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Assessing Conflict Potential Between Birds and Small-Scale Fisheries in Lake Titicaca

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ABSTRACT

Globally, fisheries bycatch constitutes a threat to aquatic birds. However, while the threat is global, the threat from bycatch in inland fisheries, and from artisanal or small-scale fishing, is particularly understudied. Here, we present the first attempt to map the vulnerability of birds to fisheries bycatch, and the conflict potential between birds and fisheries, in Lake Titicaca. We use field surveys, Bayesian Joint Species Distribution Modelling and a trait-based weighting approach to prioritise conservation efforts, to map bird vulnerability to fisheries bycatch and conflict potential between birds and fisheries in Lake Titicaca. Fisheries in Lake Titicaca are exclusively smallholder, but fisheries bycatch has been listed as a major threat to biodiversity in the lake in past surveys. We found that the region with the greatest concentration of 1×1 km pixels in the uppermost quantile of vulnerability to fisheries was in the Lago Menor, Puno Bay and Ramis River Mouth regions of Lake Titicaca, but that the greatest concentration of 1×1 km pixels in the uppermost quantile of conflict occurs in the Lago Menor region. Further work to reduce fisheries bycatch should focus on regions with the greatest existing conflict and the greatest conflict potential but can only plausibly be done with the full consent and collaboration of local fishermen.

1 | Introduction

Fisheries are a major driver of declines in aquatic bird populations worldwide (Tasker 2000; Žydelis et al. 2009; Žydelis et al. 2013). An estimated 400,000 birds die as a result of fisheries bycatch on the high seas yearly, with an unknown number being caught in inland fisheries (Žydelis et al. 2013). This is without accounting for the indirect consequences of fisheries on bird populations, such as the nonlethal effects of bycatch (Wilson et al. 2014), disruptions of fishing vessels to bird breeding colonies (Burger et al. 2019) and the reduction of prey numbers for piscivorous birds (Furness 2003). Bycatch from fisheries has been singled out as a significant conservation threat for several bird species (Dias et al. 2019; Marchowski et al. 2020) and is a major cause of population declines for a number of South American species, including the Titicaca Grebe (*Rollandia microptera*) (Martinez et al. 2006), the Junin Grebe (*Podiceps taczanowskii*) (Dinesen et al. 2019) and the Humboldt Penguin (*Spheniscus humboldti*) (Majluf et al. 2002). Despite its importance, direct information on fisheries bycatch is rare (Richards et al. 2024), and basic information needed for conservationists, such as the number of individuals being caught as bycatch and the geography of bycatch, is missing even in many well monitored parts of the world (Žydelis et al. 2013; Morkūnas et al. 2022), let alone in areas which are often neglected by fisheries research, such as the tropics (Fluet-Chouinard et al. 2018).

The lack of information on fisheries bycatch from small-scale and inland fisheries has been highlighted by several reviews on the topic (Raby et al. 2011; Žydelis et al. 2013; Lewison et al. 2014; Pott and Wiedenfeld 2017). Small-scale fisheries, also referred to as artisanal fisheries, are defined as fishing 'involving fishing households (as opposed to commercial companies), using a relatively small amount of capital and energy, relatively small fishing vessels (if any), making short fishing trips, close to shore, mainly for local consumption' by the Food and Agricultural Organisation (FAO) of the UN (FAO 2005). They account for an estimated 40% of the global fisheries catch (FAO 2023), but this figure is extremely imprecise due to the lack of information on its extent and consequently on its environmental effects. Evidence from places where artisanal fisheries have been studied suggests that they can have significantly deleterious effects on local ecosystems, including through bycatch (Shester and Micheli 2011; Pott and Wiedenfeld 2017), yet almost all of these studies have focused on artisanal fisheries and bycatch in saltwater contexts. So extreme is the lack of information on fisheries bycatch from small-scale inland fisheries that a review of freshwater bycatch found that no study had attempted to estimate bycatch rate for a small-scale inland fishery anywhere in the world (Raby et al. 2011). However, since that review, at least one study has been conducted which sought to estimate the rate of fishery bycatch of an inland water (Villar et al. 2025).

Fisheries have existed in Lake Titicaca since the first humans settled in the region (Capriles et al. 2014; Erickson 2000). However, for most of the history of human settlement in the Altiplano, fishing was a marginal industry and usually was resorted to in times of increased violence or prolonged poor harvests (Miller et al. 2021; Young 1997). When the first ethnographic accounts of Lake Titicaca life were written, fishing was primarily mentioned for its conspicuous absence, save among the Uro ethnic group who live on floating islands and rely on fishing and hunting waterfowl for subsistence (Forbes 1870; La Barre 1941). What fishing was done was primarily by using nets made of totora (*Schoenoplectus californicus* subp. *tatora*) sedge and in dried totora canoes which could not venture much beyond the shallow wetland portions of the lake (La Barre 1941). This situation changed in the mid-20th century with the introduction of salmonids and pejerrey (*Odonesthes bonariensis*) to Lake Titicaca (Everett 1973; Laba 1979). The introduction of these commercially important species precipitated a fishing boom, leading many agriculturalists to start fishing. It also led to the modernisation of the fishing fleet of Lake Titicaca, including the move to monofilament gill nets (Orlove 2002; Laba 1979) which remain the principal net-type used in Lake Titicaca to this day. Since the 1960s, the commercial fisheries of Lake Titicaca have experienced a periodic boom-and-bust cycle as the invasive species get overfished and then restocked from government programmes and escapees from pisciculture (Villar 2025). The fisheries are an important part of the local economy and a major driver of cultural (Villar, Thomsen, Gutiérrez Tito, et al. 2024) and environmental (Monroy et al. 2014) change in the region.

Fisheries bycatch has been highlighted as a major conservation concern for both the endemic Titicaca Grebe (Martinez et al. 2006; Villar et al. 2023) and the critically endangered Titicaca Water Frog (*Telmatobius coleus*) (Angulo 2008). It is likely that it is also a cause of excess anthropogenic-caused mortality among other species in Lake Titicaca, such as noncommercial fish and other diving birds such as the Andean Duck (*Oxyura ferruginea*). Conflict between fishermen and birds is also a serious barrier to practical conservation action, as many fishermen dislike birds as competitors for fish and for breaking nets when they get caught as bycatch (Quispe et al. 2023; Villar, Thomsen, Paca-Condori, et al. 2024; Villar, Yanes, Gutiérrez Tito, and Gosler 2024). This study is the first to map the conflict potential between birds and the fisheries of Lake Titicaca. Its goal is to provide a map of avian-fisheries conflict and avian vulnerability to fisheries bycatch to aid policy making on where to prioritise anti-bycatch conservation efforts.

2 | Methods

2.1 | Site Description

Lake Titicaca is a large, high-altitude, freshwater lake in the Central Andes, shared between Peru and Bolivia (Wirrmann 1992, and see site description below). It is home to 135 bird species, of which 95 are nonmigratory (Pulido Capurro 2018), and about 60 can be said to be aquatic in the broadest sense (Aparicio 1957), of which 23 regularly dive (Ibid.). This previous work identifying the birds of Lake Titicaca did not attempt to map individual species' distributions across the lake and only made qualitative assessments of species frequency. It also included vagrant species in its total of species found in Lake Titicaca. The birds of Lake Titicaca include the endangered endemic Titicaca Grebe (*R. micropter*a) (Martinez et al. 2006; Villar et al. 2023; Villar, Velásquez-Noriega, Gutiérrez Tito, et al. 2024). Several bird species in Lake Titicaca, including the Titicaca Grebe, the Silvery Grebe (*Podiceps occipitalis*) and the Andean Duck (*O. ferruginea*), are piscivorous, diving birds or both, which make them particularly vulnerable to fisheries bycatch. The wetlands of Lake Titicaca are RAMSAR listed (RAMSAR 2023) in part due to their role as bird habitats (Sarmiento 1998), and the region has been named an Important Bird Area by BirdLife International (BirdLife International 2024a, 2024b). In addition to its ornithological diversity, Lake Titicaca is a continent-wide important site for both ichthyological (Albert et al. 2011) and invertebrate (Strong et al. 2008) biodiversity. Despite its biological importance, Lake Titicaca has suffered neglect from researchers and from funders. Lake Titicaca suffers from a variety of anthropogenic pressures, including pollution from untreated human wastewater and mining runoff (Archundia et al. 2017; Cáceres Choque et al. 2013), invasive species (Hall and Mills 2000) and climate change (Quenta-Herrera et al. 2022). The humid puno ecoregion that includes Lake Titicaca has been identified as an under-represented region for conservation in Bolivia (López and Zambrana-Torrelio 2006), and the one protected area in the

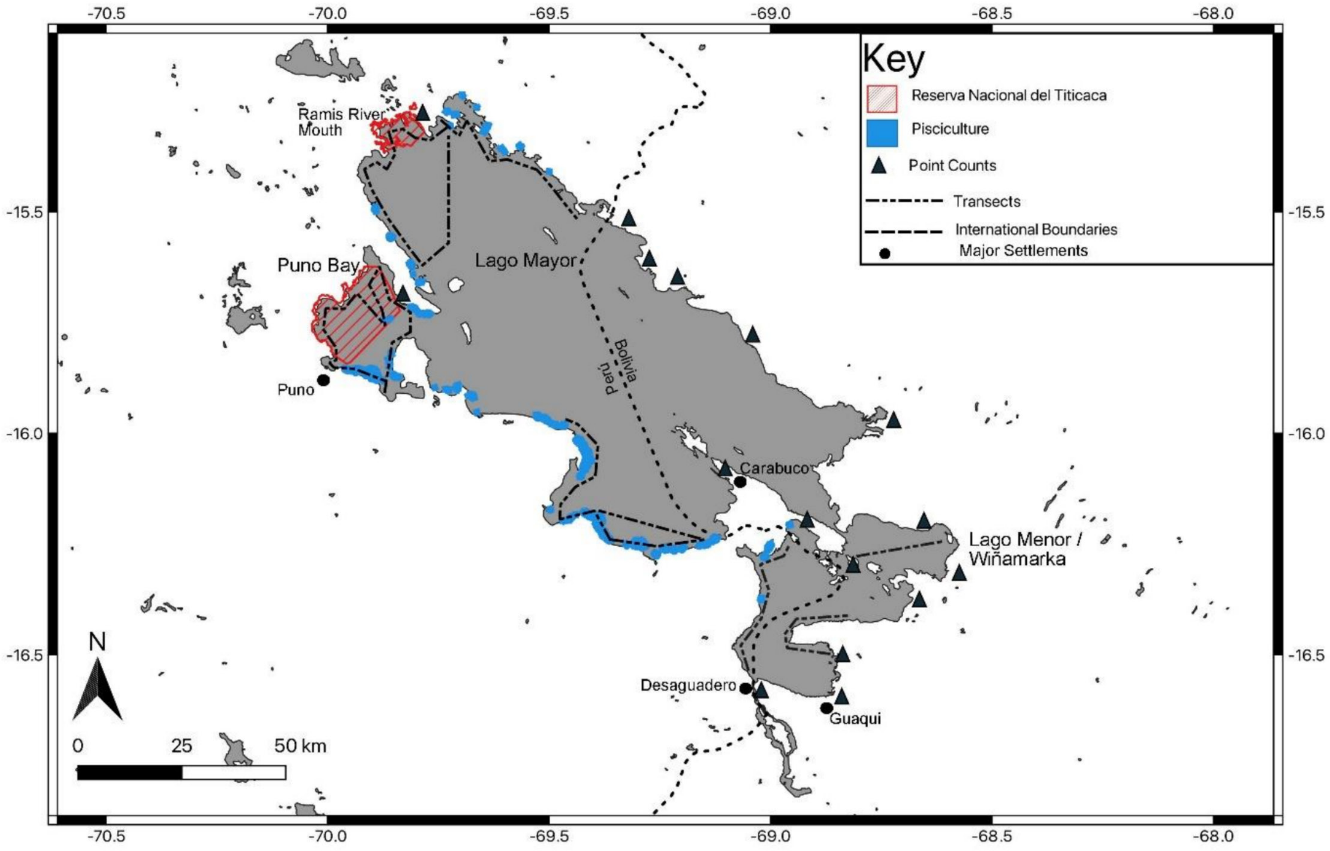

FIGURE 1 | Map of the Lake Titicaca region, with sites of point and transect counts and of pisciculture and protected areas highlighted.

Peruvian region, the Reserva Nacional del Titicaca, has experienced significant disruption in its work due to civil strife in the region (Kent 2006).

Most of the area around Lake Titicaca is rural, but several moderate to large settlements exist in the region, including the provincial capital of Puno Province in Peru, the city of Puno, the border town of Desaguadero and the main base of the Bolivian navy in Guaqui (Figure 1). Lake Titicaca is the habitat of a variety of endemic species, including over 90% of the native fish species (Albert et al. 2011), the endangered Titicaca Grebe (Villar et al. 2023) and the critically endangered Titicaca Water Frog (Angulo 2008).

2.2 | Field Methods

Surveys were conducted in 2022, between March and June in Bolivia and between July and September in Peru. We used a variety of transect and point count methods, depending on the equipment, time available and weather at each site. The coordinates of our surveys ranged from −69.99 to −68.63 in latitude and from −15.25 to −16.58 in longitude. Surveys were conducted with different groups of observers, although all observation teams included author D.A.V. Work on the Bolivian side was primarily conducted from the shore, while work in Peru was primarily conducted along transects from fishers' boats (Figure 1). An attempt was made to conduct repeated counts of all points, but this did not prove possible in all cases. The Bayesian Occupancy Model enacted by the *spOccupancy* package is robust to uneven repetitions.

2.3 | Modelling Fisheries

Previous work assessing conflict potential between birds and fisheries has relied on data on fish fleet locations from onboard observers (Norriss et al. 2020) or legally mandated logs from fishers themselves (Sonntag et al. 2012; Luck et al. 2020) or on preexisting fishing intensity maps (Cleasby et al. 2022). Even studies which included observations from researchers who joined fishing fleets (Norriss et al. 2020; Luck et al. 2020) have to rely principally on self-reported fisheries bycatch. Given the tendency of fishers to under-report bycatch across the world (Burns and Kerr 2008; Bremner et al. 2009; Soto et al. 2023), this means that even these studies probably present low-end estimates for bycatch from fisheries. However, at least these methods provide the modicum of an empirical baseline for research on fisheries bycatch in their respective regions. These options were not available in Lake Titicaca, where fisheries are subsistence and artisanal. Fisheries in Lake Titicaca are usually operated with plastic monofilament nets, with plastic twine to make the nets purchased in either Juliaca in Peru or El Alto in Bolivia. These nets are placed overnight by fishermen, who usually row in small one-to-two-person wooden boats to their preferred fishing spots. Nets are kept horizontal, weighed down by stones on one end and suspended from cork or plastic bottles on the other end, with brightly coloured plastic bottles often being tied to the end of the net to help the fishermen find them the next morning. The mesh size and length of these nets are highly irregular, though they tend to converge with the former being 3–5 cm in diameter and the latter being 100 m in length. The lack of formal logbooks in artisanal fisheries has been highlighted previously as a major barrier to studying artisanal fisheries, especially inland fisheries and those in the developing world (Raby et al. 2011). We therefore had to rely on a model of fisheries intensity. An existing model of fisheries probability in Lake Titicaca was used to assess conflict potential between birds and fisheries. Full methods can be found in Villar et al. 2025. In summary, this study presented a maximum entropy model (Phillips et al. 2006) based on the observation of fishing nets in the waters of Lake Titicaca, using the same environmental variables as we use to model bird distributions. This map has been validated using both the area-under-the-curve and the true-test statistic (Zweig and Campbell 1993; Allouche et al. 2006) and through ground truthing. The model has a TSS of over 0.5 and an AUC of over 0.8, which indicates a good fit to the existing data (Allouche et al. 2006; Komac et al. 2016).

2.4 | Modelling Bird Distributions

We used a Bayesian spatially explicit joint distribution occupancy model with information on imperfect detection. This was done using Polya-Gamma latent variables using a nearest neighbour Gaussian process (Doser et al. 2023). A benefit of this approach is that it includes information on spatial autocorrelation within the model and therefore helps alleviate issues associated with spatial biases in sampling. The occupancy covariates used to construct this model were the same as those used to construct the fisheries model, namely, coverage in totora wetlands, water depth, distance from shore, distance from protected areas, distance from population centres, presence of pisciculture and distance from pisciculture. Totora wetlands, dominated by totora sedge (*Schoenoplectus californicus* subp. *tatora*), constitute a major aquatic phytogeographic region of Lake Titicaca (Banack et al. 2004). Environmental covariates were tested for covariance using the variance inflation factor (VIF) (Akinwande et al. 2015), with a VIF below 10 indicating a lack of problematic collinearity of explanatory variables (Chatterjee and Hadi 2006). The detection covariates used in this model included the following: start time of each survey, the duration in minutes of each survey, the survey team involved, and the length of the transect, with 0 for point counts. Nonfactorial variables related to detection and occupancy were scaled.

Information on coverage in totora wetlands was obtained from Villar et al. (2023), and water depth was estimated from a bathymetric map created by the Autoridad Binancional Atonoma del Sistema Hidrico del Lago Titicaca, which was shared with author D.A.V.; information on distance from shore was obtained from Messager et al. (2016), distance from population centre from Lloyd et al. (2017) and distance from protected areas from SERNANP (2023).

Bird distribution models were assessed using Bayesian *p*-values based on the Freeman–Tukey test statistics (Freeman and Tukey 1950). Models were considered to fit the data well if the Bayesian *p*-value was between 0.1 and 0.9 (Hobbs and Hooten 2015); species for whom the model fell outside that range were excluded from further analysis. Both the fisheries and bird distribution models were constructed using 1 km × 1 km grids. Data on pisciculture came from an unpublished internal report from the Laboratorio Continental del Instituto Peruano del Mar, based on satellite imagery.

Both the fisheries and the bird distribution models were run in R version 4.3.0, using the *sdm*, *terra*, *raster*, *spOccupancy*, *dplyr* and *permGS* packages (Hijmans 2010; Wickham et al. 2014; Naimi and Araujo 2016; Brueckner et al. 2017; Hijmans 2020; Doser et al. 2022). Maps and figures were made either in QGIS or using the *tmap* R package (Tennekes 2014).

2.5 | Assessing Bird Vulnerability and Conflict Potential With Fisheries

Each species was given a species-specific weighing factor (WF). This was used to account for differences in ecology and behavioural traits which might impact vulnerability to fisheries bycatch. It could also account for the overall population size and conservation status of the species in question. The WF was calculated using a modified version of that used by Sonntag et al. (2012) to assess vulnerability of birds to gillnet bycatch in the North Sea, which itself is a modification of the vulnerability index of birds to wind farms developed by Garthe and Hüppop (2004). Our modifications have been driven by the Parkerian shortfall (Lees et al. 2020) or relative lack of detailed natural history knowledge related to species-specific ecologies and life history traits, in neotropical ornithology compared to temperate ornithology.

Our WF was calculated using five factors, the results of which can be seen in Table 1:

a. Diving behaviour. We used a 3-point scale, where 1 represents the *least vulnerable* and 3 the *most vulnerable* to fisheries bycatch. Species that dive horizontally were given a 3, species that dive vertically a 2 and species that dabble or forage near the surface a 1. Data on diving behaviour came from field observations of the species by researchers.

b. Aggregation behaviour. As in previous studies, we assumed that species which lived in denser congregations were more vulnerable to bycatch than those which lived more solitary existences. However, while previous studies considered aggregation behaviour as a binary, with it either existing or not existing, in this study, we included the original population density of the species across our sites as the measure of aggregation behaviour. Population densities were estimated from the effective range of the 8 × 42 binoculars used in the surveys, estimated to be 500 m. This results in a total surveyed area of

$$A_s = N_t(2 \times t) + 0.5N_t(\pi \times 0.5^2) + N_p(\pi \times 0.5^2)$$

where N_t is the total number of transects where a species occurred, N_p is the total number of point counts where the species occurred, t is transect length in m and A_s is the total area in km^2 surveyed for that species. Average population density was then calculated by dividing the total number of individuals of the species in question by A_s. This provides a more accurate image of both population and population density than previous work but leads to aggregation becoming more important in calculating

TABLE 1 | Variables used to calculate WF for each species and WF for each species included in this study.

Species	Diving behaviour	Square-rooted aggregation behaviour	Adult survival rate	Biogeographic population size	IUCN status	*WF*
Anas georgica	1	1.302	0.664	0.001	1	0.639
Ardea alba	1	0.686	0.740	0.0001	1	0.489
Chroicocephalus serranus	1	5.008	0.797	0.070	1	1.870
Fulica ardesiaca	1	5.575	0.716	0.0417	1	1.926
Oxyura ferruginea	2	2.712	0.710	0.015	1	1.355
Pardirallus sanguinolentus	1	1.314	0.681	0.010	1	0.652
Plegadis ridgwayi	1	2.159	0.637	0.012	1	0.868
Podiceps occipitalis	2	1.056	0.767	0.002	2	1.410
Rollandia microptera	3	2.969	0.824	1	4	5.794
Rollandia rolland	2	1.404	0.769	0.006	1	1.007
Spatula puna	1	2.189	0.569	0.034	1	0.851

WF than in previous studies. We therefore used the square root of the population density as our measure of aggregation for the purposes of calculating *WF*.

c. Adult survival rate, assuming that species with higher adult survival rates will be more affected by the excess mortality caused by mortality due to fisheries bycatch. Where we lacked adult survival rate information on individual species, we used the average of known adult mortality of members of the same genus or, if this was absent, of the most closely related genus. Data for adult survival rates come from Avibase (Lepage et al. 2014). Species for whom mortality rates were assessed from other members of the genus are highlighted with an asterisk in Table 1.

d. Biogeographic population size, taken as the percentage of the global population estimated to live in Lake Titicaca. We assumed that species for whom a higher percentage of the global population was found in Lake Titicaca would be more vulnerable to fisheries bycatch. Full details on calculating population size for birds in Lake Titicaca can be found in Villar, Thomsen, Paca-Condori, et al. 2024. In summary, the continuous distribution rasters of each species were converted to binary presence/absence rasters using the lowest presence threshold (Pearson et al. 2007). Then, following the logic of Mladenoff and Sickley (1998), Long et al. (2008) and Villar et al. (2023), the average population density, as calculated in *b* above, was multiplied by the area above the lowest presence threshold. This figure was then divided by the total estimated global population of the species, as estimated by Callaghan et al. (2021). In the case of the Titicaca Grebe, the estimated population from our survey is greater than the estimated population of the entire species. Given the species is endemic to Lake Titicaca, we gave it a value of 1 for this variable.

e. IUCN conservation status. We assumed that more endangered species would be more vulnerable to the effects of fisheries bycatch on their population than less endangered species. We used the status as of IUCN, 2023. We converted IUCN status into a descending numerical value, from 6 for critically endangered to 1 for least concern.

The variables included in *WF* were moderately collinear, with IUCN conservation status and biogeographic population size having a VIF of over 10 (10.28 and 13.64 respectively). This is to be expected, as the range size of a species is incorporated into IUCN assessments of the species' vulnerability. However, for two reasons, we still consider it appropriate to include both in our measure of *WF*: first, to make our results consistent with previously adopted measures of vulnerability to fisheries bycatch and make our results broadly comparable to them and second, because while they are somewhat collinear, they capture distinct aspects of vulnerability which conservationists and wildlife managers need to take into account; IUCN conservation determines the global conservation priority of a species, while the biogeographic range in Lake Titicaca reflects the relative importance of Lake Titicaca to that species. To examine the sensitivity of our results to the inclusion of both biogeographic range and IUCN threat status, we also calculated WF_1 which included IUCN status and excluded biogeographic range and WF_2, which did the opposite. The results of this analysis can be found in the supplemental materials.

To calculate the vulnerability of each species to fisheries bycatch, following the example of previous vulnerability indices (Sonntag et al. 2012; Garthe and Hüppop 2004), we divided these five variables into 'behavioural' variables and 'status' variables. The former are the diving behaviour (a_i) and the aggregation behaviours (b_i), and the latter are adult survival rate (c_i), biogeographic population size (d_i) and IUCN conservation status (e_i). Thus, the WF for each species *i* was calculated as follows:

$$WF_i = \frac{a_i + b_i}{2} \times \frac{c_i + d_i + e_i}{3}$$

In each pixel of the map, overall vulnerability to fisheries bycatch was calculated as follows:

$$V_s = \sum_i \left(WF_i \times P_i\right)$$

where P_i is the probability of species *i* occurring in the given pixel. V_s is a measure of the potential vulnerability to fisheries bycatch of the birds in a given grid, but it tells us nothing about the degree of conflict between birds and fisheries. Conflict intensity was therefore calculated as follows:

$$C_s = V_s \times P_f$$

where P_f is the probability of fisheries occurring. For ease of interpretation, C_s values were then divided into four categories: 'Minimal Conflict', 'Low Conflict', 'Moderate Conflict', and 'High Conflict', while V_s values were divided into 'Minimal Vulnerability', 'Low Vulnerability', 'Moderate Vulnerability' and 'High Vulnerability'. The pixels were divided into these four categories using Fisher's linear discriminant (Fisher 1936).

Although distributions were assessed of all species identified in our surveys, we only included species which met the following criteria when assessing conflict potential: (1) the species was identified in at least 10 surveys and (2) the Bayesian *p*-value of the occupancy model for the species was between 0.1 and 0.9. We include the relevant code for all species, as well as the relevant predictions, in the supplemental materials.

3 | Results

In total, 41 species were identified across 36 point-counts and 115 transects, including repeated surveys, of which 16 point-counts and three transects were in Bolivia and 20 point-counts and 112 transects were in Peru (Table 2). Fifty-four sites were surveyed twice, three were each surveyed three times and the remainder surveyed only once. Only 11 species met the criteria for inclusion in the rest of our study (see above).

As no occupancy variables had VIF greater than 10, we continued to use all variables in the model. The overall model had a Bayesian *p*-value of 0.398, indicating a good fit.

Of the total lake surface area of 8372 km^2, 2920 km^2 can be classified by our model as areas of minimal potential vulnerability ($0 < V_s < 3.45$), 2244 km^2 as areas of low potential vulnerability ($3.45 < V_s < 7.35$), 1058 km^2 as areas of moderate potential vulnerability ($7.35 < V_s < 12.04$) and 1837 km^2 as areas of high potential vulnerability ($12.05 < V_s < 16.41$) (Figure 2). The Lago Menor, the mouth of the Ramis River and Puno Bay are all areas where species with high vulnerability to bycatch have a high probability of being found. The latter two overlap significantly with the portion of the Reserva Nacional del Titicaca in Puno Bay.

TABLE 2 | Number of times in which included species were observed and Bayesian *p*-values of species-specific occupancy model.

Species	Number of observations	Bayesian *p*-value of model
Anas georgica	33	0.756
Ardea alba	13	0.512
Chroicocephalus serranus	134	0.641
Fulica ardesiaca	132	0.675
Oxyura ferruginea	88	0.497
Pardirallus sanguinolentus	10	0.219
Plegadis ridgwayi	35	0.267
Podiceps occipitalis	42	0.117
Rollandia microptera	153	0.321
Rollandia rolland	72	0.445
Spatula puna	60	0.203

Note: Only species observed in at least 10 surveys with a Bayesian *p*-value of the occupancy model of 0.1–0.9 were included.

Further to the assessment of vulnerability, of the total lake are, 5933 km^2 constitute areas of minimal conflict potential ($0 < C_s < 2.26$), 904 km^2 are classified as areas of low conflict potential ($2.26 < C_s < 6.17$), 591 km^2 as areas of moderate conflict potential ($6.17 < C_s < 10.07$) and 630 km^2 (just 9.4%) as areas of high conflict potential ($10.07 < C_s < 15.12$) (Figure 3). High conflict areas were concentrated near the shore, save for in Lago Menor where conflict areas extend to the centre of that section of the lake. This is probably because fishermen fish further from shore in that much shallower part of Lake Titicaca. Areas with low and moderate conflict potential between fisheries and birds extend into protected areas of the Reserva Nacional del Titicaca in both the Ramis River and the Puno Bay section.

4 | Discussion

This study is the first attempt to map the conflict potential between fisheries and birds in Lake Titicaca. We find that most of Lake Titicaca has minimal conflict potential. This is driven by two forces: Firstly, very few bird species are found far from shore, and secondly, very few fishers go more than a few kilometres from shore. While the highest concentration of birds was in Puno Bay, including in the area included in the Reserva Nacional del Titicaca, conflict potential was concentrated in the Lago Menor region of Lake Titicaca. This is primarily due to fishing being less common in Puno Bay than in equally shallow parts accessible from the villages of Lago Menor. This is probably because dense totora wetland of Puno Bay makes it difficult to lay gillnets compared to Lago Menor and due to the prominence of tourism in Puno Bay (Kent 2006; Villar, Thomsen, Gutiérrez Tito, et al. 2024). Despite recent moves from a fishing to a tourism based economy, much of Puno Bay is still guarded

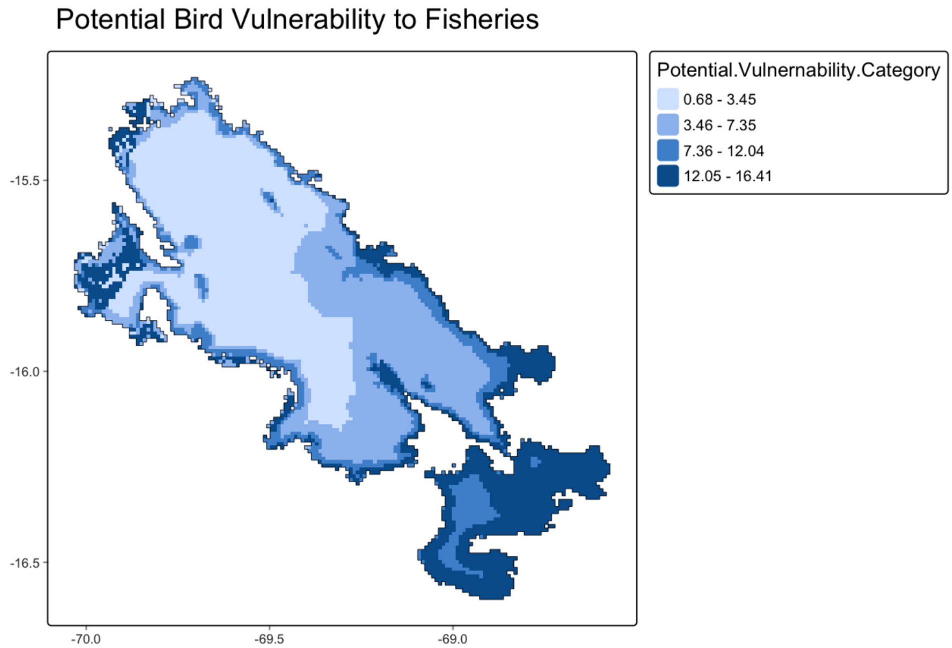

FIGURE 2 | Map of vulnerability to fisheries (V_s) of birds in Lake Titicaca as per our model.

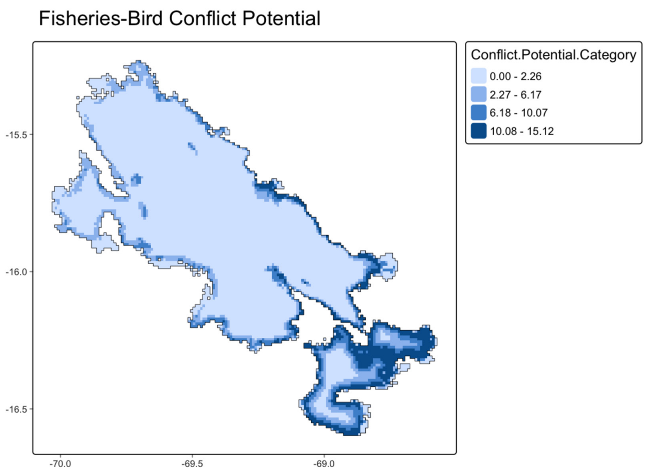

FIGURE 3 | Map of conflict potential (C_s) between fisheries and birds in Lake Titicaca as per our model.

by Uros as traditional territory. This has led to conflict with both the Reserva Nacional del Titicaca (Kent 2006; Kent 2008) and residents of non-Uro villages who attempt to fish in the region, such as the Chimu people (A. B. C., pers. obvs.). However, this also means that the bird populations there could be vulnerable if the income from tourism were to cease, as occurred during the COVID-19 pandemic (E. R. G. T., *pers. comm.*)

Technical fixes have been implemented to reduce bycatch from several types of fisheries (Campbell and Cornwell 2008). However, despite some promise that certain changes to net colour and thickness can reduce bird bycatch in some circumstances (Martin and Crawford 2015; Hanamseth et al. 2018), no technical fix reduces bycatch of a variety of bird species without reducing the fish catch (Field et al. 2019; Cantlay et al. 2020). Fishery closure, as undertaken in other regions, can be effective (Regular et al. 2013). However, given the poverty of the Altiplano and the difficulty of enforcing already existing fishery restrictions, we do not believe that closing parts of Lake Titicaca for fishing are a viable option. While restrictions exist in some areas, such as in the Reserva Nacional del Titicaca, and times, such as a seasonal size restriction legislated by the Peruvian Ministry of Production, they are rarely enforced. On Lake Titicaca, relatively minor fisheries regulations, such as minimum catch size, are not consistently enforced, and poverty in the region requires access to fishing for food and an income. In short, enforcement may require a degree of administrative capacity which would stretch the limited resources for environmental regulations which exist in the region, and its ethical challenge would constitute a radical political change liable to face significant resistance from local communities, as has been the case in the past when environmental regulations have impinged on local natural resource use (Kent 2006). To find solutions, therefore, leaves just the possibility of working with local small-scale fishermen (Psuty and Całkiewicz 2021) across Lake Titicaca. This should be focused especially near areas which have been highlighted as having the highest conflict potential, to reduce fisheries bycatch on terms which are acceptable to the fishermen themselves. Workshops to reduce fisheries bycatch are needed across Lake Titicaca. Building up the trust of fishermen to convince them to change fishing practices to help reduce bycatch will take time, especially in the context of pre-existing conflict between fishermen and birds (Quispe et al. 2023; Villar, Thomsen, Paca-Condori, et al. 2024; Villar, Yanes, Gutiérrez Tito, and Gosler 2024). Across the world (Gosler et al. 2013), and specifically in the Lake Titicaca region (Villar, Thomsen, Paca-Condori, et al. 2024), educating the human elements of the human-wildlife conflict about the global significance of the species with which they feel to be in conflict can turn negative perceptions into pride. Many residents of Lake Titicaca are unaware of the importance of their part of the world for global biodiversity, and become keener to protect it when told (Villar, Thomsen, Paca-Condori, et al. 2024).

The fact that only 11 species fit our criteria for inclusion in our model, out of the 135 which reside in Lake Titicaca and 60 which are broadly aquatic, indicates that further fieldwork is required to assess populations of birds properly in Lake Titicaca. The Reserva Nacional del Altiplano already undertakes regular bird surveys in the area under its jurisdiction and publishes the results of these in undigitized reports (E. R. G. T., pers. obvs.). The Peruvian government should provide funds for the digitization of these reports so that their data become available to researchers and conservationists elsewhere. Further regular surveys of bird populations in Lake Titicaca will be essential. Fisheries monitoring is also required to understand better the geography of fisheries in Lake Titicaca.

We understand that there are neither the funds from central governments nor the willingness from fishermen to engage in logbook or observer programmes which exist in other regions of the world, such as Europe. Instead, therefore, we suggest that short-term observer programmes, similar to those undertaken in Peruvian coastal fisheries (Alfaro-Shigueto et al. 2010; Pott and Wiedenfeld 2017), could be implemented to provide a better understanding of fisheries bycatch in Lake Titicaca. These could be combined with interviews with fishermen, as has been used to assess fisheries bycatch in other contexts (Karris et al. 2013). The potential also exists to use satellite data to monitor fishing, as has been done for other regions with little on the ground monitoring (Rowlands et al. 2019), but this would need to be undertaken to provide a control and in parallel with ground-truthing and efforts to raise awareness among locals of the status and significance of their regional endemic avifauna.

Author Contributions

D. A. Villar: conceptualization, investigation, writing – original draft, writing – review and editing, visualization, methodology, software, formal analysis, data curation, project administration, validation, funding acquisition. **Paola Velásquez-Noriega:** investigation, writing – review and editing, resources. **Bastian Thomsen:** supervision, writing – review and editing. **Edwin R. Gutiérrez Tito:** investigation, project administration. **Anahi Cosky Paca-Condori:** resources, writing – review and editing. **Edmundo G. Moreno Terrazas:** writing – review and editing, resources, supervision. **Ronald Hinojosa Cárdenas:** investigation. **Alfredo Balcón Cuno:** investigation. **Carmen Villanueva:** investigation. **Patrick Chapman:** investigation, visualization. **Jhazel Quispe:** investigation, validation, writing – review and editing. **Andrew Gosler:** supervision, writing – review and editing.

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Ethics Statement

We obtained research permits from the Peruvian (no. 004-2022-SERNANP-RNT-J) and Bolivian governments (MNHN°020/202) and received permission from all local communities where work was conducted. Local authorities do not issue formal research permits but rather permitted researchers to conduct research after discussions with them.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

Data for this study can be found in the following Dryad link (http://datadryad.org/share/5XDFkRdHf12ysZhcyClPX7SbtOFMK9NaXZrKuZJ-_0s).

Use of Artificial Intelligence

We did not use AI.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Data S1:** Supporting Information. **Data S2:** Supporting Information. **Data S3:** Supporting Information.