



Range-wide monitoring reveals demographic vulnerability in Lahille's bottlenose dolphins

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ARTICLE INFO

Keywords:

Monitoring
Mark-recapture models
Population viability analysis
Metapopulation framework
Anthropogenic threats

ABSTRACT

Robust demographic estimates are critical for effective conservation but often lacking for threatened species, demanding long-term, range-wide monitoring to generate data and assess risk. Here, we quantify key demographic parameters and evaluate extinction risk at a metapopulation scale to inform conservation planning for Lahille's bottlenose dolphin, an endangered dolphin endemic to the coastal waters of the Southwest Atlantic Ocean. Using Robust Design mark-recapture models on data from a coordinated photo-identification program (2018–2023) across five Management Units (MUs) in the southern Brazil-Uruguay subpopulation, we estimated abundance and survival. Best estimates indicate a total subpopulation abundance of 341 individuals (95% CI: 333–348), with annual survival varying among MUs from 0.934 (95% CI: 0.856–0.971) to 0.968 (95% CI: 0.926–0.986). Meta-population-based Population Viability Analysis (PVA) projects a 2% decline in subpopulation size over three generations (63 years) under current conditions, with ~4% extinction risk for two MUs. Under a worst-case scenario with increased bycatch and habitat degradation, a 94% decline in the subpopulation is projected. Findings highlight the vulnerability of Lahille's bottlenose dolphins and the urgent need for habitat protection and bycatch mitigation. We recommend maintaining and expanding coordinated, long-term monitoring to support adaptive management.

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<https://doi.org/10.1016/j.biocon.2026.111865>

Received 9 December 2025; Received in revised form 23 March 2026; Accepted 16 April 2026

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1. Introduction

Large mammals are particularly vulnerable to extinction due to slow population growth (Pimm et al., 2014). Coastal cetaceans face even greater risks, given their exposure to entanglement in fishing gear (or bycatch), habitat degradation, and pollution (e.g., Taylor et al., 2007; Kiszka et al., 2022). Monitoring is essential for assessing such threats and guiding conservation, but in the long term, large-scale programs remain logistically demanding (Moussy et al., 2022). Moreover, monitoring alone is insufficient. Without effective action, pervasive threats persist, eroding conservation efforts (Taylor et al., 2007). Overcoming these barriers requires collaborative strategies integrating monitoring with adaptive decision-making (Hemming et al., 2022). Well-coordinated, wide-scale monitoring has proven transformative, enabling recovery of the Iberian lynx (*Lynx pardinus*; Delibes-Mateos et al., 2022); bolstered anti-poaching measures for African elephants (*Loxodonta africana*; O'Donoghue and Rutz, 2015); and advancing community-based conservation of chimpanzees (*Pan troglodytes*; Chancellor et al., 2021).

In marine systems, international cooperation, such as the moratorium on commercial whaling (IWC, 1982), was pivotal for the recovery of humpback whales (*Megaptera novaeangliae*) and supported rebounds in blue (*Balaenoptera musculus*) and southern right whales (*Eubalaena australis*; Zerbini et al., 2019; Agrelo et al., 2021). In contrast, the fate of the Yangtze River dolphin/baiji (*Lipotes vexillifer*) and vaquita (*Phocoena sinus*) highlights how delayed or ineffective action, even with monitoring, can lead to collapse or extinction (Dudgeon, 2005; Rojas-Bracho et al., 2019; del Monte-Luna et al., 2025). While the baiji vanished with minimal intervention, the vaquita depends on effective enforcement of fishing exclusion zones (Robinson et al., 2022). These challenges are amplified for small, mobile, and patchily distributed aquatic populations, such as Lahille's bottlenose dolphins (*Tursiops truncatus gephyreus*¹ or *Tursiops gephyreus*). Endemic to southern Brazil, Uruguay, and Argentina, these coastal dolphins are structured into two demographically distinct subpopulations: one occurring in northern Argentina and another spanning southern Brazil and Uruguay (Fruet et al., 2014; Fig. 1). The southern Brazil-Uruguay (SB-U) subpopulation faces multiple anthropogenic threats, including bycatch, pollution, habitat degradation, and prey depletion (Fruet et al., 2021; Bezamat et al., 2021). Listed as Endangered (Fruet et al., 2025) and legally protected (Brasil, 2012, 2019, 2022; CITES, 2023; CMS, 2023), the small number of individuals necessitates urgent assessment.

Over the past decade, studies on the SB-U subpopulation identified five nested Management Units (MUs) based on ecological and genetic criteria (Fruet et al., 2014). Individual movement is limited and gene flow asymmetric; therefore, these MUs align with the concept of Demographically Independent Populations (DIPs) (Taylor et al., 2010). Robust demographic data are available for the estuarine MUs, Patos Lagoon Estuary (PLE; Fruet et al., 2015a, 2015b) and Laguna (LGN; Bezamat et al., 2020), where long-term research yielded detailed insights (e.g., Simões-Lopes and Fabian, 1999; Fruet et al., 2012, 2015a, 2015b, 2021; Daura-Jorge et al., 2012; Di Tullio et al., 2015; Secchi et al., 2017; Genoves et al., 2020; Bezamat et al., 2020, 2021). However, data for coastal MUs remain limited (e.g., Flores and Fontoura, 2006; Laporta et al., 2016; Di Giacomo and Ott, 2016; Menchaca et al., 2019), constraining comparative demographic analyses. Regional collaboration

is essential to bridge these gaps and improve conservation planning across the subpopulation's range.

Here, our primary objective is to quantify key demographic parameters and assess extinction risk at a metapopulation scale for the SB-U subpopulation in order to inform conservation planning and management decisions. Stemming from the multi-institutional Gephyreus Project, we analysed data from a standardised range-wide photo-identification monitoring program (2018–2023) covering all MUs. We applied Robust Design mark-recapture models to estimate abundance, survival, and temporary emigration. These estimates were then integrated with long-term demographic data from well-studied MUs to inform a Population Viability Analysis (PVA) evaluating extinction risks under alternative scenarios. Although a smaller and geographically distinct subpopulation occurs in northern Argentina (Fruet et al., 2014), with signs of population decline (Vermeulen and Bräger, 2015), current data are insufficient for a comparable demographic assessment of this group. Given that the SB-U subpopulation likely comprises the vast majority of all known Lahille's dolphins, this study establishes the first range-wide demographic baseline required for science-based conservation planning for this endangered dolphin.

2. Methods

2.1. Study area and sampling design

The SB-U of Lahille's bottlenose dolphin extends from the Itajaí River (26.91°S/48.66°W) in southern Brazil to the Santa Lucía River, Uruguay (34.49°S/56.25°W) (e.g., Fruet et al., 2014), situated within the Pelotas and Punta del Este Basins (Fig. 1). This region includes sandy open coasts, bays, and estuaries influenced by freshwater discharge from the Patos Lagoon and La Plata River, driving coastal productivity (Marques et al., 2010; Marín et al., 2020; Nicolodi et al., 2021). The area sustains major artisanal and industrial fisheries, yet several stocks, including key dolphin prey, are overfished or declining (Haimovici and Cardoso, 2017; Sant'Ana et al., 2017). Ongoing industrial expansion, including ports and wind farms, poses further threats to marine ecosystems and traditional livelihoods (e.g., Morales et al., 2017; Rivas et al., 2025).

We conducted coordinated photo-identification surveys across ten sites in all five MUs: Itajaí and Baía Norte (Florianópolis MU, FLN), Laguna (Laguna MU, LGN), Torres, Tramandaí, and the northern open coast of Patos Lagoon (North Patos Lagoon MU, NPL), Patos Lagoon Estuary (Patos Lagoon Estuary MU, PLE), the southern open coast of Patos Lagoon, La Coronilla-Santa Teresa, and the stretch from Oceanía del Polonio to Punta Palmar along Rocha department coast in Uruguay (South Patos Lagoon and Uruguay MU, SPLU) (Fig. 1). Together, these sites span approximately 1000 km of coastline across the SB-U subpopulation range. Surveys were conducted semi-annually from October 2018 to January 2023, excluding early 2020 due to COVID-19. Each year was divided into two sampling periods (January–June and July–December), and synchronised campaigns lasted up to four months each period, ensuring temporal consistency and facilitating data integration (Table S1). Surveys were coordinated across all MUs within the same temporal window to maximize demographic closure at the subpopulation scale.

Boat-based surveys followed pre-defined routes designed to cover core habitat areas within each MU (see Table S1 for details on survey coverage). In FLN, NPL, and SPLU, photo-identification was also conducted from fixed land-based vantage points located at narrow river mouths and channel inlets (often <200 m wide), which function as predictable transit or foraging corridors. These geomorphological features allow clear lateral views of dorsal fins from shore, facilitating reliable individual identification. In areas where sea conditions limited boat-based effort, land-based monitoring served as an effective complementary approach. Encounters followed standard protocols (Würsig and Würsig, 1977), using digital SLR cameras equipped with telephoto lenses suitable for high-resolution dorsal fin photography. During boat-

¹ Lahille's bottlenose dolphins are currently classified as a subspecies of *Tursiops truncatus* by the Society for Marine Mammalogy (Committee on Taxonomy, 2021). Multiple lines of evidence—genetic, morphological, and ecological—have been used to argue for their recognition either as a distinct species (e.g., Wickert et al., 2016) or as a subspecies (e.g., Costa et al., 2016; Oliveira et al., 2019). This taxonomic status remains under debate, and no attempt is made here to resolve it. For consistency and clarity, we refer to this form as “Lahille's bottlenose dolphin”.

based surveys, groups were followed until high-quality images of all individuals were obtained. When environmental or behavioural conditions prevented complete photographic coverage, the encounter was terminated and surveys resumed along the predefined routes to search for additional groups. Survey effort was standardised by defining a sampling day as ≥ 3 h of effective photo-ID work to ensure sufficient detection opportunity.

2.2. Photo-identification data

Photographs were quality-graded (Urian et al., 2015), retaining only sharp, perpendicular images of dorsal fins. Individuals were identified by long-lasting natural marks (Würsig and Würsig, 1977) and assigned unique IDs (e.g., PLE#008 for individual 8 from PLE). Identifiers were compiled into site- and period-specific catalogues stored in a shared repository. Three trained analysts independently reviewed all catalogues, incorporating opportunistic photo-ID records, to verify matches and inter-MU movements. Note that opportunistic data were used solely for matching; mark-recapture models relied strictly on standardised surveys. Capture histories were merged into a single dataset. For analytical consistency across sites, sampling days were pooled into 15-day capture occasions. Intervals outside these windows were excluded, and sites not surveyed within a specific window had capture probabilities fixed at zero (see modelling procedures).

2.3. Mark-recapture modelling

We estimated demographic parameters using a Robust Design framework (Kendall et al., 1997). This approach includes two nested temporal levels: secondary occasions (sampling days) within primary occasions (sampling periods). To ensure comparability across the study period, time intervals between primary occasions were adjusted in the model to estimate all demographic parameters on an annual scale.

Between primary occasions, we estimated apparent survival (Φ ; reflecting true survival and permanent emigration) and temporary emigration probabilities (γ' and γ''), representing the likelihood of individuals leave and return between periods. Within primary occasions, secondary occasions were used to estimate capture (p), and recapture (c) probabilities, assumed equal, and the abundance (N) of marked individuals. Candidate models varied in structure: Φ was considered constant (\cdot), time-dependent, or MU-specific; γ' and γ'' were set as none ($\gamma'' = 0$), random ($\gamma'' = \gamma'$), or Markovian ($\gamma'' \neq \gamma'$), and either constant or MU-specific; and $p = c$ was treated as constant, time-dependent (by primary period), or MU-specific. The abundance (N) of marked animals was estimated for each primary period and MU. To estimate the corrected abundance for each MU (N_c), we adjusted N by the proportion of marked animals (θ). For PLE, θ was calculated as the marked-to-total ratio per sampling group; for LGN, as the ratio per capture occasion (cf. Wilson et al., 1999). For FLN, NPL, and SPLU, where sparse

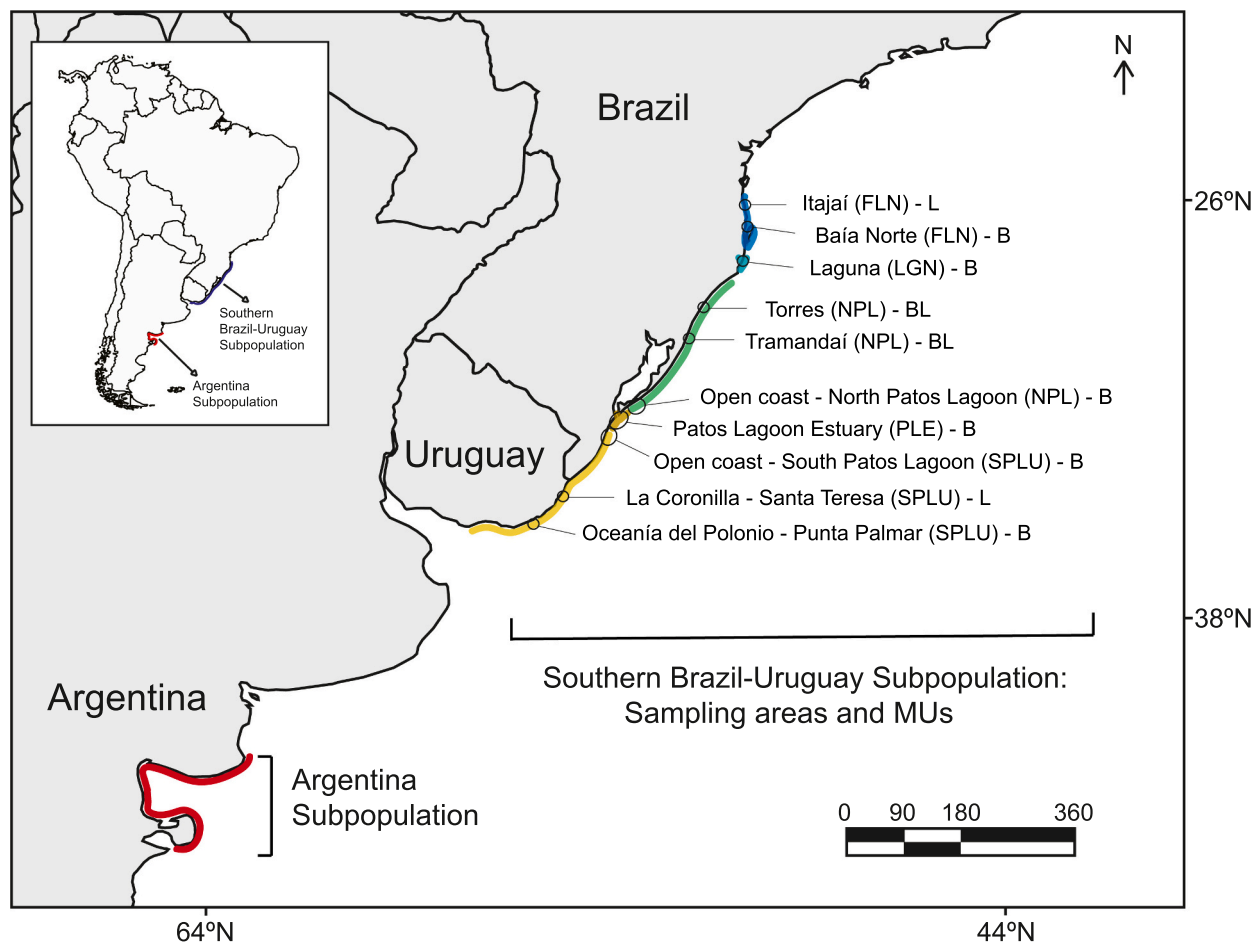


Fig. 1. Sampling sites along the range of the southern Brazil-Uruguay subpopulation of Lahille's bottlenose dolphins, grouped by Management Units (MUs, in parentheses): Itajaí (FLN), Florianópolis (FLN), Laguna (LGN), Torres (NPL), Tramandaí (NPL), North Patos Lagoon open coast (NPL), Patos Lagoon Estuary (PLE), South Patos Lagoon open coast and Uruguay (SPLU), La Coronilla-Santa Teresa (SPLU), and from Oceaía del Polonio to Punta Palmar (SPLU). Sampling method is indicated as land-based only (L), boat-based only (B), or both (BL). Circles schematically represent the approximate sampling areas within each MU. In red, the current known distribution of the Argentina subpopulation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

encounters precluded reliable θ estimates, we adopted the estimate derived from PLE open-coast areas, assuming similar marking rates due to habitat similarity. For all cases, we calculated the period-specific mean θ and its standard deviation. The corrected abundance for each MU (N_c) was then estimated as $N_c = N / \theta$, and the standard error (SE) of N_c followed the procedure in [Urian et al. \(2015\)](#):

$$SE(\hat{N}_c) = \sqrt{\hat{N}_c^2 \times \left[\left(\frac{SE(\hat{N}^2)}{\hat{N}^2} \right) + \left(\frac{SE(\theta^2)}{\theta^2} \right) \right]} \quad (1)$$

Log-normal 95% confidence intervals (CI) for N_c were constructed using a bias-correction factor (C) calculated as:

$$C = \exp \left[z \frac{\alpha}{2} \times \sqrt{\ln \{ 1 + [CV(\hat{N}_c)]^2 \}} \right] \quad (2)$$

where z is the standard normal critical value ($\alpha = 0.05$) and CV is the coefficient of variation of N_c ([Fletcher, 2008](#)).

Total subpopulation abundance (N_t) per period was estimated by summing N_c across all MUs. The combined standard error (SE) was calculated using Eq. (3) and confidence intervals followed Eq. (2) using the combined CV.

$$SE_{combined} = \sqrt{\sum SE_{MUi}^2} \quad (3)$$

As Robust Design models lack formal goodness-of-fit (GOF) tests ([White and Burnham, 1999](#)), we collapsed secondary occasions into primary occasions to evaluate Cormack–Jolly–Seber (CJS) assumptions. Initial GOF tests were conducted using Program RELEASE (via MARK; [White and Burnham, 1999](#)), applying TEST 2 (testing the assumption of equal capture probabilities) and TEST 3 (equal survival probabilities). To further investigate potential sources of lack of fit, we decomposed TEST components using the R2ucare package in R ([Gimenez et al., 2017](#)), examining transience (Test 3.SR) and trap-dependence (Test 2. CT) both globally and by Management Unit (MU). The overdispersion parameter (\hat{c}) was calculated from the pooled RELEASE results as the ratio between the sum of TEST 2 and TEST 3 components and their respective degrees of freedom. Model selection was performed using QAICc to account for this overdispersion ([Burnham and Anderson, 2002](#)). When competing models within $\Delta QAICc < 2$ embodied structurally distinct biological assumptions (e.g., alternative formulations of temporary emigration), we prioritized interpretability and biological coherence over model averaging. All analyses were performed in R version 4.4.2 ([R Core Team, 2024](#)), using the RMark package ([Laake, 2013](#)) as an interface to Program MARK. Data wrangling and plotting were performed with tidyverse, including dplyr ([Wickham et al., 2023](#)) and ggplot2 ([Wickham, 2016](#)).

2.4. Population viability analysis (PVA)

We performed a metapopulation-level PVA for all five MUs of the SB-U subpopulation using VORTEX (version 10, [Lacy, 1993; Lacy, 2000](#)), an individual-based model incorporating demographic, environmental, and genetic stochasticity to project population dynamics. Outputs were summarised with VortexR package (version 1.1.0; [Pacioni and Mayer, 2017](#)) in R (version 4.4.0; [R Core Team, 2024](#)). Scenario design and parameters relied on empirical estimates from this study and long-term data from LGN and PLE.

2.5. PVA inputs for the baseline scenario

Scenarios followed standard procedures (e.g., [Lacy, 2019; Pacioni and Mayer, 2017](#)): 5000 iterations over 100-year (365-day years); extinction defined as fewer than one individual (per VORTEX); a polygynous mating system with all adult males available for breeding; single-calf reproduction per event; no extreme events; and adult-only

dispersal among MUs, with 100% survival during dispersal. Annual reproductive probabilities were based on observed birth rates (see below), rather than assuming one calf per female per year. Dispersal rates were derived from genetic migration rates ([Fruet et al., 2014](#)) divided by generation time (21 years) ([Taylor et al., 2007](#)). Inbreeding effects were included by converting coefficients from [Fruet et al. \(2014\)](#) into lethal equivalents (a VORTEX requirement), following [Nietlisbach et al. \(2018\)](#). Density-dependent constraints (e.g., [Kanaji et al., 2024](#)) were not implemented due to software limitations and lack of data; however, as populations remain well below carrying capacity (K), overestimation of growth is likely negligible.

Three key parameters informed baseline simulations: (i) annual percentage of reproductive females, (ii) age-specific mortality rates, and (iii) carrying capacity. Reproductive rates were derived using longitudinal photo-ID data (2018–2022) from LGN and PLE. Assuming a 1:2 male-to-female adult ratio ([Fruet et al., 2015a](#)), we calculated the annual probability of reproduction as observed calves divided by 66% of the marked population, yielding a population-level metric that reflects both maturity and interbirth intervals. This value, used as the proportion of adult females breeding per year, does not imply annual reproduction for each female, but instead captures the observed reproductive output over time. Site-specific means and standard deviations were used directly in VORTEX, and averages were extrapolated to FLN, NPL, and SPLU.

Mortality for ages 0–2 came from long-term data in LGN ([Bezamat et al., 2020](#)) and PLE ([Fruet et al., 2015b](#)), with PLE values applied subpopulation-wide, given its broader ecological representativeness (including both coastal and estuarine individuals). Juvenile mortality (ages 3–7) was drawn from PLE ([Fruet et al., 2015a](#)) and applied uniformly across all MUs. Strandings at PLE ([Fruet et al., 2012](#)) and LGN (PMP-BS; [Noronha, 2025](#)) revealed a 3:1 male bias, used to adjust juvenile mortality post hoc (0.255 for males, 0.085 for females). For adults (>7 years), survival came from our best-supported mark-recapture model ([Table 1](#)), with a 2% adjustment increasing male mortality and decreasing female mortality, based on prior evidence of higher female survival ([Bezamat et al., 2019; Fruet et al., 2015a](#)). Beta distributions ([White, 2000](#)) incorporated stochasticity in the survival estimates.

Initial abundance was set as the mean of our mark-recapture estimates across sampling periods and MUs. Carrying capacity (K) was defined using a mid-range density of 1 individual/km², typical for coastal *T. truncatus* ([Durden et al., 2021](#)), applied to the mapped dolphin habitat areas in LGN (120 km²) and PLE (180 km²) to calculate local K. For MUs without well-defined open-coast habitat boundaries, we calculated an abundance-to-K ratio based on LGN and PLE and applied this ratio to abundance estimates from FLN, NPL, and SPLU to derive their K values. A 10% standard deviation was assumed to reflect uncertainty, and K was included in the sensitivity analysis (see below). All PVA parameters are summarised in [Table S2](#).

2.6. Sensitivity analyses for the PVA

We evaluated how uncertainty in key parameters affected projected population growth (stochastic r). Parameters selected, due to high uncertainty or potential vulnerability to local pressures, included inbreeding, first- and second-year mortality, juvenile and adult female mortality, annual reproductive probability (i.e., proportion of adult females birthing per year), and carrying capacity (K). For each parameter, we ran 500 iterations across 100 values sampled uniformly from a $\pm 10\%$ range around the baseline (except inbreeding, which ranged from zero to the VORTEX default). Spearman correlations and linear regressions assessed parameter influence on r , with higher correlations and r^2 values indicating stronger effects. Since VORTEX does not support elasticity analyses, future extensions could refine the assessment of the relative parameter importance.

2.7. PVA alternative scenarios and parameters decisions

We simulated trajectories for each MU and for the SB-U subpopulation under six scenarios reflecting current and potential environmental and anthropogenic stressors: (i) current conditions (“Baseline”), (ii) increased bycatch (“Additive-bycatch”), (iii) reduced bycatch (“Reduced-bycatch”), (iv) declining prey availability (“K-declining”), (v) habitat degradation affecting calf survival (“Habitat-declining”), and (vi) a “Triple-threat” scenario combining the three stressors acting synergistically on distinct demographic parameters. The “Baseline” scenario assumed current natural mortality and observed bycatch. “Additive-bycatch” scenario incorporated additional mortality rates based on LGN monitoring data (PMP-BS; Noronha, 2025), indicating one female and three male bycatch events every three years, mostly juveniles (4–5 yrs.; Noronha, 2025). Then, we applied 2% additional mortality to females and 6% to males in LGN. This mortality was split evenly between juveniles and adults (≥ 9 years for females, ≥ 11 years for males), and implemented across all MUs using VORTEX “harvest” function every three years. Conversely, the “Reduced-bycatch” scenario used the VORTEX “supplementation” function to offset 50% of current bycatch. This was implemented by adding 1% of adult females and 3% of adult males (same age classes) to each MU triennially.

“K-declining” scenario simulated a 0.5% annual reduction in carrying capacity over 63 years (three generations), reflecting prey declines. This rate was based on a 1.6% annual drop in Lebranche mullet (*Mugil liza*) biomass between 2000 and 2016 (Sant’Ana et al., 2017), a key energetic prey species that represents ~35% of Lahille’s bottlenose dolphin diet (Secchi et al., 2017; Teixeira et al., 2021). Given limited data for other prey and the assumption that fishery catches reflect stock size, we conservatively used *M. liza* trends alone to define the K-decline rate. The “Habitat-declining” scenario addresses pollutant-driven reductions in habitat quality, especially due to polychlorinated biphenyls (PCBs), known to impair first-year calf survival (Reddy et al., 2001; Wells et al., 2005). PCB levels exceeding toxicity thresholds were reported in LGN and PLE (Righetti et al., 2019). Following Schwacke et al. (2002) the risk of reproductive failure from PCB exposure is 0.79 for primiparous and 0.03 for multiparous females. Assuming a 1:5 primiparous-to-multiparous ratio (Wells et al., 2025), the resulting excess population-level reproductive risk was estimated at ~15%. Accordingly, the first-year survival was reduced by 15% (i.e., survival multiplied by 0.85). While PCB effects may already be ongoing, we treated this as a forward-looking scenario. Table S3 summarises all scenarios, including the “Triple-threat” case combining Additive bycatch, K-declining and Habitat-declining conditions.

3. Results

3.1. Sample and data profile

We conducted 57 coordinated capture occasions across all sites over seven sampling periods. Effort variation among MUs reflected area size, while differences between surveys were mainly weather-related. Sampling was generally synchronous across MUs within the defined effort window (Table S1); when a MU was not sampled during that window, its capture probability was set to zero. In total, 21,417 photos were taken, identifying 344 marked animals (Fig. S4): 33 in FLN, 37 in LGN, 127 in NPL, 65 in PLE, and 82 in SPLU. Cumulative discovery curves approached asymptotes for most MUs and for the overall subpopulation (Fig. S4), indicating substantial sampling coverage during the study period. In contrast, FLN and NPL exhibited less pronounced asymptotic trends, suggesting that additional effort may further increase individual detection in these areas. We detected no inter-MU movements, but occasional intra-MU movements occurred—e.g., between Torres and Tramandaí (~80 km; NPL) and between Patos Lagoon’s marine coast and Uruguayan waters (~240 km; SPLU).

3.2. Modelling mark-recapture data

Goodness-of-fit (GOF) tests based on collapsed primary occasions indicated mild but significant lack of fit in the pooled dataset (TEST 2 + TEST 3: $X^2 = 62.597$, $df = 44$, $p = 0.034$), driven mainly by the TEST 3 component ($X^2 = 47.282$, $df = 27$, $p = 0.0092$). Decomposition using R2ucare revealed that this deviation was not uniformly distributed across MUs. Specifically, the NPL MU exhibited strong evidence of transience (Test 3.SR: $X^2 = 34.33$, $df = 5$, $p < 0.001$), indicating the presence of individuals observed once and not subsequently re-encountered. In contrast, FLN, LGN, PLE, and SPLU showed no significant global lack of fit ($p > 0.05$) and no consistent evidence of transience. Trap-dependence (Test 2.CT) was generally weak and non-significant across MUs. These results indicate spatially structured heterogeneity in apparent survival, likely driven by movement dynamics in NPL. Because temporary emigration can mimic mortality when not explicitly modelled (see Discussion), we incorporated alternative temporary emigration structures into the candidate model set. To account for mild overdispersion in the pooled analysis, we applied $\hat{c} = 1.42$ when computing QAICc for model selection.

Accordingly, we fitted 52 candidate models within the Robust Design framework to evaluate alternative survival and temporary emigration structures. The best-supported model (M31; weight = 0.61) assumed constant apparent survival across MUs and time, MU-specific Markovian temporary emigration, and capture probability varying by MUs and primary periods (Table 1). The second-ranked model (M52; weight = 0.27) differed primarily in the treatment of temporary emigration, assuming a single emigration structure across MUs. In the Robust Design framework, temporary emigration was parameterized through the probabilities γ' (probability of remaining unavailable) and γ'' (probability of becoming unavailable), with the Markovian formulation allowing availability in a given period to depend on the individual’s availability in the previous period. Despite moderate model uncertainty ($\Delta QAIC < 2$), we considered both M31 and M52 rather than applying model averaging, because these two top models embodied distinct biological interpretations of survival-movement dynamics. M31 provides a movement-adjusted survival estimate, whereas M52 reflects apparent survival that does not fully separate survival from availability.

Model M52 estimated apparent survival (i.e., the probability that an individual remained alive and available) varying across MUs: 0.651 (95% CI: 0.441–0.816) in FLN, 0.934 (95% CI: 0.856–0.971) in LGN, 0.787 (95% CI: 0.704–0.851) in NPL, 0.968 (95% CI: 0.926–0.986) in PLE, and 0.900 (95% CI: 0.772–0.960) in SPLU (Fig. 2A shows estimates for LGN and PLE from M52). However, these values integrate survival and site fidelity. The low estimate in open-coastal MUs (particularly NPL, where GOF decomposition indicated strong transience) is consistent with movement-driven heterogeneity and incomplete separation between survival and availability under M52. Thus, M52 likely underestimates survival in MUs with fluid space use.

In contrast, M31 explicitly accounted for MU-specific temporary emigration through the estimation of γ' and γ'' , yielding a single, less movement-biased apparent survival estimate (Φ) of 0.956 (95% CI: 0.905–0.980) across all MUs and periods (Fig. 2A shows M31 estimates for FLN, NPL and SPLU). For inferential clarity, we used this movement-adjusted survival estimate for demographic projection (PVA), while retaining MU-specific apparent survival from M52 to describe spatial patterns. Temporary emigration probabilities (γ'') were low in estuarine MUs (LGN: 0.051, 95% CI: 0.021–0.120; PLE: 0.001, 0.000–0.009) but substantially higher in open-coastal systems (FLN: 0.311, 95% CI: 0.162–0.513; NPL: 0.456, 95% CI: 0.369–0.545; SPLU: 0.254, 95% CI: 0.156–0.386) (Fig. 2B), consistent with the greater spatial openness and mobility of the latter. Because such movement dynamics can bias apparent survival when not explicitly modelled, the movement-adjusted survival estimate from M31 was considered more appropriate for open-coastal MUs.

Return probability (the complement of γ' under the Markovian

Table 1

Summary of the top 10 mark-recapture models (out of 52) fitted under the Robust Design framework for Lahille's bottlenose dolphins (2018–2023) across all systematically monitored areas. Models are ranked by increasing Quasi-Akaike Information Criterion (QAICc) and report the number of parameters (npar), Δ QAICc (relative to the top-ranked model), model weight, and quasi-deviance. Model structures indicate whether survival (Φ), capture probability (p), and temporary emigration (γ) were constant (\cdot), time-varying (period), or varied among Management Units (MU).

Model	Parameters	npar	QAICc	Δ QAICc	weight	QDeviance
M31	$\Phi(\cdot) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU} \cdot \text{period})$	53	4239.23	0	0.61	6616.18
M52	$\Phi(\text{MU}) \gamma''(\cdot) = \gamma'(\cdot) p = c(\text{MU} \cdot \text{period})$	52	4240.86	1.63	0.27	6619.89
M51	$\Phi(\text{MU}) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU} \cdot \text{period})$	57	4242.49	3.26	0.12	6611.10
M32	$\Phi(\cdot) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU} \cdot \text{period})$	48	4251.51	12.28	0.00	6638.84
M26	$\Phi(\cdot) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU} + \text{period})$	29	4260.34	21.12	0.00	6686.79
M47	$\Phi(\text{MU}) \gamma''(\text{MU}) = \gamma'(\text{MU}) p = c(\text{MU} + \text{period})$	28	4260.65	21.42	0.00	6689.14
M46	$\Phi(\text{MU}) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU} + \text{period})$	33	4264.23	25.00	0.00	6682.49
M27	$\Phi(\cdot) \gamma''(\text{MU}) = \gamma'(\text{MU}) p = c(\text{MU} \cdot \text{period})$	24	4270.16	30.93	0.00	6706.81
M42	$\Phi(\text{MU}) \gamma''(\text{MU}) = \gamma'(\text{MU}) p = c(\text{MU})$	22	4284.99	45.75	0.00	6725.70
M21	$\Phi(\cdot) \gamma''(\text{MU}) \gamma'(\text{MU}) p = c(\text{MU})$	23	4286.35	47.11	0.00	6725.03

formulation) also varied across MUs: 0 for PLE (consistent with the absence of temporary emigration), 0.09 (95% CI: 0.016–0.394) for FLN, 0.31 (95% CI: 0.052–0.776) for LGN, 0.25 (95% CI: 0.168–0.335) for NPL, and 0.69 (95% CI: 0.192–0.950) for SPLU (Fig. 2C). These values reflect individuals resuming site fidelity rather than permanent movements between MUs. Since our analyses assume only temporary movements among MUs, high return probabilities reflect reappearance within the same MU after one or more missed sampling periods. Capture probability (p) varied across MUs and declined over time, possibly due to reduced effort or changes in detectability. Still, LGN and PLE maintained consistently high capture probabilities throughout the study (Fig. 2D), reinforcing their role as core long-term monitoring sites.

Abundance of marked individuals varied among MUs and across sampling periods (Table S5). Estuarine MUs (PLE and LGN) consistently supported the largest numbers of marked individuals and showed relatively stable estimates through time. In contrast, open-coastal MUs (FLN, NPL, and SPLU) exhibited greater temporal variability, particularly SPLU, where estimates fluctuated markedly among periods. After correcting for the proportion of marked individuals, total MU abundance (N_c) ranged from fewer than 40 individuals in FLN to over 100 individuals in PLE during peak periods (Fig. 3A; full estimates in Table S5). NPL and SPLU showed wider ranges than the estuarine units, consistent with greater movement dynamics and variable detection. At the subpopulation scale, total abundance (N_t) ranged from 252 individuals (95% CI: 243–260) in the first sampling period to 341 individuals (95% CI: 333–348) in the sixth period (Fig. 3B). These fluctuations likely reflect the variation in survey effort and the temporary movement, particularly in open-coastal MUs, rather than true demographic fluctuations. The highest upper confidence limit (348 individuals) provides a conservative maximum estimate of current subpopulation size.

3.3. Final PVA input decisions and sensitivity analysis

Baseline input parameters are presented in Table S2. Final parameter choices for the PVA were informed by the mark-recapture results described above. Adult mortality rates varied by MU. For open-coastal MUs (FLN, NPL, and SPLU), we used survival estimates from model M31, which accounts for MU-specific temporary emigration and therefore provides movement-adjusted estimates suitable for long-term demographic projection. For the estuarine MUs (LGN and PLE), where GOF diagnostics indicated good fit and low temporary emigration, we used MU-specific survival estimates from model M52 (Fig. 2), as movement-related bias was minimal in these systems. This approach ensured that survival inputs reflected biologically realistic movement dynamics while preserving spatial specificity where movement bias was negligible. To explore uncertainty, we conducted sensitivity analyses by varying input parameters $\pm 10\%$ from baseline, except for inbreeding, which ranged from 0 to 6.29 lethal equivalents per diploid individual (VORTEX

default) assuming 50% due to recessive lethal alleles. Table S6 presents effects on population growth (r). Inbreeding and carrying capacity had minimal influence, whereas early-life mortality had a moderate effect. Variation in the percentage of breeding females and female mortality (juvenile and adult) had the strongest impact on growth.

3.4. PVA projections of the current scenario and potential threats

Under the Baseline scenario (i.e., current bycatch levels), 100-year projections indicated negative stochastic growth rates (r -stoch) in all MUs except NPL and PLE (Table 2). At the subpopulation level, growth was effectively stable (0.0005 ± 0.0016), suggesting near demographic equilibrium. Over one generation (21 years), the population increased slightly (+1%, to 306 individuals), followed by a modest decline over three generations (63 years, –2%, to 297 individuals). Despite this apparent stability at the aggregate level, local trajectories diverged. LGN and FLN exhibited the steepest long-term declines (–71% and –43%, respectively), each projected to fall below 15 individuals, with extinction risks of 3–4% (Table 2; Fig. 4; Fig. S7).

In the Reduced-bycatch scenario, stochastic growth rates became positive for all MUs except LGN. The subpopulation showed r -stoch = 0.0106 ± 0.0188 , increasing 27% (to 385 individuals) over one generation and 34% (to 407 individuals) over three generations. However, local trajectories differed among MUs. LGN continued to decline (–39% over three generations, to 29 individuals), and FLN showed a slight reduction (–3%, to 22 individuals), despite a positive r -stoch (0.0049 ± 0.0183). Extinction risks in these two units remained very low (<0.1%), suggesting that partial bycatch mitigation substantially reduces extinction risk but may not reverse declining trajectories in smaller MUs (Table 2; Fig. 4; Fig. S6). In contrast, the Additive-bycatch scenario resulted in strong negative growth across all MUs (subpopulation r -stoch = -0.0565 ± 0.0282). The subpopulation declined 39% over one generation (to 184 individuals) and 91% over three generations (to 26 individuals). The sharpest reductions occurred in LGN (–98%), FLN (–95%), and SPLU (–94%), each approaching quasi-extinction levels (1, 1, and 3 individuals, respectively). Extinction risk increased markedly, reaching 59% in LGN, 53% in FLN, and 30% in SPLU (Table 2; Fig. 4; Fig. S7).

Under the K-declining scenario (0.5% annual reduction in carrying capacity), stochastic growth was slightly negative at the subpopulation (r -stoch = -0.0018 ± 0.0017). Although a small short-term increase was projected (+3% over one generation, to 291 individuals), the subpopulation declined 24% over three generations (to 230 individuals). LGN and FLN again showed the steepest contractions (–72% and –64%, respectively), reaching 13 and 8 individuals, with extinction risks of 4–6% (Table 2; Fig. 4; Fig. S6). In the Habitat-declining scenario (reduced first-year survival due to PCB exposure), all MUs exhibited negative growth (subpopulation r -stoch = -0.0069 ± 0.0020 ; Table 2).

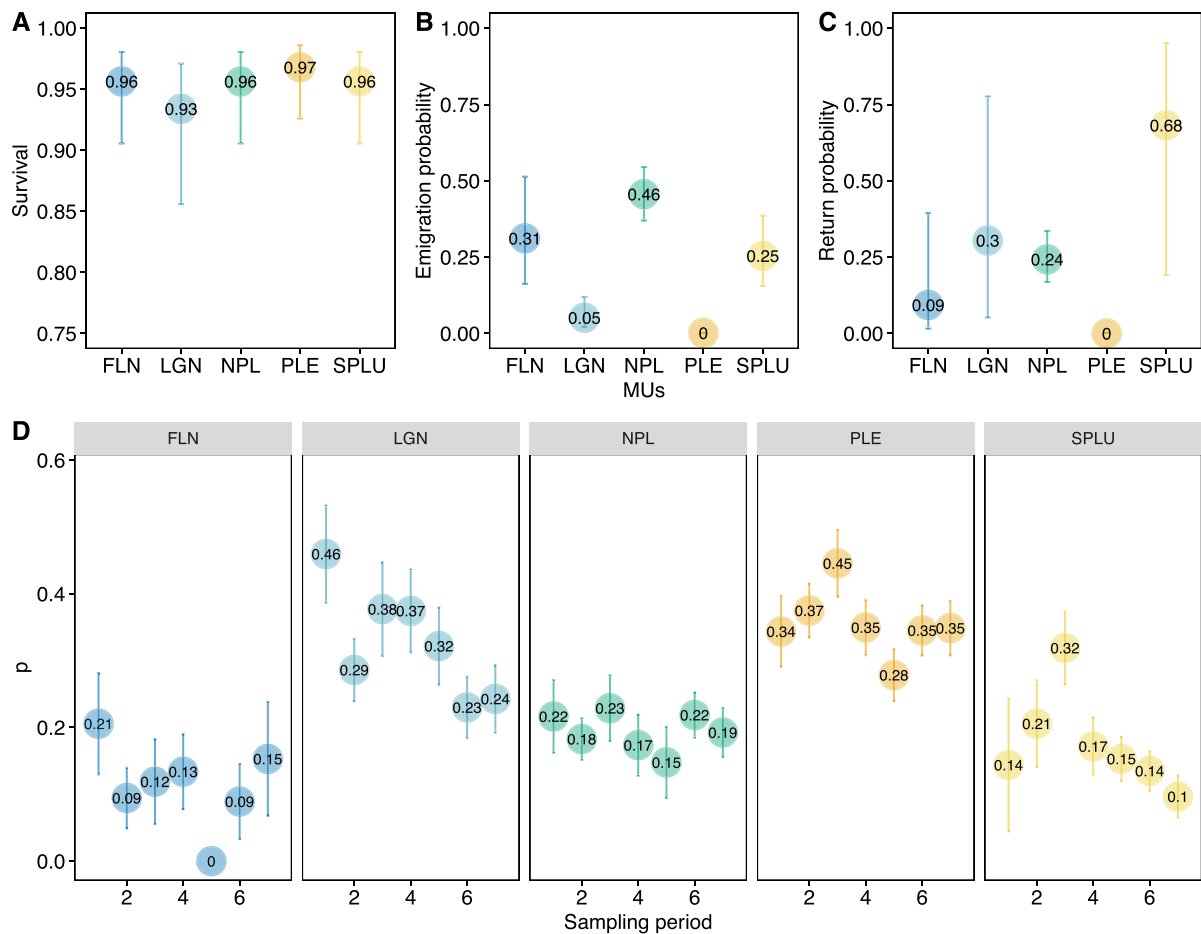


Fig. 2. Estimated demographic parameters for Lahille's bottlenose dolphins from the southern Brazil and Uruguay subpopulation, systematically monitored between 2018 and 2023: (A) apparent survival probabilities, (B) temporary emigration probability (γ'), (C) return probabilities ($1 - \gamma'$), and (D) capture probabilities (p) by Management Unit (MU) and sampling period. MUs include: Florianópolis (FLN), Laguna (LGN), North Patos Lagoon (NPL), Patos Lagoon Estuary (PLE), and South Patos Lagoon and Uruguay (SPLU). Sampling periods: 1 (10/2018–01/2019), 2 (03/2019–06/2019), 3 (09/2019–01/2020), 4 (10/2020–01/2021), 5 (09/2021–12/2021), 6 (03/2022–07/2022), and 7 (10/2022–01/2023). Estimates are presented with 95% confidence intervals. Apparent survival estimates in (A) derive from Model M31 (for FLN, NPL, and SPLU) and Model M52 (for LGN and PLE). Capture probability for FLN was fixed at zero during period 5 due to lack of effort.

The subpopulation declined 13% over one generation (to 263 individuals) and 34% over three generations (to 199 individuals). LGN (–85%) and FLN (–61%) were most affected, declining to 7 and 9 individuals, with extinction risks of 7–10% (Table 2; Fig. 4; Fig. S6). The Triple-threat scenario produced the most severe outcome. The subpopulation declined 46% over one generation (to 163 individuals) and 94% over three generations (to 18 individuals; $r\text{-stoch} = -0.0608 \pm 0.0281$). All MUs, including PLE, exhibited strong negative growth. LGN and FLN dropped by 99% and 96%, respectively, each reduced to a single individual, with extinction risks exceeding 60% (Table 2; Fig. 4; Fig. S7).

4. Discussion

Our findings provide the first multi-site demographic assessment of the endangered Lahille's bottlenose dolphin (LBD), endemic to the southwestern Atlantic. Coordinated monitoring across five Management Units (2018–2023) yielded robust estimates for the southern Brazil–Uruguay subpopulation (SB-U), revealing distinct spatial patterns: dolphins in semi-enclosed areas (e.g., PLE, LGN) exhibited high residency and estuarine dependence, while those in open systems showed high emigration rates. Mark-recapture models revealed variation in survival and capture probabilities across MUs, but low abundance overall. Integrating these estimates with existing demographic and genetic data, we conducted a rare metapopulation-based Population Viability Analysis (PVA) for a small cetacean. Only the most optimistic

PVA scenario, bycatch reduction, predicted subpopulation growth, though some MUs (FLN and LGN) still declined. Under current conditions, the subpopulation is projected to decline, with marked reductions in some MUs despite low extinction risk. In more pessimistic but plausible scenarios, several MUs face high extinction probability and the subpopulation contracts further. Our results highlight the precarious status of LBD and demonstrate how coordinated, multi-site monitoring can inform effective conservation.

4.1. Survival probabilities and temporary emigration

Apparent survival was notably affected by temporary emigration, introducing uncertainty into model selection. While apparent survival conceptually combines true survival and permanent emigration, unmodelled temporary absence can bias estimates downward, particularly in areas with high movement. GOF decomposition further indicated that this heterogeneity was spatially structured, with strong evidence of transience concentrated in NPL. By estimating MU-specific γ' (probability of remaining unavailable) and γ'' (probability of becoming unavailable), the best-supported model (M31) explicitly separated survival from availability for detection, revealing that the variation across MUs was driven primarily by movement dynamics rather than differences in true survival. This highlights the need to model temporary emigration, especially when movement scales match sampling intervals. In such contexts, models that explicitly parameterise temporary

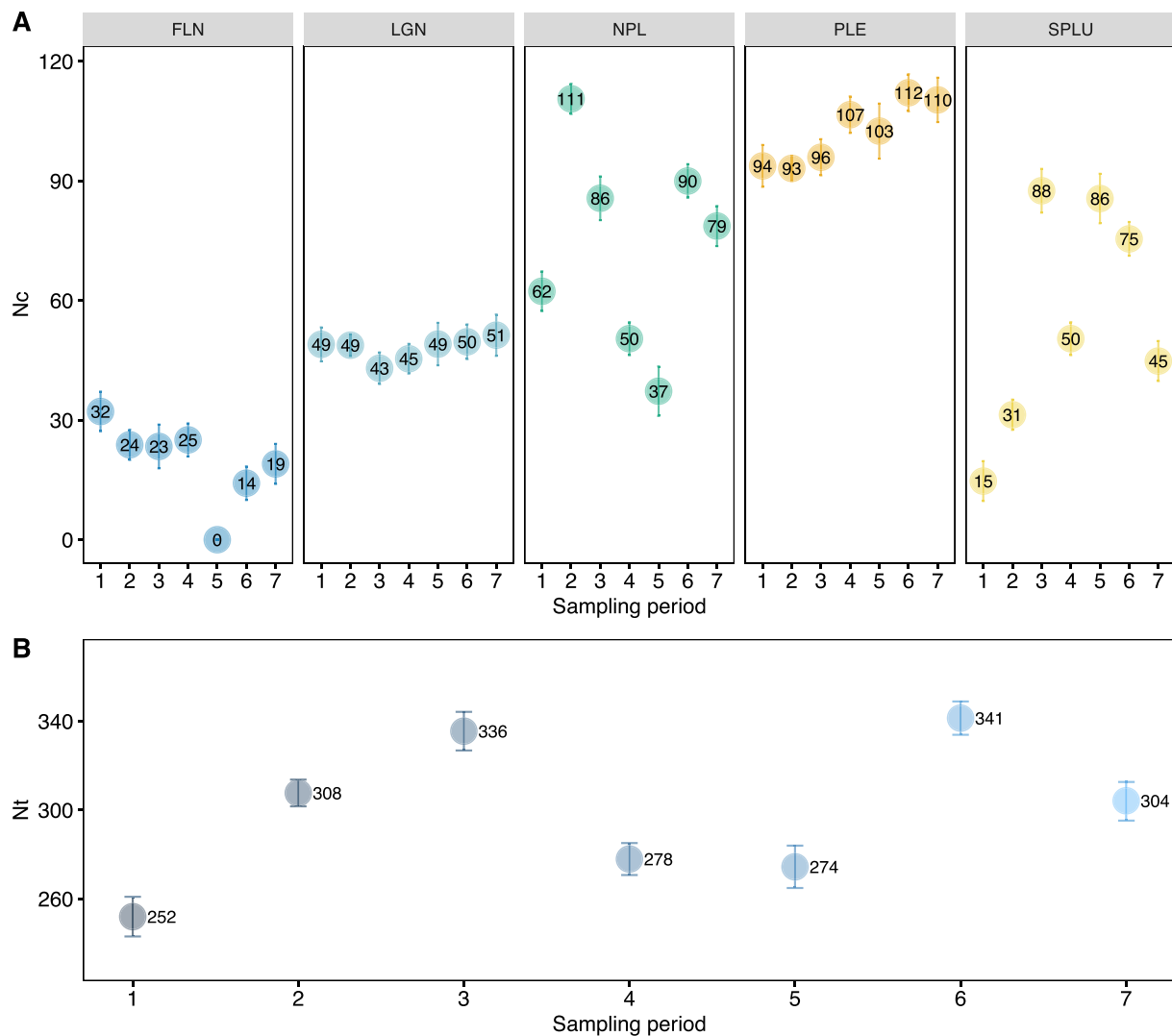


Fig. 3. Trends in abundance of Lahille's bottlenose dolphins from the southern Brazil and Uruguay subpopulation, systematically monitored between 2018 and 2023: (A) Corrected abundance estimates (N_c) by Management Unit (MU) and sampling period for Lahille's bottlenose dolphins in southern Brazil and Uruguay subpopulation between 2018 and 2023, and (B) total subpopulation abundance (N_t) by period. Management Units: Florianópolis (FLN), Laguna (LGN), North Patos Lagoon (NPL), Patos Lagoon Estuary (PLE), and South Patos Lagoon and Uruguay (SPLU). Sampling periods: 1 (10/2018–01/2019), 2 (03/2019–06/2019), 3 (09/2019–01/2020), 4 (10/2020–01/2021), 5 (09/2021–12/2021), 6 (03/2022–07/2022), and 7 (10/2022–01/2023). Estimates are shown with 95% confidence intervals.

emigration are essential to avoid confounding movement-driven absence with mortality. We note that high temporary emigration likely reflects genuine movement dynamics in open-coastal systems; however, differences in sampling efficiency among MUs—particularly those with less asymptotic discovery curves (e.g., FLN and NPL)—may have introduced additional heterogeneity in capture probability.

Model M52 (second-best) estimated MU-specific survival but applied a single temporary emigration structure across the subpopulation, thereby overlooking spatial variation in movement. Consequently, it yielded lower survival estimates for open systems (FLN, NPL, and SPLU) compared to estuarine ones (PLE and LGN), likely an artefact of unmodelled site-specific emigration biasing survival downward (Kendall et al., 1997; Schaub et al., 2004). However, the relative patterns within M52 aligned with previous studies (Daura-Jorge et al., 2012; Fruet et al., 2015a), reconfirming earlier estimates for the two long-term estuarine MUs, with PLE exhibiting higher apparent survival than LGN. In contrast, model M31 assumed constant survival and MU-specific temporary emigration, thereby reducing movement-related bias in apparent survival estimates. By explicitly accounting for spatial variation in availability, it yielded a robust subpopulation-level survival

rate of 0.96, consistent with other coastal-estuarine *Tursiops truncatus* populations (Currey et al., 2009; Speakman et al., 2010; Ludwig et al., 2021).

4.2. Abundance of Lahille's bottlenose dolphins of SB-U subpopulation

Our regional and local estimates reveal critically low numbers of Lahille's bottlenose dolphins (LBD), highlighting vulnerability to stochastic events and human pressures. The SB-U subpopulation was estimated at 252–341 individuals (upper 95% CI = 348). Assuming 60% maturity (Taylor et al., 2007), this corresponds to ~215 mature dolphins. This small number is further constrained by low connectivity among MUs. Across five years, no individual was observed moving between MUs, aligning with genetic evidence of restricted dispersal (Fruet et al., 2014).

Despite high temporary emigration in some units, the absence of cross-MU recaptures suggests that movements typically occur within, not across, MUs. Dolphins likely leave core survey areas without dispersing more broadly. While occasional inter-MU movements occur (Simões-Lopes and Fabian, 1999; Laporta et al., 2016), they appear rare.

Table 2

Population Viability Analysis (PVA) results for Lahille's bottlenose dolphins in southern Brazil and Uruguay. Scenarios modelled stochastic population growth rates (r -stoch) and their standard deviation (SD_r), population change rates (Δ), final abundance (N), and extinction probability (PE) after one (21 yr) and three (63 yr) generations. Results are presented by scenario and Management Unit (MU): Florianópolis (FLN), Laguna (LGN), North Patos Lagoon (NPL), Patos Lagoon Estuary (PLE), South Patos Lagoon and Uruguay (SPLU), and the entire southern Brazil-Uruguay subpopulation (Sub).

Scenario	MU	Mean r -stoch	SD_r	Δ (%) in 21 yr	N after 21 yr	PE (%) in 21 yr	Δ (%) in 63 yr	N after 63 yr	PE (%) in 63 yr
Baseline	FLN	-0.0136	0.0079	-12	20	<0.1	-43	13	4
	LGN	-0.0279	0.0062	-31	33	0	-71	14	3
	NPL	0.0020	0.0033	+11	82	0	+22	91	0
	PLE	0.0053	0.0024	+17	120	0	+35	138	0
	SPLU	-0.0087	0.0036	-10	50	0	-28	40	<0.1
	Sub	0.0005	0.0016	+1	306	0	-2	297	0
Reduced bycatch	FLN	0.0049	0.0183	+7	24	0	-3	22	<0.1
	LGN	-0.0110	0.0181	-13	41	0	-39	29	<0.1
	NPL	0.0118	0.0194	+41	105	0	+69	125	0
	PLE	0.0166	0.0190	+47	150	0	+60	163	0
	SPLU	0.0024	0.0190	+12	63	0	+19	67	0
	Sub	0.0106	0.0188	+27	385	0	+34	407	0
Additive-bycatch	FLN	-0.0442	0.0211	-50	11	<0.1	-95	1	53
	LGN	-0.0519	0.0254	-63	17	0	-98	1	59
	NPL	-0.0499	0.0244	-33	49	0	-90	7	7
	PLE	-0.0500	0.0294	-25	76	0	-87	13	1
	SPLU	-0.0529	0.0210	-48	29	0	-94	3	30
	Sub	-0.0565	0.0282	-39	184	0	-91	26	<0.1
K-declining	FLN	-0.0178	0.0095	-16	19	0	-64	8	6
	LGN	-0.0303	0.0077	-33	32	0	-72	13	4
	NPL	-0.0004	0.0030	+6	79	0	-3	72	0
	PLE	0.0030	0.0027	+11	114	0	+0.5	102	0
	SPLU	-0.0113	0.0045	-13	48	0	-38	34	<0.1
	Sub	-0.0018	0.0017	+3	291	0	-24	230	0
Habitat-declining	FLN	-0.0233	0.0096	-22	18	0	-61	9	7
	LGN	-0.0380	0.0086	-44	26	0	-85	7	10
	NPL	-0.0049	0.0035	-3	71	0	-17	62	0
	PLE	-0.0021	0.0031	+2	104	0	-4	98	0
	SPLU	-0.0199	0.0066	-24	42	0	-58	24	0.4
	Sub	-0.0069	0.0020	-13	263	0	-34	199	0
Triple-threat	FLN	-0.0437	0.0204	-53	11	0.2	-96	1	60
	LGN	-0.0516	0.0277	-68	15	0	-99	1	69
	NPL	-0.0524	0.0240	-41	43	0	-93	5	11
	PLE	-0.0530	0.0291	-33	68	0	-91	9	2
	SPLU	-0.0538	0.0228	-55	25	0	-96	2	38
	Sub	-0.0608	0.0281	-46	163	0	-94	18	0.1

Asymmetric gene flow, with PLE as a hub (Fruet et al., 2014), likely reflects infrequent dispersal insufficient to affect short-term demographics. In this context, losing key units like PLE could disproportionately erode both demographic and genetic cohesion, while peripheral, low-abundance units like FLN remain especially vulnerable to local extinction due to limited recolonization potential.

No MU supported more than 120 individuals. PLE, the largest unit (30–35% of the subpopulation), likely plays a key role in overall viability, as highlighted by PVA results. In contrast, FLN and LGN had fewer than 55 individuals each. These differences suggest that factors beyond habitat type (coastal vs. estuarine), such as prey availability, social structure, or estuarine size, shape local abundance. FLN, at the northern limit, may also reflect low densities expected under the Abundant-centre hypothesis, which predicts decreasing population density toward the margins of a species' geographic range (Panter et al., 2023). Abundance fluctuations in open-coastal MUs (FLN, NPL, and SPLU) stem from variable survey effort and temporary emigration rather than demographic shifts. Insufficient effort in specific periods (e.g., FLN: 6–7; SPLU: 1–2), likely reduced capture probability and increased temporary unavailability, reflecting the short-term movements within each MU rather than across MUs.

The open nature of some MUs and adverse weather reduced detectability, likely accounting for abundance fluctuations (as noted by Lee et al., 2014), especially in dynamic units like FLN, and potentially contributing to underestimates. Similarly, the wide SPLU estimate range likely reflects effort differences across periods. At the subpopulation

scale, apparent changes in total abundance between the first and last periods are unlikely to represent real demographic shifts. Instead, they likely stem from variation in effort and spatial dynamics, particularly in open MUs where temporary emigration is more common, affecting the number of individuals available for capture.

Despite uneven sampling, results consistently indicate a very small subpopulation. Complementary aerial and boat-based surveys targeting other coastal cetaceans (e.g., franciscana *Pontoporia blainvillei* and southern right whale) along southern Brazil and Uruguay, including areas not sampled in this study, have recorded only occasional LBD sightings (Karina Groch pers. comm; GEMARS, unpubl. data). Similarly, opportunistic photographic records beyond core areas yielded high recapture rates of known individuals and few new identifications, reinforcing that the subpopulation is both small and demographically isolated.

4.3. PVA projections and uncertainties

Baseline projections indicate consistent population decline under the current baseline scenario for all MUs except NPL and PLE. FLN and LGN showed the steepest declines, followed by SPLU. PLE emerged as the most demographically stable unit, declining only under the most pessimistic scenarios, likely reflecting stronger demographic performance and more reliable monitoring. While suggesting resilience, particularly relative to LGN, interpretation requires caution, as estimates for open systems may be affected by sampling uncertainty.

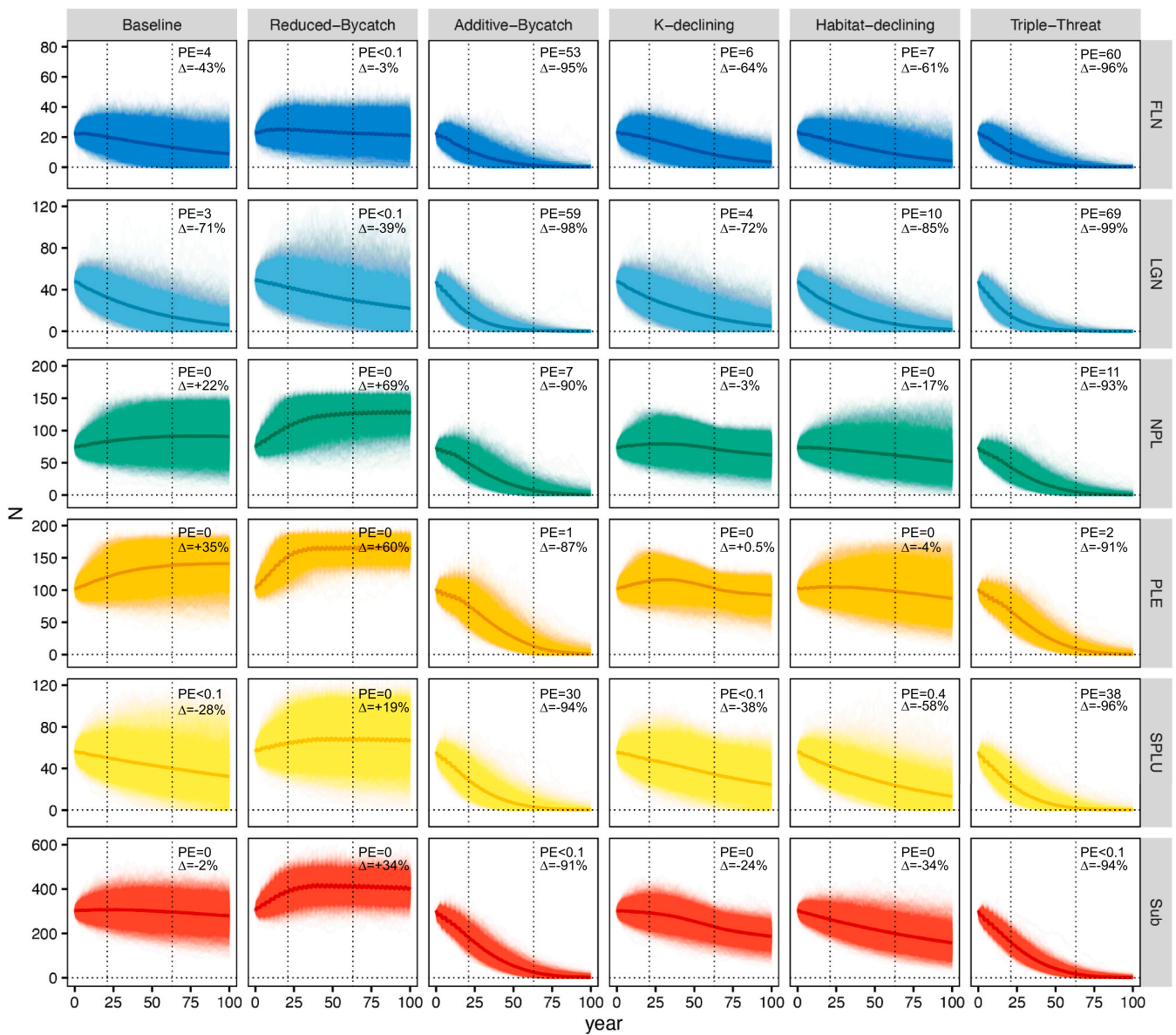


Fig. 4. Projected population trajectories over 100 years for each Management Unit (MU) and the full southern Brazil-Uruguay subpopulation (Sub) of Lahille's bottlenose dolphins under different scenarios: MUs include Florianópolis (FLN), Laguna (LGN), North Patos Lagoon (NPL), Patos Lagoon Estuary (PLE), and South Patos Lagoon and Uruguay (SPLU). Each line represents an iteration ($n = 5000$); the bold line shows the mean trajectory. Dashed black line indicates the extinction threshold ($N = 0$). Vertical dashed lines mark one (21 years) and three (63 years) generations. Each panel displays the extinction probability (PE) and population change (Δ) after three generations.

Bycatch was the most impactful threat, with sensitivity analyses confirming adult female mortality as a key demographic driver, consistent with findings for large vertebrates (Caughley, 1966) and LBD specifically (Fruet et al., 2021). Under current conditions, the subpopulation remains nearly stable over 100 years, but only the Reduced-bycatch scenario (50% reduction) led to meaningful recovery (+34% over three generations). Yet, FLN and LGN still declined (-3% and -39%), indicating partial mitigation may not ensure local persistence for low-abundance units where extinction risk is inherently higher. Conversely, doubling bycatch caused sharp declines in all MUs, resulting in a 50% subpopulation reduction within three generations and high extinction risks for FLN (53%) and LGN (59%). Resource decline and habitat degradation also reduced numbers, the latter more severely. The K-declining scenario led to a 24% drop over three generations, while habitat degradation to a 34% decline. As these pressures likely intensify and co-occur (Fruet et al., 2012; Sant'Ana et al., 2017; Righetti et al.,

2019), the Triple-threat scenario offers a plausible projection of future risks, predicting a 94% subpopulation reduction over three generations and marginal extinction risk. Critically, even this severe projection relies solely on established stressors, excluding, due to data limitations, emergent threats like infectious diseases.

Despite advances in monitoring and modelling, key uncertainties remain. Long-term datasets from LGN and PLE enabled robust demographic estimates, but parameters for other MUs were extrapolated from LGN, PLE, or literature. Sensitivity analyses identified the proportion of breeding females and juvenile/adult mortality as main drivers of population growth (Table S6), emphasizing the need to refine these parameters for FLN, NPL, and SPLU. Carrying capacity (K) also remains uncertain; though it becomes relevant mainly under growth scenarios, observed only with reduced bycatch. We based K on density estimates from similar systems, incorporated demographic stochasticity, and assumed a 0.5% annual K decline to reflect cumulative habitat

degradation, pollution, fishing, and prey depletion (Fruet et al., 2012, 2021; Secchi et al., 2017; Righetti et al., 2019). Continued coordinated monitoring is essential to refine all inputs, validate outputs, and our PVA projections.

4.4. Conservation implications

Globally, some cetacean populations are as small and fragmented as Lahille's bottlenose dolphins (LBD). Notable parallels include the vaquita, which plummeted from ~600 to ~10 individuals due to bycatch (Jaramillo-Legorreta et al., 2019); Maui's dolphin (*Cephalorhynchus hectori maui*), with fewer than 60 individuals (Hamner et al., 2014); and the Burrnun dolphin (*Tursiops cf. australis*) in southern Australia, estimated at under 150 individuals (Beddoe et al., 2024). While population thresholds like "<500 individuals" highlight demographic vulnerability (Traill et al., 2010), some small populations persist long-term under favourable conditions, through ecological buffering or local adaptations. For instance, vaquitas may have persisted for millennia at low effective sizes, possibly purging deleterious alleles (Robinson et al., 2022). Genomic analysis of LBD could clarify whether they represent a historically rare lineage with limited inbreeding depression, offering critical insights for conservation planning under persistent isolation and low numbers.

Unlike naturally rare species, LBD face declines driven primarily by human activities. Intensifying threats, such as bycatch, habitat degradation, and prey depletion (Fruet et al., 2012; Righetti et al., 2019), could push even a historically small population towards extinction. Evidence suggests a geographic contraction of their range, particularly in the northern portion (Itajaí–Florianópolis) (Carrion, 2014) and parts of northern Argentina (Vermeulen and Bräger, 2015). Our PVA indicates that cumulative habitat degradation alone could drive substantial declines, while bycatch poses an acute risk: losing even a few reproductive females significantly elevates extinction probability (Harvey Sky et al., 2022). Although historically frequent in southern Brazil, at least five bycatch events occurred in Uruguay since 2019, highlighting the urgency of binational mitigation. Even under optimistic scenarios, FLN and LGN remain highly vulnerable due to small size and limited recovery capacity, leaving them exposed to unusual mortality events.

Acute mortality can have disproportionate impacts on small, slow-growing populations like LBD. A morbillivirus outbreak in Rio de Janeiro, for instance, killed at least 277 Guiana dolphins (*Sotalia guianensis*) within five months, primarily females and calves (Groch et al., 2018). Emerging pathogens like avian influenza (H5N1) (e.g., Leguia et al., 2023; Uhart et al., 2024) raise concern for sudden, large-scale mortality that could rapidly erode reproductive capacity. Chronic or sublethal infections also compromise health; in other cetaceans, skin lesions from viral or fungal pathogens (e.g., poxvirus, *Brucella ceti*, *Lacazia loboii*) are linked to immunosuppression and reduced reproductive success (Van Bresse et al., 2009). Similar lacaziosis-like lesions are increasingly recorded in LBD, particularly in FLN and LGN (Daura-Jorge and Simões-Lopes, 2011; Sacristán et al., 2016), the units most susceptible to decline in our PVA. Integrating disease effects into future PVAs is critical for accurate risk assessment.

This vulnerability of LBD emphasizes the need for bycatch mitigation, habitat protection, and health monitoring. Lessons from other small cetaceans (see Manlik et al., 2016) confirm that persistence hinges on adult female survival and reproductive success, and juvenile survival. The likely extinction of the baiji, driven by bycatch and habitat degradation, highlights how anthropogenic pressures can cause collapse despite evolutionary persistence (Turvey et al., 2007). Although LBD have not yet reached this tipping point, their small numbers, fragmentation, and chronic exposure to multiple threats demand immediate, coordinated and sustained conservation actions across their entire range. Strengthening regional coordination through the Gephyreus Project will be key to maintaining long-term monitoring of MUs, understanding threats, and guiding effective management to secure the

conservation of LBD.

CRediT authorship contribution statement

Fábio G. Daura-Jorge: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Paula Laporta:** Writing – review & editing, Supervision, Resources, Investigation, Funding acquisition, Data curation, Conceptualization. **Ignacio B. Moreno:** Writing – review & editing, Supervision, Resources, Investigation, Funding acquisition, Data curation, Conceptualization. **Paulo H. Ott:** Writing – review & editing, Supervision, Resources, Investigation, Data curation, Conceptualization. **Paulo A.C. Flores:** Writing – review & editing, Supervision, Resources, Investigation, Data curation, Conceptualization. **Pedro Volkmer de Castilho:** Writing – review & editing, Supervision, Resources, Investigation, Data curation, Conceptualization. **André S. Barreto:** Writing – review & editing, Supervision, Resources, Investigation, Conceptualization. **Rodrigo Cezar Genoves:** Writing – review & editing, Investigation, Data curation, Conceptualization. **Carolina Bezamat:** Writing – review & editing, Investigation, Data curation, Conceptualization. **Alexandre M.S. Machado:** Writing – review & editing, Visualization, Software, Investigation. **Eduardo R. Secchi:** Writing – review & editing, Supervision, Resources, Funding acquisition, Conceptualization. **Silvina Botta:** Writing – review & editing, Supervision, Resources, Conceptualization. **Dan Jacobs Pretto:** Writing – review & editing, Investigation, Data curation, Conceptualization. **Rodrigo Machado:** Writing – review & editing, Investigation, Conceptualization. **Juliana C. Di Tullio:** Writing – review & editing, Supervision, Conceptualization. **Luciano Dalla Rosa:** Writing – review & editing, Supervision, Conceptualization. **Paulo C. Simões-Lopes:** Writing – review & editing, Supervision, Conceptualization. **Martin Sucunza Perez:** Writing – review & editing, Investigation. **Elisa B. Ilha:** Writing – review & editing, Investigation. **Yuri R. Camargo:** Writing – review & editing, Investigation. **Carolina Menchaca:** Writing – review & editing, Investigation, Funding acquisition. **Cecilia Laporta:** Writing – review & editing, Investigation, Funding acquisition. **Mariana Carrion:** Writing – review & editing, Investigation. **Liane Dias:** Writing – review & editing, Investigation. **Fernanda F. Peverari:** Writing – review & editing, Investigation. **Federico Sucunza:** Writing – review & editing. **Daniel Danilewicz:** Writing – review & editing, Conceptualization. **Lorenzo Von Fersen:** Writing – review & editing, Funding acquisition. **Pedro F. Fruet:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Ethical statement

Fieldwork was conducted in full compliance with Brazilian environmental regulations under the appropriate research permits issued by the Ministry of Environment (SISBIO licenses no. 24407-9 and 640/2015). As this study relied solely on non-invasive observational data (photo-identification) and did not involve animal capture, handling, or experimentation, specific institutional ethical committee approval was not required.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work, as non-native English speakers, the author(s) used GRAMMARLY strictly for language polishing and grammatical correction. After using this tool/service, the author(s) reviewed and edited the content as needed and take(s) full responsibility for the content of the published article.

Funding

This study was conducted within the scope of the Long-Term Ecological Research Programs PELD-SELA and PELD-ELPA, funded by CNPq (grants 446014/2024-9, 442206/2020-8), FAPESC (2021TR387, 2025TR001518), FAPERGS (21/2551-0000774-5) and CNPq (444601/2024-4). The Gephyreus Project received key support from Fundação Grupo Boticário, YAQU PACHA, Nuremberg Zoo, Zoomarine-Algarve, Zoo Duisburg, ICMBio, and the Whitley Fund for Nature. Portos RS funded monitoring efforts in the Patos Lagoon Estuary. The Projeto Botos da Barra (CECLIMAR) received financial support from Fundação Grupo Boticário and Propesq UFRGS. Research fellowships were provided by CNPq to F.G.D.-J. (308913/2022-0), E.R.S. (304616/2025-7) and P.C.S.-L. (303229/2024-1), and by FAPESC to A.M.S.M. (post-doctoral fellowship). L.P. was supported by ECCOSUR through Yaqu Pacha Uruguay. We also acknowledge CAPES for PROEX funding and access to the Portal de Periódicos.

Declaration of competing interest

None.

Acknowledgments

We thank Dr. Barbara Taylor for her insightful comments and careful review of an earlier version of this manuscript. We are deeply grateful to the many researchers, students, and volunteers from Brazil and Uruguay whose dedication to fieldwork and data collection made this large-scale, collaborative monitoring effort possible.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2026.111865>.

Data availability

All R code and anonymized datasets are available in a view-only repository on the OSF: https://osf.io/qanh9/overview?view_only=b2b1f94204bc40979814c74a01625b6d

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