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Habitat preference and abundance of *Coscoroba coscoroba* and *Cygnus melancoryphus* in Petrel wetland (O'Higgins region, Chile): Implications in the conservation

Preferencia de hábitat y abundancia de *Coscoroba coscoroba* y *Cygnus melancoryphus* en el humedal Petrel (Región O'Higgins, Chile): Implicaciones en la conservación

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ABSTRACT

Wetlands are environments with a high diversity and global importance due to the ecosystem services they provide. In Chile, most wetlands are threatened, including their avifauna, especially the two southern hemisphere swans (*Coscoroba coscoroba* and *Cygnus melancoryphus*). Here, we made three years of monitoring of abundance of both species in the Petrel wetland. Our results show that the Petrel wetland is an important site for the conservation of South American swan populations, with an abundance average of 38.5 ± 18.8 individuals of *C. coscoroba* (peak of 60 ind.) and 11.6 ± 12.4 individuals of *C. melancoryphus* (peak of 40 ind.). These swan species avoid sites close to urban areas and show a preference for habitats distant from anthropogenic activities. The detection probability of swans was influenced by minimum and maximum temperature and wind speed. The vegetation aquatic and riverine are key variables to the abundance of these swans. Our large monitoring highlights the importance of Petrel wetland for the conservation of southern swans and provide value information about their abundance patterns and the population dynamics of *C. coscoroba* and *C. melancoryphus*. These data support the need to develop conservation and management plans for this wetland.

Keywords: Anserinae, Black neck swan, Coscoroba swan, temporal seasonality, urban wetlands

RESUMEN

Los humedales son ambientes con una alta diversidad e importancia global debido a los servicios ecosistémicos que proporcionan. En Chile, la mayoría de los humedales están amenazados, incluyendo su avifauna, destacando los dos cisnes del hemisferio Sur (*Coscoroba coscoroba* y *Cygnus melancoryphus*). Realizamos tres años de monitoreo de la abundancia de ambas especies en el humedal Petrel. Nuestros resultados muestran que el humedal Petrel es un sitio importante para la conservación de las poblaciones de cisnes sudamericanos, con un promedio de abundancia de 38,5 ± 18,8 individuos de *C. coscoroba* (máximo de 60 ind.) y 11,6 ± 12,4 individuos de *C. melancoryphus* (máximo de 40 ind.). Estas especies de cisnes evitan los sitios cercanos a las áreas urbanas y muestran preferencia a los hábitats distantes de las actividades antrópicas. La probabilidad de detección de los cisnes estuvo influenciada por la temperatura mínima y máxima y la velocidad del viento. La vegetación acuática y ribereña son variables claves para la abundancia de estas especies. Nuestro amplio monitoreo destaca la importancia del humedal Petrel para la conservación de los cisnes de Sudamérica y proporciona información valiosa sobre sus patrones de abundancia y la dinámica poblacional de *C. coscoroba* y *C. melancoryphus*. Estos datos respaldan la necesidad de desarrollar planes de conservación y manejo para este humedal.

Palabras clave: Anserinae, cisne coscoroba, cisne cuello negro, estacionalidad temporal, humedales urbanos.

INTRODUCTION

Wetlands are environments with a high diversity and global importance due to the ecosystem services they provide (Estades *et al.* 2012). However, they are considered within the most threatened ecosystem in the world, especially by anthropogenic disturbances such as urbanization, drainage and intensive crops (Gagné & Fahrig 2007). In Chile, most wetlands are threatened (Saavedra & Villarroel 2019), especially their avifauna (Estades & Vukasovic 2013). Also, the public conservation policies have not been really effective, therefore, many wetlands are currently threatened and without legal protection figures (Mella-Romero *et al.* 2018). Most of these wetlands are associated to river estuaries and have an intertidal regime, and therefore human activities such as shell fishing, artisanal fishing or salt production are abundant (Paredes *et al.* 2015).

The subfamily Anserinae is composed by 33 species, most of them present in temperate zones of the north hemisphere, while being scarce in the south hemisphere (Johnsgard 2010). In South America, the presence of two species stands out, coscoroba swan (*Coscoroba coscoroba* Molina, 1782) and black-necked swan (*Cygnus melancoryphus* Molina, 1782), the only species of swans in this region of the continent. Despite their restricted distribution, the information regarding them is limited and unlike the species of swans of the Northern Hemisphere, of which it has been achieved establishes that their populations are increasing or are stable in number, for these two species, the information on the size and trends of the populations is quite imprecise, therefore the conservation status of South American swan species is not well known. However, currently like all swan species worldwide according to the International Union for Conservation of Nature (IUCN), these species are classified as “Least Concern” (IUCN 2016) globally, but the lack of data on their population trends and the poor quality of existing data regarding their population ecology, such as their habitat requirements, makes this assessment unsafe (Rees *et al.* 2019).

Both species are highly dependent on wetlands to maintain their populations and are recognized as susceptible to rapid habitat loss (Rees *et al.* 2019), especially in Chile, where they have an irregular distribution (Rees & Brewer 2005), and many of these ecosystems are unprotected. Despite this, studies on their habitat preferences and the seasonal variation of their populations are scarce (Vilina *et al.* 2002; Figueroa-Fábrega *et al.* 2006; Gibbons *et al.* 2007; Silva & García 2007; González *et al.* 2011) especially in urban wetlands. Therefore, the knowledge about abundance patterns and seasonality of these species is essential to develop appropriate conservation and management plans.

Therefore, to contribute to improve their conservation status, it is important to know the environments to which they are directly associated within the wetlands and to understand the relationships with their habitat. Thus, we hypothesize that population variations of both swan species depend on vegetation cover and influenced by anthropogenic threats in urban wetlands. For this, throughout several years of monitoring, we assessed the habitat preference and abundance of *C. coscoroba* and *C. melancoryphus* in an urban wetland. Specifically, 1) to estimate the abundance patterns of their populations; and 2) to determine the habitat preference.

MATERIAL AND METHODS

Study area

Petrel wetland is located in the estuary of San Antonio river in the O’Higgins region, Chile (34°23’01’’ S-71°59’58’’ W; Fig. 1). The wetland has a size of 69 ha approx. and it mainly receives fresh water from San Antonio river and two tributaries (Quebrada El Retiro and Quebrada El Lingue). During winter and due to the rains, this wetland connects with the Pacific Ocean, suffering therefore a strong seasonality (Andrade & Grau 2005). The climate is maritime Mediterranean with oceanic influence (Luebert & Pliscoff 2017). The riparian vegetation is mainly represented by the presence of the species *Sarcocornia neei*, *Distichlis spicata*, *Lupinus arboreus*, *Conium maculatum*, *Raphanus sativus*, *Carpobrotus aequilaterus* (García 2007). Petrel wetland has high biodiversity levels (Cornejo *et al.* 2018a), however, and in despite of the great abundance of bird communities with respect to other national wetlands, it never received a protection figure and it is not considered within the Ramsar Convention (Mella-Romero *et al.* 2018). This space is highly threatened by eutrophication and contamination due to its proximity to an urban core (Vargas 2017).

C. coscoroba and *C. melancoryphus* monitoring

We monitored, monthly, the abundance of *C. coscoroba* and *C. melancoryphus* during 3 phenological years, from April of 2016 to March of 2019 (n = 36). We selected 6 survey sites at a distance of approx. 400 meters between them, and with an observation radius of 200 meters. These points allowed for a good visibility to watch the species and to cover the whole wetland (Bibby *et al.* 2000, Fig. 1). All the samplings began 30 minutes after sunrise, ensuring that the meteorological conditions are adequate for observation, that is, without rain, fog or strong wind (De la Maza & Bonacic 2013). Two researchers participated in each session, one of whom observed and reported, and the other recorded what was observed, staying for 5 minutes per point. The number of individuals per species was recorded through visual records (supported by Bushnell falcon 10x50mm binoculars and Celestron ultima 65 18-55x 65mm spotting scope) and auditory over the body of water and the riverside of the wetland (Bibby *et al.* 2000). The order of the census points (start and end) was inverted monthly to avoid bias caused by the starting hour. This way, if the order of a transect was from point 1 to point 6 one month, the next month it would be from point 6 to point 1.

Temporal and environmental variables

For each survey point, we delimited a buffer area of 200 m radius to identify landscape variables (Fig. 1). Using Google Earth® (2018) images and ArcGIS 10.3 software (ESRI, Redlands, California, USA), we draw all habitat types and structures as vector shapes. We classify land use into the following types: 1) Wetland; 2) Riverine vegetation; 3) Aquatic vegetation; 4) Forest plantation; 5) Grasslands; 6) Sea; 7) Beaches and bare soil; 8) Urbanized land. For the analysis, each land cover was represented as the percentage of occupied area (Table 1; Supplementary Material 1). Additionally, to explore the influence of anthropogenic activities, we calculated the distance from each sampling point to the urban core (Table 1). In addition, at the beginning of each census, we record the following temporary climatic variables: Temperature (°C); minimum and maximum temperature (°C), accumulated precipitation of the previous day (mm) and wind speed (km/h) (Supplementary Material 1). These meteorological data were obtained from the "Puente negro" meteorological station, which is close (<2 km) to the study site and belongs to the "INIA Agrometeorological Network (www.agrometeorologia.cl)". These variables were incorporated as predictor variables in our models.

Data analysis

To analyse the differences between years in the species abundance, we used a regression model with Poisson distribution and then we applied a two-tailed ANOVA with Tukey multiple comparison post hoc test (Zuur *et al.* 2007), where we treated the wetland as a single site. We used a significance level of 5% (α = 0.05). We implemented these analyses using the *emmeans* package in R software (R CORE TEAM 2019).

To determine the habitat preference of the two swan species, we calculated the selection/avoidance for each survey point using the preference index (*E*) by Ivlev (1961), which has the following equation:

$$E = (Uj - Aj)/(Uj + Aj)$$

where *Uj* is the proportion of presences in each site *j* and *Aj* is the proportion of availability in the environment. This index ranges between -1 and +1, where -1 is total avoidance and +1 is total selection. This way, it is possible to know the habitat preference for each swan species within the wetland.

We used *N*-mixture models to estimate the local population size of the swan species while accounting for imperfect detection (Royle 2004). To estimate the detection probability and local population size, the *N*-mixture model uses count data that are obtained during a period of time in which the local populations are assumed to be closed (i.e. there aren't any births, deaths, and movement) which corresponds to the months comprised between April and September of every year. The data recorded during those months for both species are the ones that were used.

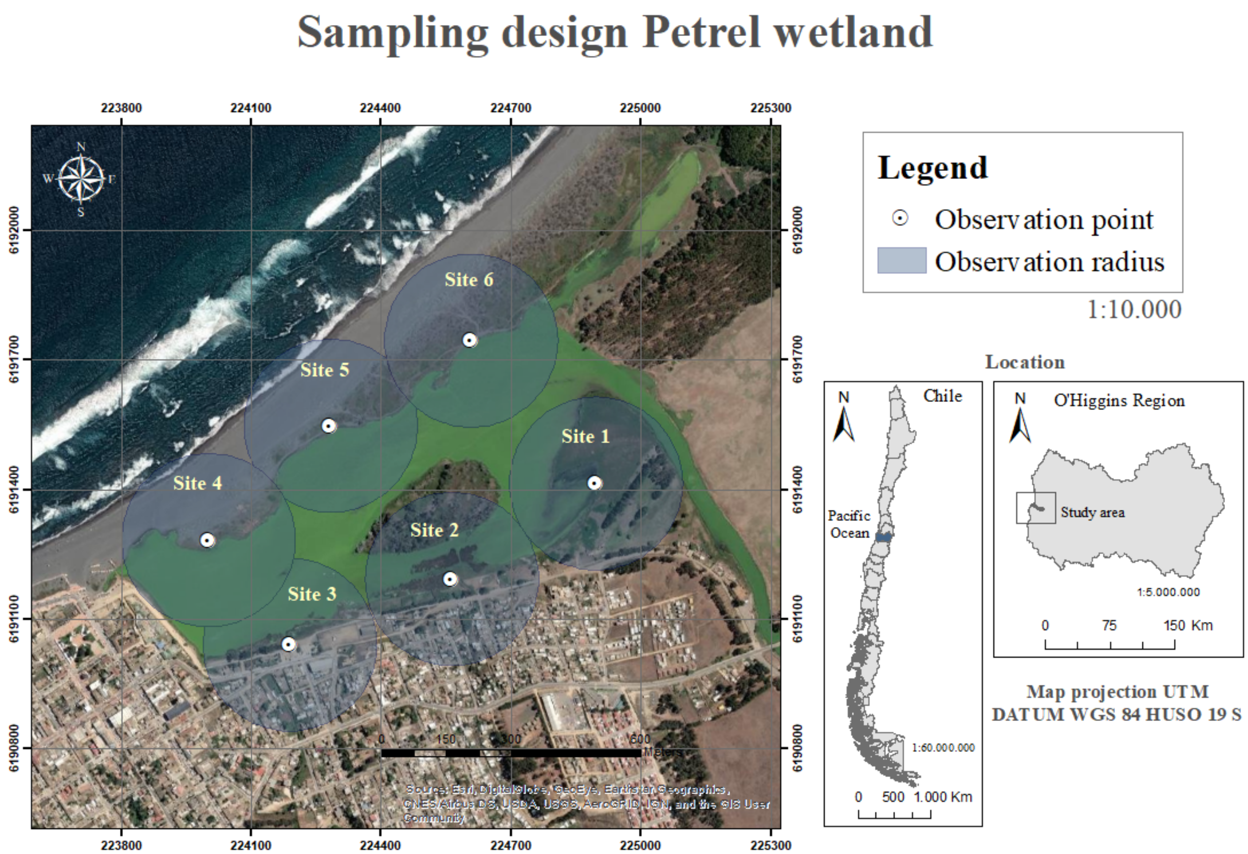

Figure 1. Study area and sampling design. / Área de studio y diseño de muestreo.

Table 1. Characterization of sampling sites. / Caracterización de los sitios de muestreo.

Sites	Wetland (%)	Riverside vegetation (%)	Aquatic vegetation (%)	Forest plantation (%)	Grasslands (%)	Sea (%)	Beaches and bare soil (%)	Urbanized land (%)	Distance to urban center (m)
Site 1	30.1	34.8	0.0	7.9	23.8	0.0	0.0	3.3	490
Site 2	20.3	4.2	8.7	16.2	12.1	0.0	2.1	36.2	200
Site 3	39.4	5.2	2.2	0.0	1.6	0.0	0.0	51.6	120
Site 4	55.2	8.0	0.0	0.0	0.0	12.1	24.7	0.0	130
Site 5	37.8	16.6	2.5	0.0	0.0	7.7	35.4	0.0	500
Site 6	41.5	26.6	3.1	0.0	0.0	0.0	28.8	0.0	650

N-mixture models were fitted using *pcount* function of the *unmarked* package in R software (Fiske & Chandler 2011). We first fit null models for each mixture type (Poisson, zero-inflated Poisson or negative binomial) in order to select the most appropriate error distribution. Then, we fit different candidate detection probability models (*p*). Using this best detection model, we calculated different candidate estimate abundance models (λ). Detection probability, as well as abundance models were fitted according to all possible additive combinations of temporal and environmental variables (Supplementary Material 1) as covariates, respectively. For both the detection and abundance modelling process, we added the year of measurement as a possible covariate. The best mixture type and the best detection model were selected using second-order Akaike's Information Criterion (AICc). Tested covariates did not show a correlation between them

(| Pearson | < 0.6), therefore there cannot be multicollinearity in the models. When two or more covariates were correlated, univariate models (i.e. models with only one covariate) were fitted to select the covariate that was best associated with the response variable (lower *p*-value). All continuous covariates were standardized by converting to z-scores before to the modelling process in order to be able the comparison of estimated parameters by the models.

We used AICc to compare candidate models, and to estimate parameter, unconditional standard errors and 95 % confidence via model averaging (Burnham & Anderson 2002). Model averaging was done using *AICcmodavg* package in R software (Mazerolle 2020). Candidate models that represented a 95 % confidence set of best-ranked regression models (i.e. the models which cumulative Akaike weight ≤ 0.95) were averaged. We used the parametric bootstrap approach to obtain *p-values* from sums of squares, Chi-square and Freeman-Tukey fit statistics that quantified the fit of a model to a data set, and as a measure of the goodness of fit of the best-ranked regression models. Model averaging was done with models that fitted well, i.e. with models that overcame the goodness-of-fit test. We simulated 10,000 bootstrap samples for each fit assessment. A dispersion parameter ($\hat{c}$) was calculated as the ratio of the observed Chi-square fit statistic to the mean of the simulated distribution. We consider evidence of overdispersion and lack of adjustment of the models when $\hat{c} \geq 2$ (MacKenzie *et al.* 2017). We also used the best-ranked regression models to estimate the latent abundance at each site and year using empirical Bayes methods through the ranef function of the unmarked package in R software (Fiske & Chandler 2011). This method estimate the posterior distribution of the latent variable (abundance in our case) given the data and the estimates of the fixed effects of N-mixture models.

RESULTS

We counted a total of 1387 *C. coscoroba* with an abundance average of 38.5 ± 18.8, where the highest abundance was registered at sites 1 and 6, while the species almost did not appear in the site 4. For *C. melancoryphus*, we counted a total of 416 individuals with an abundance average of 11.6 ± 12.4, where the species highest abundance was at sites 1 and 2 (Table 2).

The annual distribution of *C. coscoroba* was its highest from November to March, with a peak of 60 ± 13.7 individuals, while *C. melancoryphus* had its highest abundance from November to December, with approx. 20 ± 7.4 individuals. The Petrel's wetland maintains a stable population of 40 individuals for *C. coscoroba* and 12 individuals for *C. melancoryphus*.

When we analysed the differences in estimated abundances through the survey years, we observed that there were significant differences between them for *C. coscoroba* and *C. melancoryphus* (Table 2, Fig. 2). However, there were a decrease in the abundance for *C. coscoroba* during year 2.

Table 2. Accumulated abundance per year and the average abundance at each sampling point for both swan species. The Ivlev's index range between -1 (avoidance) to +1 (preference). / Abundancia acumulada por año y la abundancia media en cada punto de muestreo para ambas especies de cisnes. El índice de Ivlev oscila entre -1 (evitación) y +1 (preferencia).

Species	Year	Accumulative individual number	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
C. coscoroba	1	533 ± 17.5	12.3 ± 5.2	13.1 ± 6.3	2.1 ± 2.8	0.8 ± 1.4	5.0 ± 4.7	11.5 ± 13.0
	2	253 ± 24.0	11.5 ± 7.7	6.3 ± 5.2	0.0 ± 0.0	0.0 ± 0.0	7.1 ± 13.7	9.2 ± 6.9
	3	435 ± 13.0	13.8 ± 5.1	5.6 ± 2.9	0.9 ± 1.1	1.0 ± 1.9	4.5 ± 2.8	10.3 ± 7.8
		Ivlev's Index	0.724	0.710	0.386	0.097	0.625	0.703
C. melancoryphus	1	212 ± 16.7	4.8 ± 6.1	8.2 ± 10.4	0.8 ± 1.0	1.0 ± 1.8	1.3 ± 2.3	1.5 ± 2.8
	2	114 ± 10.0	2.3 ± 2.3	3.5 ± 4.6	0.4 ± 0.9	0.3 ± 0.8	2.0 ± 3.1	0.9 ± 2.3
	3	90 ± 6.6	2.6 ± 2.7	2.9 ± 2.3	0.7 ± 0.9	0.7 ± 1.4	0.3 ± 0.8	0.3 ± 0.6
		Ivlev's Index	0.654	0.754	0.338	0.247	0.376	0.190

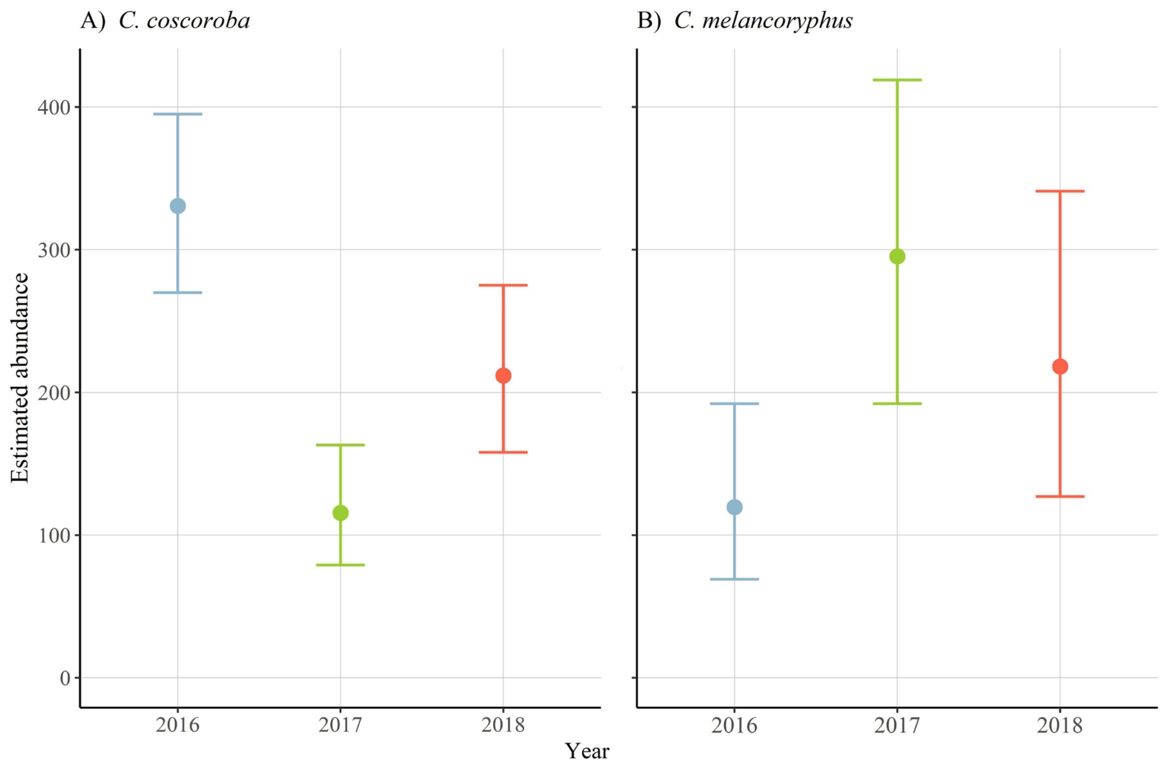

Figure 2. Estimated abundance per year for *C. coscoroba* (A) and *C. melancoryphus* (B) considering the Petrel wetland as a single site. / Abundancia estimada por año para *C. coscoroba* (A) y *C. melancoryphus* (B) considerando el humedal Petrel como un solo sitio.

Ivlev's habitat preference index showed that *C. coscoroba* had a similar preference for sites 1, 2, 5 and 6, while *C. melancoryphus* showed preference for sites 1 and 2. Both species avoided the sites 3 and 4 (Table 2, Fig. 1).

With the exception of one model, all the best-ranked regression models successfully passed the goodness-of-fit and dispersion tests (Supplementary Material 2, 3 and 4).

When we analysed the abundance of both species, we found that *C. coscoroba* only showed differences between years in the site 1, while *C. melancoryphus* only showed them in the site 6 (Supplementary Material 5).

The detection probability of *C. coscoroba* is determined by minimum and maximum temperature and wind speed (Supplementary Material 6), where the most important variable is the maximum temperature (Supplementary Material 7). For *C. melancoryphus* also the precipitation level is a key factor (Supplementary Material 6), but the maximum temperature is the most important variable (Supplementary Material 7). The abundance models of *C. coscoroba* showed that presence of riverine and aquatic vegetation is important for this species (Supplementary Material 6) where aquatic vegetation is the most relevant variable (Supplementary Material 7). However, for *C. melancoryphus* the landscape variables were not important (Supplementary Material 6).

The probability of detection of *C. coscoroba* is positive with an increase in the maximum temperature and wind speed, while is negative for an increase in the minimum temperature (Fig. 3A). The models for *C. melancoryphus* show that this species have a negative detection probability when the variables of maximum temperature, wind speed or precipitation increase. However, the detection probability is positive if the minimum temperature increases (Fig. 3B).

The estimated abundance of *C. coscoroba* is positive with regards to the percentage of aquatic and riverine vegetation cover (Fig. 4A and 4B) and for *C. melancoryphus* showed a positive response with the percentage of grassland cover (Fig. 4D).

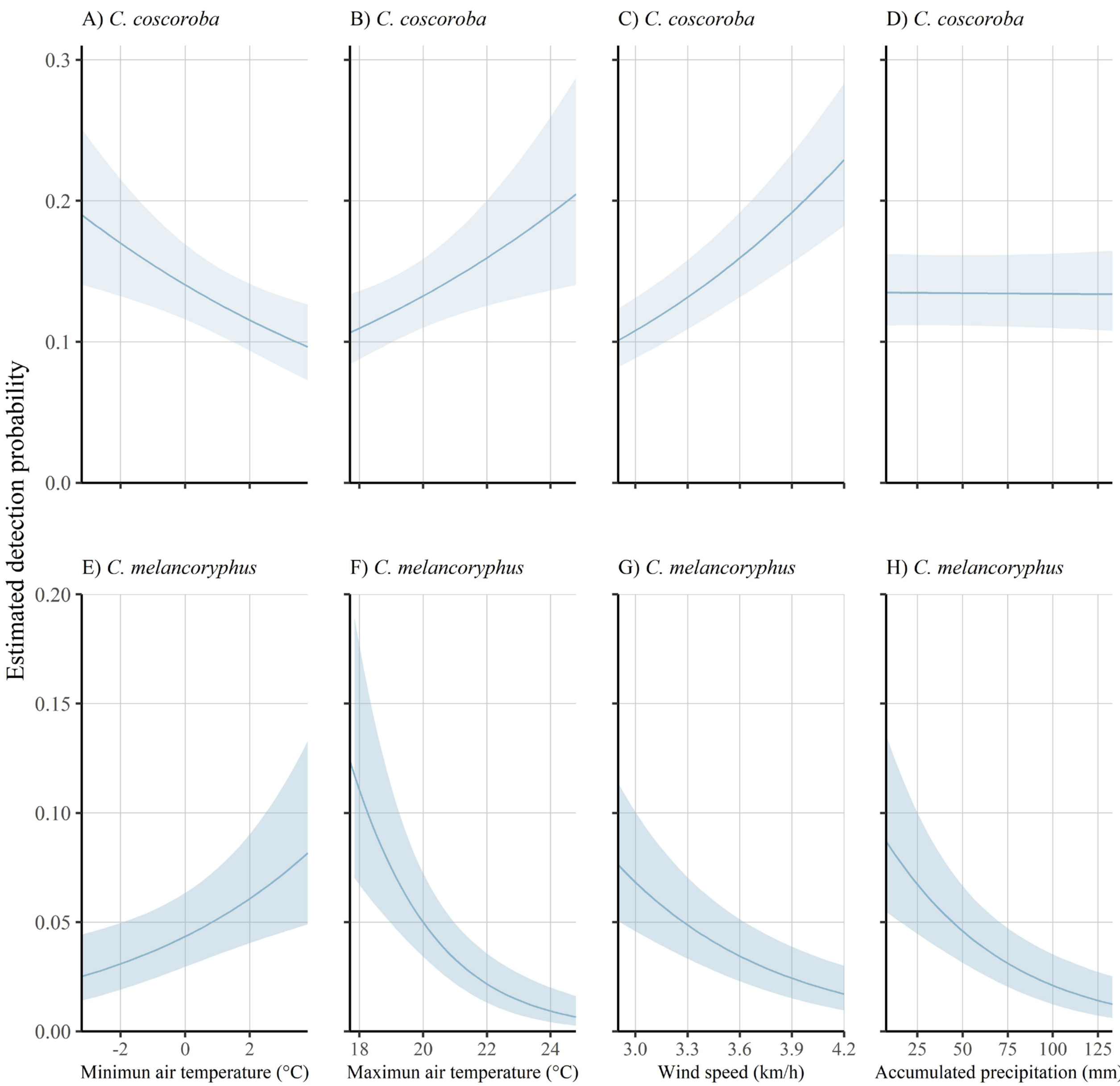

Figure 3. Detection probability prediction using model averaging from the best-ranked regression models by species (See Supplementary Material 7). The relation between the best-adjusted temporal variables and the detection of *C. coscoroba* (A) and *C. melancoryphus* (B). / Predicción de la probabilidad de detección utilizando el promedio de los modelos de regresión mejor clasificados por especie (Véase Material Suplementario 7). Relación entre las variables temporales mejor ajustadas y la detección de *C. coscoroba* (A) y *C. melancoryphus* (B).

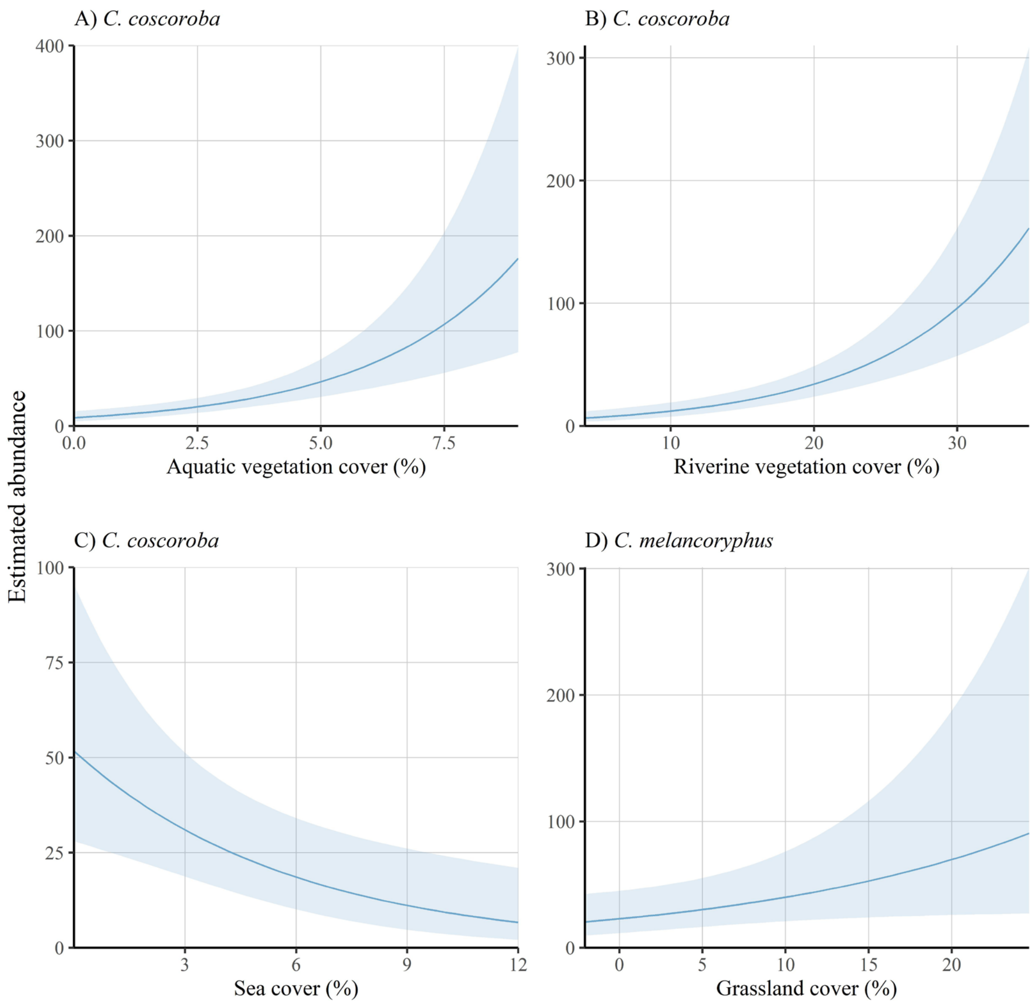

Figure 4. Abundance prediction using model averaging from the best-ranked regression models by species (See Supplementary Material 7). The relation between the best-adjusted environmental variables and the abundance of *C. coscoroba* (A, B y C) and *C. melancoryphus* (D). / Predicción de la abundancia utilizando el promedio de los modelos de regresión mejor ajustados por especie (Véase Material Suplementario 7). Relación entre las variables ambientales mejor ajustadas y la abundancia de *C. coscoroba* (A, B y C) y *C. melancoryphus* (D).

DISCUSSION

In spite of their high biodiversity level, the ecosystem services provided and their importance for bird conservation (Rojas *et al.* 2003; Ten Brink *et al.* 2013), in Chile, wetlands are barely protected and there is not a monitoring in place of threatened bird species (González-Acuña *et al.* 2002; Canepa *et al.* 2006; Silva & Brewer 2007; González *et al.* 2011; Estades & Vukasovic 2013).

Our results show that Petrel wetland is an important site for the conservation of South American swans, in accordance with the results obtained in previous aquatic bird census in Chile between 2009-2011 (Matus *et al.* 2010; Schmitt *et al.* 2011), in wich it is mentioned that a site is nationally important if it harbors 5 % of the estimated Chilean population of a species. At least for the case of *C. coscoroba* the threshold value of 5 % (according to the latest published reports of national waterbird censuses - CNAA) corresponds to 23 individuals in 2009 (Matus *et al.* 2010) and 29 individuals in 2010 (Schmitt *et al.* 2011). Our results record an average of 39 individuals, thus exceeding the 5 % threshold. Therefore, Petrel's wetland is a site of national importance for the conservation of these swans. Additionally, our results highlight that the population density in this area are the highest recorded in comparison with other Chilean wetlands within the Ramsar Convention (Schlatter *et al.* 1991; Rojas & Tabilo 2004; Matus *et al.* 2010; Muñoz-Pedreros & Merino 2014; Mella-Romero *et al.* 2018; Miranda-García *et al.* 2021).

Even so, the populations of both species could range through years due to environmental conditions. In our study, we observed that the abundance of both species decreased significantly in 2017 (year 2). This happened because, during that year, the decision makers opened a channel between the wetland and the ocean, causing the entrance of saltwater. In others studies, it was found that anseriforms are not tolerant to high levels of salt (Chapman & Kimstach 1996; Soto & Bert 2010), and therefore, this decision could cause the swans to abandon the wetland. Similar information was found in others studies with *C. coscoroba* and other anatids (Cornejo *et al.* 2018b).

Regarding the detectability of the species, our results provide relevant information for the development of monitoring programs suitable for the species, since considering the factors that influence the probability of detection can improve the precision of the estimates (Vanausdall & Dinsmore 2020). Furthermore, this is the first study that has examined the importance of considering factors that may influence the probability of detection when estimating the density or abundance of South American swans.

The probability of detection varied between both species. The minimum and maximum temperature and the wind speed significantly influenced the probability of detection with a linear relationship. These variables influenced inversely between both species. Wind speed, for example, tends to negatively influence the detectability of some species, such is the case of *C. melancoryphus*, since windier conditions make it difficult for birds to hear (Harms & Dinsmore 2012), however, being large birds, their detection does not depend to a large extent on their vocalization. On the contrary, the wind speed was positively related to the detectability of *C. coscoroba*. On the other hand, the maximum temperature showed a negative effect on the detection of *C. melancoryphus*, but not for *C. coscoroba*, since it has been shown that temperature has a slight positive effect on some breeding wetland birds, as it can increase bird activity (Verner 2019), but others have found little or no effect (Conway & Gibbs 2011; Harms & Dinsmore 2012). By virtue of this, in order to explain these differences between both species, more specific studies are required that would facilitate the analyzes to increase our understanding of the factors that affect the probability of detection of South American swans and would improve our ability to estimate population trends.

On the other hand, our results shown that both species have preferences for those habitats far away from urban areas, and with abundant riverine and aquatic vegetation. In this last context, it has been described that South American swan, due to their condition as aquatic herbivores, depend on the dominance and accessibility of macrophyte banks in the shallow areas of coastal and limnetic wetlands of southern South America (Velasquez *et al.* 2019). Therefore, these sites provide for feeding areas (Couve *et al.* 2016) and breeding and rest areas (Gibbons *et al.* 2007; Silva & Brewer 2007). In turn, we found that the sites 3 and 4 were avoided by the swans since those sites have high levels of anthropic impact such as cattle, rubbish, feral dogs and horse races (Cornejo *et al.* 2018b; Mella-Romero *et al.* 2018; Bravo-Naranjo *et al.* 2019) representing a serious threat for the swans.

Also, our data shown that the wetland maintains stable populations of both swan species but with an important seasonality during the summer months (Fig. 2). Some authors affirm that this phenomenon is typical in the centre of Chile and it is a process named "*local dispersion in warm period*" (Schlatter & Sielfeld 2006; Mella 2008; Simeone *et al.* 2008; Vilina & Cofré 2008; Estades & Vukasovic 2013). These studies indicate that the swans commute from small lakes and ponds of the central valley, which dry up during the summer and, therefore, cannot provide food or breeding areas (Estades *et al.* 2012).This phenomenon increases the importance of the coastal wetlands to maintain and conserve swan populations and highlights the need to study the commuting routes of these swans with the objective to create a network of wetlands.

Our study stands out the sensibility of *C. coscoroba* y *C. melancoryphus* populations to the anthropogenic disturbances and the need to maintain long period monitoring to understand the abundance changes of these species. Also, it shows the importance of Petrel wetland in the conservation and maintenance of the *C. coscoroba* y *C. melancoryphus* populations. There must urgently be developed conservation plans for this important wetland, increasing the riverine vegetation and decreasing the anthropic disturbances such as cattle, feral dogs or opening channels to the ocean.

In conclusion, this study provides valuable information to know the habitat use and abundance patterns and the population dynamics of *C. coscoroba* y *C. melancoryphus* in urban wetlands. Therefore, we highlight the importance of Petrel wetland as a priority site for the conservation of both species. Indeed, it is necessary to increase the educational programs to valorise the conservation of urban wetlands.

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