

RESEARCH ARTICLE

A range-wide full-annual-cycle model informs conservation of a declining migratory shorebird

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Abstract

1. Untangling the spatial and temporal processes that influence the population dynamics of migratory species is challenging, because changes in abundance are shaped by variation in demographic rates across differing environments throughout the annual cycle. Population and demographic data available on migratory species are often fragmentary, providing only local, season-specific perspectives. To be effective, conservation strategies require range-wide, full-annual-cycle knowledge.
2. We developed a novel approach to link data from different segments of the population in a unified framework using stage- and season-specific population growth rates to assess population trends and conservation interventions for the New Zealand endemic migratory tōrea (South Island pied oystercatcher, *Haematopus finschi*). We combined demographic data collected at breeding and non-breeding sites between 1980 and 2022 in an integrated population model and population viability analysis to (a) determine population dynamics over time, (b) identify underlying drivers of change by estimating stage- and season-specific demographic rates and (c) evaluate future conservation interventions and interacting threats that are likely to primarily affect survival by comparing future population trajectories through a range of scenarios.
3. The tōrea population has been declining over the past 42 years by an average of 1.8% annually. Summer survival probabilities were generally higher than winter survival probabilities. Adult survival probability fluctuated less over time than the other demographic rates. Variation in population growth rates was most strongly associated with subadult winter survival ($r=0.29$).
4. The population viability analysis showed that conservation interventions aimed at improving productivity would only be beneficial if survival probabilities remain

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at current levels. It is unlikely that additional mortality due to future, increased threats on survival can be offset by increased productivity.

5. *Synthesis and applications.* The tōrea population decline suggests the need for a higher IUCN threat classification and a conservation strategy (e.g. habitat protection and restoration) targeting multiple demographic rates across the annual cycle of tōrea. Our novel approach using population growth rates to link fragmented local, season-specific data to model range-wide full-annual-cycle dynamics has the potential to guide the conservation of migratory species where existing data are similarly fragmented.

KEYWORDS

Haematopus finschi, integrated population model, management strategies, population dynamics, population viability analysis, seasonal survival, South Island pied oystercatcher

1 | INTRODUCTION

Populations of many migratory species are in decline as individuals encounter multiple threats within and between their seasonal breeding and non-breeding sites (Cooke et al., 2024; Saunders et al., 2025). These threats include degradation or loss of breeding and non-breeding habitats and mortality during migration (Cooke et al., 2024). Impacts of these threats at one site have the potential to cascade to population-wide impacts as migratory species move and depend on multiple sites throughout their annual cycles (Erickson et al., 2017). Consequently, it is difficult for conservation managers to know where to focus management efforts to halt declines (Faaborg et al., 2010).

The need for a full-annual-cycle approach to evaluate conservation strategies for migratory species is increasingly recognised (Hostetler et al., 2015), because it offers the opportunity to determine which seasons and locations are limiting migratory populations. Conservation managers can then make more informed decisions to avoid wasted conservation resources and reduced population viability (Martin et al., 2007). However, examples of such approaches are rare (Brown et al., 2016; Davis et al., 2023; Marolla et al., 2023). It is challenging to develop demographic models for migratory species because they move over large areas, and different segments of the population (e.g. age classes or sexes) often occupy different parts of the range throughout the annual cycle (Hostetler et al., 2015). Few survey methods provide the geographical and temporal coverage needed for range-wide monitoring of demographic rates, and this results in localised and temporally disjunct data from which it is challenging to infer broad-scale population trends (Saunders et al., 2019; Zipkin & Saunders, 2018).

Integrated population models (IPMs) are a recent advance that addresses some of the challenges of demographic models for migratory species (Schaub & Kéry, 2022). In an IPM, detailed demographic data at fine scales can be combined with broad-scale survey data (Ahrestani et al., 2017), and seasonal survival can be estimated by combining data sets collected at different times during the annual

cycle (Rushing et al., 2017). IPMs can be easily expanded to population viability analyses (PVAs), which have the advantage that all information in various data sets, and related uncertainty, is propagated through management scenarios (Saunders et al., 2018).

Here we demonstrate a novel approach in an IPM which uses fragmentary, season-specific monitoring data of the migratory New Zealand tōrea (South Island pied oystercatcher, *Haematopus finschi*), to assess population dynamics and inform conservation strategies. The tōrea population is thought to have been declining since the mid-2000s based solely on counts at non-breeding sites but underlying demographic drivers of the decline remain unknown (Riegen & Sagar, 2020). In a full-annual-cycle IPM, we combined monitoring data capturing different stage classes during breeding and non-breeding seasons and linked these data through the population growth rates. We then extended the IPM to a PVA to evaluate future population trajectories. Our goal was to assess how the combination of threats that primarily impact survival throughout the annual cycle affects tōrea population trajectories. Climate change-induced habitat loss and degradation have been identified as a key threat to tōrea (Brumby et al., 2025). Furthermore, renewable energy infrastructure is also expected to expand significantly in New Zealand in the future (MBIE, 2024), and survival of tōrea may decrease due to collisions during movement and the habitat loss and degradation associated with such infrastructure (Powlesland, 2009). Although it can be expected that productivity is less influential than adult survival on the population dynamics of a long-lived shorebird (Sæther & Bakke, 2000), present conservation actions to compensate for increased tōrea mortality at windfarms focus on increasing productivity through predator control at breeding sites (Bennet et al., 2022; Craig et al., 2015).

To improve our understanding of current and future population trends in tōrea, we aim to:

1. determine range-wide population dynamics by combining seasonally specific data from different stage classes in a unified modelling framework,

2. estimate stage- and season-specific demographic rates to identify how demographic rates change over time and influence population dynamics and then
3. assess future population trajectories under a range of scenarios encompassing conservation management and future threats. We specifically test whether increased mortality, which is expected following windfarm development, can be compensated by increased productivity at breeding sites.

2 | MATERIALS AND METHODS

2.1 | Focal species and study region

Tōrea are short-distance migrants endemic to New Zealand. Breeding occurs in the austral spring (August to December) in inland riverbeds or farmland, primarily in the South Island (Figure 1), followed by movements to coastal estuaries in the North and South Islands for the austral autumn and winter resulting in low migratory connectivity (January to July; Sagar & Veitch, 2014). Individuals start to breed at 4–6 years old (Sagar et al., 2000), and subadults generally remain coastal over the breeding season (Sagar & Geddes, 1999).

2.2 | Data sets

Four primary data sets were available for tōrea populations (Figure 1; Figure S1):

1. productivity data from breeding areas,
2. mark-resight data from banding on breeding and non-breeding areas,
3. coastal non-breeding counts from throughout New Zealand,
4. river counts of breeding adults at key breeding sites on South Island rivers.

2.2.1 | Productivity data

Nests were monitored annually (August–November) across farmland around Mayfield (1987–2000 [14 years]; Sagar et al., 2000; $n=421$ broods), and across both farmland and riverbed in the Rangitata valley (2020–2022 [3 years]; $n=184$ broods; Table S1). While only a small percentage of breeding pairs have been monitored each breeding season (Table S1), we believe they represent an unbiased sample of productivity from the centre of their range. Nests of tōrea were located by observing breeding pairs. Nest contents and pairs with chicks were checked at 2–5-day intervals. Tōrea lay up to two replacement clutches if breeding attempts fail early but are not double-brooded (Sagar et al., 2000). Chicks were considered successfully fledged at 21 days, at which point it was possible to mark individuals with bands. We estimate productivity as the number of fledglings produced per monitored female per season.

2.2.2 | Banding mark-resight data

Nesting adults were captured with drop traps at two breeding sites: Mayfield (1987–2004 [18 years]; Sagar et al., 2000; $n=245$) and Rangitata valley (2020–2022 [3 years]; $n=77$). In the Rangitata valley, chicks were captured by hand ($n=70$). Full-grown individuals of all ages were captured by cannon netting at one of the main non-breeding sites (January–February) in the Tasman and Golden Bay area (2008–2018 [11 years]; Figure 1; Figure S2; Table S1; $n=177$).

All individuals were banded with metal and either coloured bands (Mayfield) or alpha-numeric flag-bands (Rangitata, and Tasman and Golden Bay). Individuals were aged as juvenile, subadult or adult through a combination of plumage, moult, primary feather wear, bare body parts and iris colouration. Thorough searches of breeding study sites to re-sight individuals banded at breeding sites were carried out annually (September–November; Figure 1). Observers reported resightings of individuals banded at breeding or non-breeding sites during the non-breeding season (January–May) from throughout New Zealand (Figure 1).

2.2.3 | Coastal non-breeding counts

Tōrea were counted at estuaries (June–July) by observers from Birds New Zealand (1984–2022 [39 years]; Figure 1). Counts were synchronised within a region to minimise errors from the movement of birds between sites (Riegen & Sagar, 2020). At tidal sites observers timed counts with high tide, when waders congregate at roosting sites, and used scopes for accurate species identification and counting. A walk-through survey was carried out at coastal lakes. We selected 15 non-breeding sites across New Zealand that were regularly surveyed (Figure S3). These data represent 38% of all sites that have ever been surveyed (Riegen & Sagar, 2020) but encompass more than 90% of counted tōrea in 2022 because all major non-breeding sites are included.

2.2.4 | River breeding counts

Breeding tōrea were counted at rivers during the breeding season (September–December) using a walk-through method (1980–2021 [42 years]; Figure 1; O'Donnell & Moore, 1983). In each survey the full width of the riverbed encompassing all potential habitat was surveyed by observers forming a line with approximately 50m spacing. The observers walked downstream, counting all birds seen as they passed them and using binoculars for accurate species identification and counting. Double counting of flying birds was avoided by communication among observers. We selected data from 23 rivers, where standard sections were surveyed at least five times during the 42-year period (Figure S4).

Handling of tōrea was carried out under Department of Conservation Animal Ethics Committee approval AEC 363

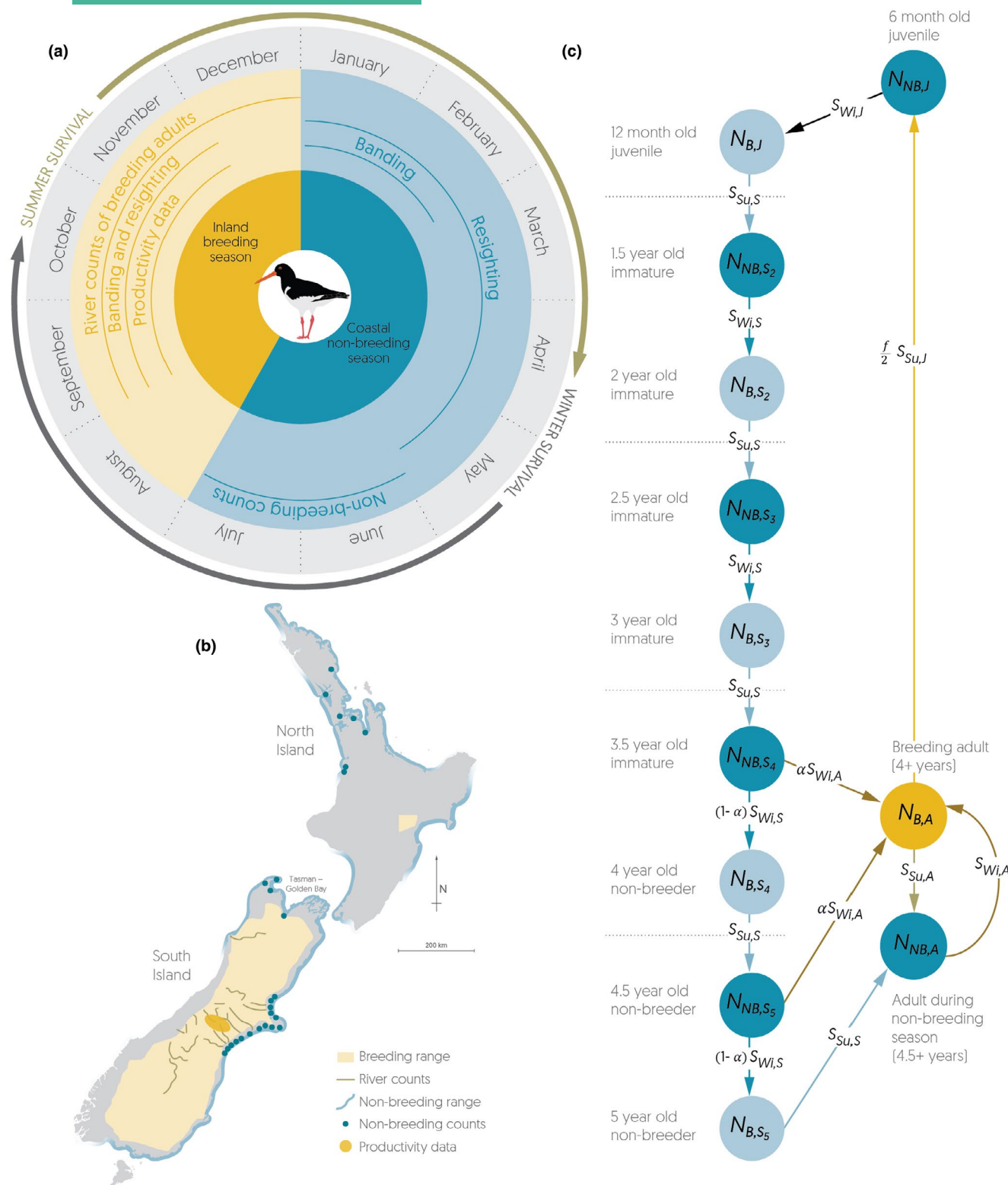


FIGURE 1 (a) The life cycle of tōrea and seasonally collected datasets used in the integrated population model. (b) Seasonal ranges of tōrea and coverage of data. Banding and resighting during the breeding season occurred only at the same sites where productivity data were collected; banding at non-breeding sites only occurred at Tasman/Golden Bay. Non-breeding resightings for all banded birds spanned the entire non-breeding range. (c) The life cycle graph. The seasonal stage-structured model distinguishes six demographic stages during the breeding (B) and non-breeding (NB) season: N_J = Juveniles <12 months old, N_{S_2} = 1.5- and 2-year-old immature subadults, N_{S_3} = 2.5- and 3-year-old immature subadults, N_{S_4} = 3.5- and 4-year-old non-breeding subadults, N_{S_5} = 4.5- and 5-year-old non-breeding subadults, N_A = 4 years or older breeding adults. Arrows show demographic processes governed by demographic rates: Seasonal (k) and stage-specific (c) survival ($S_{k,c}$), productivity (f) and the proportion of subadults that mature to become adults (α). Please note that the year index of the parameters has been omitted for simplicity.

and Wildlife Act authorisations issued by the Department of Conservation to the Ornithological Society of New Zealand.

2.3 | Integrated population model

We estimated productivity, seasonal survival probabilities, proportion of subadults that start to reproduce, population sizes and growth rates for tōrea by developing an IPM with a stage-structured formulation with both biological and observational processes (Schaub & Kéry, 2022). We distinguished six demographic stages per season (Figure 1): juveniles, subadults (>1–3-year-old immatures and 4–5-year-old non-breeders) and adults. Because our IPM is a seasonal model, the age of the individuals in these stages changes seasonally as shown in Figure 1.

The IPM consisted of four conditionally related submodels (Figure S5) that link the demographic data to seasonal abundance data through shared parameters in the biological process model.

2.3.1 | Biological process model

We described seasonal abundances of tōrea ($N_{t,k,c}$) for each of the six demographic stages in two time periods annually (coastal non-breeding season and inland breeding season; Figure 1). The index t represents year, k represents season (coastal non-breeding = NB, inland breeding = B) and c represents the stage class (J = Juvenile, S_i = Subadult with $i = 2, 3, 4$ or 5 depending on age, A = Adult). For the season index, k , we defined the non-breeding season as the period from 1 January to 31 July and the breeding season as the period from 1 August to 31 December. We indicate seasonal survival from the midpoints: 15 April to 14 October as winter ($s_{t,Wi,c}$) survival including the migration of breeders from coastal non-breeding areas to inland breeding grounds and 15 October to 14 April as summer ($s_{t,Su,c}$) survival including the migration of breeders and fledged juveniles from inland breeding grounds to coastal non-breeding areas.

Seasonal population sizes of juveniles were estimated as:

$$N_{t,NB,J} = \frac{N_{t-1,BA}}{2} \times f_t \times s_{t,Su,J} \quad (1)$$

$$N_{t,B,J} = N_{t,NB,J} \times s_{t,Wi,J} \quad (2)$$

Juvenile non-breeding season abundance ($N_{t,NB,J}$) is the product of the number of breeding pairs during the previous breeding season ($\frac{N_{t-1,BA}}{2}$), productivity (f_t), and summer survival of juveniles ($s_{t,Su,J}$). We assume that all adults attempt breeding and a 1:1 sex ratio (Sagar et al., 2002; Sagar & Veitch, 2014). Juvenile abundance the following breeding season ($N_{t,B,J}$) is a result of winter survival ($s_{t,Wi,J}$) of those juveniles ($N_{t,NB,J}$).

Seasonal population sizes of subadults were estimated for each subadult age class separately (Figure 1). Population sizes during the non-breeding season were estimated as:

$$N_{t,NB,S_i} = N_{t-1,B,S_{i-1}} \times s_{t,Su,S_i} \quad (3)$$

Abundance for given subadult class i (2–5) during the non-breeding season (N_{t,NB,S_i}) is the product of the number of the juveniles ($N_{t-1,NB,J}$ for $i = 2$) or lower breeding subadult class for older ($i > 2$) stages ($N_{t-1,B,S_{i-1}}$) of the previous year (Figure 1) and subadult summer survival (s_{t,Su,S_i}).

None of the 2- or 3-year-old subadults are breeding and their population size during the breeding season (N_{t,B,S_i} ; $i = 2, 3$) was estimated as the product of the number of that subadult class during the non-breeding season (N_{t,NB,S_i}) and subadult winter survival (s_{t,Wi,S_i}):

$$N_{t,B,S_i} = N_{t,NB,S_i} \times s_{t,Wi,S_i} \quad (4)$$

The 4- and 5-year-old subadults may start to breed. Population sizes of 4- or 5-year-old non-breeding subadults during the breeding season (N_{t,B,S_i} ; $i = 4, 5$) were estimated as:

$$N_{t,B,S_i} = N_{t,NB,S_i} \times (1 - \alpha_t) \times s_{t,Wi,S_i} \quad (5)$$

where $1 - \alpha_t$ is the proportion of subadults that remains as non-breeding subadults (N_{t,B,S_i}) and s_{t,Wi,S_i} is subadult winter survival (Figure 1). The proportion of subadults maturing to breeders (α_t) is a latent variable estimated from the model based on changes in overall population size, changes in adult breeding population size and survival rates.

We consider the adult population to consist of mixed ages of individuals from 4 years or older (Figure 1) that all attempt to breed during the breeding season and consider 4- and 5-year-old individuals that do not attempt to breed as subadults.

Adult population sizes during the non-breeding season ($N_{t,NB,A}$) were estimated as:

$$N_{t,NB,A} = (N_{t-1,BA} \times s_{t,Su,A}) + (N_{t-1,B,S_5} \times s_{t,Su,S_5}) \quad (6)$$

where $N_{t-1,BA}$ and N_{t-1,B,S_5} are the previous inland breeding season adult and coastal non-breeding 5-year-old subadult abundance, respectively, and $s_{t,Su,A}$ and s_{t,Su,S_5} are adult and subadult summer survival, respectively.

Adult population sizes during the breeding season ($N_{t,B,A}$) was estimated as:

$$N_{t,B,A} = ((\alpha_t \times (N_{t,NB,S_4} + N_{t,NB,S_5})) + N_{t,NB,A}) \times s_{t,Wi,A} \quad (7)$$

It includes adults ($N_{t,B,A}$) and 3.5- and 4.5-year-old subadults that recruited into the breeders ($\alpha_t \times (N_{t,NB,S_4} + N_{t,NB,S_5})$) during the non-breeding season in the same year, and winter survival ($s_{t,Wi,A}$; Figure 1).

The total seasonal (k) abundance in each year ($N_{t,k}$) is the sum of all stage classes:

$$N_{t,k} = N_{t,k,J} + N_{t,k,S_2} + N_{t,k,S_3} + N_{t,k,S_4} + N_{t,k,S_5} + N_{t,k,A} \quad (8)$$

2.3.2 | Estimation of life-cycle parameters

Productivity submodel

We estimated productivity (f_t) from the ratio of the number of chicks reaching 21 days of age (J_t) to the number of pairs monitored (B_t), from birds monitored at two South Island sites. We assume

$$J_t \sim \text{Poisson}(B_t \times f_t) \quad (9)$$

and estimated productivity with a year-specific temporal random effect as

$$\log(f_t) \sim \text{normal}(\bar{f}, \sigma_f^2) \quad (10)$$

where $\bar{f}$ is mean productivity and σ_f^2 is the temporal variation of productivity.

Cormack-Jolly-Seber (CJS) submodel

We used banding and resighting data from breeding and non-breeding sites to estimate season- and stage-specific apparent survival (hereafter survival) and detection probabilities within a CJS model (Lebreton et al., 1992). Seasonal resight data were collected for individuals banded at breeding sites as resight surveys were carried out at breeding sites and across non-breeding sites. However, for individuals banded at non-breeding sites, only annual survival could be estimated as no systematic surveys were carried out across all possible breeding sites. For individuals banded at Tasman and Golden Bay, seasonal survival was therefore estimated as latent survival. Because of the broad spatial coverage of non-breeding sites with regular surveys (Figure 1), the probability of permanent emigration by individuals to other sites is low and estimates of survival should be close to true survival here.

We used the state-space formulation of the CJS model, and defined the combination of year and season as an occasion (j , two per calendar year):

$$z_{ij} \sim \text{Bernoulli}(z_{i,j-1,k,c} \times s_{j-1,c}) \quad (11)$$

$$y_{ij} \sim \text{Bernoulli}(z_{ij} \times p_{j,c}) \quad (12)$$

where z_{ij} is the true state (0=dead, 1=alive) of individual i in occasion j , $s_{j-1,c}$ is the survival probability of the given stage c (Juvenile, Subadult, Adult) from occasion $j-1$ to occasion j , y_{ij} is the observed state (0=not observed, 1=observed) of individual i in occasion j , and $p_{j,c}$ is the probability of detecting an individual of stage c in occasion j , given that it is alive and at the site. We estimate a single survival probability across subadult ages (Figure 1) as data for known-age subadults was limited (Table S1). All even and all odd occasions refer to one of the two seasons, respectively, from which we estimated a year-specific temporal random effect of seasonal stage-specific survival and respective detection probability as

$$\text{logit}(s_{t,k,c}) \sim \text{normal}(\bar{s}_{k,c}, \sigma_{s_{k,c}}^2) \quad (13)$$

$$\text{logit}(p_{t,k,c}) \sim \text{normal}(\bar{p}_{k,c}, \sigma_{p_{k,c}}^2) \quad (14)$$

where $\bar{s}_{k,c}$ is mean seasonal (k) survival probability of stage c , $\sigma_{s_{k,c}}^2$ is the temporal variation of seasonal survival, $\bar{p}_{k,c}$ is the mean seasonal detection probability, and $\sigma_{p_{k,c}}^2$ is the temporal variation around it.

For birds banded at the coastal sites, we estimated annual survival ($s_{t,An,c}$) for stages, c , using a similar state-space formulation of the CJS model as described above. We expressed annual survival as the product of latent seasonal survival which enabled us to link the estimates originating from the datasets of birds banded at the breeding and at the non-breeding sites:

$$s_{t,An,c} = s_{t,Su,c} \times s_{t,Wi,c} \quad (15)$$

Non-breeding count state-space submodel

To inform the winter population size we used annual non-breeding count data ($C_{t,NB,m}$) of tōrea collected across 15 sites (m). We estimated an annual non-breeding index ($I_{t,NB}$) as

$$C_{t,NB,m} \sim \text{normal}(I_{t,NB}, \sigma_{t,NB}^2) \quad (16)$$

where $\sigma_{t,NB}^2$ is the annual variance of the index. We linked $I_{t,NB}$ to the annual population growth rate ($\lambda_{t,NB}$) of the total non-breeding population (Equation 8) by assuming that both estimators follow similar trajectories:

$$\frac{I_{t+1,NB}}{I_{t,NB}} = \frac{N_{t+1,NB}}{N_{t,NB}} = \lambda_{t,NB} \quad (17)$$

Breeding count state-space submodel

To inform the adult breeding population size, we estimated an annual index ($I_{t,B}$) of breeding tōrea from annual braided river survey data ($C_{t,B,n}$) collected across 23 rivers (n):

$$C_{t,B,n} \sim \text{normal}(I_{t,B}, \sigma_{t,B}^2) \quad (18)$$

where $\sigma_{t,B}^2$ is the annual variance of $I_{t,B}$. We linked $I_{t,B}$ to the annual population growth rate of the adult breeding population ($N_{t,B,A}$) by assuming that both estimators follow similar trajectories:

$$\frac{I_{t+1,B}}{I_{t,B}} = \frac{N_{t+1,B,A}}{N_{t,B,A}} = \lambda_{t,B,A} \quad (19)$$

2.3.3 | Model implementation

We used the programme JAGS (Plummer, 2003) implemented through R using the package jagsUI (Kellner, 2021; R Core Team, 2024) to estimate posterior distributions of demographic parameters. We ran 12 independent chains of 200,000 iterations, a burn-in of 20,000 iterations and an adaptation phase of 1000 iterations, thinning chains by 250, yielding a total of 8640 samples from the joint posterior. We used vague prior distributions for all parameters, except for the initial population size and the proportion of subadults that start to reproduce, for which we used informative priors (see Supporting Information S1 for details). Model convergence was confirmed visually and using Rhat-statistics <1.05 (Gelman & Hill, 2006). We assessed the ability of the IPM to produce

biologically relevant estimates of demographic rates and a realistic representation of *tōrea* population dynamics, first by plotting predictions of abundance and productivity against relevant observational data to ensure predictions were not substantially biased (Figure S6), and second by comparing the posterior distributions of demographic rates obtained from the IPM and from models of the demographic rates independently (Schaub & Kéry, 2022). Medians and 95% Bayesian credible intervals (BCIs) of posterior distributions were calculated.

2.4 | Correlation of population growth with demographic rates

We evaluated the relative contribution of demographic rates (i.e. seasonal stage-specific survival, as well as productivity) to variation in population growth of breeding adults during the breeding season

($\lambda_{t,B,A}$; Equation 19). We calculated the Pearson's correlation coefficient r and 95% BCI between the demographic parameters and the population growth rates using the full posterior samples to appropriately propagate errors. In addition, we calculated the probability that the correlations were positive ($P(r > 0)$; Schaub & Kéry, 2022).

2.5 | Assessment of management alternatives and possible future threats

We compared five future scenarios in our PVA based on achievable management interventions and/or increasing threats and current compensatory measures (Table 1). Under each scenario we simulated future population trajectories of the total non-breeding population as part of the model fitting process by directly manipulating demographic rates for future years as detailed in Table 1 (Schaub & Kéry, 2022). Under each scenario, we projected 30 years (i.e. about

TABLE 1 Details of five scenarios of management alternatives and future threats used in the PVA for *tōrea*.

Scenario	Explanation	Justification	References
1. No change (control)	Predator control of varying intensity at a small subset of priority breeding sites with the aim of increasing productivity.		O'Donnell et al. (2016)
2. Increase productivity by 15%	Landscape-level nest and chick predator management across breeding areas targeting the entire suite of invasive mammals and native avian predators. Increase available nesting habitat through clearance and control of invasive weeds in braided riverbeds and ensure adequate river flows. Where river flows are controlled through hydro dams, avoid highly fluctuating water levels during the breeding season. Prohibit 4WD, quad and motorcross bike access to rivers and limit disturbance by people and dogs during the breeding season.	Level of increase is within the range of productivity observed for <i>tōrea</i> and for other oystercatcher species.	Ens and Underhill (2014); O'Donnell et al. (2016); Norbury et al. (2021)
3. Increase subadult and adult survival by 5% and juvenile survival by 10%	Increased current minimum level of conservation management at coastal non-breeding sites to increase survival through buffer zones to halt development and pollution around stopover and non-breeding season sites, limiting disturbance by people and dogs at stopover and non-breeding sites, restoration and enhancement of habitat quality within estuary networks, including sustaining food supplies through shellfishery management.	Level of increase for survival is within the range of survival observed for other oystercatcher species.	Schmechel (2001); Dowding and Moore (2006); Ens and Underhill (2014); Santos et al. (2023); Schlesselmann et al. (2024)
4. Decrease subadult and adult survival by 1% and juvenile survival by 2%	Habitat loss and degradation of breeding, non-breeding and feeding areas through increased development, climate change impacts such as reduced food availability at coastal sites, weather extremes reducing high tide roost availability and increased risk of collision with accelerated development of solar and wind infrastructure.	Possible cumulative effect on survival from a range of future threats that may be localised or widespread.	Powlesland (2009); Thaxter et al. (2017); Griffin et al. (2023); MBIE (2024); Brumby et al. (2025)
5. Decrease subadult and adult survival by 1% and juvenile survival by 2% while increasing productivity by 15%	Current compensation of possible decreased survival targets increasing productivity; see (2) and (4)	As for (2) and (4)	Craig et al. (2015); Bennet et al. (2022)

three generations given the estimated generation time based on the population model of 10.6 years; Supporting Information S1) into the future. We calculated (i) the probabilities that scenarios with management or increased impact of threats would result in larger population sizes in 30 years (i.e. 2052) as compared to Scenario 1—No change and (ii) the cumulative probability of a population size reduction by 50% over 30 years as this threshold would trigger a Red List classification of Endangered (Criterion A3; IUCN, 2024).

3 | RESULTS

3.1 | Population abundance and trend

Our IPM shows that the *tōrea* population declined over the 42-year study period. The estimated posterior median of the total population size at the surveyed sites during the coastal non-breeding season declined from 87,889 individuals (BCI: 66,089–119,271) in 1980 to 48,588 (BCI: 27,388–83,963) by 2022, with the pattern mirrored in numbers of breeding adults (Figure 2a). The annual rate of population change ($\lambda_{t,B,A}$) of adults during the breeding season ranged

from 0.91 (BCI: 0.81–1.13) to 1.15 (BCI: 0.90–1.74) over the study period (Figure 2b), and the long-term geometric mean of the annual population growth rates suggests declines of 1.8% per year over the 42-year study period (0.982, BCI: 0.982–0.983). See Tables S2–S17 for details on population sizes of all stage classes and parameter estimates.

3.2 | Population demographic rates

Summer survival probabilities were generally higher compared with winter survival probabilities of *tōrea* (Figure 3; Table 2). While adult survival probability was relatively stable across seasons, winter survival probabilities of juveniles and subadults were more variable than their summer survival probabilities (Figure 3a–f; Table 2). As a result, median annual survival probabilities for the 42-year study period were 0.87 (BCI: 0.85–0.89) for adults, followed by 0.78 (BCI: 0.65–0.90) for subadults and lowest for juveniles with 0.67 (BCI: 0.45–0.93; Table 2). Credible intervals around survival probabilities were wide for juveniles and subadults, and particularly wide for winter survival, in part due to their lower resighting probability (Table 2).

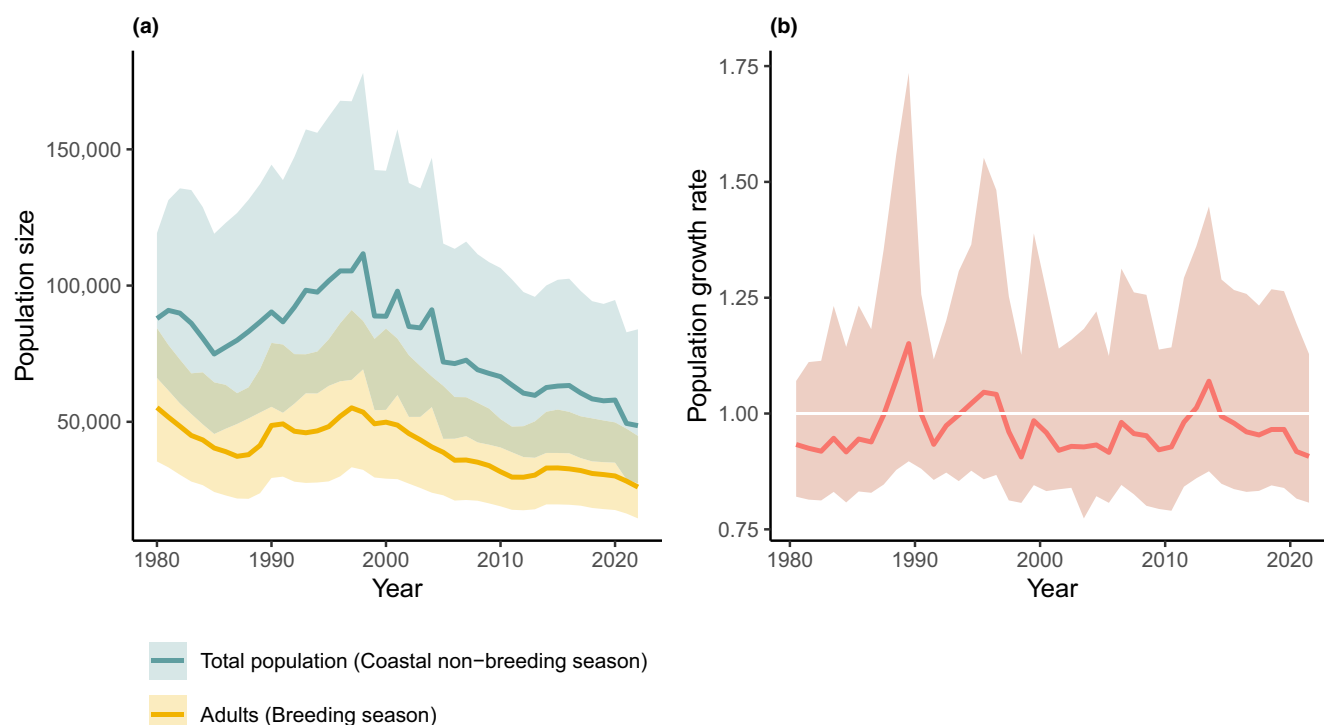


FIGURE 2 (a) Estimates of the total (combined adults, subadults and juveniles) during the non-breeding season and breeding adult population size; and (b) annual population growth rates of the breeding adult population of *tōrea* ($\lambda_{t,B,A}$) in New Zealand, 1980–2022. Shading represents 95% credible intervals. $\lambda = 1$ is represented by the white line.

FIGURE 3 Estimates of (a) adult summer survival, (b) subadult summer survival, (c) juvenile summer survival, (d) adult winter survival, (e) subadult winter survival, (f) juvenile winter survival, (g) adult annual survival, (h) subadult annual survival, (i) juvenile annual survival and (j) productivity *tōrea* in New Zealand, 1980–2022. Shading represents the 95% Bayesian credible interval. The duration of the two seasonal periods is 6 months.

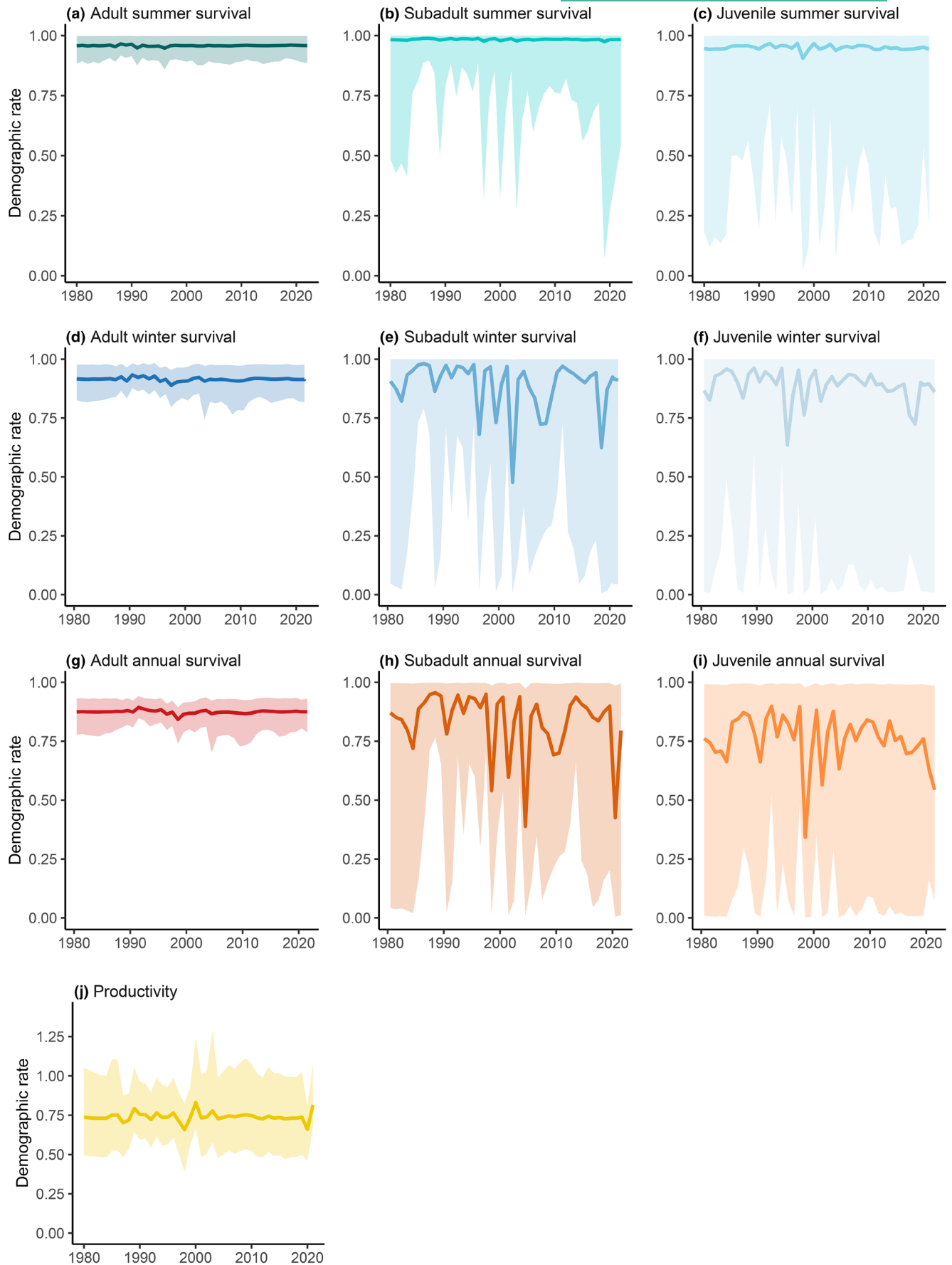


TABLE 2 Estimated medians and 95% Bayesian credible intervals of demographic parameters of tōrea.

Parameter		Median	Lower 95% BCI	Upper 95% BCI
Summer survival probability				
Adults	$\overline{s_{Su,A}}$	0.96	0.92	1.00
Subadults	$\overline{s_{Su,S}}$	0.98	0.91	1.00
Juveniles	$\overline{s_{Su,J}}$	0.95	0.72	1.00
Winter survival probability				
Adults	$\overline{s_{Wi,A}}$	0.92	0.87	0.96
Subadults	$\overline{s_{Wi,S}}$	0.92	0.75	0.99
Juveniles	$\overline{s_{Wi,J}}$	0.89	0.51	1.00
Annual survival probability				
Adults	$\overline{s_{An,A}}$	0.87	0.85	0.89
Subadults	$\overline{s_{An,S}}$	0.77	0.65	0.90
Juveniles	$\overline{s_{An,J}}$	0.67	0.45	0.93
Productivity	$\bar{f}$	0.74	0.64	0.83
Proportion of first-time breeders	$\bar{\alpha}$	0.54	0.24	0.81
Summer resight probability				
Adults	$\overline{p_{Su,A}}$	0.17	0.10	0.26
Subadults	$\overline{p_{Su,S}}$	0.28	0.01	0.89
Juveniles	$\overline{p_{Su,J}}$	0.25	0.04	0.81
Winter resight probability				
Adults	$\overline{p_{Wi,A}}$	0.50	0.27	0.72
Subadults	$\overline{p_{Wi,S}}$	0.30	0.09	0.60
Juveniles	$\overline{p_{Wi,J}}$	0.24	0.02	0.81

The median probability of subadults maturing to breed during the study period was 0.54 (BCI: 0.25–0.81; Table 2). Across the study period, a median of 0.74 (BCI: 0.64–0.83) offspring were produced per female per year with some year-to-year variation (Figure 3; Table 2).

3.3 | Drivers of population change

There was weak correlation of subadult winter survival ($r=0.29$, BCI: 0.00–0.55) with breeding adult population growth rates (Figure 4e). The probability of a positive correlation was also highest for subadult winter survival ($P(r>0)=0.97$) compared with other demographic rates. All other demographic rates were not correlated with population growth rates (Figure 4).

3.4 | Comparison of management alternatives and threats

In comparison to no change in conservation management (Scenario 1), increased productivity (Scenario 2) and increased survival

(Scenario 3) resulted in 0.99 and 0.97 probabilities, respectively, of achieving larger population sizes in 30 years. Decreased survival (Scenario 4) resulted in a low probability of 0.21 of achieving larger population sizes in 30 years compared to the no change scenario, and this probability of achieving larger population sizes increased to only 0.37 when decreased survival was offset by increased productivity (Scenario 5).

The probability of a 50% population reduction 30 years into the future with no change in management (Scenario 1) was 0.58 (Figure 5). In comparison, this probability was lowest (0.05) in the scenario with increased survival (Scenario 3) and second lowest (0.40) in the scenario with increased productivity (Scenario 2; Figure 5). Decreased survival with increased productivity (Scenario 5) and decreased survival without further mitigation (Scenario 4) both resulted in higher probabilities of a 50% population reduction (0.69 and 0.85, respectively; Figure 5).

4 | DISCUSSION

We established that the tōrea population declined by >1% annually over the 42-year study period. Our result is consistent with

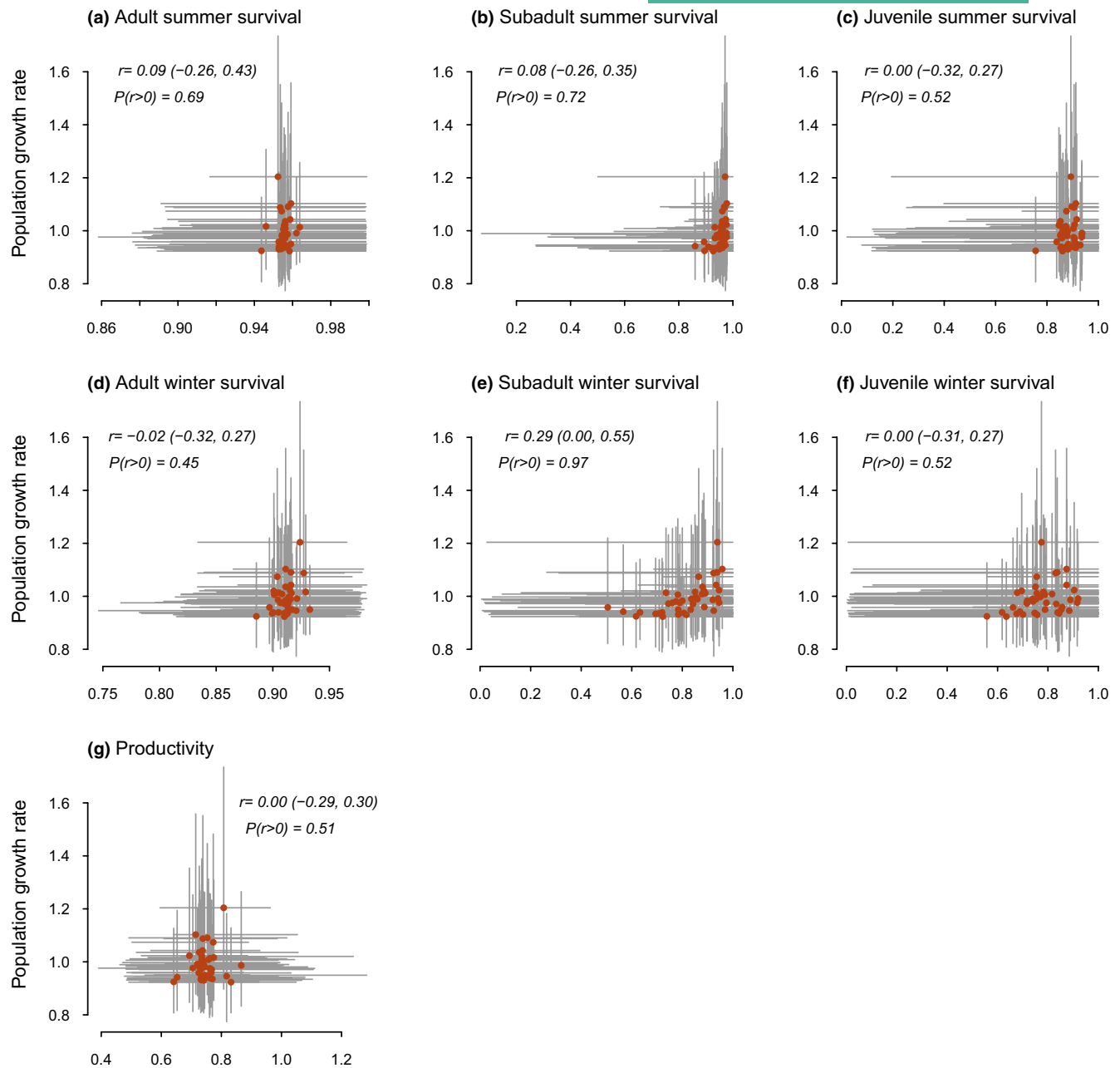


FIGURE 4 Annual posterior means of the estimated population growth rates ($\lambda_{t,B,A}$) plotted against posterior means of the estimates for (a) adult summer survival, (b) subadult summer survival, (c) juvenile summer survival, (d) adult winter survival, (e) subadult winter survival, (f) juvenile winter survival and (g) productivity from an integrated population model of tōrea, 1980–2022. Posterior means of the correlation coefficients (r), associated 95% credible intervals and probabilities that estimates are positive ($P(r>0)$) are given. Horizontal and vertical lines show the limits of the 95% credible intervals for each estimate.

previous studies showing a reduction in inland breeding range occupancy based on national surveys comparing 1969–1979 with 1999–2004 (Occupancy: 40.5 [38.0, 44.1] cf. 32.6 [30.9, 34.1]; Walker et al., 2020) and declines in abundance during the non-breeding season from 2005 to 2019 (1.2% per year; Riegen & Sagar, 2020). The decline in tōrea meets the IUCN Red List criterion of a documented population reduction of $\geq 30\%$ over three generations and tōrea could be considered Vulnerable rather than Least Concern (IUCN, 2024).

Our results suggest that the long-term decline of tōrea is most likely driven by survival. Our PVA highlighted that increased productivity will be less effective at changing population trajectories than management that increases survival. Moreover, we found that increasing productivity improved the current population trajectory only if survival remained at current levels. Long-lived species are thought to experience demographic buffering, that is selection against temporal variance in the demographic rate that most strongly influences population growth measured by sensitivity or

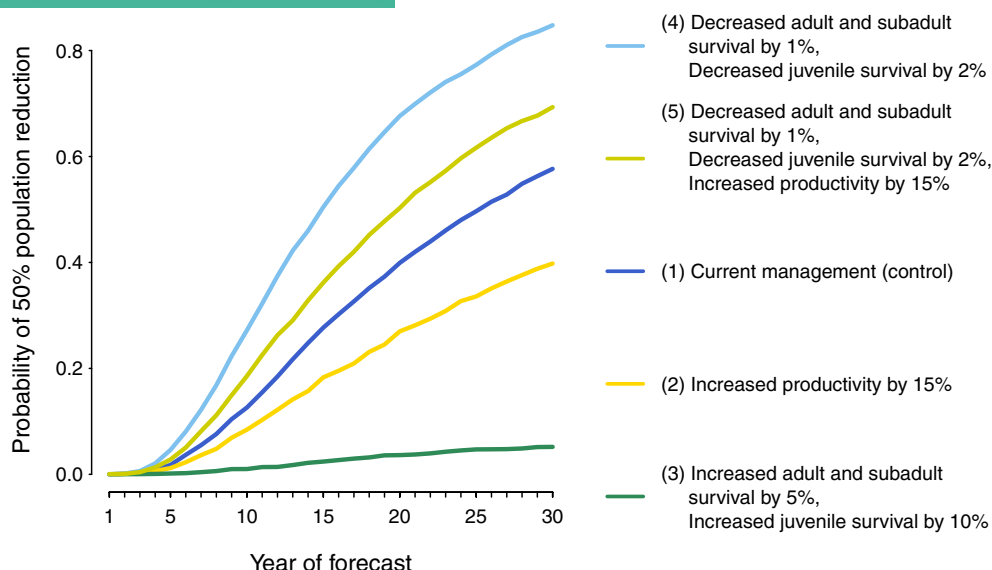


FIGURE 5 Comparison of the cumulative probabilities of a 50% population reduction of the tōrea population 30 years into the future under five different intervention and threat scenarios as described in Table 1.

elasticity (Le Coeur et al., 2022). Our results suggest that similar to other oystercatcher species (Van de Pol et al., 2010), temporal variation in adult survival was less than in other demographic rates (i.e. juvenile survival, subadult survival or productivity), yet population growth rates were more closely associated with subadult survival likely because of its greater variability over time. Hence, while in practice, an increase in adult survival may be difficult to achieve because of demographic buffering (Le Coeur et al., 2022), impacts that result in additional mortality are likely to have a strong impact on the population trajectory.

Our study suggests that emerging threats may accelerate the decline of tōrea. Climate change-induced habitat loss and degradation has already resulted in decreased adult survival in oystercatcher species elsewhere (Griffin et al., 2023). Furthermore, shorebirds are highly vulnerable to collision mortality at windfarms (Powlesland, 2009; Thaxter et al., 2017) which are expected to proliferate with a transition to more renewable energy generation in the future. While the present-day approach to mitigating the effects of windfarms on shorebirds in New Zealand is to compensate for such additional mortality through increased productivity (Bennet et al., 2022; Craig et al., 2015), we show here that even best-case outcomes of predator management at breeding sites will not offset even small decreases in survival. This problem has been reported for Eurasian oystercatchers (*Haematopus ostralegus*) where population decline caused by ongoing reductions in adult survival in the 2000s has not reversed, despite increases in productivity in the 2010s (Allen et al., 2022). Hence, New Zealand will need a conservation strategy targeting multiple demographic rates across the annual cycle if the decline of the tōrea population is to be halted.

Our IPM allowed us to estimate seasonal survival for the tōrea population for the first time. We integrated seasonal count data, covering different segments of the population, using a novel approach

that linked these data using population growth rates rather than population size, as is usually done. This approach yielded annual estimates of survival and productivity of tōrea comparable with earlier analyses of individual data sets of tōrea (Sagar et al., 2000, 2002), and estimates of annual survival similar to those for other oystercatcher species worldwide (Juvenile survival: 0.50–0.80; subadult survival 0.80–0.92 and adult survival: 0.87–0.96; Ens & Underhill, 2014; Allen et al., 2022). At the seasonal level, we found that median survival is lower in winter (including the spring migration for breeding adults), and, in lower age classes, winter survival is also much more variable. In other migratory species, seasonal survival has also been found to be lower in periods that include spring migrations (e.g. Allen et al., 2019). Although our IPM suggests that subadult winter survival is the strongest driver of population dynamics, it is possible that other demographic rates could influence population dynamics, especially those rates fitted with high uncertainty (i.e. juvenile survival or proportion of maturing subadults). Two improvements in future data collection could decrease the uncertainty around seasonal demographic rates and thereby improve our assessment of underlying drivers of population dynamics. First, increased effort to band chicks on breeding grounds would enable monitoring of known-age individuals and thus more precise estimates of survival in younger stage classes. Second, if resighting surveys were carried out four times a year, targeting the start and end periods of both breeding and non-breeding seasons, it would be possible to decompose survival estimates into seasonal stationary and movement periods.

Our approach can inform and improve conservation management in three ways. First, the IPM provides a new platform to reveal both the overall population trends and key demographic drivers of a species by combining different monitoring data sets collected from breeding and non-breeding seasons. Uncertainty in each data-set, capturing different segments of the population, can be fully

accounted for in this full-annual-cycle approach. Second, combining our IPM with a PVA can inform the conservation and monitoring strategies for a species. For example, our IPM-PVA revealed the need to expand the focus of tōrea management beyond breeding sites to address threats at non-breeding sites and during migration. In addition, our model highlighted the need for targeted monitoring to obtain survival estimates of younger stage classes and by season to decrease uncertainty about demographic rates, particularly those of younger age classes. Third, high-resolution animal-borne GPS tags have the potential to give precise data on the timing of losses, dispersal and migratory connectivity. Our approach has the potential to better investigate the interaction between movements, threats, and conservation actions for migratory species allowing spatially targeted actions as such data are easily incorporated into an IPM.

AUTHOR CONTRIBUTIONS

Ann-Kathrin V. Schlesselmann, Adrian Monks, Susan Walker, Emma Williams and Colin F. J. O'Donnell conceived the ideas; Ann-Kathrin V. Schlesselmann, Adrian Monks, Susan Walker and Michael Schaub refined the ideas and methodology; Ann-Kathrin V. Schlesselmann, Paul Sagar, David S. Melville, Rob Schuckard and Sam Krause collected and/or managed data; Ann-Kathrin V. Schlesselmann led the analysis and writing of the manuscript. All authors gave critical input on the drafts and final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

Data and code are provided via Datastore at: <https://datastore.landcareresearch.co.nz/dataset/full-annual-cycle-ipm-pva-for-torea> (Schlesselmann et al., 2025).

REFERENCES

Ahrestani, F., Saracco, J. F., Sauer, J. R., Pardieck, K. L., & Royle, J. A. (2017). An integrated population model for bird monitoring in North America. *Ecological Applications*, 27, 916–924.

Allen, A. M., Ens, B. J., Van de Pol, M., Van der Jeugd, H., Frauendorf, M., Oosterbeek, K., & Jongejans, E. (2019). Seasonal survival and

migratory connectivity of the Eurasian oystercatcher revealed by citizen science. *Auk*, 136, 1–17.

Allen, A. M., Jongejans, E., van de Pol, M., Ens, B. J., Frauendorf, M., van der Sluijs, M., & de Kroon, H. (2022). The demographic causes of population change vary across four decades in a long-lived shorebird. *Ecology*, 103, e3615.

Bennet, D., Smith, V., Payne, R., & Houston, S. (2022). *Operational management plan for the enhancement of avifauna habitat at three sites in the lower Hakatere/Ashburton River*. Contract report No. 6228. Prepared for Environment Canterbury, Christchurch.

Brown, D. J., Ribic, C. A., Donner, D. M., Nelson, M. D., Bocetti, C. I., Deloria-Sheffield, C. M., & Thompson, D. (2016). Using a full annual cycle model to evaluate long-term population viability of the conservation-reliant Kirtland's warbler after successful recovery. *Journal of Applied Ecology*, 54, 439–449.

Brumby, A., Marshall, J., Murray, T., O'Donnell, C. F. J., & Richards, R. (2025). *Trait-based climate change vulnerability assessments of terrestrial taxa in Aotearoa New Zealand*. Science for Conservation, Department of Conservation.

Cooke, S. J., Piczak, M. L., Singh, N. J., Akesson, S., Ford, A. T., Chowdhury, S., Mitchell, G. W., Norris, D. R., Hardesty-Moore, M., McCauley, D., Hammerschlag, N., Tucker, M. A., Horns, J. J., Reisinger, R. R., Kubelka, V., & Lennox, R. J. (2024). Animal migration in the Anthropocene: Threats and mitigation options. *Biological Reviews*, 99, 1242–1260.

Craig, J. L., Kessels, G., Langlands, P., & Daysh, S. (2015). Planning for net biodiversity gains: A case study of Hauāuru mā raki Wind Farm, New Zealand. In C. Hull, E. Bennett, E. Stark, I. Smales, J. Lau, & M. Venosta (Eds.), *Wind and wildlife* (pp. 69–91). Springer.

Davis, K. L., Saunders, S. P., Beilke, S., Ford, E. R., Fuller, J., Landgraf, A., & Zipkin, E. F. (2023). Breeding season management is unlikely to improve population viability of a data-deficient migratory species in decline. *Biological Conservation*, 283, 110104.

Dowding, J. E., & Moore, S. J. (2006). *Habitat networks of indigenous shorebirds in New Zealand*. Science for Conservation, Department of Conservation.

Ens, B. J., & Underhill, L. G. (2014). Synthesis of oystercatcher conservation assessments: General lessons and recommendations. *International Water Studies*, 20, 5–22.

Erickson, R. A., Diffendorfer, J. E., Norris, D. R., Bieri, J. A., Earl, J. E., Federico, P., Fryxell, J. M., Long, K. R., Mattsson, B. J., Sample, C., Wiederholt, R., Thogmartin, W. E., & Fuller, R. (2017). Defining and classifying migratory habitats as sources and sinks: The migratory pathway approach. *Journal of Applied Ecology*, 55, 108–117.

Faaborg, J., Holmes, R. T., Anders, A. D., Bildstein, K. L., Dugger, K. M., Gauthreaux, S. A., Jr., Heglund, P., Hobson, K. A., Jahn, A. E., Johnson, D. H., Latta, S. C., Levey, D. J., Marra, P. P., Merkord, C. L., Nol, E., Rothstein, S. I., Sherry, T. W., Sillett, T. S., Thompson, F. R., III, & Warnock, N. (2010). Conserving migratory land birds in the new world: Do we know enough? *Ecological Applications*, 20, 398–418.

Gelman, A., & Hill, J. M. (2006). *Data analysis using regression and multi-level/hierarchical models*. Cambridge University Press.

Griffin, C. P., Brush, J. M., & Schwarzer, A. C. (2023). Decline in annual survival of American oystercatchers wintering in Florida linked to extreme high tides. *Journal of Wildlife Management*, 87, e22399.

Hostetler, J. A., Sillett, T. S., & Marra, P. P. (2015). Full-annual-cycle population models for migratory birds. *Auk*, 132, 433–449.

IUCN. (2024). *Guidelines for using the IUCN Red List categories and criteria. Version 16*. Standards and Petitions Committee. <https://www.iucnredlist.org/documents/RedListGuidelines.pdf>

Kellner, K. (2021). *jagsUI: A wrapper around "rjags" to streamline "JAGS" analyses. R package version 1.5.2*. <https://cran.r-project.org/package=jagsUI>

Le Coeur, C., Yoccoz, N. G., Salguero-Gómez, R., & Vindenes, Y. (2022). Life history adaptations to fluctuating environments: Combined effects of demographic buffering and lability. *Ecology Letters*, 25, 2107–2119.

- Lebreton, J. D., Burnham, K. P., Clobert, J., & Anderson, D. R. (1992). Modeling survival and testing biological hypotheses using marked animals. *Ecological Monographs*, 62, 67–118.
- Marolla, F., Aarvak, T., Hamel, S., Ims, R. A., Kéry, M., Mellard, J. P., Nater, C. R., Schaub, M., Vougioukalou, M., Yoccoz, N. G., & Øien, I. J. (2023). Life-cycle analysis of an endangered migratory goose to assess the impact of conservation actions on population recovery. *Biological Conservation*, 281, 110028.
- Martin, T. G., Chadès, I., Arcese, P., Marra, P. P., Possingham, H. P., & Norris, D. R. (2007). Optimal conservation of migratory species. *PLoS One*, 2, e751.
- MBIE. (2024). Electricity demand and generation scenarios: Results summary. <https://www.mbie.govt.nz/assets/electricity-demand-and-generation-scenarios-report-2024.pdf>
- Norbury, G. L., Price, C. J., Latham, M. C., Brown, S. J., Latham, A. D. M., Brownstein, G. E., Ricardo, H. C., McArthur, N., & Banks, P. B. (2021). Misinformation tactics protect rare birds from problem predators. *Science Advances*, 7, eabe4164.
- O'Donnell, C. F. J., & Moore, S. J. (1983). *The wildlife and conservation of braided river systems in Canterbury. Fauna survey unit report*. New Zealand Wildlife Service.
- O'Donnell, C. F. J., Sanders, M., Woolmore, C., & Maloney, R. F. (2016). *Management and research priorities for conserving biodiversity on New Zealand's braided rivers*. Department of Conservation.
- Plummer, M. (2003). JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. In *Proceedings of the 3rd international workshop on distributed statistical computing* (pp. 1–10).
- Powlesland, R. G. (2009). *Bird species of concern at wind farms in New Zealand. DOC research & development series 317*. Department of Conservation.
- R Core Team. (2024). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing.
- Riegen, A. C., & Sagar, P. M. (2020). Distribution and numbers of waders in New Zealand, 2005–2019. *Notornis*, 67, 591–634.
- Rushing, C. S., Hostetler, J. A., Sillett, T. S., Marra, P. P., Rotenberg, J. A., & Ryder, T. B. (2017). Spatial and temporal drivers of avian population dynamics across the annual cycle. *Ecology*, 98, 2837–2850.
- Sæther, B.-E., & Bakke, Ø. (2000). Avian life history variation and contribution of demographic traits to the population growth rate. *Ecology*, 81, 642–653.
- Sagar, P. M., Barker, R. J., & Geddes, D. (2002). Survival of breeding Finsch's oystercatchers (*Haematopus finschi*) on farmland in Canterbury, New Zealand. *Notornis*, 49, 233–240.
- Sagar, P. M., & Geddes, D. (1999). Dispersal of South Island pied oystercatchers (*Haematopus ostralegus finschi*) from an inland breeding area of New Zealand. *Notornis*, 46, 89–99.
- Sagar, P. M., Geddes, D., Banks, J., & Howden, P. (2000). Breeding of South Island pied oystercatchers (*Haematopus ostralegus finschi*) on farm land in mid-Canterbury, New Zealand. *Notornis*, 47, 71–81.
- Sagar, P. M., & Veitch, D. (2014). Conservation assessment of the South Island oystercatcher *Haematopus finschi*. *International Wader Studies*, 20, 155–160.
- Santos, C. D., Catry, T., Dias, M. P., & Granadeiro, J. P. (2023). Global changes in coastal wetlands of importance for non-breeding shorebirds. *Science of the Total Environment*, 858, 159707.
- Saunders, S. P., Cuthbert, F. J., Zipkin, E. F., & Rhodes, J. (2018). Evaluating population viability and efficacy of conservation management using integrated population models. *Journal of Applied Ecology*, 55, 1380–1392.
- Saunders, S. P., DeLuca, W. V., Bateman, B. L., Deppe, J. L., Grand, J., Knight, E. J., Meehan, T. D., Michel, N. L., Seavy, N. E., Smith, M. A., Taylor, L., Witko, C. J., & Wilsey, C. B. (2025). Multispecies migratory connectivity indicates hemispheric-scale risk to bird populations from global change. *Nature Ecology & Evolution*, 9, 491–504.
- Saunders, S. P., Farr, M. T., Wright, A. D., Bahlai, C. A., Ribeiro, J. W., Jr., Rossman, S., Sussman, A. L., Arnold, T. W., & Zipkin, E. F. (2019). Disentangling data discrepancies with integrated population models. *Ecology*, 100, e02714.
- Schaub, M., & Kéry, M. (2022). *Integrated population models: Theory and ecological applications with R and JAGS*. Academic Press.
- Schlesselmann, A.-K. V., Monks, A., Innes, J., McArthur, N., & Walker, S. (2024). Conservation challenges in mobile birds: What do we know and need to know for effective conservation of endemic inland migrants? *New Zealand Journal of Ecology*, 48, 3575.
- Schlesselmann, A.-K. V., Monks, A., Walker, S., Sagar, P., Melville, D. S., Schuckard, R., Williams, E., Krouse, S., O'Donnell, C. F. J., & Schaub, M. (2025). Data from: A range-wide full-annual-cycle model informs conservation of a declining migratory shorebird. *Datastore*. <https://datastore.landcareresearch.co.nz/dataset/full-annual-cycle-ipm-pva-for-torea>
- Schmechel, F. (2001). *Potential impacts of mechanical cockle harvesting on shorebirds in Golden and Tasman Bays, New Zealand. DOC science internal series*. Department of Conservation.
- Thaxter, C. B., Buchanan, G. M., Carr, J., Butchart, S. H. M., Newbold, T., Green, R. E., Tobias, J. A., Foden, W. B., O'Brien, S., & Pearce-Higgins, J. W. (2017). Bird and bat species' global vulnerability to collision mortality at wind farms revealed through a trait-based assessment. *Proceedings of the Royal Society B: Biological Sciences*, 284, 2017082.
- Van de Pol, M., Vindenes, Y., Sæther, B.-E., Engen, S., Ens, B. J., Oosterbeek, K., & Tinbergen, J. M. (2010). Effects of climate change and variability on population dynamics in a long-lived shorebird. *Ecology*, 91, 1192–1204.
- Walker, S., Monks, A., & Innes, J. (2020). National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004. *Notornis*, 67, 677–691.
- Zipkin, E. F., & Saunders, S. P. (2018). Synthesizing multiple data types for biological conservation using integrated population models. *Biological Conservation*, 217, 240–250.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Supporting Information S1. Additional methods.

Supporting Information S2. Additional results.

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