

RESEARCH ARTICLE

Habitat Expansion, but Population Isolation. The Impact of Climate Change on an Endemic Frog of Patagonia

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ABSTRACT

The Rio Negro Frog (*Atelognathus nitoi*) is an amphibian species endemic to Patagonia, shared between Argentina and Chile. Currently, it is classified as Vulnerable (IUCN) and is mainly threatened by anthropogenic pressures such as habitat alteration and climate change. Due to its life history and habitat requirements, primarily determined by temperature and precipitation, climate change may particularly influence the species' distribution. We evaluated how climate change may affect geographic range by producing a Species Distribution Model (SDM), generating a habitat suitability map and calculating its Extent of Occurrence (EOO). Projections were made for the period 2041–2060 under the Shared Socioeconomic Pathways SSP1-2.6 and SSP5-8.5, representing optimistic and pessimistic climate change scenarios, respectively. A set of five atmosphere–ocean general circulation models was used to reduce uncertainty. We evaluated the degree of connectivity of the lagoons with records of the species by applying Lin's functional linkage index, allowing comparisons in terms of their relative importance for the system's connectivity. Our results indicate an increase in suitability area over time, with a westward distribution shift in future scenarios. However, local extinctions are projected at the northern and southern margins of the species' range. Therefore, although the EOO is expected to expand, the species' life-cycle characteristics, limited dispersal capacity and low connectivity between the populations analysed should be taken into account when planning conservation actions.

1 | Introduction

Climate change is causing significant alterations in species distribution worldwide due to its impact on the physiological and ecological processes that are key to the development of organisms (Parmesan and Yohe 2003; Hansen et al. 2006; Pecl et al. 2017). If emissions continue at the current rate, it is projected that global temperatures will increase by 1.5°C above pre-industrial levels by 2050 (IPCC 2023). In this context, it is important to study current distributions of the organisms and project future changes as a fundamental basis for planning and ultimately mitigating the impacts of climate change (Araújo and

Rahbek 2006). Therefore, in recent decades, the scientific community has emphasised the urgent need for a new framework for biodiversity conservation that considers changes in species distribution as a consequence of climate change (CBD 2022).

The Species Distribution Model (SDM), also known as ecological niche modelling, utilises species occurrence data combined with environmental predictors through statistical or machine-learning techniques to generate spatial predictions about the distribution of species (Franklin 2011; Peterson et al. 2011). One of its applications is to estimate habitat suitability under current environmental conditions and then project these models onto

future climate change scenarios (e.g., SSP1-2.6 and SSP5-8.5). Such projections enable the prediction of potential changes in species distributions and the evaluation of their degree of protection (Araújo and Rahbek 2006; Guisan et al. 2013). This is especially valuable for little-known, endemic or threatened species (van Proosdij et al. 2016) and has been widely utilised to plan conservation strategies by identifying priority areas (Di Pietro et al. 2025) and anticipating biological invasions (Vera, Berkunsky, et al. 2025).

However, the use of SDM faces significant methodological challenges. Among the most relevant are biases in occurrence data (e.g., incomplete sampling or sampling concentrated in accessible areas), the choice of environmental variables that are not representative of the ecological niche, model overfitting, lack of independent validation and the frequent assumption that species are in equilibrium with their environment (Araújo et al. 2019). Despite these challenges, SDM remains a key tool in conservation biogeography, especially when complemented by field validation and local ecological knowledge. Even overpredictions may indicate possible species expansions beyond their native ranges or under different climate change scenarios.

Amphibians are considered good models for assessing the potential impacts of climate change, as their biphasic life cycle depends on aquatic habitats during larval stages and on terrestrial environments with humid microhabitats in adulthood to sustain efficient cutaneous gas exchange (Duellman and Trueb 1994; Stuart et al. 2004; Wells 2007; Taylor et al. 2021). Today, this taxon is among the most threatened vertebrates globally, with 36% of the 8051 assessed species classified as internationally threatened and at least 37 species considered extinct (IUCN 2025).

In the southern cone of South America, approximately half of a total of 57 amphibian populations assessed between Argentina and Chile are experiencing decline, and one-third are undergoing local extinction events, meaning the disappearance of local populations or subpopulations (Kacoliris et al. 2022). The Patagonia region is shared between Argentina and Chile. Although it has a low richness of anuran species compared to the tropical or subtropical regions of South America, many of them are endemic, threatened, and have an important conservation value (Úbeda and Grigera 2007; Kacoliris et al. 2022; Rolón et al. 2024). In Argentina, the Patagonian Steppe is the ecoregion with the highest percentage (75%) of threatened taxa, followed by the Patagonian Forests (65%) (Vaira et al. 2017).

The Rio Negro frog (*Atelognathus nitoi*; Barrio 1973) preferentially inhabits wetlands around small or temporary ponds (Basso 1998; Díaz-Páez et al. 2011; Mella-Romero et al. 2022). The species is mainly threatened by livestock grazing, fires and habitat fragmentation, while projected climatic changes may further exacerbate habitat loss and isolation among populations. Currently, it is categorised as Vulnerable both internationally (IUCN SSC Amphibian Specialist Group 2019; hereafter IUCN) and nationally in Argentina (Úbeda and Basso 2012). In addition, the species has been included in Argentina within the 'Species of native vertebrates of

special value' (Administración de Parques Nacionales 1994). In Chile, this species, currently the only representative of the genus in the country, has not yet been assessed under the Species Classification Regulation of the Ministerio del Medio Ambiente de Chile (Barrasso and Basso 2019; Correa 2019; Figueroa-Ponce and Moya 2023).

At the time of assessment, *A. nitoi* was considered endemic to Argentina, as it had only been documented at a single locality, *Laguna Verde*, in the Río Negro province (IUCN). Subsequently, it was synonymised with three other species (*A. cei*, *A. salai*, *A. jeinimenensis*). This means that conservation status assessments are outdated, as they do not incorporate recent information on taxonomy and population trends (Barrasso and Basso 2019; Bonino et al. 2022; Figueroa-Ponce and Moya 2023). Assessing the degree of connectivity among these populations is crucial for determining whether individuals move or disperse between them, thereby contributing to gene flow, demographic stability and the long-term viability of the species (Ray et al. 2002; Semlitsch 2008; Velasco et al. 2019).

In Patagonia, climate projections indicate a general increase in temperature and a regional decrease in precipitation, although eastern areas could experience more variable trends (Barros et al. 2015; IPCC 2023). While reduced precipitation would generally limit species that depend on water, warmer temperatures could allow colonisation of areas that were previously too cold.

Within this framework, the present study aims to assess the current and future distribution of *A. nitoi* and evaluate the functional connectivity between its populations. Specifically, we analyse whether habitats located between known populations present environmental conditions suitable for the species, which can act as natural corridors or even support populations. To achieve this, we set the following objectives: (1) identify the bioclimatic variables that influence the current distribution of suitable habitats; (2) project these models onto future climate scenarios (SSP1-2.6 and SSP5-8.5, 2041–2060) and evaluate the expected changes in suitable areas, including shifts in distribution (latitude, longitude) and changes in total extent; (3) estimate the Extent of Occurrence (EOO) under current and future conditions; and (4) assess the degree of functional connectivity between populations.

Based on the ecology of *A. nitoi*, we hypothesise that: (H1) the extent and spatial configuration of suitable habitat will change under future climate scenarios due to increasing temperatures and decreasing precipitation across Patagonia. Some lagoons currently occupied by the species are expected to become climatically unsuitable because of reduced rainfall, whereas a westward expansion of suitable habitats may occur as rising temperatures enable colonisation of areas that are currently too cold (Barros et al. 2015; IPCC 2023); (H2) intermediate areas between populations will include climatically suitable habitats, potentially acting as dispersal corridors or sustaining populations; and (H3) the functional connectivity among lagoons will remain low, given the geographic distances between known populations, which may restrict gene flow and reduce long-term persistence (Ray et al. 2002; Semlitsch 2008; Velasco et al. 2019).

2 | Methods

2.1 | Study Area

The study area encompasses the temperate Patagonian Forest ecoregion, which extends widely in Chile, reaching a narrow strip in western Argentina in association with the Andes Mountain Range (Paruelo et al. 1998) (Figure 1). This ecosystem is adjacent to the extra-Andean Patagonian steppe (Morrone 2015; Roig-Juñent et al. 2018). The region is characterised by cold, dry climates, with precipitation concentrated mainly in the winter season. The water bodies are located on an elevation gradient from 210m (*Puerto Chacayal*) to 1538m (*Laguna Verde*), with

some lagoons located in lower areas near the steppe and others surrounded by mountains, resulting in high climatic variability (Villalba and Veblen 1997).

2.2 | Occurrence Data

Species occurrence data were obtained from the exhaustive review of the scientific literature carried out by Figueroa-Ponce and Moya (2023), which was expanded by searching for records in the citizen science databases Global Biodiversity Information Facility (GBIF—<https://www.gbif.org/>), iNaturalist (www.inaturalist.org) and Vertnet (<https://vertnet.org>). We used 27

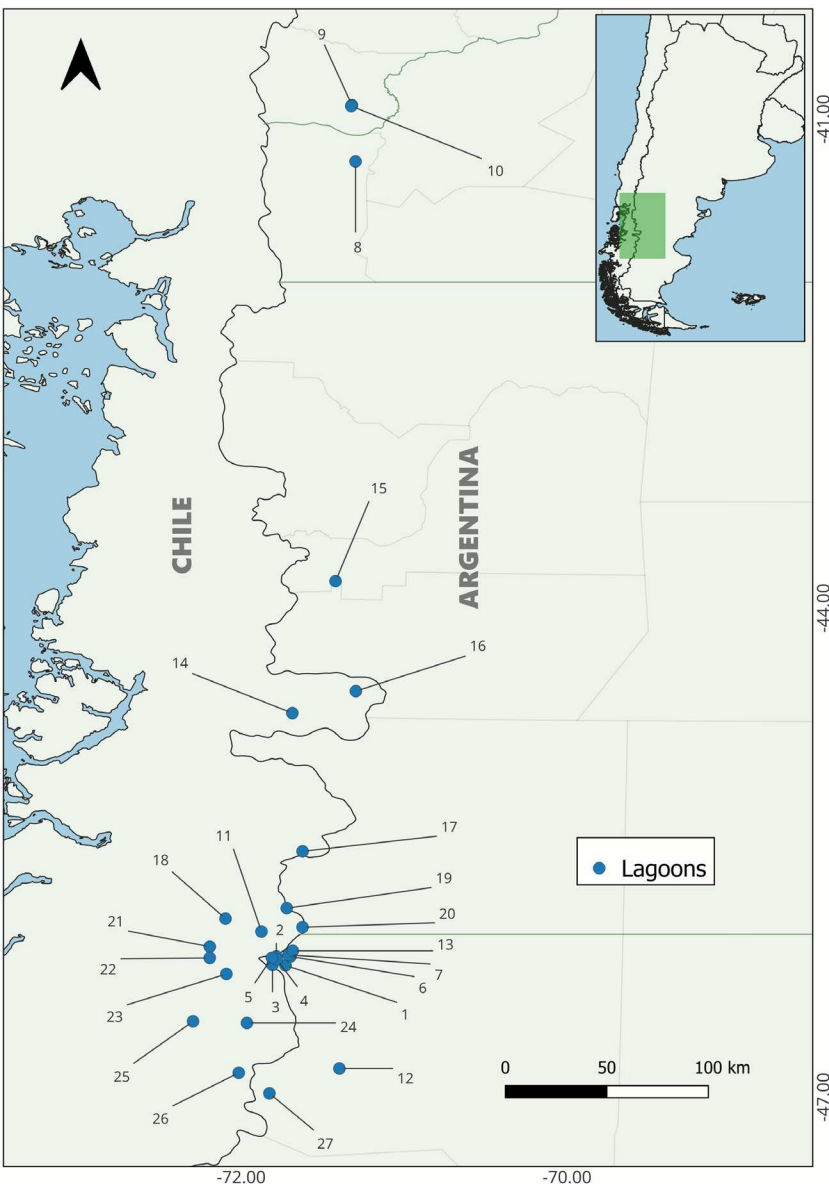


FIGURE 1 | Study area. The border with Chile and the limits of the Argentine provinces included in the study are shown. Blue dots indicate the lagoons: 1–7. Without name (near El Portezuelo); 8. Laguna Verde (Bariloche); 9 and 10. Without name (near Cerro Saihueque); 11. Laguna Chiguay (Villa Cerro Castillo); 12. Laguna del Sello (Meseta Lago Buenos Aires); 13. Laguna de los Gendarmes (near Paso Río Mosco); 14. Without name (near La Tapera); 15. Without name (near Lago General Vintter); 16. Without name (near Alto Río Cisnes); 17. Without name (near Coyhaique Alto); 18. Without name (near La Paloma); 19 and 20. Without name (near Balmaceda); 21. Without name (near Cerro Castillo); 22. Without name (near Las Ardillas); 23. Lago Lapparent (Río Ibáñez); 24. Without name (near Chile Chico); 25. Without name (near Puerto Chacayal); 26. Lago Jeinimeni (Chile Chico); 27. Without name (near Monte Zeballos).

TABLE 1 | Points of occurrence of *Atelognathus nitoi* used for models.

Latitude	Longitude	References
−46.1907	−71.7279	GBIF; VERTNET
−46.1359	−71.7846	GBIF; VERTNET
−46.1881	−71.8083	GBIF; VERTNET
−46.1551	−71.7812	GBIF; VERTNET
−46.1441	−71.8116	GBIF; VERTNET
−46.1362	−71.6975	GBIF; VERTNET
−46.1265	−71.7098	GBIF; VERTNET
−40.9201	−71.3244	GBIF
−41.3050	−71.2243	Barrio (1973); Figueroa-Ponce and Moya (2023); VERTNET; GBIF
−40.9146	−71.3208	GBIF; Figueroa-Ponce and Moya (2023)
−45.9848	−71.8752	Núñez and Gálvez (2015), Figueroa-Ponce and Moya (2023)
−46.8236	−71.3974	GBIF; iNaturalist; Figueroa-Ponce and Moya (2023)
−46.1014	−71.6834	Cei (1984); Figueroa-Ponce and Moya (2023)
−44.6443	−71.6841	Basso (1998); Figueroa-Ponce and Moya (2023); VERTNET
−43.8341	−71.4208	Barrasso and Basso (2019); Baldo et al. (2020); Figueroa-Ponce and Moya (2023)
−44.4961	−71.2744	Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023)
−45.4833	−71.6166	Raimilla (2015); Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023); GBIF
−45.8583	−71.9375	iNaturalist; Figueroa-Ponce and Moya (2023)
−45.8855	−71.7013	Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023)
−45.9349	−71.6424	GBIF, iNaturalist; Figueroa-Ponce and Moya (2023)
−46.0763	−72.1929	Díaz-Páez et al. (2011); Figueroa-Ponce and Moya (2023)
−46.1491	−72.1963	Núñez and Gálvez (2015); Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023)
−46.2411	−72.0919	Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023)
−46.5333	−72.0000	Alveal and Díaz-Páez (2021); Mella-Romero et al. (2022); Figueroa-Ponce and Moya (2023)

(Continues)

TABLE 1 | (Continued)

Latitude	Longitude	References
−46.5333	−72.2666	Díaz-Páez et al. (2011); Figueroa-Ponce and Moya (2023)
−46.8496	−72.0152	Meriggio et al. (2004); Figueroa-Ponce and Moya (2023)
−46.9774	−71.8282	Scolaro et al. (2008); Barrasso and Basso (2019); Figueroa-Ponce and Moya (2023)

Note: The coordinates (latitude, longitude) of the points of occurrence of *Atelognathus nitoi* are provided.

verified records, comprising both museum collections and citizen science databases that provided photographs suitable for reliable identification (see Table 1). To develop the species distribution models (see below), we removed duplicate presence points within each cell (~1 km²) to avoid model overfitting and reduce sampling biases (Phillips et al. 2006). The geographic coordinates used in this study are expressed in the WGS84 geodetic reference system (EPSG:4326).

2.3 | Selection and Processing of Environmental Variables

Nineteen bioclimatic variables from the WorldClim database version 2.1 (<http://www.worldclim.org>) were used, with the highest resolution available (~1 km² per cell), representing annual, seasonal trends and extreme climatic values.

All variables were scaled and cropped to the area of interest using QGIS software version 3.40 (QGIS Development Team 2024). To reduce multicollinearity among predictors, we applied a Variance Inflation Factor (VIF) analysis using the function `correct_colinvar` from the R package `flexsdm` (Naimi and Araújo 2016; Velazco et al. 2022), with a threshold value of 10. Variables with VIF > 10 were excluded iteratively until all remaining predictors met the collinearity criteria. Seven uncorrelated bioclimatic variables were retained for the species distribution models: Annual Mean Temperature (Bio1), Isothermality (Bio3), Temperature Seasonality (Bio4), Mean Temperature of Wettest Quarter (Bio8), Mean Temperature of Driest Quarter (Bio9), Precipitation of Driest Month (Bio14) and Precipitation Seasonality (Bio15) (Table 2).

2.4 | Species Distribution Modelling and Potential Sampling Areas

We generated species distribution modelling in current and future scenarios with Maxent software version 3.4.4 (Phillips et al. 2006; Phillips and Dudik 2008), which employs an algorithm based on the concept of maximum entropy to predict the suitable habitat for species in each landscape unit. Compared to other techniques, Maxent has demonstrated quantitatively superior performance (Elith et al. 2006; Hernandez et al. 2006; Elith and Leathwick 2009). Although presence-absence methods have become increasingly prominent due to their demonstrated

TABLE 2 | Bioclimatic variables and VIF results.

Code	Variable	Units	Mean	Range	Deviation	VIF
				min—max		
Bio1	Annual Mean Temperature	°C*10	11.25	−4.76 to 20.33	4,39	9.28
Bio2	Mean Diurnal Range	°C*10	12.55	4.96 to 17.66	2,76	> 10
Bio3	Isothermality	°C*10	48.16	35.16 to 59.42	2,99	2.25
Bio4	Temperature Seasonality	°C*10	505.14	232.74 to 677.58	95,18	5.29
Bio5	Max Temperature of Warmest Month	°C*10	25.20	5.40 to 35.20	6,37	> 10
Bio6	Min Temperature of Coldest Month	°C*10	−0.76	−14.50 to 7.90	2,99	> 10
Bio7	Temperature Annual Range	°C*10	25.96	12.10 to 34.80	5,24	> 10
Bio8	Mean Temperature of Wettest Quarter	°C*10	10.95	−8.55 to 27.25	8,27	3.60
Bio9	Mean Temperature of Driest Quarter	°C*10	10.66	−5.95 to 21.90	4,50	7.26
Bio10	Mean Temperature of Warmest Quarter	°C*10	17.41	0.06 to 27.25	5,22	> 10
Bio11	Mean Temperature of Coldest Quarter	°C*10	4.97	−9.63 to 13.28	3,74	> 10
Bio12	Annual Precipitation	mm	610.68	122 to 4660	659,29	> 10
Bio13	Precipitation of Wettest Month	mm	86.12	16 to 460	82,67	> 10
Bio14	Precipitation of Driest Month	mm	24.79	0 to 27.80	38,80	5.74
Bio15	Precipitation Seasonality	mm	42.79	7.76 to 116.05	22,04	3.45
Bio16	Precipitation of Wettest Quarter	mm	237.07	41 to 1330	236,92	> 10
Bio17	Precipitation of Driest Quarter	mm	85.47	0 to 1002	130,58	> 10
Bio18	Precipitation of Warmest Quarter	mm	138.43	0 to 1300	157,47	> 10
Bio19	Precipitation of Coldest Quarter	mm	173.68	6 to 1133	232,37	> 10

Note: We indicate code, variable description, units, mean, range, standard deviation, and variance inflation factor (VIF) values for each variable. Variables retained for modelling had VIF < 10, whereas variables not selected are indicated as > 10. Bold values indicate the predictor variables selected for the final distribution models and their corresponding VIF scores.

predictive performance in recent comprehensive evaluations (e.g., Norberg et al. 2019), Maxent remains one of the most widely used tools for species distribution modelling when only presence data are available (Elith and Leathwick 2009; Franklin 2011). Notably, field validations, such as those in West et al. (2016) and Johnson et al. (2023), demonstrate that predictions produced by Maxent often align with independent field surveys, making it especially valuable for studies of rare, cryptic or data-deficient species.

The predictive capacity of the models was maximised by selecting linear and quadratic adjustment features (polynomials) for the environmental variables, a maximum of 2500 background points, a regularisation multiplier of 0.25, and by removing duplicate presence records within each cell. We assessed variable importance in SDM by using the Jackknife test implemented in Maxent, which estimates the relative contribution of each predictor to model performance. The value of the regularisation multiplier and the types of features were chosen after an initial assessment of model performance using the tune_max function of the flexsdm package (Velazco et al. 2022). We considered both the area under the curve (AUC) statistic (Phillips et al. 2006; Phillips and Dudik 2008; Anderson and Gonzalez 2011; Shcheglovitova and Anderson 2013) and the True Skill Statistic (TSS; Allouche et al. 2006; Hirzel et al. 2006) to evaluate model

performance. While AUC provides a threshold-independent measure of discriminatory power, TSS incorporates sensitivity and specificity, allowing us to minimise false negatives that could lead to overlooking potentially suitable habitats. In addition, we applied null models through the nullRandom function of the dismo package in R (R Core Team 2025; Hijmans et al. 2023). We obtained the AUC for the null models (10 replicates) and then calculated the *p* value using the *t*-test (Sillero et al. 2021). To convert the continuous suitability predictions into a binary classification (suitable vs. unsuitable habitat) and estimate the extent of potential habitat, we used the ‘equal test sensitivity and specificity’ criterion as the cut-off threshold in R. This approach was chosen because it symmetrically balances omission and commission errors, is computed using independent test data (thus reducing the risk of overfitting), and is easily reproducible (Liu et al. 2005).

To model future scenarios for 2041–2060 under Shared Socioeconomic Pathways (SSP1-2.6 and SSP5-8.5), to develop optimistic and pessimistic climate change scenarios, respectively (e.g., Minoli and Avila 2017; Gómez Cruz et al. 2021), we considered multiple Atmosphere–Ocean General Circulation Models (AOGCMs) from the WorldClim database version 2.1, which provides CMIP6-based climate projections. These models were used to account for variability and reduce

TABLE 3 | Model performance for present and future distributions of *Atelognathus nitoi*.

	TSS	AUC	<i>t</i> -statistic	df	<i>p</i>	IC (95%)	Mean AUC of the null model replicates	AUC mean of model replica
Current	0.706	0.839	−11.174	9	<0.001	[−inf—0.57]	0.522	0.839

uncertainty (Araújo and New 2007). Specifically, we projected the distribution of *A. nitoi* under five GCMs (ACCESS-CM2, BCC-CSM2-MR, CMCC-ESM2, HadGEM3-GC31-LL, MPI-ESM1-2-HR). The selection of these GCMs was based on their frequent application in ecological and biogeographical studies across South America and on their evaluated performance in simulating regional climate variables at global and continental scales (Eyring et al. 2016; Müller et al. 2018; Bi et al. 2020; Reboita et al. 2024). We then integrated the results using an ensemble approach, considering as suitable those areas predicted by the majority of models (≥ 3 out of 5). To evaluate changes in the total area of suitable habitat, we overlaid the current and future ensemble projections (SSP1-2.6 as optimistic and SSP5-8.5 as pessimistic scenarios) in QGIS. Overlapping regions indicate persistent presence from the present to 2060, whereas non-overlapping regions represent potential future habitat loss or gain (Zank et al. 2014). A detailed summary of the modelling process following the ODMAP protocol (Zurell et al. 2020) is provided in the [Supporting Information](#).

We followed the IUCN (2012) recommendation to assess the potential conservation of the species under current and future climate change scenarios. For this purpose, we based our assessments on the EOO, assuming that it is equivalent to the total area of suitable habitat derived from the models in the current and future scenarios.

Based on the area of suitable habitat modelled for the current scenario in Maxent, a list of localities with no species records was drawn up, selected for their proximity to known records of the species and their location.

2.5 | Connectivity Assessment

We assumed each known presence point within a 1 km² cell as a population, occupying a discrete habitat patch separate from other groups of individuals. Coordinates were then adjusted by relocating each point to the nearest lagoon (Figure 1) using Google Earth Pro. Additionally, the area of each lagoon was calculated.

To assess the relative degree of connectivity of each lagoon, we applied Lin's (2009) functional linkage index, modified by Kacoliris et al. (2019): $CI = \sum Ai*(1/Di)$; where Ai represents the areas of the lagoons neighbouring the habitat patch of interest, and Di is the distances to those lagoons. This index assumes that connectivity is more significant when the distance between lagoons is smaller and the size of the lagoon is larger; negative values reflect lower connectivity (more isolated gaps), and positive values indicate higher connectivity (presence of larger and/or closer gaps). The analysis was conducted using the geosphere package (Hijmans 2024) in R. To facilitate interpretation and allow comparisons across sites,

TABLE 4 | Percentage contribution of the environmental variables under current conditions and future scenarios (2041–2060).

Code	Variables	%
Bio1	Annual mean temperature	17.4
Bio3	Isothermality	14.4
Bio4	Temperature seasonality	10.3
Bio8	Mean temperature of wettest quarter	12.5
Bio9	Mean temperature of driest quarter	5.1
Bio14	Precipitation of driest month	34.5
Bio15	Precipitation seasonality	5.7

the index was normalised by dividing each value by the maximum observed value, producing a relative scale from 0 to 1. In this way, the connectivity of each lagoon with all other lagoons was calculated.

3 | Results

3.1 | Species Distribution Modelling and Potential Sampling Areas

The models calibrated under current climatic conditions showed a good predictive performance, with an average AUC of 0.839 (± 0.120) and a TSS of 0.706 (± 0.080). These values indicate that the models achieved a robust capacity to discriminate between suitable and unsuitable habitats for *A. nitoi*. The null models showed significant differences in the *t*-test ($p < 0.05$), highlighting the robustness of the species distribution models obtained (Table 3).

The Jackknife analysis identified the environmental variables contributing most to the model (Table 4). The most informative variables for generating potential distribution models were Bio14, which contributed 34.5%, and Bio1, which contributed 17.4%.

The models generated in Maxent showed suitable habitat areas for *A. nitoi* belonging to the Andean-Patagonian Forest ecoregion distributed across Argentina and Chile. The suitable habitat area (0.5992 threshold), which we assumed to represent the species' EOO, is 18 575.27 km² under the current scenario (Figure 2A).

The ensemble projections under the optimistic (SSP1-2.6) and pessimistic (SSP5-8.5) scenarios revealed clear spatial differences in the extent of suitable habitat (Figure 2B,C). The comparison between the scenarios showed an increase of about 24.7% under the optimistic scenario with respect to the current

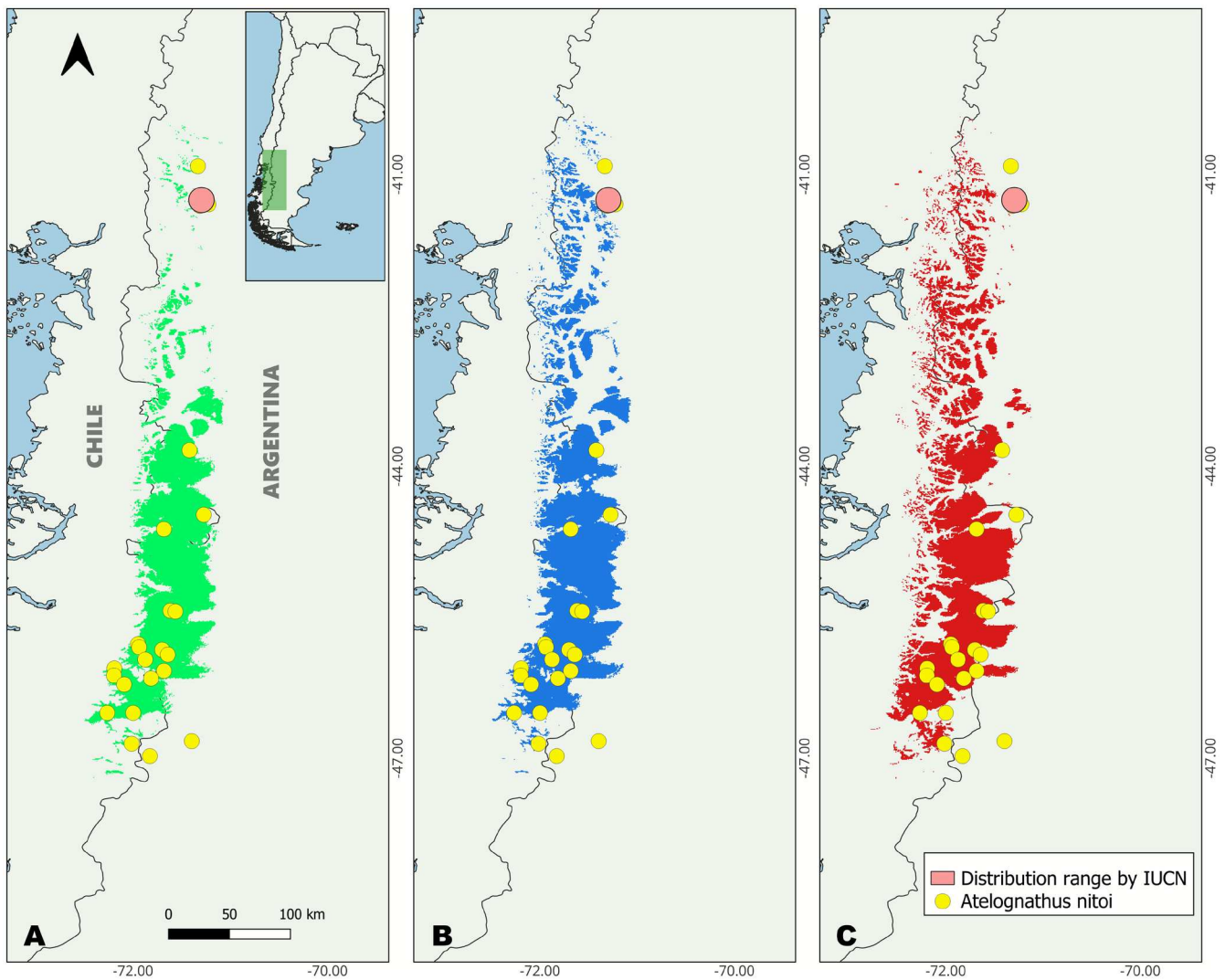


FIGURE 2 | Suitable habitats under current and future scenarios. (A) Current scenario (indicated in green). Yellow dots indicate known occurrence records of the species. (B) Optimistic SSP1-2.6 scenario in blue. (C) Pessimistic SSP5-8.5 scenario in red.

suitable habitat, whereas under the pessimistic scenario, suitable habitat is expected to increase by 17.9% by 2060.

Based on the generated models, under the current scenario and SSP1-2.6, the southern populations of *A. nitoi*, corresponding to *Laguna del Sello* (lagoon 12; 46.82°S, 71.39°W) and lagoon 27 (near Monte Zeballos; 46.97°S, 71.82°W), are predicted to be lost. Under SSP5-8.5, additional losses are expected, including lagoon 16 (near Alto Río Cisnes; 44.50°S, 71.29°W), as well as the northernmost populations of the species, such as lagoons 9 and 10 (near Cerro Saihueque; 40.91°S, 71.32°W; 40.92°S, 71.32°W) and *Laguna Verde* (lagoon 8; 41.26°S, 71.29°W; Table 5).

We obtained 14 potential sampling localities provided by the results of the SDM on suitable habitat (Figure 3). In these localities, there are no records of the species, so it would be interesting to focus sampling efforts in these suggested localities, considering the preference of the species for wetlands around small or temporary ponds.

3.2 | Connectivity Assessment

We obtained 27 lagoons, assumed to be local populations. The studied lagoons presented different trend results, reflecting the degree of functional connectivity of each one with the rest of the system (Table 5). *Laguna Verde* (lagoon 8), the type locality, is the most isolated from the system (CI=0), together with lagoons 9 and 10 (located in Cerro Saihueque) located further north of the geographic distribution of the species. *Laguna Chiguay* (lagoon 11) is the most connected of the system (CI=1), corresponding to what was formerly known as *A. salai*.

4 | Discussion

Our results indicate that the current modelled suitable habitat (18 575 km²) for this species does not exceed the 20 000 km² threshold used by the IUCN to categorise the species as threatened. This contrasts with other studies based on minimum

TABLE 5 | Connectivity index for *Atelognathus nitoi* calculated according to Lin (2009), with modifications by Kacolis et al. (2022). For each lagoon, the province of origin, the size of the area, the coordinates and the normalised connectivity index (CI) are described as a measure of the contribution of each lagoon to the system. Lagoon numbers correspond to those cited in the text.

	Lagoons	Province	Area (m ²)	Latitude	Longitude	CI
1	Without name (near El Portezuelo)	Santa Cruz	24	−46.1907	−71.7279	0.6
2	Without name (near El Portezuelo)	Santa Cruz	53	−46.1359	−71.7846	0.7
3	Without name (near El Portezuelo)	Santa Cruz	90	−46.1881	−71.8083	0.7
4	Without name (near El Portezuelo)	Santa Cruz	40	−46.1551	−71.7812	0.7
5	Without name (near El Portezuelo)	Santa Cruz	34	−46.1441	−71.8116	0.7
6	Without name (near El Portezuelo)	Santa Cruz	34	−46.1362	−71.6975	0.6
7	Without name (near El Portezuelo)	Santa Cruz	35	−46.1265	−71.7098	0.7
8	Laguna Verde (Bariloche)	Río Negro	3603	−41.2603	−71.2982	0
9	Without name (near Cerro Saihueque)	Neuquén	5301	−40.9146	−71.3208	0
10	Without name (near Cerro Saihueque)	Neuquén	2007	−40.9201	−71.3244	0
11	Laguna Chiguay (Villa Cerro Castillo)	Aysén	40 508	−45.9832	−71.8758	1
12	Laguna del Sello (Meseta Lago Buenos Aires)	Santa Cruz	127	−46.8236	−71.3974	0.4
13	Laguna de los Gendarmes (near Paso Río Mosco)	Santa Cruz	67 081	−46.1014	−71.6834	0.6
14	Without name (near La Tapera)	Aysén	3742	−44.6436	−71.6861	0.1
15	Without name (near Lago General Vintter)	Chubut	42 858	−43.8341	−71.4208	0.1
16	Without name (near Alto Río Cisnes)	Aysén	1565	−44.5087	−71.2963	0.1
17	Without name (near Coyhaique Alto)	Aysén	14 773	−45.4910	−71.6235	0.3
18	Without name (near La Paloma)	Aysén	7 620 000	−45.9041	−72.0964	0.2
19	Without name (near Balmaceda)	Aysén	319 683	−45.8400	−71.7207	0.6
20	Without name (near Balmaceda)	Aysén	78 871	−45.9564	−71.6237	0.6
21	Without name (near Cerro Castillo)	Aysén	194 317	−46.0763	−72.1929	0.9
22	Without name (near Las Ardillas)	Aysén	25 766	−46.1451	−72.1932	0.8
23	Lago Lapparent (Río Ibáñez)	Aysén	78 478	−46.2433	−72.0911	0.7
24	Without name (near Chile Chico)	Aysén	4 000 000	−46.5439	−71.9649	0.4
25	Without name (near Puerto Chacayal)	Aysén	8 459	−46.5338	−72.2955	0.7
26	Lago Jeinimeni (Chile Chico)	Aysén	3 810 536	−46.8496	−72.0152	0.3
27	Without name (near Monte Zeballos)	Santa Cruz	1 270	−46.9757	−71.8268	0.6

Note: Bold values indicate the highest CI values among lagoons.

convex polygons (27 662 km²; Figueroa-Ponce and Moya 2023), which have long been criticised because they consider a continuous and homogeneous habitat between presence records (Burgman and Fox 2003).

However, in both future scenarios (SSP1-2.6 and SSP5-8.5), the distribution undergoes a marked spatial change. We project an increase in suitable habitat (23 165 km² under SSP1-2.6; 21 913 km² under SSP5-8.5) consistent with previous studies indicating that, in certain regions, amphibians will increase their ranges due to increased temperature and precipitation (Toranza et al. 2016), although in Patagonia this expansion is likely driven primarily by rising temperatures. Under the

SSP1-2.6 scenario, the southern populations of *A. nitoi*, corresponding to *Laguna del Sello* and lagoon 27 (near Monte Zeballos), are predicted to be lost. In the pessimistic SSP5-8.5 scenario, suitable areas shift markedly westwards towards the Andes mountain range, leading to the predicted loss of populations in the southern (e.g., *Laguna del Sello* and lagoon 27) and northern margins of the species' range (e.g., lagoons 9 and 10 near Cerro Saihueque, lagoon 16 near Alto Río Cisnes, and *Laguna Verde*). This pattern aligns with the broader expectation that climate change drives directional changes in range producing movements towards the poles or upslope (Chen et al. 2011; VanDerWal et al. 2013; Lenoir and Svenning 2015). It is worth mentioning that the pessimistic

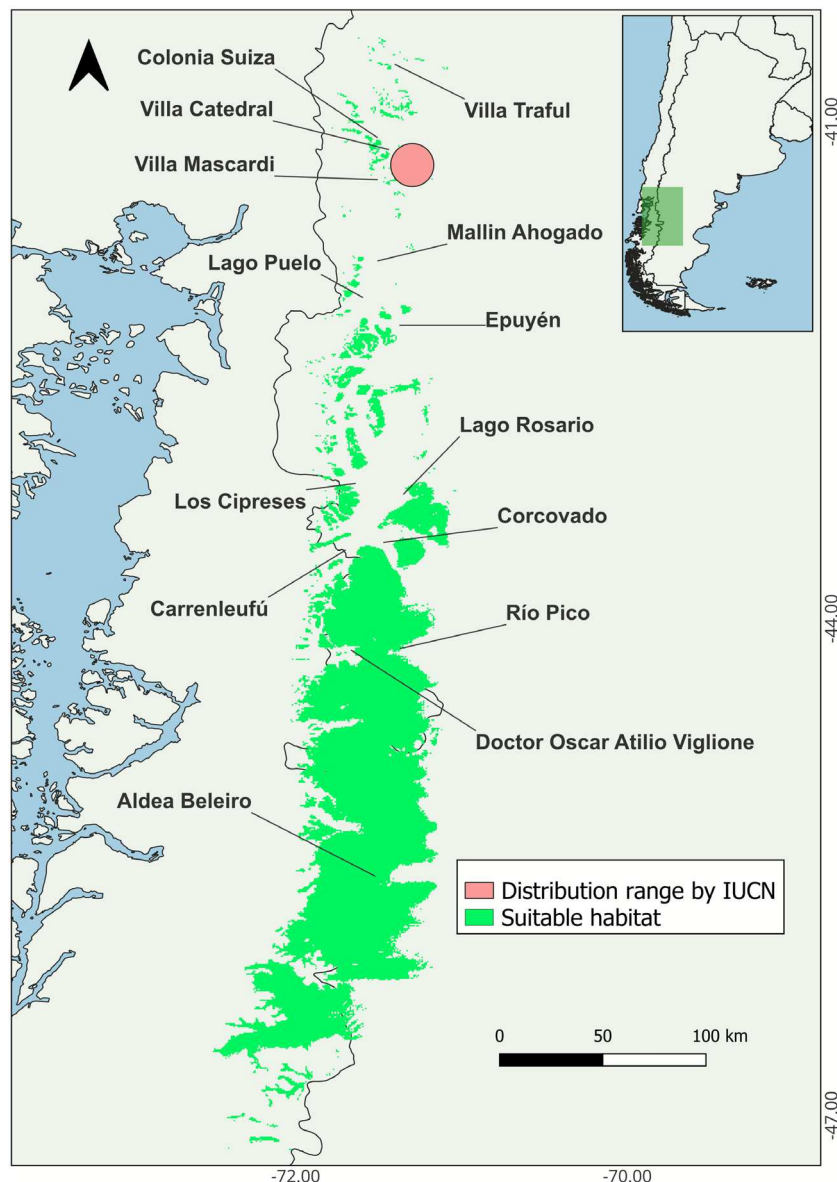


FIGURE 3 | Potential sampling sites. Suggested localities for sampling in search of new populations of *A. nitoi*, projected on the generated model of suitable habitat in the current scenario (indicated in green).

scenario has become a conservative model at present, as it includes several downward biases (e.g., indirect emission sources), making it more likely to reflect the expected climate change in the region compared to the optimistic scenario (see Ferrante et al. 2023).

The main explanation for the predicted expansion in amphibian distribution lies in its close relationship with climate. The projected temperature increase would create favourable conditions for geographic expansion into areas that were previously too cold for the species (Carey and Alexander 2003), as predicted in our analysis. Due to numerous factors (such as geographical barriers to dispersal, biotic interactions and human modification of the environment), few species occupy all the areas compatible with their niche requirements (Ray et al. 2002; Araújo et al. 2006). The modelling does not take into account the extent of urban sprawl and land use, which may negatively influence population survival. Considering this, two critical situations must be

addressed despite the increase in the estimated area of suitable habitat: (1) the dispersal capacity of individuals and their ability to colonise new habitats, and (2) the relative contribution of each population to the system.

Various hypotheses have been formulated regarding species dispersal (Holt 1990). For this taxon, in general, the most appropriate hypothesis is that the species will not disperse in the face of increasing habitat fragmentation (e.g., Araújo et al. 2006). Due to the distinctive characteristics of amphibian life cycles, breeding sites are surrounded by areas where migration is potentially possible. These areas may consist of a network of suitable terrestrial habitat patches interspersed with unsuitable patches that must be crossed. Successful migration depends on the presence of additional suitable breeding pools within this area (Ray et al. 2002). However, isolated populations, as currently understood for *A. nitoi*, may face challenges depending on the rate of environmental change and threats. Therefore, it should

be considered that an adequate net area in km² in the model will not necessarily be aligned with the success of the species in colonising new territory over time and maintaining a gene flow that allows for the perpetuation of the species or their populations over time. Species interactions under climate change are likely to be altered, potentially affecting potential distributions rather than the species' self-ecology (Peterson 2006).

Water bodies with high connectivity values play a key role in the metapopulation structure of the species (Velasco et al. 2019). Conservation and monitoring efforts should be differentially prioritised towards those populations that will be most affected by climate change to protect the species efficiently. On the other hand, water bodies with low connectivity may limit the flow of individuals between different populations, limiting the ability to recolonise from other populations, which increases the risk of local extinction and low genetic variability. Therefore, ponds with a low CI require more attention and specific management strategies to mitigate their isolation and, if possible, improve their connectivity, allowing the dispersal of individuals and increasing the resilience of the population.

We recommended increased sampling efforts aimed at identifying new populations, with a focus on areas of suitable habitat identified in this study, particularly those near currently known populations with low connectivity. Among the 14 localities situated within the predicted suitable habitat, we particularly highlight Carrenleufú, Corcovado and Río Pico, as these sites lie in intermediate areas between known distribution points and could provide important data on the connectivity of the system and the expansion of the species. Sampling efforts at these localities would be fundamental to confirm the presence or absence of the species, which would provide key information for future conservation strategies.

In summary, our results support hypotheses H1 and H3, indicating that the extent and spatial configuration of suitable habitat for *A. nitoi* vary significantly under future scenarios and that current connectivity among lagoons is low. Hypothesis H2 is only partially supported: although climatically suitable areas emerge in intermediate regions that could act as dispersal corridors or sustain populations, the pronounced westward shift of suitable conditions, together with the species' limited dispersal capacity, may restrict the colonisation of new habitats and lead to the potential loss of populations.

Finally, we emphasise the importance of considering local populations, rather than species, for conservation efforts because the threats faced by them vary significantly among local populations of the same species and because a population-level conservation approach is less influenced by taxonomic changes (Lindenmayer and Burgman 2005). For example, *Laguna Verde*, the population at the type locality, is the only permanent water body within a Natural Protected Area in Argentina where *A. nitoi* tadpoles are observed annually (Úbeda et al. 1999), emerging as a critical habitat for the viability of the species. However, this lagoon faces several threats, including impacts from livestock, forest fires, increased ultraviolet radiation, firewood extraction, and the presence of horses, underlining the urgent need for specific management measures (Bonino et al. 2022). For each lagoon, alternative conservation strategies could be evaluated using the

Conservation Priorities Method (Kacoliris et al. 2016, 2019; Vera, Jones, et al. 2025). This approach focuses on prioritising area-based units, such as individual lagoons, to guide conservation efforts at a local scale. It also includes complementary conservation actions such as habitat restoration, environmental education, legislation and the design of species recovery programmes, in addition to the planning of natural reserves.

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Ethics Statement

The authors have nothing to report.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Supporting Information

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