CI</th>
<th>Upper CI</th>
<th>p-value</th>
<th>Rhat</th>
</tr>
</thead>
<tbody>
<tr>
<td>λ</td>
<td>N. pumilio - Monkey puzzle tree</td>
<td>0.48</td>
<td>0.2</td>
<td>0.08</td>
<td>0.87</td>
<td>0.01</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>N. antarctica - Monkey puzzle tree</td>
<td>0.01</td>
<td>0.34</td>
<td>-0.69</td>
<td>0.65</td>
<td>0.53</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>N. antarctica</td>
<td>-0.09</td>
<td>0.23</td>
<td>-0.56</td>
<td>0.35</td>
<td>0.56</td>
<td>1.0</td>
</tr>
<tr>
<td>p</td>
<td>Year 1996</td>
<td>0.02</td>
<td>3.19</td>
<td>-6.26</td>
<td>6.24</td>
<td>0.49</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1997</td>
<td>0</td>
<td>3.16</td>
<td>-6.26</td>
<td>6.18</td>
<td>0.51</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1998</td>
<td>0.02</td>
<td>3.15</td>
<td>-6.17</td>
<td>6.14</td>
<td>0.5</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Postreproductive season</td>
<td>0.04</td>
<td>3.15</td>
<td>-6.1</td>
<td>6.18</td>
<td>0.49</td>
<td>1.0</td>
</tr>
</tbody>
</table>

Model 2 (ΔDIC = 1.58)

<table>
<thead>
<tr>
<th>Function</th>
<th>Variable</th>
<th>Mean</th>
<th>SD</th>
<th>Lower CI</th>
<th>Upper CI</th>
<th>p-value</th>
<th>Rhat</th>
</tr>
</thead>
<tbody>
<tr>
<td>λ</td>
<td>PSRI</td>
<td>1.9</td>
<td>1.08</td>
<td>-0.01</td>
<td>4.03</td>
<td>0.04</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1996</td>
<td>0.26</td>
<td>0.22</td>
<td>-0.15</td>
<td>0.69</td>
<td>0.33</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1997</td>
<td>-0.04</td>
<td>3.16</td>
<td>-6.26</td>
<td>6.1</td>
<td>0.52</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1998</td>
<td>-0.01</td>
<td>3.16</td>
<td>-6.21</td>
<td>6.18</td>
<td>0.51</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>N. pumilio - Monkey puzzle tree</td>
<td>0.37</td>
<td>0.21</td>
<td>-0.01</td>
<td>0.78</td>
<td>0.03</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>N. antarctica - Monkey puzzle</td>
<td>-0.1</td>
<td>0.34</td>
<td>-0.8</td>
<td>0.55</td>
<td>0.58</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>N. antarctica</td>
<td>-0.19</td>
<td>0.24</td>
<td>-0.67</td>
<td>0.27</td>
<td>0.62</td>
<td>1.0</td>
</tr>
<tr>
<td>p</td>
<td>Year 1996</td>
<td>-0.03</td>
<td>3.17</td>
<td>-6.28</td>
<td>6.14</td>
<td>0.51</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Year 1997</td>
<td>0.02</td>
<td>3.17</td>
<td>-6.2</td>
<td>6.25</td>
<td>0.5</td>
<td>1.0</td>
</tr>
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<td></td>
<td>Year 1998</td>
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<td>6.12</td>
<td>0.5</td>
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</tr>
<tr>
<td></td>
<td>N. pumilio - Monkey puzzle tree</td>
<td>0.03</td>
<td>3.17</td>
<td>-6.13</td>
<td>6.22</td>
<td>0.49</td>
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<tr>
<td></td>
<td>N. antarctica - Monkey puzzle tree</td>
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<td>-6.16</td>
<td>6.18</td>
<td>0.5</td>
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<tr>
<td></td>
<td>N. antarctica</td>
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<td>3.17</td>
<td>-6.2</td>
<td>6.21</td>
<td>0.5</td>
<td>1.0</td>
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<tr>
<td></td>
<td>Postreproductive season</td>
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<td>3.19</td>
<td>-6.2</td>
<td>6.22</td>
<td>0.49</td>
<td>1.0</td>
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</tbody>
</table>

DISCUSSION

Persistence of woodpecker populations in protected areas requires sufficient availability of old-growth forests (e.g., Lammertink 2004) that provide woodpeckers with coarse woody debris, snags, as well as large living trees suitable for foraging and nesting (Angelstam and Mikusinski 1994, Vergara and Schlatter 2004, Drever and Martin 2010, Lorenz et al. 2016). Although systematic and regular time-series data are required to assess population dynamics and demography, the results of this study largely contribute to understanding the bidecadal variation of local woodpecker populations inhabiting two national parks (Conguillío and Nahuelbuta National Parks). Despite significant bidecadal variation in woodpecker populations living in central-southern Chilean national parks, these parks had contrasting population trends, with woodpeckers in the Nahuelbuta and Conguillío National Parks exhibiting increasing and declining trends, respectively. The pattern of population decline in Conguillío National Park suggests that this park may be falling short of its goal to protect viable populations of forest wildlife. At this point, the causes of the declines are uncertain, but we note that the Conguillío National Park has experienced large recent forest disturbances. Indeed, recent fires have destroyed at least 35 km² of native forest in Conguillío. Thus, our findings paired with what we have learned about the ecology of woodpeckers suggest that populations may be limited by availability of suitable habitat. However, the declining status of the woodpecker population in the Conguillío park should be confirmed through identifying proximate mechanisms of population change, such as survival and reproduction success.

Our study suggests that woodpeckers responded to forest conditions, and that remotely-sensed estimates of habitat quality were not homogeneous within or among national parks. Old-
Table 4. Bayesian estimates of the yearly mean density and detection probability of Magellanic Woodpeckers (Campephilus magellanicus; Mean ± SD) for two national parks (NP), as fitted by the best-supported model (Table 2). Different letters indicate significant differences between years for the same park, as evaluated through their Bayesian credible intervals (CI). Bidecadal (1990s vs. 2016) variation (%) in density (± SD) and its p-values are also shown for each park (*p < 0.05; ***p < 0.001).

<table>
<thead>
<tr>
<th>Year</th>
<th>Conguillio NP</th>
<th>Nahuelbuta NP</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Density (n/km²)</td>
<td>Detection probability (p)</td>
</tr>
<tr>
<td>1996</td>
<td>3.21±0.17 a</td>
<td>0.39±0.22</td>
</tr>
<tr>
<td>1997</td>
<td>3.87±0.14 b</td>
<td>0.47±0.24</td>
</tr>
<tr>
<td>1998</td>
<td>2.79±0.18 c</td>
<td>0.39±0.24</td>
</tr>
<tr>
<td>1990's</td>
<td>3.29±0.10 a</td>
<td>0.42±0.23</td>
</tr>
<tr>
<td>2016</td>
<td>1.90±0.23</td>
<td>0.47±0.25</td>
</tr>
<tr>
<td>20-y change in density (%)</td>
<td>-42.53±9.64***</td>
<td>34.28±20.65*</td>
</tr>
</tbody>
</table>

According to the Bayesian estimates, there is a significant decrease in density between 1996 and 2016 for Conguillio NP, with a 42.53% decrease in density, while Nahuelbuta NP shows a 34.28% increase in density. The p-values indicate that these changes are statistically significant (***p < 0.001).
Responses to this article can be read online at:

Acknowledgments:
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LITERATURE CITED


