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Research article

Bird and bat mortality at wind farms in South America: Lessons from monitoring and mitigation practices in Chile

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ABSTRACT

Wind energy can help mitigate global CO_2 emissions; however, it also has adverse effects on biodiversity, particularly through collision-related mortality among flying vertebrates. While these impacts have been extensively studied in North America and Europe, information from South America remain limited. In this study, we assessed bird and bat mortality, along with monitoring and mitigation practices at wind farms in Chile, one of the leading countries in wind energy development in South America. We analyzed 15 years of post-operational monitoring data from 47 wind facilities and examined the drivers of wildlife mortality and evaluate the methods used to monitor, estimate, and mitigate these impacts. We documented a total of 1218 bird fatalities representing 80 different species, and 1250 bats fatalities from 6 species. The only threatened species recorded was the Andean condor (*Vultur gryphus*), with nine casualties across three wind farms, all located in the north-central Chile. While bird collisions showed no clear seasonal pattern, bat mortality peaked during spring and autumn. Mortality rates were influenced by a range of factors, including environmental, biotic, geographic, and turbine-related characteristics. Our study revealed that monitoring strategies are often inconsistently reported and lack standardization. Carcass removal trials, essential for correcting detection and persistence biases, are rarely conducted. Only 56 % of the wind farms implemented mitigation measures, with passive measures more commonly adopted than active ones. These findings highlight the need to standardize monitoring protocols and apply appropriate bias correction methods in mortality estimates at the wind farm scale. These improvements are essential for drawing reliable conclusions about wildlife impacts and for designing effective mitigation strategies at regional and national levels.

1. Introduction

The ongoing efforts to mitigate anthropogenic climate change effects and meet international commitments have boosted the development of renewable energy use worldwide. Wind energy stands as the renewable energy that has experienced the highest expansion in recent years, with almost 779.841 GW produced in 2022, which is expected to increase to 2400 GW by 2027 (IEA, 2023). Both offshore and onshore wind energy production is led mainly by North American and Northern European countries, together with others from the Asian continent, such as India and China (IRENA, 2025). However, in the last decade, emerging countries, such as those in the Global South, have shown interest in including this energy source to supply the population with this sustainable energy (Matthäus and Mehling, 2020). Although wind energy remains relatively expensive in many low- and some middle-income countries in the Global South, expectations of the reduction of financial costs associated with it will likely trigger a major wave of wind energy development in the coming years (Matthäus and Mehling, 2020; Snyder, 2020).

Among countries in the Global South, South American nations are expected to expand their wind energy production by 122 % by 2032, mainly in Argentina, Brazil, Chile, and Peru (Solaun and Cerdá, 2019). These countries are considered major biodiversity hotspots due to their functional diversity, species richness and abundance (Myers et al., 2000), and unique ecological realms (Dinerstein et al., 2017), making them a priority target for conservation. Despite wind energy being a

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sustainable alternative to carbon-driven energies, it still has adverse impacts, particularly on bird and bat populations (Katzner et al., 2025), through direct mortality (i.e., population-level effects) and displacement (i.e., functional habitat loss) (Thaxter et al., 2017; Heuck et al., 2019; Marques et al., 2020). Given the knowledge of the threat that turbines pose to flying vertebrates, evaluating their impact on the Global South countries (e.g., South America) where wind energy production is still incipient is highly important to anticipate potential effects on sensitive species populations (Agudelo et al., 2021). Identifying the species groups most affected by wind energy and shedding light on the potential factors influencing fatalities in turbines are essential to guide management actions at a regional scale. Far beyond, this information could help establish standardized monitoring protocols and implement effective mitigation measures by selecting the most appropriate biomonitoring strategies and adopting cost-effective measures that reduce bias in mortality rate estimation and halt direct mortality due to collision in turbines in these species-rich areas.

In this study, we evaluate the impact of wind farms on birds and bats in Chile, a country at the forefront of the energy transition and ranked first in renewable energy investment in Latin America and the Caribbean (Bloomberg, 2018; IRENA, 2025). Wind energy development in Chile began in 2001 and has expanded to approximately 1360 turbines with an installed capacity of 4.6 GW. This rapid growth has occurred in a biodiversity-rich region that supports globally threatened species such as the Andean condor (*Vultur gryphus*) and the rufous-tailed hawk (*Buteo ventralis*) (Petit et al., 2018; Martínez-Harms et al., 2021). Despite this, the ecological impacts of wind energy in Chile remain largely unassessed. Existing studies from other South American countries, such as Brazil, Colombia, and Uruguay, have typically been limited to single sites or short-term monitoring, focusing mainly on species inventories or broad collision risk estimates rather than systematic post-construction assessments (Agudelo et al., 2021). In contrast, national-scale syntheses combining multi-year monitoring, mortality estimation, and evaluation of mitigation practices are still lacking across the region (Rebolo-Ifrán et al., 2025). Our study helps address this gap by providing the first large-scale assessment of bird and bat mortality, monitoring protocols, and mitigation measures across Chilean wind farms.

Our objectives are threefold. First, we review the methods used to assess bird and bat mortality at wind farms in Chile, compile multi-year mortality data, and identify the species most affected. Second, we evaluate the influence of seasonality, environmental variables, and turbine characteristics on mortality rates, providing evidence for drivers that have rarely been quantified in South American contexts. Finally, we assess the range and effectiveness of mitigation measures implemented at wind farms, filling a regional gap in comparative analyses of practical responses to wildlife impacts. Overall, our study provides applied tools based on scientific evidence to guide the design of standardized monitoring protocols and effective mitigation strategies, advancing efforts to reconcile wind energy development and wildlife conservation in Chile

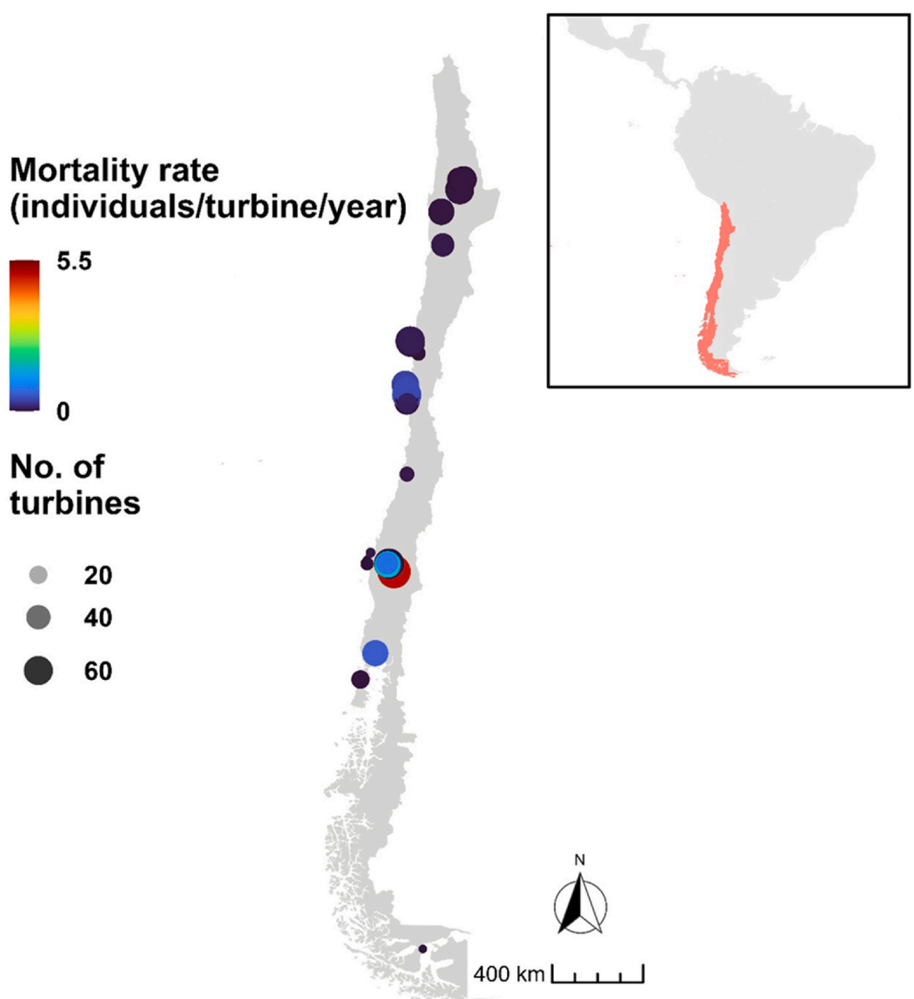

Fig. 1. Distribution and mortality rates of wind farms with reported fatalities in Chile at 2023 (n = 36). The circle size represents the number of turbines at each wind farm, and the color intensity represents the mortality rate as individual per turbine and year.

and across South America.

2. Methods

2.1. Study area

Wind farms in Chile span from northern edge of the Atacama Desert (22°28′S 68°47′O) to southern Patagonia (52°56′S 70°50′O). However, most wind farms are gathered in the Atacama Desert and the Mediterranean area of central Chile (Fig. 1). By the end of 2023, a total of 1266 had been installed across the country. A variety of turbine models have been deployed depending on the developer, but the most common capacity per turbine is 2 MW, with installed capacities across wind farms ranging from 0.82 MW to 850 MW. The most frequent turbine hub height is 90 m (range: 71–200 m), and rotor diameter span from 40 to 150 m.

2.2. Compilation of monitoring procedures and mortality data

Information on post-construction monitoring practices and mitigation measures associated with bird and bat collisions adopted in wind farms in Chile was compiled from the Environmental Impact Study (EIS) and Environmental Approval Resolution (*Resolución de Calificación Ambiental* EAR) for each project. These data were sourced from the public database of the *Servicio de Evaluación Ambiental* (SEA) of Chile (MINSEGPRES, 2010).

In June 2023, we consulted the SEA database (https://www.sea.gob.cl) and selected 47 EIS and their corresponding EAR documents corresponding for wind farm projects approved and developed between 2006 and 2018. These represent all available studies for wind farms approved and operational up to 2022. For each project, we compiled the monitoring methodology and the mitigation measures developed. A comprehensive review of all EIS documents was conducted to determine whether mitigation measures for birds, bats, or both were included. None of the Environmental Impact Assessment determined a potential significant impact on these groups, in accordance with Chilean environmental legislation. As a result, wildlife mortality monitoring was generally carried out under voluntary environmental commitments rather than as a regulatory requirement. In those cases, where monitoring methodologies were reported, we gathered details on the timing of carcass searches, whether carcass persistence trials were conducted (including the type of carcass used), and whether other corrections (i.e. detection bias) were applied to estimate turbine-related mortality more accurately.

Our final dataset included complete post-construction monitoring surveys from 36 wind farms, and mortality data extracted from 558 reports corresponding to 40 operational wind farms across the country from 2009 to 2022. Then, we reviewed all available reports submitted to the Superintendence of the Environment (SMA – *Superintendencia del Medio Ambiente*), the agency responsible for environmental compliance and oversight, for projects that included monitoring as part of their environmental commitments. Each report was downloaded and reviewed individually to extract data on mortality, as well as other relevant variables such as monitoring frequency, sampling effort, and methodological details. Monitoring reports were accessed through the SMA public database (https://snifa.sma.gob.cl/SeguimientoAmbiental/RCA). These documents detail the mitigation measures actually implemented during operation. In cases that mitigation measures were reported, we classified then into two types: active measures, when they involve human active intervention (e.g., turbine shutdown on demand or carrion removal), and passive measures, which do not require human intervention (e.g., automated shutdown systems, light-based or acoustic deterrent devices).

2.3. Estimation of observed mortality rates

Mortality monitoring data collected by environmental consultants at each wind farm were compiled. The information included species identification, number of individuals, date of collision, and total monitoring period. Based on these data, the annual observed mortality rates per turbine for each species at each wind farm were estimated. Mortality rate was calculated by dividing the total number of fatalities by the total number of monitoring years and the total number of turbines at each wind farm.

The detection of fatalities is affected by several sources of bias, including search efficiency, carcass disappearance due to scavenging, and crippling bias (Ravache et al., 2024). These biases are influenced by various factors such as habitat type, the composition of the scavenger community, the searcher's experience, and the time interval between searches (Barrientos et al., 2018; Smallwood, 2007, 2017). Unfortunately, no previous estimates of these biases are available for Chile or for South America as a whole. For this reason, we used raw data (i.e., observed mortality), acknowledging that these values represent minimum mortality estimates for each wind farm. However, we consider that comparisons among wind farms within the same ecoregion remain valid, as the scavenger communities and vegetation structure, key determinants of detection and persistence biases, are likely to be similar, provided that comparable monitoring methodologies were applied.

The annual observed mortality rates per wind farm were used to evaluate the effect of factors that could influence mortality at the country scale. However, to justify using the overall mortality of each wind farm, we assessed whether the mortality rates at each wind farm changed over the operational years (see, for example, Ferrer et al., 2012). To do this, we analyzed using linear models to analyze whether there was a relationship between the annual mortality rates of birds and bats and the number of years the wind farms had been in operation (from 1 to 9 years). These results yielded no significant differences in mortality rates between wind farms despite the difference in operation years (Supplementary Material Table S1 and Fig. S1).

2.4. Variables selection

A set of environmental and technical variables were selected for which have been described in previous studies as contributing to observed mortality (see Marques et al., 2020). We included factors as: ecoregion, distance to coast, turbine height and turbine rotor diameter, number of turbines per wind farm, and species richness for birds and bats. The ecoregion included the predominant habitat types present within the country. According to Dinerstein et al. (2017), ecoregions encompass the genuine ecological realms present in a given territory and are also indicating of the health state of the ecosystems and biodiversity within them. In the case of Chile, due to the physiognomy of the country, ecoregions were ordered in a latitudinal gradient. The ecoregion data was obtained from Dinerstein et al. (2017), and the corresponding category for each wind farm was acquired by intersecting the georeferenced points of wind farm locations and ecoregion polygons by using the "st_intersection" function of the "sf" package (Pebesma, 2018). Hence, we obtain a unique ecoregion category for each wind farm. Similarly, we selected the distance to the coast as an indicator of species abundance and richness due to the particular country characteristic (i.e., it expands in length more than in width). The distance to the coast was estimated by measuring the distance from each wind farm using the "st_distance" function within the "sf" package (Pebesma, 2018). Data on bird and bat species richness in Chile were obtained from the Biodiversity Mapping platform (https://biodiversitymapping.org/; Jenkins et al., 2013), provided in raster format at a 10 km resolution. To estimate species richness around each wind farm, we intersected the raster data with a 20 km buffer surrounding each project, extracting the corresponding bird and bat richness values. Regarding technical characteristics, we selected turbine height and rotor diameter key factors determining collision risk for birds and bats (see e.g. Santos et al., 2022). Finally, we included the presence of mitigation measures as a factor with three levels: active, passive and both, due to their potential impact on collision reduction (e.g., McClure et al., 2021; Ferrer et al., 2022).

2.5. Statistical analyses

First, we computed the Kruskal-Wallis test to assess the effect of seasonality on bird and bat fatalities separately. Secondly, we investigated the factors contributing to variation in observed mortality rates among wind farms. To do this, we conducted separate generalized linear models (GLM) with negative binomial distribution for birds and bats, using mortality rate as the response variable. These analyses were performed independently for each group, recognizing that birds and bats may respond differently to collision-related factors (Thaxter et al., 2017). The ecoregion type and presence of mitigation measures were included as factors, while distance to coast, turbine height, turbine rotor diameter, number of turbines, and presence of mitigation measures were included as covariates in the model. We included the presence of mitigation measures as a two-level factor variable in the models. To avoid issues with quasi-separation due to sparse data, ecoregion categories with fewer than three observations were combined into aggregated groups using a lumping approach, ensuring more robust and stable model estimates. The latter covariates were mean-centered by using the scale function. Sampling effort was standardized and included in the models as offset by calculating an offset term as the product of the number of turbines and weights corresponding to sampling frequency categories, thereby explicitly accounting for variable monitoring intensities across datasets. Quantitative covariates were scaled using scale function due to the differences in magnitude orders. Spearman correlation matrices of predictor variables for bats and birds were computed and visualized (see Supplementary Material Fig. S2) to diagnose potential multicollinearity problems. Due to the detection of a high correlation ($|r| > 0.7$) between rotor diameter and turbine height, rotor diameter was excluded from subsequent analyses to mitigate variance inflation and improve model interpretability.

To assess the effect of the different monitoring methodologies and mitigation actions, we tested differences in observed mortality rates between wind farms with different sampling monitoring schedules and mitigation measures (active or passive) by running two independent Kruskal-Wallis tests for non-normal data.

Models were compared using the Akaike Information Criterion corrected for small sample sizes (AICc; Burnham and Anderson, 2002). The model with the lowest AICc value was considered the best fit for our data, but models with a difference of ΔAICc <2 were also considered alternatives (Burnham and Anderson, 2002). In case that two or more models showed ΔAICc <2 we computed model average by using “mod.avg” function from the “MuMIn” package (Barton, 2022). Variable importance (from 0 to 1) was calculated to show how much each predictor contributed to the models based on their overall support across all averaged models.

To ensure the best models were reliable, we checked if the spread of the data around the model predictions (homogeneity of variance) and the distribution of the errors (normality of residuals) met the necessary statistical assumptions. We used the "ggResidpanel" package in R for these checks (Goode and Rey, 2019). All tests were two-tailed, and the statistical significance was set at $\alpha = 0.05$. All results were shown as mean ± standard deviation. Spatial and statistical analyses were done in R version 4.2.2 (R Core Team, 2022).

3. Results

3.1. Mortality estimation and driving factors

Based on data collected over 15-year period (2009–2023), consultants recorded a total of 1218 bird casualties belonging to 80 different species, and 1250 bats from 6 species (Fig. 2A). Among the birds, Thraupidae (Passeriformes) were the most impacted family, followed by Columbidae, Charadridae and Accipitridae (Fig. 2B and D). The most frequent species were the grassland yellow finch (*Sicalis luteola*) and the southern lapwing (*Vanellus chilensis*), with 12 % and 10 % of the total bird fatalities recorded, respectively. As for the bats, Molossidae was the most impacted family (Fig. 2C), being the Brazilian free-tailed bat (*Tadarida brasiliensis*), the species exhibiting the largest number of casualties in turbines (57.8 %, Fig. 2E). The only threatened species recorded was the Andean condor, with nine casualties in three wind farms, all located in the north-central Chile.

The observed mortality rate for bats did not vary between operational years of wind farms (Supporting information Table S1; Fig. S1A). Similarly, no clear temporal trend was observed in bird mortality, except for a slight increase during the fifth and sixth years (Supporting information Table S1; Fig. S1B). However, these differences were minimal considering the full dataset.

The mean observed mortality rate at wind farms was 0.29 ± 0.68 fatalities per turbine per year (range = 0–2.8), with the bats the group showing higher mortality rates (0.33 ± 0.79) compared to birds (0.26 ± 0.56). Bat mortality between seasons showed a significant seasonal variation ($\chi^2 = 10.18$, df = 3, $p = 0.017$) with spring and autumn exhibiting the highest mortality peaks. In contrast, bird mortality remained similar across seasons ($\chi^2 = 3.39$, df = 3, $p = 0.335$) (Fig. 3).

The best model for observed mortality rate for birds included distance to coast, ecoregion type, and turbine height, and explained 24.3 % of the variability in our data (Tables 1 and 2). In the case of the bat mortality rate, the best model included ecoregion type and distance to coast, and explained 26.6 % of the variability in our data (Tables 1 and 2). Observed mortality rates showed a positive response to distance to the coast in both birds and bats (Fig. 4A and D). In birds, mortality rates were also positively associated with turbine height and were higher in Chilean Matorral and Valdivian temperate forests compared to other ecoregions (Fig. 4B and C). In the case of bats, mortality rates were higher in the Chilean scrubland, followed by the Valdivian forest, Atacama Desert, and Magellanic subpolar forest (Fig. 4B). Turbine height was included in the bird model but not in the bat model. Neither model found significant effects of mitigation measures (bird model did not include this variable; bat model: $p = 0.369$).

3.2. Mortality monitoring procedures

Of the 36 post-construction monitoring surveys reviewed, 62 % did not report the sampling schedule used to monitor wildlife mortality. In 23.24 % of the wind farms, mortality samplings were carried out continuously, during all the operational phase. Mortality sampling schedules showed significant differences between wind farms (Kruskal-Wallis $\chi^2 = 25.28$, df = 7, p = <0.001) on the commitments of each project, ranged from daily to half-yearly sampling periods (Fig. 5A). The most common was quarterly samplings (13 %), followed by daily surveys (8.51 %) and surveys carried out on a bimonthly and fortnightly basis (6.38 % each of them). Wind farms performing sampling daily showed the highest mortality rates, while for the rest of the sampling schedules the mortality rate was similar (Fig. 5B). Each mortality monitoring campaign generally took one week to complete (100 % of the monitoring), although this duration varied depending on the number of wind turbines and the number of surveyors involved (2–4). The search methodology was common to all projects assessed and included systematic transects beneath each wind turbine, extending up to twice the length of the turbine blade. No trained dogs were used in any case.

Although all reviewed post-construction monitoring projects included systematic carcass searches beneath wind turbines, only 52 % of the facilities (26 out of 47) reported bird and bat mortality data throughout the operational phase. Carcass persistence trials were conducted in just 6.38 % of the projects. However, we found no evidence that trials were performed to estimate searcher efficiency in any of the

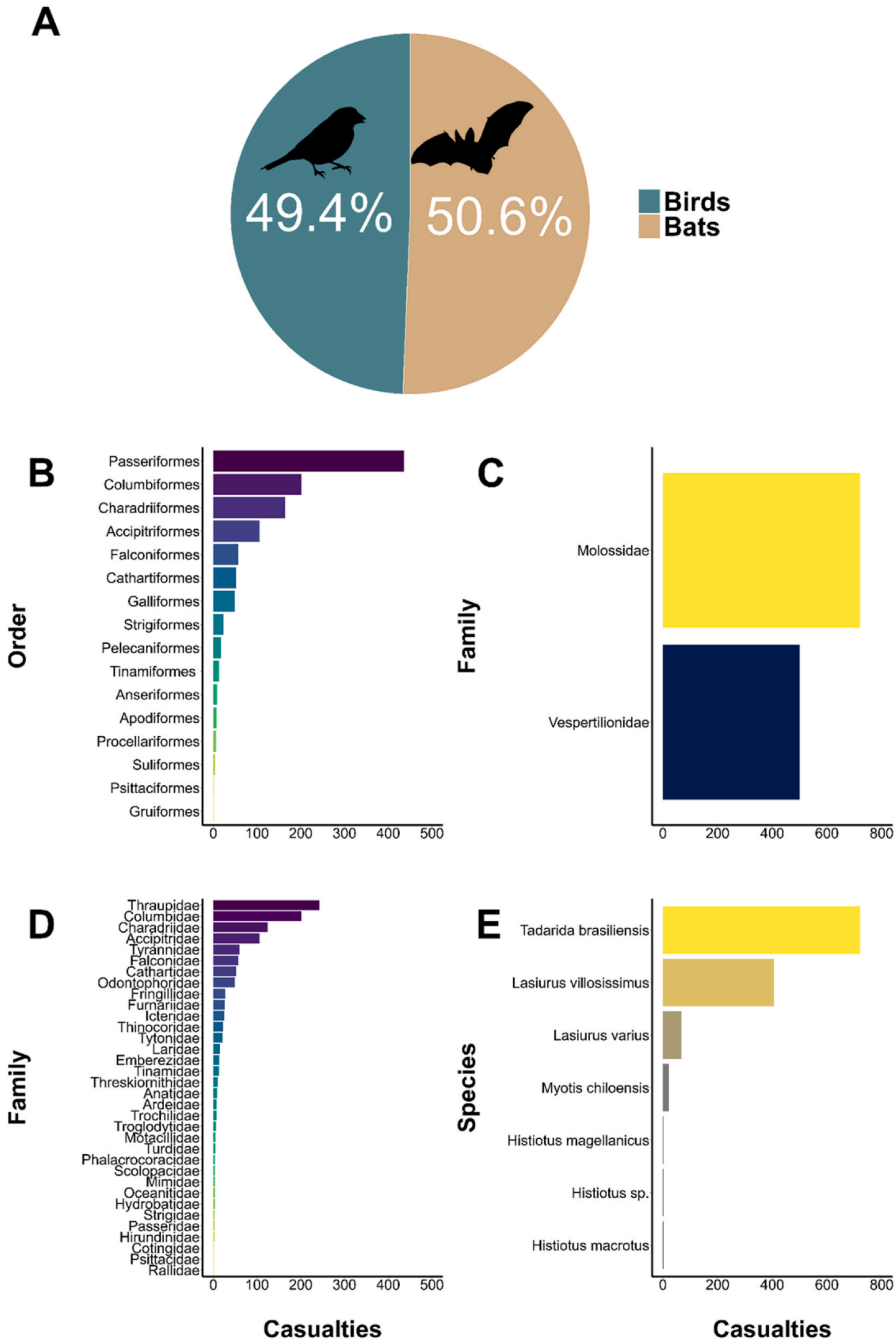

Fig. 2. Wildlife mortality at wind farms in Chile collected over 15-year period. Panel A shows the percentage of bird and bat casualties. Panels B and D display the total number of bird casualties by order and family, respectively. Panels C and E present the total number of bat casualties by family and species. Note that bat casualties are not shown by order, as all recorded species belong to a single order (*Chiroptera*). Silhouettes were obtained from PhyloPic (https://www.phylopic.org/).

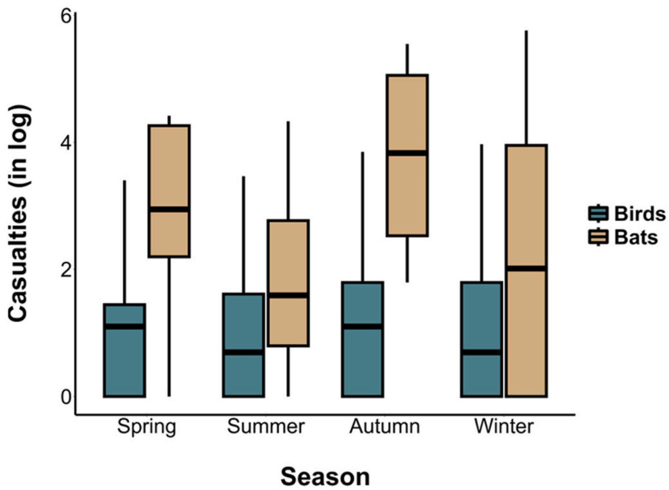

Fig. 3. Seasonal variation in the total number of observed bird and bat casualties at wind farms in Chile.

projects. In the few projects that conducted persistence trials, most bird carcasses used were domestic chickens (*Gallus gallus*), with some also using captive-bred ducks and feral pigeons (*Columba livia var. domestica*). None of the studies used carcasses of wild birds. Furthermore, there was no indication that the results of these persistence trials were applied to calculate corrected mortality estimates.

3.3. Mitigation measures

44.68 % of wind farms (n = 21) had no mitigation measures. From the rest of wind farms with mitigations measures 12.77 % (n = 6) had one mitigation measure and 42.55 % had two or more mitigation measures. A total of 8 different mitigation measures were recorded, 9.8 % active and 90.2 % passive (Fig. 5C). Active measures included stopping on demand of turbines (4.25 %) and carcass removal (6.38 %), while passive measures were mainly associated with increasing turbine visibility like blades or towers painted with colored patterns or reflective paint, perch deterrents, ultrasonic deterrents for bats and light deterrents that displace or avoid attracting birds. The color bands on the blades was the most popular measure (33 % of wind farms), followed by STROBE red lights (13.7 %) and anti-reflective painting (12 %, Fig. 5C).

The mitigation measures implemented in Chilean wind farms showed substantial variation in associated bird mortality rates. Stopping on demand exhibited the lowest and least variable mortality values (Fig. 5D). Carcass removal and ultrasound deterrents also presented relatively low median mortality, with slightly greater dispersion. Measures such as anti-reflective painting, STROBE red lights, and dissuasive or diversion devices showed intermediate mortality levels (Fig. 5D). In contrast, white-flashing beacons and colour bands displayed higher and more variable mortality rates. Anti-perching devices showed the highest overall variability despite generally low median values. No significant differences were detected between turbines with and without mitigation measures ($\chi^2 = 4.19$, df = 8, p = 0.84). Considering only wind farms that implemented mitigation, those applying only active measures showed higher mean mortality rates (0.60 ± 1.02 fatalities per turbine per year) compared with those using only passive measures (0.24 ± 0.54 fatalities per turbine per year), although this difference was not statistically significant due to high variability among wind farms and the limited number of sites with active measures ($\chi^2 = 1.55$, df = 1, p = 0.21).

Table 1
Generalized linear model with negative binomial distribution (GLM) selection explaining bird and bat mortality rates (individuals per turbine per year). Note that only models within <2 AICc were shown together with the Null model. df (degrees of freedom), logLik (log-likelihood), AICc (corrected Akaike Information Criterion), ΔAICc (difference in AICc relative to the best model), and weight (Akaike weight, indicating the relative likelihood of each model).

Group	Model	df	logLik	AICc	ΔAICc	weight
Birds	Distance to coast + Ecorregion	6	−21.83	57.76	0	0.38
	Turbine Height	3	−26.5	59.56	1.81	0.15
	Distance to coast + Ecorregion + Mitigation Measures	7	−21.81	60.48	2.73	0.1
	Turbine Height + Distance to coast + Ecorregion	7	−21.83	60.52	2.77	0.09
	Turbine Height + Distance to coast	4	−26.23	61.41	3.66	0.06
	Turbine Height + Mitigation Measures	4	−26.26	61.47	3.71	0.06
	Distance to coast	3	−27.69	61.94	4.18	0.05
	Null	2	−29.52	63.32	5.56	0.02
	Turbine Height + Distance to coast + Ecorregion + Mitigation Measures	8	−21.81	63.4	5.65	0.02
	Turbine Height + Distance to coast + Mitigation Measures	5	−26.01	63.48	5.73	0.02
	Distance to coast + Mitigation Measures	4	−27.69	64.33	6.58	0.01
	Turbine Height + Ecorregion	6	−25.18	64.46	6.7	0.01
	Mitigation Measures	3	−29.48	65.53	7.77	0.01
	Turbine Height + Ecorregion + Mitigation Measures	7	−25.06	66.99	9.24	0
	Ecorregion	5	−27.88	67.21	9.46	0
	Ecorregion + Mitigation Measures	6	−27.85	69.8	12.04	0
Bats	Distance to coast + Ecorregion	6	−22.47	59.04	0	0.56
	Distance to coast + Ecorregion + Mitigation Measures	7	−22.03	60.93	1.9	0.22
	Turbine Height + Distance to coast + Ecorregion	7	−22.47	61.81	2.77	0.14
	Turbine Height + Distance to coast + Ecorregion + Mitigation Measures	8	−22.02	63.83	4.8	0.05
	Turbine Height	3	−30.22	66.99	7.96	0.01
	Turbine Height + Mitigation Measures	4	−29.3	67.56	8.52	0.01
	Turbine Height + Distance to coast	4	−29.69	68.32	9.29	0.01
	Turbine Height + Ecorregion	6	−27.38	68.86	9.82	0
	Turbine Height + Distance to coast + Mitigation Measures	5	−28.83	69.13	10.1	0
	Distance to coast	3	−31.37	69.3	10.27	0
	Turbine Height + Ecorregion + Mitigation Measures	7	−26.88	70.64	11.6	0
	Distance to coast + Mitigation Measures	4	−31.24	71.44	12.4	0
	Null	2	−34.58	73.43	14.4	0
	Ecorregion	5	−31.12	73.71	14.68	0
	Mitigation Measures	3	−34.57	75.69	16.65	0
	Ecorregion + Mitigation Measures	6	−31.1	76.31	17.27	0

Table 2
Result for the averaged GLM models with negative binomial distribution for the mortality rate. Note that averaged coefficients from the best models are shown. The level of reference for Ecoregion and Mitigation measures was the Atacama Desert and No, respectively. Significant values are highlighted in bold. Abbreviations: coeff. Coefficients; SE = Standard Error, Adj SE = Adjusted Standard Error, Z = z-value, P = significance, Import = variable importance (0–1).

Group	Term	coeff	SE	Adj SE	Z	P	Import
Bird	(Intercept)	−5.11	3.04	3.08	1.658	0.097	
	Distance to coast	1.05	0.78	0.79	1.327	**0.003**	0.71
	Ecorreg: Chilean Matorral	3.57	3.06	3.10	1.152	**0.045**	0.71
	Ecorreg: Valdivian temperate forests	3.44	3.08	3.12	1.102	0.067	
	Ecorreg: Other	−20.81	5.6 e^7	5.8 e^7	0.00	0.998	
	Turbine Height	0.30	0.51	0.54	0.557	**0.032**	0.29
Bat	(Intercept)	−35.51	1.9 e^7	2.03 e^7	0.00	1.000	
	Distance to coast	1.87	0.52	0.53	3.524	**<0.001**	1.00
	Ecorreg: Chilean Matorral	34.23	2.0 e^7	2.0 e^7	1.685	1.000	1.00
	Ecorreg: Valdivian temperate forests	33.25	1.9 e^7	2.0 e^7	1.636	1.000	
	Ecorreg: Other	4.683	6.0 e^7	6.2 e^7	7.543	1.000	
	Mitigation Measures Yes	−0.142	0.369	0.376	0.377	0.369	0.28

4. Discussion

4.1. Operational wildlife fatalities sampling

Despite the Chilean government providing a guide with the methodology for survey programs at wind farm projects (SAG, 2015; GIZ & Myotis Chile, 2025), there remains a strong need to standardize monitoring frequency and survey protocols. Although the official guidance specifies procedures for carcass searching, it does not ensure consistent application across projects. Monitoring is proposed to last five years or the entire operational phase, yet in practice, survey frequency and extent vary widely (from daily to half-yearly), with most projects (70 %) conducting searches only once per season. Our results clearly show that daily surveys detect substantially more carcasses than less frequent schedules, underscoring that the current non-standardized approach systematically underestimates mortality. Consequently, this variation in effort not only hampers accurate impact estimation but can also create the misleading impression that projects have limited effects on wildlife, since the absence of detected carcasses does not necessarily indicate the absence of fatalities (Huso et al., 2015).

An essential part of environmental impact monitoring programs is to systematically assess the wildlife affected and provide corrected estimates of mortality to be able to compare between infrastructures (Conkling et al., 2020, 2022; Martins et al., 2023). In this sense, almost half of the projects reviewed did not provide this information. Furthermore, it is necessary to stress the obligation to implement these studies to determine the impact and assess whether the mitigation measures are effective. Correction for detection bias (which is influenced by observer experience, percentage of areas searched and habitat visibility) and scavenging bias, are not widely employed. For example, although in all wind farms, the searches for bird and bat carcasses covered all wind turbines, the search area in all projects was associated with the clear zone used for the turbine installation or an extension of the blade radius with a maximum distance of 100 m around the turbine. However, the search area in all projects was limited to the cleared area used for installing the turbine or to an extension of the blade radius. This does not contemplate that detection probability can often be increased by improving searcher efficiency, increasing the area searched under each turbine, searching more frequently or using well-trained dogs (Huso et al., 2015; Barrientos et al., 2018). As we have noted, in terms of search bias and scavenging bias, habitat characteristics and landscape configuration should be considered during monitoring design to obtain more precise estimates of mortality rates (Bernardino et al., 2013; Kitano et al., 2023). Furthermore, the percentage of unsearched area was not included in any of the reviewed documents. This information is essential for the reliable estimation of the parameters of the model (Huso and Dalthorp, 2014).

Finally, to assessing carcass persistence, it is crucial to acknowledge interspecific variation in removal rates. For instance, larger avian species, particularly raptors, tend to exhibit lower scavenging rates and thus persist longer in the field (Wilson et al., 2022). To obtain more accurate and region-specific detection probability estimates, future research should incorporate disappearance trials using a diverse range of wild bird and bat species, with a specific focus on including raptors that may be subject to differential scavenging pressures. This approach will enhance the ecological realism of mortality assessments at wind energy facilities.

4.2. Species mortality

Our findings showed a wide range of bird and bat species affected by wind turbines in Chile, including threatened species such as the Andean condor. Although the number of birds and bat fatalities was similar, the species diversity and community composition were very different. While a total of 80 bird species were reported dead in wind farms, in contrast with only six species of bats, this represented up *c.a.* 46 % of the total species reported for this country while that bird community represents only *c.a.* 19 % of the total bird species recorded to Chile. In spite of this, only two bat species account for the 92 % of bats reported: Brazilian free-tailed bat and South American hoary bat (*Lasiurus villosissimus)*. These species are the most abundant and generalist in Chile, they are distributed in a wide variety of environments and are frequently found in cities and urban environments (Iriarte, 2008; Rodríguez-San Pedro et al., 2016). Otherwise, other affected species, such as *Myotis* sp. and *Histiotus* sp., which have a wide distribution, overlapping with the location of wind farms, although by preferring coastal environments, close to bodies of water or wooded and shrubby environments (Iriarte, 2008), could reduce the number of accidents. These ecological and behavioral differences, like home range size and migration of Brazilian free-tailed bat and South American hoary bat, could explain the highest number of carcasses reported over the other widespread species (Lloyd et al., 2023).

According to other studies in North America (Erickson et al., 2014; Zimmerling et al., 2013) or in Europe (Morant et al., 2025) passerine birds were the most common species affected by wind turbines, accounting for up to 36 % of all bird carcasses recorded in Chile. Shorebirds and raptors were the following groups more impacted, each comprising approximately 33 % of recorded bird fatalities. Despite wind farms being located in northern coastal areas of Chile, the presence of migratory shorebirds was minimal. Instead, the most frequently reported shorebird was the southern lapwing, a resident species whose habits are more associated with grassland and crop areas than coastal or wetland areas, and which is often found near human settlements. While previous studies have identified raptors and vultures as particularly sensitive to wind energy infrastructure (e.g., Drewitt and Langston, 2006; Thaxter et al., 2017; Marques et al., 2020), we found a surprisingly low number of carcasses for the two most abundant vulture species in the region, the turkey vulture (*Cathartes aura*) and the black vulture

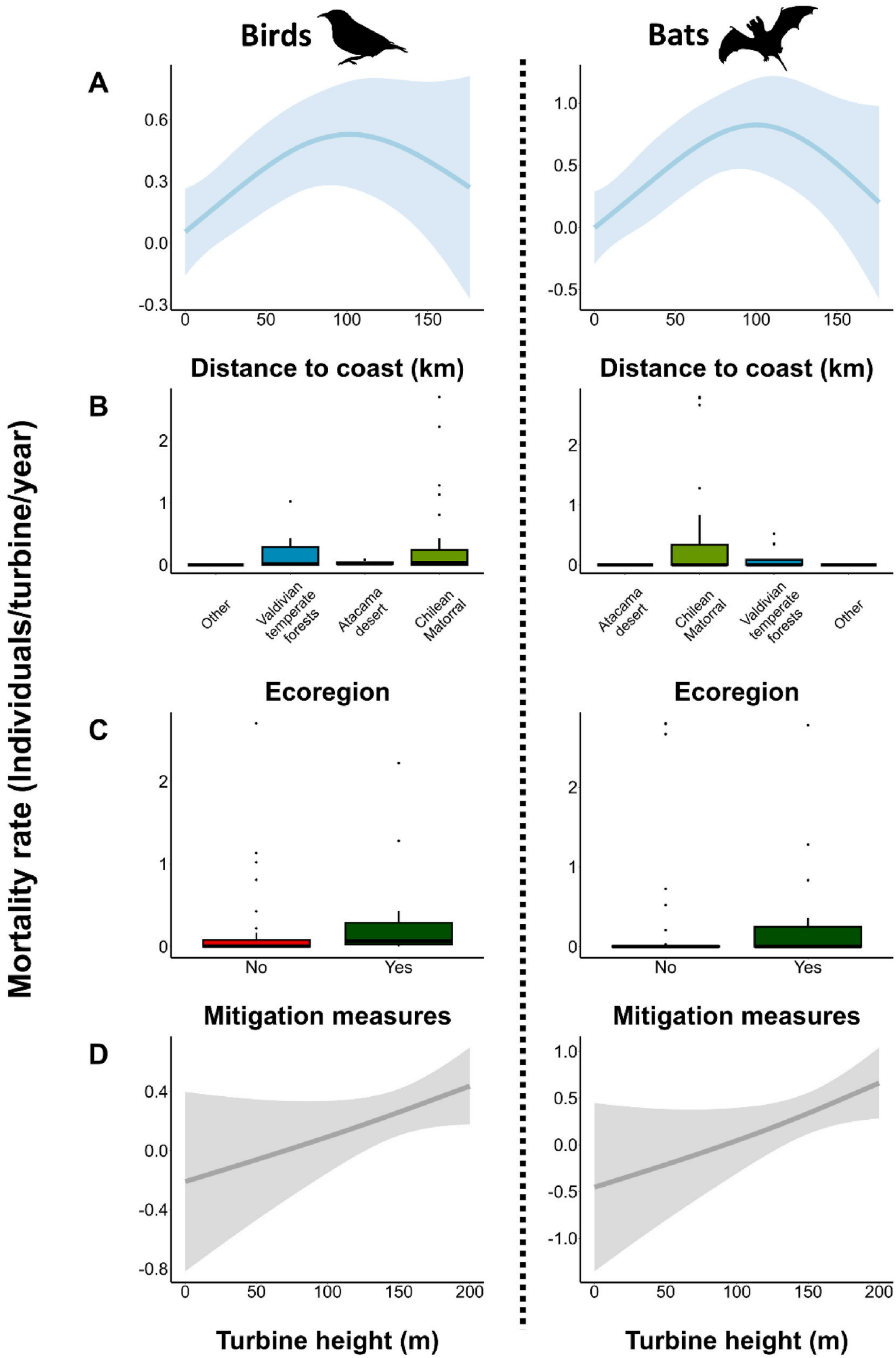

Fig. 4. Effects of the main variables included in the best generalized linear model (GLM) explaining bird and bat mortality rates (individuals per turbine per year) at wind farms in Chile.

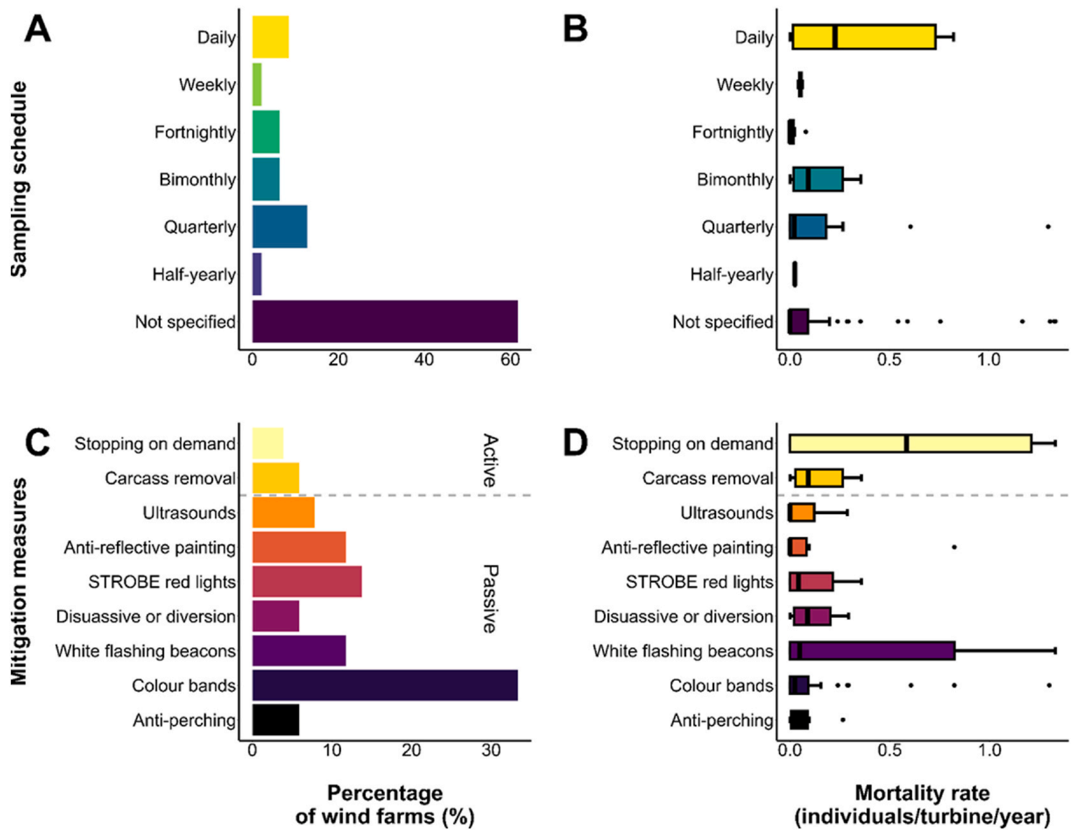

Fig. 5. Mortality monitoring schedules and mitigation measures implemented in Chilean wind farms. Panel A shows the percentage of wind farms reporting the type of sampling schedule used to estimate mortality rates, while Panel B presents the corresponding mortality rates for each sampling type. Panels C and D display the percentage of wind farms implementing mitigation measures and the associated mortality rates, categorized into active and passive approaches. The category "Dissuasive or diversion" refers to cases where reports did not specify the type of passive method used.

(*Coragyps atratus*). In contrast, the variable hawk (*Geranoaetus polyosoma*) was the most commonly recorded raptor. Notably, the Andean condor, one of the rarest species and classified as Vulnerable by the IUCN, was recorded on three wind farms. Although these facilities are located outside the species' known breeding range, winter movements from the Andes to the Chilean coast have been documented, especially among juveniles and immature individuals (Lambertucci et al., 2018). These observations suggest that the Andean condor could be a highly sensitive to wind turbine collisions. This is an important concern for the conservation of the species, as even low levels of mortality can have significant effects on population dynamics (Carrete et al., 2009; Duriez et al., 2023), and these young and dispersing individuals may play a key role in the expansion of the species. Therefore, due to their wide-ranging movements (e.g. Lambertucci et al., 2014; Guido et al., 2023), the development of a transboundary conservation strategy is essential. Such a framework would help identify priority areas for renewable energy development while minimizing biodiversity impacts across South America. (Lambertucci et al., 2014). Some initiatives in this direction have already been carried out (e.g. Perrig et al., 2020).

It is important to acknowledge that the mortality data presented in this study are uncorrected and do not account for potential sources of bias such as carcass detection rates, scavenger removal, or search effort variability (Bernardino et al., 2013). As a result, our comparisons between different regions of Chile rely on the assumption that such biases are consistent across sites. While this allows for relative assessments within the country, extrapolating these findings to other geographic areas should be approached with caution. Differences in monitoring protocols, environmental conditions, and species composition may significantly influence mortality patterns elsewhere (Ravache et al., 2024).

These species mortality patterns must be interpreted in the context of Chile's unique ecological and geographic characteristics. Chile lies along the Pacific Flyway, and several northern and central coastal wetlands host migratory shorebirds from the Northern Hemisphere (Senner et al., 2017). However, despite the coastal location of many wind farms, the presence of migratory species in carcass records was low. This may be due to habitat use patterns that do not overlap with turbine areas, or flight behaviors (i.e. flying height during migration) that reduce collision risk. The predominance of resident and generalist species suggests that collision risk is more closely linked to local ecology and prolonged exposure than to large-scale migratory dynamics. Incorporating these landscape and biogeographic elements is essential to understanding regional variation in collision risk and highlights the need to tailor mitigation strategies to local ecological conditions.

4.3. Seasonality and driving factors

Our study found no evidence of association between the years of wind farm operation and mortality rates. Therefore, variations in mortality rates between years within each farm would probably be related to variations in the exposure of sensitive birds. Also, the absence of a general pattern of declining mortality over the years of wind farm operation could support the idea of a lack of habituation of resident birds to the presence of the wind farm (Stewart et al., 2007; Sassi et al., 2024) or a long-term decline of local populations through continued mortality.

While bats showed two seasonal peaks of mortality concentrated in spring and autumn, bird mortality rates showed almost equal fatalities across all seasons, with a slight increase also in winter and spring. The seasonal pattern of bat fatalities may be due to increased activity and home ranges of bats during the reproductive season and the previous months to the restricted season. These changes in home ranges and increase in time activity have been observed in desert bats in response to seasonally harsh conditions and resource scarcity (Connena et al., 2009). Given these clear seasonal peaks in bat mortality, we strongly recommend that future monitoring programs implement mandatory high-frequency surveys (e.g., every 3 days) during spring and autumn migration windows to adequately capture mortality events and assess population-level impacts. The limited knowledge regarding the basic ecology, such as movement patterns, population abundance, and habitat use, of most bat species present in Chile represents a significant limitation to understanding their collision risk with wind turbines.

In the case of birds, the absence of seasonal variations seems to indicate that there is no change due to the seasonal movements of the birds, and therefore, could suggest that resident birds are the most affected by wind turbines, as our data indicate. The importance of migration as a risk factor for birds has been noted previously (Erickson et al., 2014; Lloyd et al., 2023), but we do not find differences between seasons, even when the largest facilities in Chile are in coastal areas of the Atacama Desert where exist a great abundance of shorebirds and migrant birds associated to Humboldt Current exist (Weichler et al., 2004) and therefore, fatalities should be greater during the migration season in spring and autumn.

The only shared covariate for bird and bat models was the distance to the coast. The observed mortality rates increased with the distance to the coast, with distance showing a significant positive effect in both taxa (birds: $\beta = 1.48$, $p = 0.003$; bats: $\beta = 1.87$, $p < 0.001$). This pattern could be related to the Andean foothills, an area characterized by high abundance and diversity of wildlife (Ormazabal, 1993). Another potential explanation is that prey availability might be higher within this distance range due to stable productivity throughout the year (Zambrano et al., 2018).

Habitat characteristics also proved to be relevant in explaining mortality rates in both taxa. Ecoregion was a significant predictor in both bird and bat models, with Chilean Matorral showing significantly higher mortality rates compared to reference ecoregions (birds: $\beta = 5.02$, $p = 0.045$; bats: model estimates showed elevated coefficients though with large standard errors indicating numerical instability). However, these ecoregion-based mortality differences must be interpreted with extreme caution due to substantial detectability bias inherent in our uncorrected data. Comparing mortality rates between the open, arid landscapes of the Atacama Desert and the dense vegetation of the Valdivian temperate forests is methodologically risky, as the observed "biological" differences may largely reflect "visibility" differences rather than true mortality patterns. Carcass detection probability almost certainly varies dramatically across these vastly different habitats, what appears as lower mortality in densely vegetated southern forests may simply reflect our inability to find carcasses rather than actual lower collision rates. The complete absence of searcher efficiency trials and the minimal implementation of carcass persistence trials (only 6.38 % of projects) in our dataset means we cannot disentangle genuine ecological patterns from detection artifacts. Therefore, absolute mortality comparisons across ecoregions should be cautiously used for conservation prioritization or regulatory decisions without first conducting standardized detectability studies. The significance of ecoregion in observed mortality rates could be attributed to the highest species diversity and abundance of bats in the two central ecoregions, namely the Chilean scrubland and Valdivian forests (CONAMA, 2008), though this interpretation remains confounded by the detection bias issue. Local field studies are therefore necessary to determine fine-scale species' habitat use, abundance, and behavioral responses to infrastructure in these areas (Rebolo-Ifrán et al., 2025). We strongly advocate for the mandatory use of trained detection dogs in high-vegetation ecoregions (particularly the Valdivian and Magellanic forests) to address the significant underestimation bias in these environments. Detection dogs have been shown to dramatically improve carcass detection rates in dense vegetation (Paula et al., 2011; Matthäus and Mehling, 2020), and their use should be a standard requirement for all wind farms operating in forested habitats.

In the case of birds, turbine height was retained in the best model and showed a significant positive effect on mortality rates ($\beta = 1.04$, $p = 0.032$), indicating that taller turbines were associated with higher bird fatalities. This finding aligns with previous studies showing increased collision risk with turbine height (Barclay et al., 2007). However, turbine height was not retained in the bat model, and other technical factors related to turbine design (such as rotor diameter, which was removed due to high multicollinearity with turbine height) were not significant predictors.

The presence of mitigation measures was not retained in the bird model during model selection, and showed no significant effect in the bat model ($\beta = -0.51$, $p = 0.369$). The absence of mitigation effects could be attributed to various contrasting reasons. On the one hand, there is considerable debate regarding the effectiveness of many commonly installed mitigation measures on wind farms. In many instances, their efficacy appears to be minimal (e.g., Marques et al., 2014), supporting the idea that the model does not select the presence of mitigation measures as a relevant factor. Furthermore, even when these measures are effective, they are often only mandated in cases where prior environmental assessments have identified a risk. Consequently, the observed mortality in wind farms without measures (i.e., low risk) might be comparable to those in high-risk farms where measures have already been installed. Beyond these statistical explanations, the lack of mitigation efficacy likely reflects deeper biological and design issues with the predominantly passive measures (90.2 % of all mitigation) implemented in Chilean wind farms. Static visual deterrents such as colored blade markings or reflective paint may lose effectiveness as birds habituate to these unchanging stimuli over time, essentially learning to ignore them as non-threatening features of their environment. More critically, these visual-based mitigation measures may not be physiologically tuned to the sensory systems and flight behaviors of key Chilean species at highest risk. For example, the Andean condor relies heavily on soaring flight and thermal detection, while the Variable hawk exhibits rapid pursuit hunting behavior—neither species may respond predictably to static visual cues designed based on studies of European or North American species. The absence of species-specific testing for Chilean taxa represents a fundamental gap in mitigation design. Future mitigation efforts should prioritize active measures (such as curtailment-on-demand or acoustic deterrents) that can adapt to real-time conditions, and should undergo rigorous efficacy testing with target species before widespread deployment.

Finally, it should be noted that both bird and bat models showed only moderate explanatory power, explaining 24.3 % and 26.6 % of the variance, respectively. This limited performance likely reflects the heterogeneous quality and reporting detail of available monitoring data, the lack of standardized sampling across projects (Conkling et al., 2021), and the potential influence of unmeasured factors such as weather conditions, topography, or species-specific behavior (Barrios and Rodriguez, 2004; Marques et al., 2014). Acknowledging these limitations is essential for correctly interpreting our results and highlights the need for more consistent data collection and standardized protocols in future assessments.

4.4. Mitigation measures

Given the well-documented environmental impacts of wind farms, project developers are often required to implement mitigation measures to reduce wildlife mortality. It is therefore notable that almost half of the reviewed projects did not apply any mitigation actions. This limited implementation may reflect the current state of scientific evidence: for many proposed measures, effectiveness remains uncertain, varies among species or contexts, or has not been rigorously tested (Conkling et al., 2022; Marques et al., 2014). In addition, some measures with more robust support, such as on-demand shutdowns operated by personnel, entail substantial operational costs that can constrain their use (Ferrer et al., 2022). These uncertainties and limitations may also explain why some developers adopt several mitigation measures simultaneously, as observed in 21 % of projects.

The absence of a significant relationship between mitigation measures and recorded mortality in our study must be interpreted in light of the constraints of the available data. The observational nature of the monitoring programs, the lack of pre-implementation baseline data, and the frequent combination of multiple measures at the same site prevented us from evaluating the effectiveness of individual actions in a standardized manner. Moreover, few wind farms implemented a single mitigation measure, reducing the number of independent replicates available for comparison. It is important to note that mitigation measures are generally imposed based on collision risk, usually being required in locations with a higher predicted or observed risk, often due to the presence of sensitive species. As a result, wind farms with mitigation actions may inherently display higher mortality values, which could also explain why projects using active measures showed higher mortality than those using passive ones. This pattern is further influenced by the strong imbalance in sample sizes, as passive measures remain more common in Chile, as in other regions (e.g., Spain), largely due to the costs associated with personnel-based shutdown procedures (Ferrer et al., 2022).

The effectiveness of mitigation measures is also likely to vary among species due to differences in behavioral responses. Acoustic deterrents may be effective for some bat species but not others, while visual deterrents often show inconsistent results among bird groups and regions. Additionally, substantial variation in site characteristics and the absence of standardized monitoring protocols further limit strong inference about mitigation performance (see also Marques et al., 2014). To improve understanding of causal mechanisms and support evidence-based management, future research would benefit from rigorous experimental or quasi-experimental designs, such as before–after–control–impact (BACI) frameworks, that can more clearly isolate the effects of mitigation measures on species-specific mortality patterns (Katzner et al., 2025).

5. Recommendations and future perspectives

Our work indicates that wind energy development in Chile could impact some groups of birds, especially bats. These two groups are also those that, given their size, tend to show greater detectability biases and disappearance due to scavenger consumption, so the corrected estimates of mortality may be significant.

The loss of these specimens negatively impacts local or regional population dynamics and the reduction of ecosystem services they provide (i.e., insect control; Ellerbrok et al., 2022). Our work points out that central and medium south Chile could be the most sensitive place to develop this renewable energy due to the concentration of diversity and abundance of species. By contrast, in other areas, there could be less impact (in terms of the number of deaths) but a strong negative effect on more endangered or exclusive species (e.g. in the high Andes or Tierra del Fuego). Constructing spatially explicit risk maps to inform of potential risk areas, particularly for sensitive and endangered species (e.g. Morant et al., 2024), is advisable. It should also be noted that some of the most sensitive species are highly mobile and therefore this planning should be oriented on a transboundary scale (Lambertucci et al., 2014). Unfortunately, the scarcity of fine-grained occurrence and abundance data for many species on the South American continent works against the possibility of large-scale planning, as the exception of an emblematic species as the Andean Condor (see Perrig et al., 2020). The lack of specific regulations, both at the local level and between countries, also works against the possibility of large-scale planning, which may generate additional risk for species with transboundary populations.

Adequate monitoring of any indicator is essential to ensure reliable environmental assessment. It is crucial to standardize sampling effort and survey frequency in order to meet the requirements of the established indicators. Insufficient monitoring can lead to underestimation of the actual impact on wildlife species affected by infrastructure projects. Monitoring protocols should account for factors such as carcass detection probability, site accessibility, and environmental variability, including vegetation type, seasonal changes, and climatic conditions. Furthermore, this process must be complemented by carcass removal experiments (to assess scavenger activity) and detectability studies, which are fundamental for accurately estimating the true mortality of species and individuals impacted by wind farms or electrical transmission projects (Ravache et al., 2024).

Finally, it is necessary to standardize methodologies to obtain comparable and corrected mortality rates. Furthermore, it is necessary to design experimental studies that provide scientific data about the effectiveness of mitigation measures, particularly for South American wildlife communities affected by wind farms.

CRediT authorship contribution statement

Francisco Santander: Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jon Morant:** Writing – review & editing, Visualization, Methodology, Investigation, Formal analysis, Conceptualization. **Juan Manuel Pérez-García:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Author declaration

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Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Francisco Santander reports a relationship with Geobitoa, an environmental consultancy that includes: employment and funding grants. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.128420.

Data availability

Data will be made available on request.

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