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Estimating Population Size and Survival for Two Subpopulations of the Critically Endangered Patagonia Frog, *Atelognathus patagonicus*, in Laguna Blanca National Park, Argentinian Patagonia

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ABSTRACT

Detailed knowledge of population dynamics and species status is essential for effective conservation planning. The Critically Endangered Patagonia frog *Atelognathus patagonicus* inhabits small, isolated lagoons in north-western Argentinian Patagonia. Using capture–mark–recapture techniques over two consecutive summer seasons (2023–2024), we estimated the abundance and survival of two subpopulations in the Antiñir and Jabón lagoons within Laguna Blanca National Park. POPAN models estimated abundances of 2937 (95% CI: 2089–4368) individuals in Antiñir and 5605 (95% CI: 4547–7041) in Jabón, including all life stages. Cormack–Jolly–Seber models revealed an apparent juvenile survival of 0.988 (95% CI: 0.981–0.993) in Antiñir, and the value ranged from 0.865 (95% CI: 0.627–0.939) to 0.966 (95% CI: 0.954–0.975) in Jabón. Adult survival was estimated at 0.956 (95% CI: 0.932–0.972) in Antiñir and 0.969 (95% CI: 0.923–0.988) in Jabón. We found albino individuals only in Antiñir, comprising 5% of the subpopulation, with an apparent survival of 0.956 (95% CI: 0.4–0.999) across all stages. Our findings suggest early signs of recovery following recent severe droughts. However, expanding monitoring efforts to additional lagoons, particularly those outside the protected area, is crucial to the assessment of broader recovery trends. This study provides a valuable foundation for comparing population dynamics across subpopulations and developing targeted conservation strategies to ensure the long-term protection of this species.

Resumen

El conocimiento detallado de la dinámica poblacional y el estado de las especies es esencial para una planificación eficaz de la conservación. La Críticamente Amenazada ranita patagónica (*Atelognathus patagonicus*) habita pequeñas lagunas aisladas en el noroeste de la Patagonia argentina. Utilizando técnicas de captura-marcado-recaptura durante dos temporadas de verano consecutivas (2023–2024), estimamos la abundancia y supervivencia de dos subpoblaciones en las lagunas Antiñir y Jabón dentro del Parque Nacional Laguna Blanca. Los modelos POPAN estimaron abundancias de 2937 (IC del 95%: 2089–4368) individuos en

Antiñir y 5605 (IC del 95%: 4547–7041) en Jabón, incluidas todas las etapas de vida. Los modelos Cormack-Jolly-Seber revelaron una supervivencia aparente juvenil de 0,988 (IC del 95%: 0.981–0.993) en Antiñir y el valor varió de 0,865 (IC del 95%: 0.627–0.939) a 0,966 (IC del 95%: 0.954–0.975) en Jabón. La supervivencia adulta se estimó en 0.956 (IC del 95%: 0.932–0.972) en Antiñir y 0.969 (IC del 95%: 0.923–0.988) en Jabón. Encontramos individuos albinos solo en Antiñir, comprendiendo el 5% de la subpoblación, con una supervivencia aparente de 0.956 (IC del 95%: 0.4–0.999) en todos los estadios. Nuestros hallazgos sugieren signos tempranos de recuperación tras las recientes sequías severas. Sin embargo, es fundamental ampliar los esfuerzos de monitoreo a otras lagunas, en particular las que se encuentran fuera del área protegida, para evaluar tendencias de recuperación más amplias. Este estudio proporciona una base valiosa para comparar la dinámica poblacional entre subpoblaciones y desarrollar estrategias de conservación específicas para garantizar la protección a largo plazo de esta especie.

1 | Introduction

Amphibians are among the most threatened vertebrates globally, with 41% of all known species in some threat category (Pimm et al. 2014). In the southern cone of South America, Argentina aligns with this global trend, with a significant proportion of amphibian species experiencing marked declines (Kacoliris et al. 2022). The Patagonian Steppe is the ecoregion with the highest percentage (75%) of threatened amphibians in the country (Vaira et al. 2017). The main threats to these species are habitat loss and degradation, chytrid fungus infection and the introduction of invasive species (Vaira et al. 2012; Kacoliris et al. 2022).

Detailed information on the population dynamics and current status of endangered Patagonian amphibians is key to making decisions and leading pragmatic management actions to promote their long-term survival (Vaira et al. 2017). However, there is a lack of data about population parameters of amphibians in Patagonia. Currently, the only available information on the population ecology of endangered Patagonian anuran species comes from studies on the Valcheta frog (*Pleurodema somuncurense*) by Velasco et al. (2016, 2019). In the case of the Critically Endangered Patagonia frog (*Atelognathus patagonicus*), there is no updated information regarding its population parameters.

The Patagonia frog is an endemic species from the lagoon system of the Laguna Blanca National Park and surrounding areas in Argentinian Patagonia (Ceï and Roig 1968; Cuello et al. 2009). It has the particularity of presenting two morphotypes: aquatic and littoral (Ceï and Roig 1968). The aquatic form is characterized by well-developed cutaneous folds, extensive interdigital membranes and a yellow-orange ventral surface. In contrast, the littoral form displays reduced cutaneous folds, less developed interdigital membranes and a greyish-white ventral surface (Ceï and Roig 1968). While the aquatic morphotype is the dominant one, these morphotypes are reversible in response to environmental conditions; that is, during drier seasons, the terrestrial morphotype increases in frequency (Cuello et al. 2008).

Laguna Blanca was the habitat of the most important subpopulation of this species, which was highly abundant during the 1970s (Péfaur and Duellman 1980) and remained so until the mid-1980s, according to observations reported by Administración de Parques Nacionales (APN, unpublished data). However, the introduction of exotic fish species—perch (*Percichthys colhuapiensis*), rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*)—into Laguna Blanca caused significant changes to the ecosystem

(Ortubay et al. 2006), leading to the extinction of *A. patagonicus* in this lagoon (Fox et al. 2005; Cuello and Perotti 2006).

The Patagonia frog remains confined to several smaller, isolated lagoons that are still fishless. Nevertheless, this species faces additional threats, including habitat degradation caused by cattle presence (Cuello et al. 2014, 2017) and emerging diseases such as ranavirus and chytridiomycosis (Fox et al. 2006; Ghirardi et al. 2014). Furthermore, the aquatic vegetation on which this species depends has been severely degraded across all lagoons due to the prolonged drought that occurred between 2010 and 2016 (Cuello et al. 2015). As a consequence of this severe drought, reproduction ceased, and there was an estimated population size reduction of over 90% (Cuello pers. comm.). The dramatic climatic cycles in the region can act synergistically with other risk factors, exacerbating the species' vulnerability (Cuello et al. 2017). Moreover, cases of albinism have been reported for the first time in one of the subpopulations (Rolón et al. 2023), and their expression could be linked to this population decline. For all these reasons, the Patagonia frog is categorized as Critically Endangered according to the IUCN Red List of Threatened Species (IUCN SSC Amphibian Specialist Group 2019) and as Endangered in the National Red List (Basso et al. 2012).

Since the last assessment and the extreme drought that affected all subpopulations, studies have yet to address the current status and situation of the species. In this work, we aim to estimate the abundance and survival of two *A. patagonicus* subpopulations in two lagoons within Laguna Blanca National Park using capture–mark–recapture (CMR) techniques. We also determined the abundance and survival of albino individuals encountered in one of these lagoons. Our results will contribute to a better understanding of this species' population dynamics, as well as its response and recovery from disturbances, aiding in the development of effective conservation strategies.

2 | Methods

2.1 | Study Area

Laguna Blanca National Park is located in north-western Argentinean Patagonia, in Neuquén Province (Figure 1). It protects 11,250 ha of a unique sector within the Patagonian Steppe ecoregion, harbouring several endemic species, including the Patagonia frog, the most threatened amphibian within the park (Rolón et al. 2024). The area is characterized by a system of unconnected basaltic lagoons of volcanic origin, where this

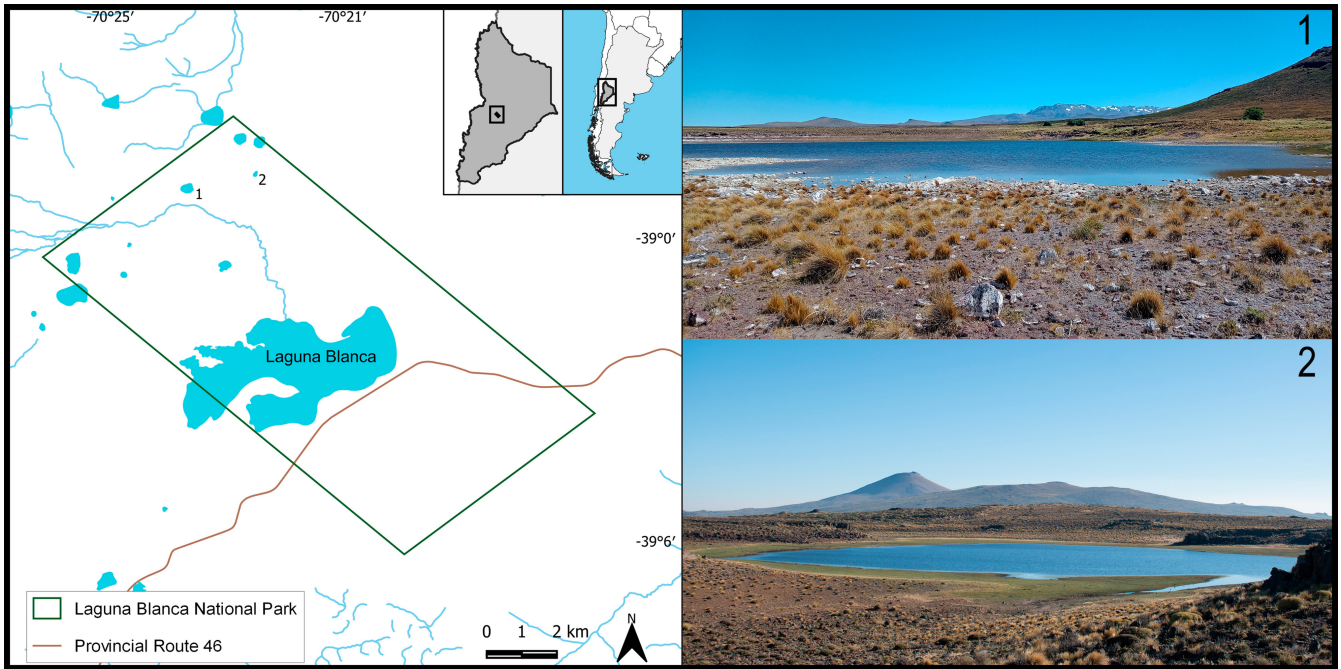


FIGURE 1 | Location of Laguna Blanca National Park in Neuquén province, Argentinian Patagonia. 1: Antiñir Lagoon. 2: Jabón Lagoon.

species inhabits (Ceï 1980). The park was declared a Ramsar site, an Important Bird and Biodiversity Area (IBA) and a Key Biodiversity Area (KBA) due to its significant importance as a wetland and its critical role as a feeding and breeding area for waterfowl (APN 2017).

The landscape is characterized by rocky, rugged terrain featuring numerous volcanic cones scattered among cliffs, basaltic walls and rocky plains (Marcolín 1968). The vegetation is typical of the shrub-grass steppe, with *Mulinum spinosum* and *Stipa* species being the dominant plants (León et al. 1998; Morello et al. 2012). The area experiences an extreme climate, marked by hot, dry summers and cold winters with snowfall (Conti 1998). The average annual temperature ranges between 8°C and 10°C, while the annual rainfall varies from 280 to 370 mm and is concentrated in autumn and winter (APN 2017). We focused our studies on two lagoons within the park (Figure 1): Antiñir Lagoon (38°59'2.46" S, 70°23'49.64" W) and Jabón Lagoon (38°58'43.94" S, 70°22'27.10" W). As part of the park's management plan, the Antiñir Lagoon was fenced to prevent livestock entry. Both small lagoons suffered a huge drought between 2010 and 2016 (Cuello et al. 2015). During the sampling sessions, the average lagoon area was 4.76 ha for Antiñir and 6.08 ha for Jabón.

2.2 | Data Collection

Between January and March 2023 and from December 2023 to March 2024, we conducted a CMR survey in both lagoons during the austral summer season. Each season, we carried out three visits per lagoon (one per month, constituting primary periods), averaging four consecutive days per visit (secondary periods). During the first sampling season, we completed 12 visits across January, February and March. In the second sampling season, we conducted 12 additional visits

in December, January and March, resulting in a total of 24 capture sessions per lagoon.

To catch the individuals, we combined visual encounter surveys (Crump and Scott 1994), looking beneath rocks both in water and on land around the lagoons, with funnel traps (Smith and Rettig 1996), which have proven effective in previous *A. patagonicus* studies for capturing different aquatic stages (Cuello et al. 2008, 2009, 2014, 2017). Sampling was conducted during the daytime, from morning until afternoon, averaging 6 h/day with three people.

We marked individuals in situ with an individual code using visible implant elastomers, a low-risk marking technique that does not interfere with the biology of the individuals (Hale and Gray 1998; Fitzgerald et al. 2004; Olsen et al. 2004). Each individual was marked on the distal area of the limbs following Velasco et al. (2018), using a unique combination of colours (Figure 2). Since there are four possible marking locations (left forelimb, right forelimb, left hindlimb and right hindlimb) and multiple colour options, this method allows for a large number of unique combinations and minimizes the likelihood of mark loss or migration. We classified individuals into larval stages (including tadpoles and metamorphs), juveniles (SVL < 34 mm) and adults (Figure 3). Whenever possible, we also identified the morphotype based on morphological features (cutaneous folds, interdigital membranes and ventral surface colouration) and determined the sex of adults by examining external secondary sexual characteristics (males exhibited nuptial pads on the first finger, while females had oocytes that were visible through the thin, transparent ventral skin; Figure 3). After data collection, each individual was carefully released at its original capture site. We ensured gentle handling throughout the process. To prevent cross-contamination of potential pathogens between sampling sites, we strictly adhered to the Declining Amphibian Populations Task Force (DAPTF) Fieldwork Code of Practice

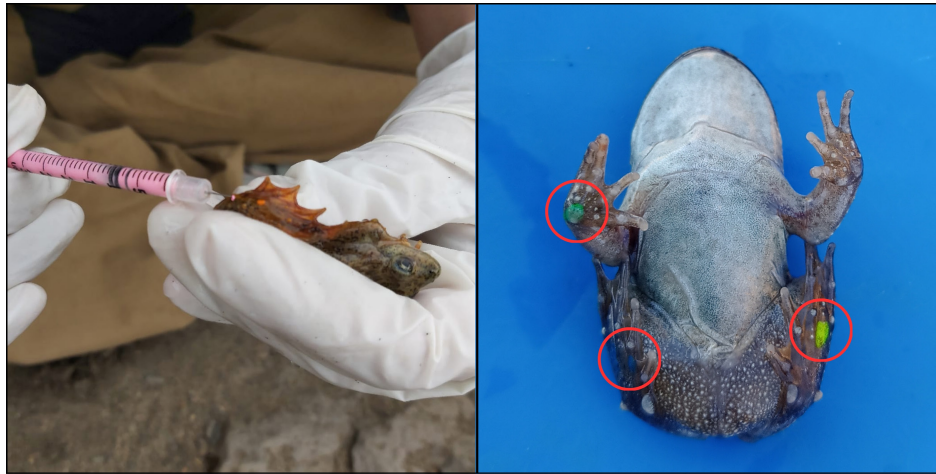


FIGURE 2 | Marking procedure. The left panel illustrates the injection of elastomers into one of the limbs. The right panel shows the marks on an individual: The right forelimb is marked in green, the right hindlimb in blue and the left hindlimb in yellow.

and the guidelines outlined by Lips et al. (2001), cleaning boots and traps with a 70% ethanol solution and rinsing them with sterilized water between sites.

2.3 | Data Analysis

To estimate abundance and apparent survival, we employed the POPAN model (Schwarz and Arnason 1996) and the classic model of Cormack–Jolly–Seber (CJS) using the program MARK 10.1 (White and Burnham 1999). For abundance analysis, we developed a set of candidate models that consider temporal variation and group effect (larval stages, juveniles and adults) to estimate monthly abundance per group. For survival analysis, parameters were calculated by season for juveniles and adults (groups). When the number of captures or recaptures was low for a group, it was removed from the analysis as the estimation was imprecise. We also estimated the abundance and survival of albino individuals encountered in the Antiñir Lagoon. We employed POPAN and CJS models to calculate the population parameters for all the sampling periods including all the stages (larvae, juveniles and adults).

Models were ranked based on the Akaike (1976) information criterion (AIC) (Burnham and Anderson 2003), and the model fit was evaluated using the c -hat value derived from Fletcher's method (Cooch and White 2024). We determined that an acceptable fit for the most general model to the data among the candidate set of models was a c -hat value ranging from 1 to 2 (Cooch and White 2024). Ultimately, the model with the best AIC ranking from the candidate set was used to estimate the population parameters.

3 | Results

We marked 857 individuals in the Antiñir Lagoon (714 during the first sampling season and 143 in the second season) with 272 recaptures. In the Jabón Lagoon, we marked a total of 1578 individuals (812 during the first sampling season and 766 in the second season) with 915 recaptures. In both lagoons, juveniles

made up the majority of marked individuals during the first sampling season. During the second season, Jabón still had a majority of juveniles, while Antiñir had adults as the dominant group (Figure 4). Of the 195 marked adults in Antiñir, 55 (28%) were recaptures of immature individuals from the previous season. In Jabón, only 19 individuals (22% of the total marked adults) were recaptured after maturing into adults from the prior season. We could determine 71 males and 55 females in Antiñir (captured male:female ratio of 1.3:1) and 29 males and 27 females in Jabón (1.1:1). In Antiñir, all adults exhibited the aquatic morphotype, whereas in Jabón, 12.8% of adults displayed the littoral form. The top-ranked models, based on AIC criteria, selected for estimating the population parameters are summarized in Table 1.

3.1 | Abundance Estimation

The total population size was estimated at 2937 (95% CI: 2089–4368) for Antiñir and 5605 (95% CI: 4547–7041) for Jabón, considering all marked individuals (larval stages, juveniles and adults). Adult abundance could only be estimated during the second sampling season (Table 2). Besides, the abundance for December could not be determined due to the low number of recaptures that month in both lagoons (one recapture in Antiñir and zero recapture in Jabón). Considering the lagoon areas, we estimated a density of 0.06 individuals·m⁻² (95% CI: 0.04–0.09) for Antiñir and 0.09 individuals·m⁻² (95% CI: 0.08–0.12) for Jabón.

3.2 | Apparent Survival Estimation

In juveniles, apparent survival was estimated at 0.988 (95% CI: 0.981–0.993) for the Antiñir subpopulation and ranged from 0.865 (95% CI: 0.627–0.939) to 0.966 (95% CI: 0.954–0.975) per season for the Jabón subpopulation (Figure 4). For adults, survival was estimated at 0.956 (95% CI: 0.932–0.972) in Antiñir and 0.969 (95% CI: 0.923–0.988) in Jabón (Figure 5). During the first season, only a few adults were encountered (five per lagoon), so survival could be estimated only for juveniles. In the second season, adult captures allowed for survival estimation;

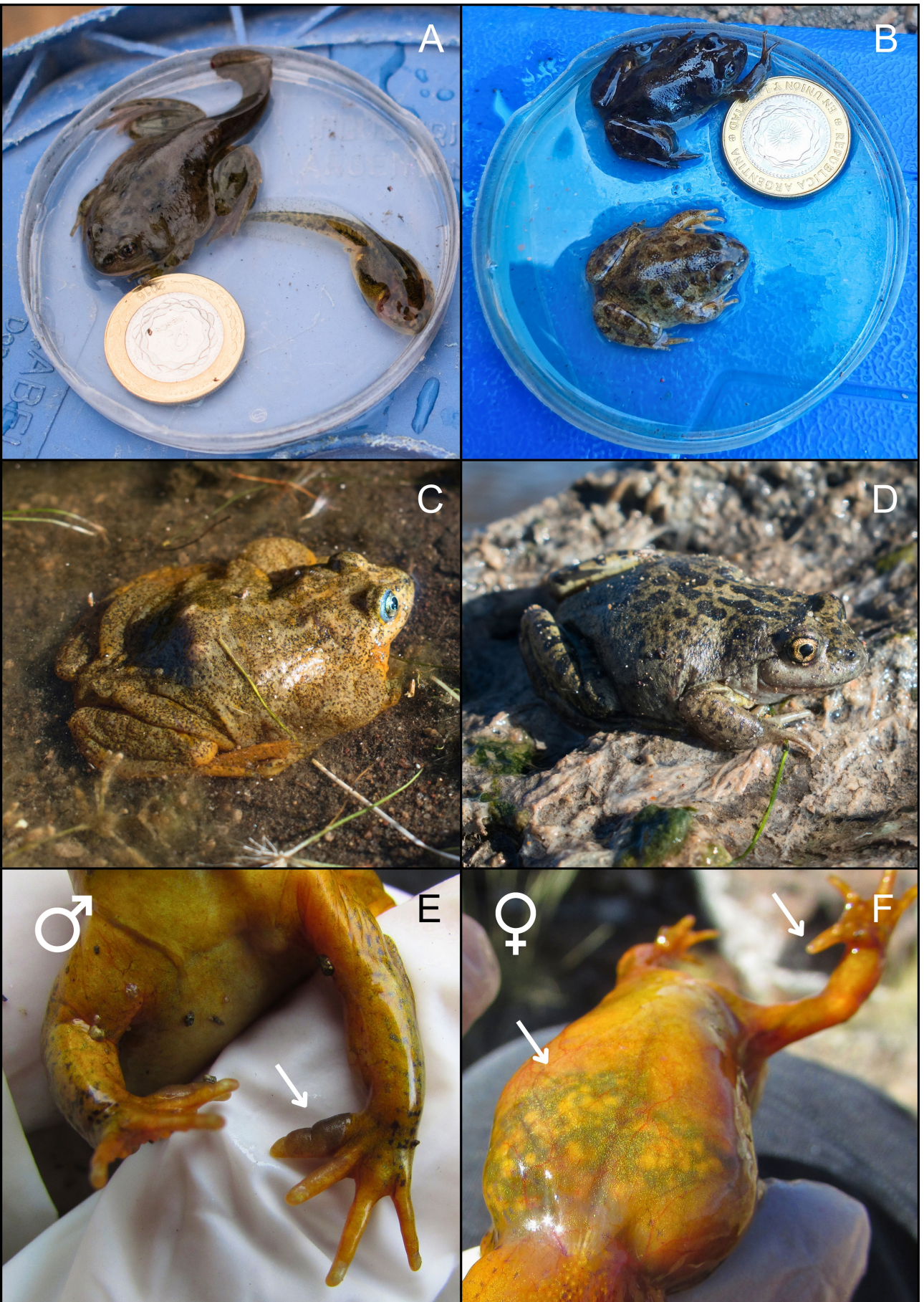


FIGURE 3 | Legend on next page.

FIGURE 3 | Life stages, morphotypes and sex differentiation of the Patagonia frog. (A) Larval stages. (B) Juveniles. (C) Adult aquatic form. (D) Adult littoral form. (E) Male, arrow indicates the nuptial pad. (F) Female, arrows show visible oocytes and an unthickened first finger. Photos A, C and D by Facundo de Los Santos; B by Tomás Álvarez; E and F by Melisa Rolón.

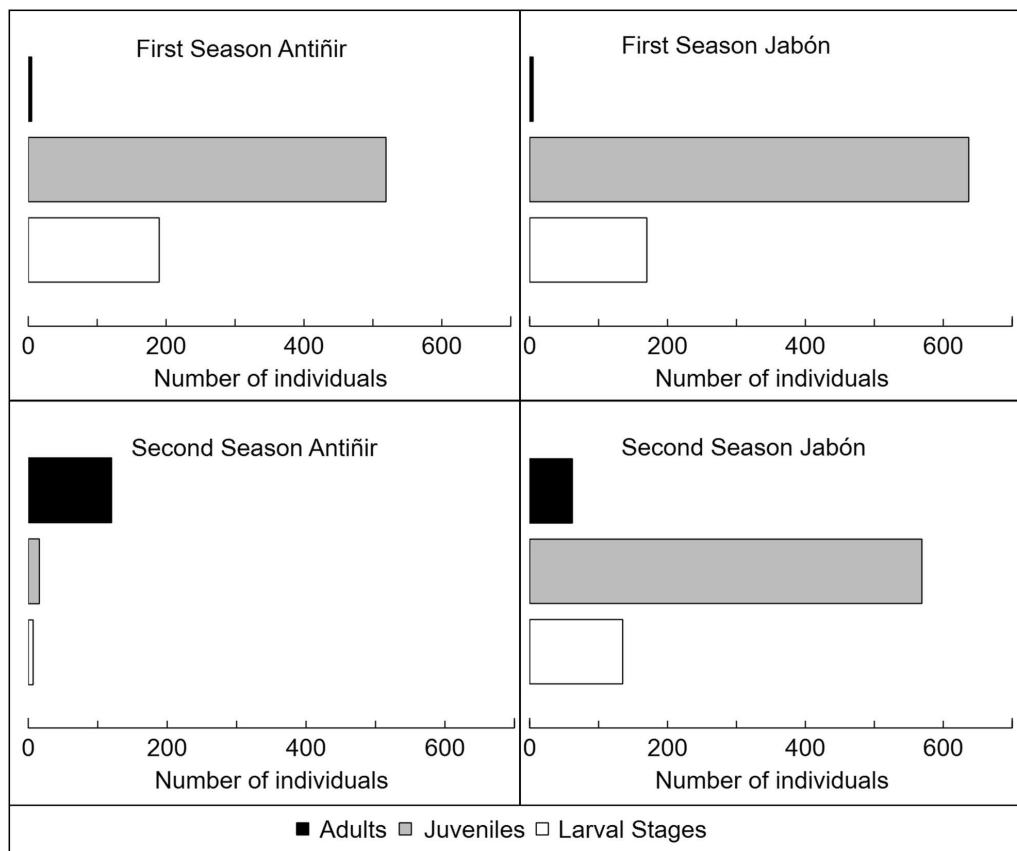


FIGURE 4 | Number of individuals marked per lagoon during each season.

however, juvenile survival in Antiñir could not be determined due to the low number of recaptures (only one).

3.3 | Albino Individuals

We recorded a total of 40 albino individuals in the Antiñir Lagoon (32 during the first sampling season and 8 in the second season), including 6 tadpoles, 2 metamorphs, 26 juveniles and 6 adults. We did not find any albino individuals in the Jabón Lagoon. The estimated abundance for both seasons was 146 (95% CI: 64–496) individuals, representing 5% of the Antiñir subpopulation. Regarding apparent survival, albino individuals exhibited a value of 0.956 (95% CI: 0.4–0.999) considering all the marked stages.

4 | Discussion

This is the first work using CMR techniques to estimate the abundance and survival of the Patagonia frog. Fox et al. (2005) estimated density using transects and quadrats in several lagoons within the species distribution range. For the Antiñir and Jabón lagoons, they reported average densities of 0.4 and 0.11

individuals·m⁻², respectively. Compared to our results, these higher values may be associated with the larger lagoon sizes during those years (by 2003, the Antiñir Lagoon was 60% larger and the Jabón Lagoon 50% larger, based on Google Earth Pro data). Besides, these authors reported the highest number of tadpoles in February; however, we found the peak in January for the two lagoons during both seasons. Additionally, they observed a density peak in March, while in our case, the peak varied by lagoon and season. In Antiñir, the peak occurred in March 2023 and January 2024, with an average of 102 and 36 captures per day, respectively. In Jabón, the highest numbers were recorded in January 2023 with an average of 148 captures per day and in March 2024 with 132 captures per day. In all cases, most of the captures were juveniles, except in Antiñir during January 2024, when nearly all captures were adults. Interestingly, in this lagoon, during the second season, the population structure shifted, showing very few larvae and juveniles but a high number of adults. Many adults captured in this season were recaptured from the larval or juvenile stages of the previous season. These adults were likely not in breeding condition.

Since the Antiñir and Jabón lagoons are located near each other, climatic conditions were similar for both during the study. However, Antiñir is shallower, making it more

TABLE 1 | Set of models selected for density estimation (POPAN) and apparent survival (Cormack–Jolly–Seber [CJS]).

Lagoon	Period	Selected model	AICc	AICc weights	Num. par.	Deviance	−2log(L)
POPAN							
Antiñir	January 2023	phi(.g) p(.g)	195.322	0.613	12	−716.865	170.153
Antiñir	February 2023	phi(tg) p(.g)	171.214	0.842	10	−273.636	150.103
Antiñir	March 2023	phi(t) p(.)	422.415	0.724	8	−796.479	406.047
Antiñir	January 2024	phi(.g) p(.g)	191.407	0.959	9	−269.618	172.063
Antiñir	All periods	phi(.) p(t)	2490.442	1.000	29	−3961.949	2430.856
Antiñir	Albinos—All periods	phi(.) p(.)	82.149	0.859	6	−83.352	67.938
Jabón	January 2023	phi(.g) p (tg)	1081.699	0.809	19	−1472.879	1042.636
Jabón	February 2023	phi(t) p(.)	713.713	0.719	6	−369.468	701.523
Jabón	March 2023	phi(.) p(.)	512.745	0.670	4	−325.872	504.638
Jabón	January 2024	phi(.g) p(.g)	315.054	0.985	16	−795.158	281.468
Jabón	March 2024	phi(.g) p(.g)	493.218	0.931	10	−1078.104	472.764
Jabón	All periods	phi(t) p(t)	5435.006	1.000	60	−7377.943	5311.996
CJS							
Antiñir	Juveniles—First season	phi(.) p(t)	1081.924	0.999	12	154.352	1057.443
Antiñir	Adults—Second season	phi(.) p(.)	241.173	0.997	2	64.943	232.981
Antiñir	Albinos—All periods	phi(.) p(.)	46.602	0.968	2	12.330	42.302
Jabón	Juveniles—First season	phi(t) p(t)	3439.313	1.000	21	978.189	3396.587
Jabón	Juveniles and adults— Second season	phi(.g) p(.g)	799.642	0.982	4	120.215	791.577

Abbreviations: −2log(L) = −2log likelihood or model deviance; AIC = Akaike's information criterion, dot (.) = constant time variation; g = group variation; Num. par. = number of modelled parameters; p = capture probability; phi = apparent survival; t = time variation.

TABLE 2 | Subpopulation abundance (95% confidence intervals) of the Patagonia frog per group by month for each lagoon. Dashes indicate groups for which estimates could not be obtained.

Lagoon	Group	January 2023	February 2023	March 2023	January 2024	March 2024
Antiñir	Larval stages	1195 (674–2258)	103 (33–511)	—	—	—
Antiñir	Juveniles	1337 (508–3811)	471 (338–709)	726 (570–986)	—	—
Antiñir	Total adults	—	—	—	377 (239–684)	—
Antiñir	Female adults	—	—	—	219 (142–376)	—
Antiñir	Male adults	—	—	—	158 (97–308)	—
Jabón	Larval stages	1285 (578–3168)	—	—	572 (284–1321)	—
Jabón	Juveniles	587 (530–667)	268 (255–289)	274 (259–299)	765 (524–1175)	842 (669–1126)
Jabón	Total adults	—	—	—	134 (56–639)	99 (31–496)

susceptible to daily temperature fluctuations. Given that the second season was warmer than the previous summer (Rolón, pers. obs.), the lower reproductive activity in Antiñir could be attributed to this higher susceptibility. In contrast, the absence of important differences in reproductive activities between seasons in Jabón could be explained by its greater depth. This may have acted as a thermal buffer during the

second season, helping to maintain the observed reproductive activity. The low recruitment could also be due to a biennial reproductive strategy, as seen in some species (Licht 1975; Muths et al. 2010). In seasonally dry or cold regions, biennial reproduction becomes common as the high energy demands of producing eggs and limited periods for energy accumulation necessitate concentrating reproductive efforts in favourable

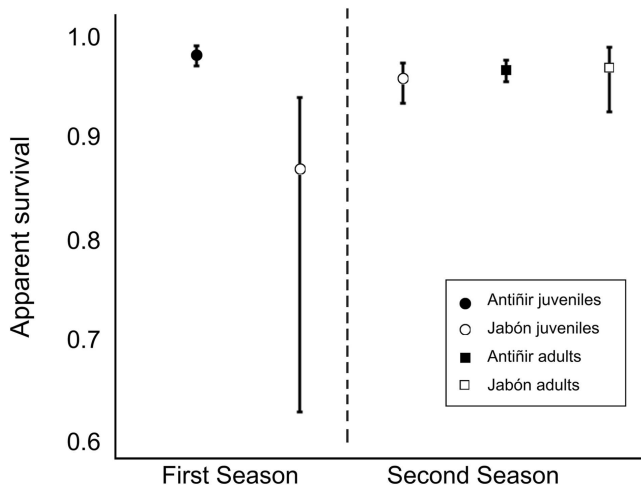


FIGURE 5 | Average and 95% confidence intervals for survival estimations in Patagonia frog subpopulations.

years to enhance survival and reproductive success (Bull and Shine 1979). In Jabón, this could be explained by alternating reproduction between at least two cohorts. Nevertheless, more comprehensive studies are needed to corroborate this hypothesis.

Regarding the effect of livestock, the Antiñir Lagoon is fenced, whereas the Jabón Lagoon is not. Among the negative impacts of livestock presence are water eutrophication caused by faeces and trampling, which alters water conditions and vegetation development (Burkholder et al. 2007). This also directly affects the Patagonia frog, as the species uses rocks as refuges, and a few individuals have been found crushed beneath them (Rolón, pers. obs.). One might expect that, given their similarities, Jabón would host a smaller subpopulation due to this threat; however, our results suggest otherwise. This discrepancy could be related to specific conditions in Jabón, such as the proportion of available microhabitats or other covariates not considered in this study. Furthermore, it is possible that livestock is not affecting the population size directly but rather influencing the species' autecology, for example, by impacting the body condition of individuals. Conducting a comparative analysis of body condition between individuals from both lagoons would be an interesting next step.

In this study, we recorded that 5% of the Antiñir subpopulation exhibited albinism, a percentage notably higher than reported in other studies on amphibians, which range from 0.000005% to 1% ($N > 5000$) as documented by Henle et al. (2017). The occurrence of a significant number of albino individuals across all life stages (larvae, juveniles and adults) may suggest that the gene responsible for albinism is frequently expressed within this subpopulation. This could be a consequence of a potential genetic bottleneck caused by the severe drought in previous years, which drastically reduced the population size (Cuellar et al. 2015), as there have been no reports of albinism in previous samplings conducted in this lagoon.

Similar to Fox et al. (2005), we also found a high percentage (97%) of frogs exhibiting the aquatic morphotype, while only 2% displayed the littoral morphotype, and the remaining individuals

showed intermediate forms, considering adult specimens from both lagoons. Given that both lagoons retained water throughout the period between seasons, it is reasonable that the aquatic morphotype was predominant. All individuals exhibiting the littoral form were found beneath rocks on land, at distances ranging from 2 to 20 m from the shoreline of the lagoon.

In Argentinian Patagonia, to date, there are no other studies on abundance and survival involving amphibian species inhabiting lentic water bodies. Velasco et al. (2019) conducted CMR studies on another Critically Endangered Patagonian species, the Valcheta frog (*P. somuncurensis*), which is micro-endemic to the Valcheta stream. They reported subpopulation abundances ranging from 220 to 4516 individuals, with an estimated density of 0.63 ± 0.15 individuals m^{-2} . While most subpopulations of *P. somuncurensis* exhibited lower abundances than those recorded for the Patagonia frog in our study, population densities were considerably higher in *P. somuncurensis*. The variability in population sizes and densities among species emphasizes the need to explore these patterns further to inform effective conservation strategies.

In other regions worldwide, Şirin et al. (2024) applied CMR methods to monitor a population of *Pelophylax bedriagae caralitanus* in Gölcük Lake, Türkiye. They estimated population sizes ranging from 4834 to 5382 individuals across successive years, comparable to our estimates for the Jabón Lagoon. However, their study focused exclusively on adult individuals, suggesting a larger population size than our results, which include larval and juvenile stages. This difference is logical, as the surface area of the lake (83 ha) is much larger than the size of the lagoons in our study. Similarly, Duarte et al. (2011) used CMR to estimate male abundance of the Endangered Houston toad (*Anaxyrus houstonensis*) in Texas, USA. These authors reported values between 201 and 307 individuals, which align with our estimates for adult individuals. Variability in population dynamics across species and life stages underscores the need for tailored conservation approaches.

Regarding survival, estimates were relatively high, with juveniles exhibiting elevated survival values, typically expected to be lower than those of adults, as observed in *P. somuncurensis* (Velasco et al. 2019). *P. bedriagae caralitanus*, a frog inhabiting lakes and ponds in Türkiye, displayed comparable high survival rates, reaching 0.98 in a specific year (Şirin et al. 2024). Nonetheless, our results were derived from the summer seasons. Between seasons, especially after winter, it is likely that many juveniles do not survive to adulthood, aligning with the low number of adults observed. In this sense, Jabón had the lowest number of adults, despite having the highest overall abundance. This difference could be attributed to the lower juvenile survival in Jabón during the first sampling season compared to Antiñir. The high survival rates of juveniles in Antiñir may have resulted in a larger number of adults from that cohort during the second sampling season. This assumption is supported by the fact that many juveniles (55 individuals) marked during the first season in Antiñir were recaptured in the second season, having matured and exhibiting secondary sexual characteristics. Males exhibited nuptial pads, and females showed visible oocytes through their skin, indicating reproductive readiness.

Although not directly evaluated in this study, the recovery of aquatic vegetation may play a significant role in individual survival and the predominance of specific life stages. Cuello et al. (2017) demonstrated that the abundance of *A. patagonicus* tadpoles and the presence of multiple ontogenetic stages were higher in areas with well-developed macrophytes (e.g., *Myriophyllum quitense*), as these provide both shelter and foraging opportunities. During our study, both lagoons exhibited reduced aquatic vegetation, which may have contributed to the lower number of individuals recorded compared to previous surveys. Individuals were frequently found in association with rocky areas, which also offer shelter and prey availability (e.g., *Hyalella* sp.), particularly along the shoreline. But with reduced aquatic vegetation, frogs may rely solely on rocks as refuge, potentially increasing their exposure to predation.

If we look ahead, the future of these two subpopulations will likely depend on climatic conditions, water availability and habitat quality. If drought events become more frequent and intense due to climate change, smaller and shallower lagoons such as Antiñir may be more severely affected. In contrast, deeper lagoons like Jabón might buffer against some environmental fluctuations but remain vulnerable to anthropogenic impacts such as livestock disturbance. Continued monitoring will be critical to detect early warning signs of decline, especially for early life stages. The current demographic trends suggest partial recovery, but the long-term viability of these subpopulations remains uncertain without sustained conservation efforts.

4.1 | Implications for Conservation

Our findings indicate signs of species recovery following the severe drought experienced in previous years (Cuello et al. 2015). However, the density values remain lower than those reported by Fox et al. (2005). Furthermore, our research focused on only two of the 23 known subpopulations of the species. It is crucial to expand monitoring efforts to include the other lagoons, especially those located outside the protected boundaries of the National Park, to assess broader recovery trends. Although the lagoons have regained water, they have not yet returned to their usual sizes, and the aquatic vegetation, which provides essential food and shelter (Cuello et al. 2017), is still in the process of recovering. In this context of habitat loss, isolation and population decline, genetic studies become particularly relevant. Martinazzo Giménez (2011) found no significant genetic differences and reported high gene flow among *A. patagonicus* subpopulations. New genetic studies would be valuable to evaluate the current status of subpopulations after the drought, particularly in Antiñir, where cases of albinism have been observed.

Currently, in collaboration with the APN, Fundación Temaikèn and the Universidad Nacional de La Plata, an ex situ breeding colony is being maintained with the goal of establishing a reproductive nucleus to supplement and reintroduce individuals into Laguna Blanca National Park. In addition, in situ efforts focus on excluding livestock from lagoons, creating threat-free sanctuaries and restoring Laguna Blanca through intensive fishing to reduce exotic fish abundance are being carried out. We strongly encourage the fencing of the Jabón Lagoon, as well as other lagoons inhabited by the Patagonia frog, to prevent

cattle entry. This management action would not only reduce disturbances caused by livestock but also promote the recovery of aquatic vegetation (e.g., Arellano et al. 2017). In turn, this would not only benefit the Patagonian frog subpopulation but also enhance the overall ecological integrity of these wetlands. The data on population structure obtained in this work provide valuable insights into the population size and survival rates during years with substantial water availability. This information serves as a foundation for comparing *A. patagonicus* abundance across other subpopulations and assessing changes over time. Finally, to ensure effective conservation actions, it is essential to develop population viability models to assess management strategies aimed at enhancing the status of this species.

Author Contributions

Melisa C. J. Rolón: investigation, writing – original draft. **Rodrigo Calvo:** writing – review and editing, investigation. **David G. Vera:** investigation, writing – review and editing. **Facundo L. De Los Santos:** writing – review and editing, investigation. **Germán Tattamanti:** investigation, writing – review and editing. **Diego O. Di Pietro:** writing – review and editing. **Federico P. Kacolis:** project administration, supervision, methodology, writing – review and editing. **Melina A. Velasco:** supervision, writing – review and editing, methodology.

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Ethics Statement

This study was conducted under Dirección Regional Patagonia Norte, APN Scientific Research Permit No. 1822. We followed the DAPTF Fieldwork Code of Practice and the guidelines outlined by Lips et al. (2001).

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The authors have nothing to report.

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