**BEFORE AN EXPERT PANEL** **FTAA-2508-1100**
**THE POINT SOLAR FARM**

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|---|---|
| Under the | FAST-TRACK APPROVALS ACT 2024 |
| In the matter of | an application for approvals to construct and operate an approximately 670-hectare solar farm to supply electricity to the national grid |
| By | **FAR NORTH SOLAR FARM LIMITED**<br>Applicant |

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**STATEMENT OF EVIDENCE OF SUSAN WALKER**

**TERRESTRIAL ECOLOGY**

19 February 2026

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**Environmental Defence Society Inc**
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Shay Schlaepfer, Chief Operating Officer and In-house Counsel
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Counsel Acting: Rob Enright
[redacted]

## Introduction

1. My full name is Susan Walker.

2. I have been asked by the Environmental Defence Society to provide expert evidence on the actual and potential terrestrial ecology effects of Far North Solar Farm's proposed 670-hectare solar farm at the northern shore of Lake Benmore between the Pukaki River and the Ohau C hydro canal.

3. I am an ecologist and Principal Researcher in the Manaaki Whenua Group within the crown owned Bioeconomy Science Institute (formed July 2025). I am based in Dunedin, where I worked for the predecessor organisation Manaaki Whenua – Landcare Research from 1997 to June 2025.

4. I have MSc (1994) and PhD (1997) degrees from the University of Otago. I have published more than 60 peer-reviewed scientific journal papers and book chapters in international and national literature and produced more than 50 internally peer-reviewed contract reports.

5. My fields of expertise relevant to this application include:

    a. The botany, ecology, and conservation management of modified indigenous ecosystems of the dry eastern rain-shadow zone of the South Island ('dry inland South Island') including the Mackenzie Basin;

    b. Biodiversity assessment, including measurement and reporting of the biodiversity and conservation outcomes and achievements of policies (and incentives);

    c. Quantitative field sampling and measurement of biodiversity components and assessment of ecological significance;

    d. National and regional long-term changes in New Zealand's land cover and indigenous bird fauna; and

    e. Movements of inland-migrant birds.

6. I have also researched, published scientific papers and written reports about the dynamics of indigenous birds and introduced rodents across New Zealand forests, the ecology and conservation of threatened plants, evolutionary patterns of plant richness, radiation and endemism and effects of climate change on New Zealand's indigenous biodiversity among other subjects.

7. I am often engaged to provide ecological advice in reports or oral presentations to central and local government agencies on matters of ecology and biodiversity assessment and protection. In 2018, I received the New Zealand Ecological Society's Ecology in Action award which recognises contribution to the application of ecological knowledge, including communication, education and transfer of ecological science.

8. I have particular expertise and field experience in the ecology of dryland ecosystems on the floor of the Upper Waitaki basin (in Waitaki and Mackenzie districts) and the basin floors of Central Otago and Queenstown Lakes districts. Since commencing my postgraduate studies in 1993 I have undertaken research and field investigations in many parts of dry inland South Island and led government-funded research programmes into the biodiversity of New Zealand 'drylands'. My recent and ongoing research includes investigation of cross-boundary effects of intensive land use on remaining areas of indigenous dryland vegetation (Canterbury Plains and Mackenzie basin) and causes of decline in inland *Lepidium* species, which are three threatened (Nationally Critical) plant species endemic to dry inland South Island basins.

9. Since 2009 I have provided expert evidence for several RMA hearing panels, the Environment Court and the High Court concerning ecology, ecological changes and development proposals in the Mackenzie Basin. I documented the large-scale clearance enabled by the 'improved pasture exemption' in the Mackenzie District Plan and provided ecological evidence which contributed to suspension of that exemption in 2016. I provided expert ecological evidence before the Environment Court for Plan Change 13 (2016) and Plan Change 18 (2019 to 2025).

## Code of Conduct

10. I have read the Environment Court code of conduct for expert witnesses, and I have prepared this evidence in accordance with that code. I confirm that my evidence is within my area of expertise, except where I state I am relying on the evidence of another person. I have acknowledged the material and expertise relied on in the preparation of this evidence and in forming my opinions. To my knowledge I have not omitted to consider any material facts known to me that alter or detract from the opinions I express in this evidence.

## The Project site

11. In this evidence I refer to The Point Solar Farm application site as 'the Project site' (Attachment 1). I refer to the area proposed to be largely covered by solar panels on the flat surface of the outwash terrace as the 'Project site footprint' and to the surrounding areas of terrace surface within the Project site boundary as the 'Project site perimeter'.

12. I also refer to 'outwash terrace edges' which lie outside but adjacent to the Project site. An outwash terrace edge is the gently sloping zone between the flat outwash terrace surface and the steeply sloping outwash terrace 'risers' which descend to the lower alluvial surfaces cut by the Pukaki and Tekapo rivers to the east and the Twizel and Ohau rivers to the west of the Project site.

13. I am familiar with the Project site footprint and its perimeter, and with the adjacent terrace edges. In November 2016 I undertook a survey around the perimeter of the site, and in January 2017 I also briefly inspected the area that is now the Project site footprint on the terrace surface[1]. On both visits I recorded flora and bird fauna. I have not visited the Project site since.

[1] My surveys were in preparation for expert ecological evidence on the effects of an application for irrigation at the Project site which was subsequently withdrawn. The terrace surface had been sown in a ryecorn crop in the year prior to my visit, although I recorded that some indigenous species remained, notably the At Risk – Declining species *Raoulia parkii*, *Carex breviculmis* and *Muehlenbeckia axillaris*. I recorded several banded dotterels and South Island pied oystercatcher (both At Risk – Declining), and black fronted terns (Threatened - Nationally Endangered) feeding on or over the terrace surface in both November and January.

14. I have collected data on the GPS locations of known populations of the Threatened – Nationally Critical plant *Lepidium solandri* across the upper Waitaki basin, including the terrace edges adjacent to the Project site.

## Scope

15. This evidence covers three matters relating to:

    a. Effects of potential bird collisions on bird populations;

    b. Inadequate ecological information for the Project site; and

    c. Potential effects on significant indigenous vegetation on adjacent terrace edges.

## Effects of potential bird collisions on bird populations

16. The Project site is in a place that puts at least eighteen Threatened or At Risk bird species at risk of additional mortality through collision with the panels or associated infrastructure. As Dr O'Donnell writes in his December 2025 report[2] "*The proposed solar farm is at one of the busiest known bird flyways in the Mackenzie basin; being located adjacent to numerous breeding sites on the Twizel, Pukaki, Tekapo and Ōhau Rivers, and comprising a complex mosaic of wetlands and braided river breeding habitats, with many breeding sites located within 100m of the proposed solar farm site.*"

17. The applicant's assessment shows that the banded dotterel, black-fronted tern, South Island pied oystercatcher, and New Zealand pipit (all Threatened or At-Risk species) use the site for feeding or breeding directly[3]. The area is therefore ecologically significant under the Canterbury Regional Policy Statement (CRPS) significance criterion 4 and also an Significant Natural Area (SNA) in Mackenzie

[2] Proposed solar farm – **The Point, Twizel.** Risks to Threatened and At-Risk bird species from construction and ongoing operation. Dr Colin O'Donnell ONZM, Principal Science Advisor, Department of Conservation, Private Bag 4715, Christchurch 8140. Dated December 2025.

[3] As also noted by Dr O'Donnell in his comment.

District[4]. The habitat for these bird species will be removed by development of the solar farm.

18. The eighteen bird species at risk of collision include six inland breeding wading birds, terns and gulls (Charadriiformes) for which the Mackenzie Basin is a national stronghold[5] (Table 1).

**Table 1. Threatened or At Risk inland breeding wading birds, terns and gulls (Charadriiformes) for which the Mackenzie Basin is a national stronghold.**

| Species name | Common name | Status 2021 |
|---|---|---|
| *Anarhynchus frontalis* | Wrybill/ngutu-pare | Threatened - Nationally Increasing |
| *Charadrius bicinctus bicinctus* | Banded dotterel/tūturiwhatu | At Risk - Declining |
| *Haematopus finschi* | South Island pied oystercatcher/tōrea | At Risk - Declining |
| *Larus bulleri* | Black-billed gull/tarapuka | At Risk - Declining |
| *Himantopus novaezelandiae* | Black stilt/kakī | Threatened - Nationally Critical |
| *Chlidonias albostriatus* | Black-fronted tern, tarapirohe | Threatened - Nationally Endangered |

19. All of the six Threatened or At Risk species in Table 1 are endemic. Wrybill (*Anarhynchus frontalis*, endemic at the genus level) and the five species-level endemics (black stilt/ kakī *Himantopus novaezelandiae*, black-billed gull *Larus bulleri*, black-fronted tern *Chlidonias albostriatus*, banded dotterel *Charadrius bicinctus*, and South Island pied oystercatcher *Haematopus finschi*) breed mainly in the South Island, on sparsely vegetated inland braided riverbeds and outwash terraces which formed in the Pleistocene. Some populations of some inland breeding species remain and overwinter inland, but others migrate to feed in coastal habitats around New Zealand and, in the case of some banded dotterel, on the coast of southeastern Australia.

[4] Criterion 4 of Appendix 3 to Policy 9.3.1 in the CRPS is "Indigenous vegetation or habitat of indigenous fauna that supports an indigenous species that is threatened, at risk, or uncommon, nationally or within the relevant ecological district". In the Mackenzie District Plan Significant indigenous vegetation and significant habitats of indigenous fauna means areas of indigenous vegetation or habitats of indigenous fauna which meet the criteria listed in the Canterbury Regional Policy Statement's Policy 9.3.1 and Appendix 3; or are listed in Appendix I as a Site of Natural Significance.
[5] Walker S, Monks A, Innes J 2020. National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004. Notornis 67(4): 677-691.

20. I am coauthor of a recently published research paper on one of the seven species (South Island pied oystercatcher, or tōrea) (Attachment 2)[6]. 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.

21. Our study established that the tōrea population declined by >1% annually over the 42-year study period. This result is consistent with our earlier published work[7] showing a reduction in inland breeding range occupancy based on national surveys comparing 1969–1979 with 1999–2004, and with another published study showing declines in abundance during the non-breeding season between 2005 and 2019 (1.2% per year)[8].

22. The study shows that the long-term decline of tōrea is driven principally by poor survival of subadult and mature birds and not by low productivity (i.e. the number of chicks produced and fledged). Furthermore, even the most optimistic scenario of increased productivity through predator management on breeding grounds was not predicted to improve the current population trajectory unless the survival of subadults and adults remained at current levels. Predator management on breeding grounds is therefore unable to offset the effect of additional loss of subadult and adults; additional losses of adults, such as through bird collisions at new renewable energy installations (whether wind or solar) will increase the rate of decline in the national tōrea population.

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[6] Schlesselmann AK, Monks A, Walker S, Sagar P, Melville DS, Schuckard R, Williams E, Krouse S, O'Donnell CF, Schaub M 2026. A range-wide full-annual-cycle model informs conservation of a declining migratory shorebird. Journal of Applied Ecology 63(1): e70228.
[7] Walker S, Monks A, Innes J 2020. National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004. Notornis 67(4): 677-691.
[8] Riegen AC, Sagar PM 2020. Distribution and numbers of waders in New Zealand, 2005–2019. Notornis, 67: 591–634.

23. Although our study did not extend to the other five species of inland breeding Charadriiformes, I would expect a similar result given their broadly similar life-histories.

24. The best measure available to reduce the risk of accelerated population declines in inland breeding Charadriiformes from bird collisions at solar farms would be to avoid siting solar farms within regions (such as the Mackenzie Basin) that are known strongholds for these species. If it is necessary to locate solar farms within known strongholds, then careful siting will be needed. That is, installations should be sited as distant as possible from known important breeding and feeding habitats, movement corridors, and/or flyways. This Project site is within "*one of the busiest known bird flyways in the Mackenzie basin*" and "*located adjacent to numerous breeding sites*"[9].

## Inadequate ecological information

25. The Project site is the lowest part of an extensive inland outwash gravel terrace landform ('the Pukaki outwash'[10]) extending from near Lake Pukaki to the Lake Benmore delta on the floor of the Mackenzie Basin. The site experiences a particularly extreme climate that is drier, warmer and frostier than most other places in the Mackenzie Basin. The overall aridity gradient across the Mackenzie basin floor is poorly represented in legally protected areas (i.e. public conservation land and covenants) which are mainly in the higher, wetter and milder northwest.

26. Inland outwash gravels are naturally uncommon ecosystems[11] ranked as Critically Endangered[12] in New Zealand. Indigenous vegetation on a naturally

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[9] Proposed solar farm – The Point, Twizel. Risks to Threatened and At-Risk bird species from construction and ongoing operation . Dr Colin O'Donnell ONZM, Principal Science Advisor, Department of Conservation, Private Bag 4715, Christchurch 8140. Dated December 2025.
[10] The Project site is mapped as a latest Late Otiran outwash surface by Barrell, D.J.A.; Andersen, B.G.; Denton, G.H.; Smith Lyttle, B. 2013: Glacial geomorphology of the central South Island, New Zealand. GNS Science Monograph 27a.
[11] Williams, P.A.; Wiser, S.; Clarkson, B.; Stanley, M.C. 2007. New Zealand's historically rare terrestrial ecosystems set in a physical and physiognomic framework. NZ Journal of Ecology 31: 119-128.
[12] Holdaway, R.J.; Wiser, S.K.; Williams, P.A. 2012. Status assessment of New Zealand's naturally uncommon ecosystems. Conservation Biology 26: 619-629.

uncommon ecosystem is ecologically significant under the Canterbury Regional Policy Statement (CRPS) significance criterion 6[13] and is therefore significant indigenous vegetation under the Mackenzie District Plan (MDP)[14].

27. In my experience arid locations (such as the Project site) in the Mackenzie Basin are typically highly modified but nevertheless usually meet the definition of indigenous vegetation in the MDP[15] and support more indigenous species than exotic species. Several characteristically drought-tolerant Threatened and At-Risk plant species are usually present, so that arid locations usually meet the CRPS significance criterion 4 and are significant indigenous vegetation[16].

28. The potential presence of Threatened and At-Risk indigenous plant species on or near the Project site is important in the context of recent trends in the Mackenzie Basin. Numbers of Threatened and At Risk indigenous vascular plant species on the Mackenzie Basin floor increased by about one third (from 83 to 109) since 2012, largely because their remaining habitats were widely converted and developed over the last two to three decades. The Mackenzie Basin remains the national stronghold for many of these species' populations.

[13] Appendix 3 to Policy 9.3.1 in the CRPS lists the criteria for determining significant indigenous vegetation and significant habitat of indigenous biodiversity. Criterion 6 is "Indigenous vegetation or an association of indigenous species that is distinctive, of restricted occurrence, occurs within an originally rare ecosystem, or has developed as a result of an unusual environmental factor or combinations of factors".
[14] In the Mackenzie District Plan Significant indigenous vegetation and significant habitats of indigenous fauna means areas of indigenous vegetation or habitats of indigenous fauna which meet the criteria listed in the Canterbury Regional Policy Statement's Policy 9.3.1 and Appendix 3; or are listed in Appendix I as a Site of Natural Significance. An appeal on this section of the Plan has recently been withdrawn, although the MDC website has not yet been updated to reflect this.
[15] In the definitions, indigenous vegetation "means a community of vascular plants, mosses and/or lichens that includes species native to the ecological district and many include exotic species."
[16] Criterion 4 of Appendix 3 to Policy 9.3.1 in the CRPS is "Indigenous vegetation or habitat of indigenous fauna that supports an indigenous species that is threatened, at risk, or uncommon, nationally or within the relevant ecological district".

**Table 2. Numbers of Threatened, At Risk and Data Deficient indigenous vascular plant species occurring on the Mackenzie basin floor in 2012, 2017 and 2023 (listed at three consecutive revisions of the New Zealand Threat Classification System or NZTCS).**

| NZTCS conservation status | 2012 | 2017 | 2023 |
|---|---|---|---|
| Extinct | 1 | | |
| Threatened – Nationally Critical | 8 | 10 | 10 |
| Threatened – Nationally Endangered | 10 | 10 | 12 |
| Threatened – Nationally Vulnerable | 11 | 14 | 13 |
| At Risk – Declining | 26 | 37 | 59 |
| At Risk – Naturally Uncommon | 26 | 12 | 12 |
| Data Deficient | 1 | 4 | 3 |
| Total | 83 | 87 | 109 |

29. The present flora and fauna values of the Project site are still largely unknown. The Project site was surveyed by Wildland Consultants on 12th December 2022 (vegetation), 13th December 2022 (lizards), 14th December 2022 (avifauna) and 2nd February 2023 (invertebrates). A further 7-hour visit was undertaken on 20 January 2026[17]. Those vegetation and invertebrate surveys were insufficient to locate and document the cryptic flora and invertebrate fauna that is potentially present in these environments[18]. I therefore have little confidence that indigenous biodiversity has been adequately described.

30. Aerial images and recent site photographs show that the eastern portion of the Project footprint and perimeter[19] has been less developed for cropping than the western portion over the last decade. The low-stature vegetation is likely to continue to provide important habitat for indigenous fauna (especially birds, but also potentially invertebrates). The eastern part of the Project site may also

[17] Wildland Consultants memo "Vegetation and habitat survey of The Point Solar Farm" dated January 2026. It states that "General vegetation patterns were observed by a drive-through survey, combined with more detailed walkthrough inspections at many points across the site. A total of seven hours was spent on site".

[18] For example, I was part of the team in a recent survey for small and cryptic dryland indigenous flora in Central Otago, which required 28 person hours per 100 hectares. An example of more adequate invertebrate survey is provided by a recent supplementary investigation at the site of the proposed Haldon Solar farm by SLR Consulting New Zealand Limited. Field surveys in December 2025 included pitfall trapping, grasshopper transects, sweep netting, and visual encounter surveys; additional survey work targeting taxa active later in summer was also planned for late January 2026.

[19] This vegetation is mapped as "Brome-hawkweed-sheep's sorrel-haresfoot trefoil grassland/herbfield" in the Wildland Consultants memo "Vegetation and habitat survey of The Point Solar Farm" dated January 2026.

continue to support indigenous plant species. In my experience, indigenous vascular and non-vascular plants may persist and recolonise after disturbance on outwash gravels where extreme environmental conditions exclude or debilitate exotic plant species that are potential competitors.

31. I understand that a further vegetation survey of the Project site has been undertaken recently, that results will be provided to parties, and that opportunity for comment on the results will be provided. Unfortunately, the summer timing of the recent additional survey is unsuitable for finding some of the indigenous plant species that may be present. In my experience several species characteristic of outwash surfaces have died back or become very difficult to detect by mid-January in most years.

32. In summary (and subject to having the opportunity to review any further ecological survey information) I consider that Threatened and At-Risk plant and/or invertebrate species and their habitats may be present (but not as yet identified) at the Project site and would be adversely affected by the development.

## Potential effects on significant indigenous vegetation on adjacent outwash terrace edges

33. The edges[20] of the broad outwash terraces in the Mackenzie Basin are botanically distinctive zones that typically support a characteristic suite of Threatened and At-Risk plant species. Exceptional aridity is a distinctive environmental feature of terrace edges: there is usually a high proportion of bare ground and rock, and relatively few exotic species can grow there.

34. I am familiar with the edges and the risers of the Pukaki outwash terrace immediately adjacent to the proposed Project site perimeter, which I surveyed in 2016. These habitats are significant indigenous vegetation under the Mackenzie

[20] Outwash terrace ‘edges’ are the gently sloping zone between the flat outwash terrace surface and the steeply sloping outwash terrace risers (which descend to lower alluvial surfaces formed by the rivers) (see.

District Plan because they are indigenous vegetation on a naturally uncommon ecosystem and support several Threatened and At-Risk plant species[21].

35. My data record several locations of the Threatened – Nationally Critical vascular plant species *Lepidium solandri* in the terrace edge zone adjacent to the Project site in 2016 (Attachment 3). At that time, the vegetation of the terrace edges and risers adjacent to the Project site had also recently been surveyed and described as potential sites of natural significance (SONS) for the Mackenzie District Council by Mr Michael Harding. Together, our data record the presence of the following 11 vascular plant species listed as Threatened or At Risk adjacent to (within 10 to 100 metres of) the Project site:

    - *Anthosachne falcis* (At Risk – Declining)
    - *Carex resectans* (At Risk – Declining)
    - *Carmichaelia petriei* (At Risk – Declining)
    - *Carmichaelia vexillata* (At Risk – Declining)
    - *Convolvulus verecundus* f *verecundus* (At Risk – Declining)
    - *Lepidium solandri* (Threatened - Nationally Critical)
    - *Luzula ulophylla* (At Risk – Declining)
    - *Muehlenbeckia ephedroides* (At Risk – Declining)
    - *Poa maniototo* (At Risk – Declining)
    - *Raoulia australis* (At Risk – Declining)
    - *Raoulia parkii* (At Risk – Declining)

36. The lichen *Xanthoparmelia semiviridis* (At Risk - Declining) was also common.

[21] My data from November 2016 show that the edges also provide feeding and breeding habitat for Threatened or At Risk bird species (banded dotterel, South Island pied oystercatcher, and black fronted tern). I expect that they are also habitats of Threatened or At Risk lizard and invertebrate species, but I am not aware of survey information for those fauna groups.

37. Ecological restoration plantings and landscape screening plantings on the perimeter of the Project site are proposed as mitigation or compensation for adverse effects of the Project.

38. I understand that a final ecological compensation proposal is not yet available for comment. However, I also understand that about 14 hectares of 'ecological restoration' is proposed within a predator proof reserve on the eastern perimeter of the Project site, and that ecological restoration here may include restoration plantings.

39. There is also a current proposal for 'landscape screening plantings' on at least part of the Project site perimeter. For example, an RFI response by RMM Landscape Architects dated 9 February 2026 shows a 1km x 40m wide strip of proposed plantings on the southern perimeter, and a 1.5km x 35m strip proposed on the eastern perimeter. The locations of these proposed screening plantings are indicated in my Attachment 1.

40. These plantings would be located either directly on, or in close proximity to[22], remaining significant indigenous vegetation both on and adjacent to the Project site perimeter. They are likely to have both direct and indirect adverse effects on any remaining significant ecological values.

41. Any planting of woody species in stony substrates such as those at the Project site requires significant ground disturbance. Screening plants will also likely require irrigation if they are to grow rapidly in this extremely arid environment. Disturbance and the addition of water in this environment will remove habitat for any indigenous species and result in proliferation of weedy non-indigenous plant species.

42. If plantings are successful, later there will also be adverse shading effects. The indigenous plant species on and around the Project site are generally light demanding and shade-intolerant. Higher humidity and shelter provided by

[22] That is, they will be mostly within 10 to 100 metres of those areas.

shading will encourage taller weeds and exotic grass growth, also transforming the habitat so that low growing, drought-adapted indigenous plants are outcompeted.

43. Important adverse effects would be likely to arise from any watering or irrigation of plantings on the terrace surface. The outwash gravel substrate is exceptionally porous and patterns and the directions of subterranean drainage poorly known. Excess water from watering or irrigation could potentially emerge onto the terrace edges and risers at substantial distances from where it is applied. There is also a risk of aerial overspray of irrigation water beyond planted belts in this characteristically windy environment. Resulting moisture seepage and/or overspray would modify the environment of the ecologically significant vegetation by removing or reducing the arid conditions that the indigenous plant species depend on.

44. Edge effects are changes in vegetation composition that are caused by environmental modification (such as increases in soil moisture or increased seed inputs) which extends beyond development footprints[23]. My research has shown that in the Mackenzie Basin both irrigation and exotic forestry can produce edge effects on adjacent vegetation; the most common effect is an increase in the cover of exotic grasses; and that effects can extend as much as hundreds of metres from the boundary of the development into adjacent indigenous vegetation[24]. The indigenous plants of the terrace edges around the Project site are very vulnerable to edge effects from the solar farm development and perimeter plantings because they are directly adjacent to them.

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[23] Walker S, Brownstein G, Monks A 2019. Avoiding cross-boundary effects of agricultural land use on indigenous dryland habitats in the Canterbury region: consenting guidelines and planning recommendations. Manaaki Whenua – Landcare Research Contract Report LC3636, prepared for Environment Canterbury.

[24] Walker, S 2020. Measured edge effects on indigenous grassland and shrubland vegetation on low-relief topography in Canterbury. Manaaki Whenua – Landcare Research Contract Report LC3866, prepared for Environment Canterbury.

45. Shading by solar panels on the terrace surface will modify the climate and create a more humid and shaded environment with zones of concentrated water runoff. There are three potential effects of this.

    a. First, there will likely be proliferation of exotic grasses. The grass sward may spread (through edge effects) to the terrace edges and risers where it may outcompete the low-growing and light demanding indigenous plant species and modify the habitats of indigenous fauna such as invertebrates and lizards.

    b. Second, there may be new moisture seepage effects on the terrace edges and risers due to the newly moist environment and concentrated runoff zones beneath and between the panels on the terrace surface. Like seepage from potential watering of plantings, extra moisture emerging distant from the panels themselves would remove the arid habitat conditions required by the indigenous plant species.

    c. Third, there may be a compounding of grass sward edge effects and moisture seepage effects together which could effectively remove the significant indigenous vegetation.

46. My recommendations for mitigating effects of the Project on the significant indigenous vegetation on the terrace edges and risers are as follows

    a. Any panels and plantings (whether for screening or ecological restoration) at the Project site should be set back at least 100m from terrace edges.

    b. To monitor any edge effects on the adjacent vegetation (and potentially enable adaptive management) baseline survey and ongoing monitoring are needed along transects running from the Project footprint, across the Project perimeter and adjacent terrace edges, and down the terrace risers.

c. Plants selected for any landscape screening should be only the most drought tolerant on the list[25] and they should be appropriately hardened (ideally over a full season at the Project site) before being planted. Any watering of screening plantings should be restricted and temporary. For example, there may be watering in by hand (or by truck)[26] immediately following planting and at most one to two follow up waterings within the following six-month period and not thereafter.

d. Ecological restoration[27] should be primarily passive, relying on natural in-situ regeneration of plants adapted to this extreme environment and hand weeding of any prominent exotic plant species that threaten to outcompete low-growing indigenous plants. Successful rehabilitation will be very slow and cannot be sped up in an extreme environment such as this. Site disturbance should be limited as far as possible.

e. If any ecological restoration planting is attempted (and I do not recommend this) there should be no irrigation beyond an initially watering in by hand. If plants are introduced to the site that are intolerant of the naturally extreme environmental conditions, then this is not ecological restoration.

**Susan Walker**
19 February 2026

**Attachments**

Attachment 1. Map of The Point Solar Farm (TPSF) Project site showing features referred to in this evidence.

Attachment 2. Schlesselmann AK, Monks A, Walker S, Sagar P, Melville DS, Schuckard R, Williams E, Krouse S, O'Donnell CF, Schaub M 2026. A range-wide full-annual-cycle

[25] In my opinion the list of potential planting species includes species that are inappropriate at this environmentally extreme location because they will either fail or require persistent watering. However, some species from moister environments than the Project site may survive in close proximity to the panel array where they are likely to benefit from the microclimate amelioration provided by the panels.
[26] The Wildlands Ecological Enhancement Plan (May 2025) mentions knapsack spraying or boom spraying from a truck for watering plantings.
[27] For example, within the proposed predator proof exclosure.

model informs conservation of a declining migratory shorebird. Journal of Applied Ecology 63(1): e70228.

Attachment 3. Locations of the Threatened – Nationally Critical vascular plant species *Lepidium solandri* in the terrace edge zone adjacent to the Project site in 2016.

**Attachment 1**

**Attachment 1. Map of The Point Solar Farm (TPSF) Project site, with the boundary shown in pink. The footprint is shown in blue and the perimeter in green. The map was derived from Appendix 10 of the Applicant response to request for information dated 9 February 2026.**

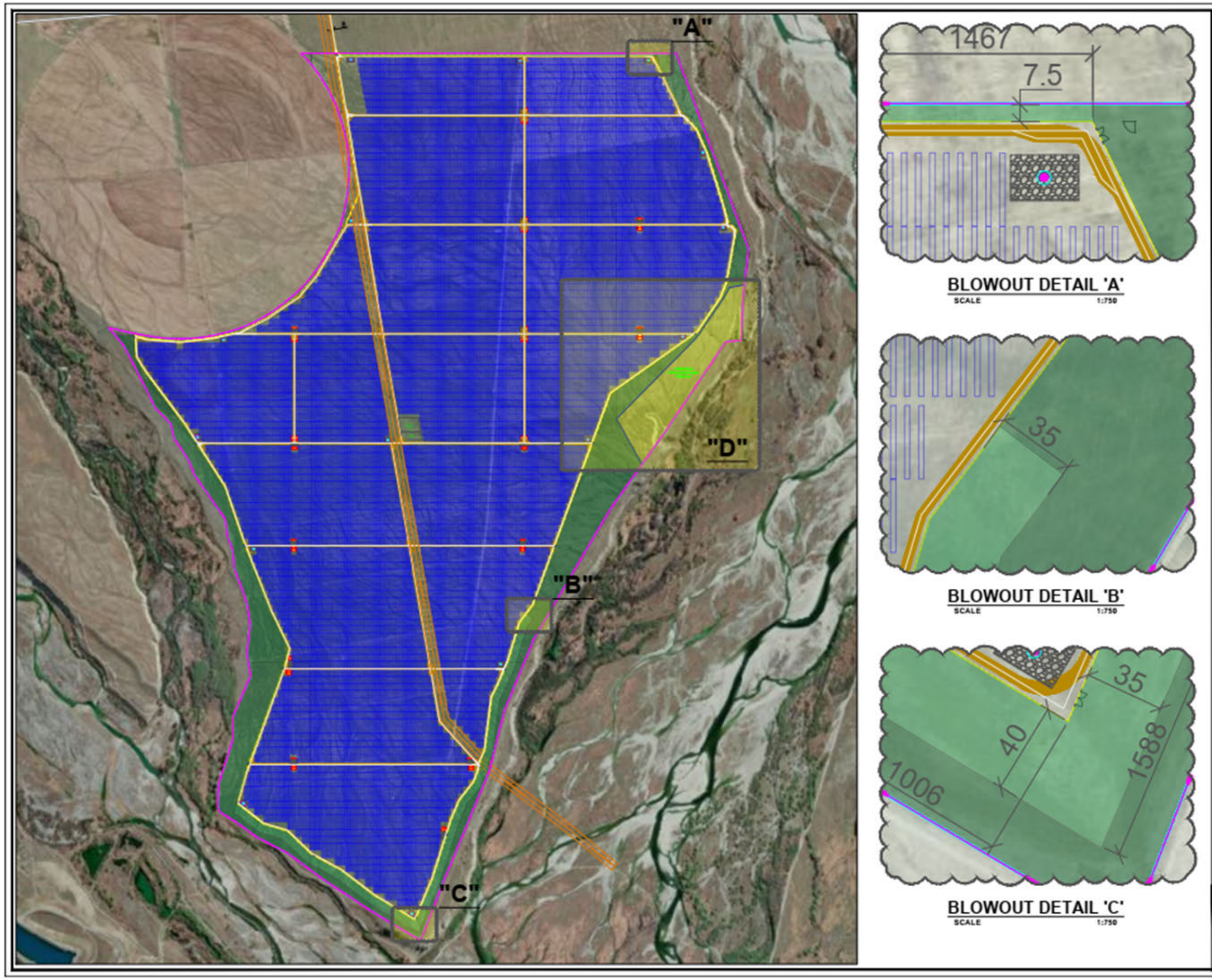
**KEY**

**Footprint**

**Perimeter**

Proposed perimeter screening

Other perimeter



**Attachment 2**

Received: 29 April 2025 | Accepted: 5 November 2025

DOI: 10.1111/1365-2664.70228

Journal of Applied Ecology

RESEARCH ARTICLE

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

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**Funding information**
Ministry of Business, Innovation and Employment (MBIE); Environment Canterbury Regional Braided River Initiative; New Zealand Department of Conservation Mobile Threatened Species Workstream

**Handling Editor:** Vitor Hugo Paiva

## 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

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 50 m 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

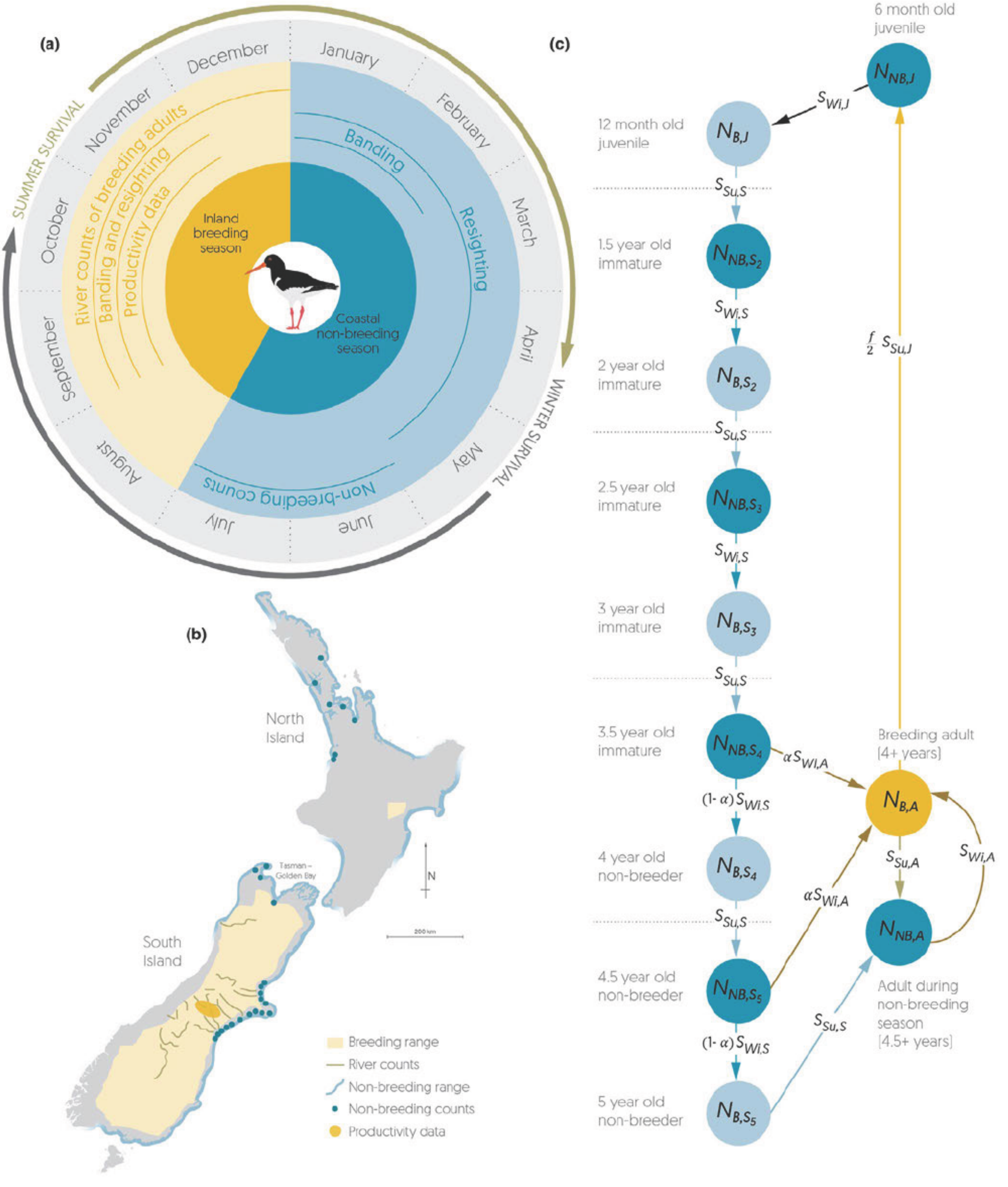


FIGURE 1 (a) The life cycle of tōrea and seasonally collected datasets used in the integrated population model. (b) Seasonal ranges of tōrea and coverage of data. Banding and resighting during the breeding season occurred only at the same sites where productivity data were collected; banding at non-breeding 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 ($\alpha$). 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:

$$N_{t,NB,J} = \frac{N_{t-1,B,A}}{2} \times f_t \times s_{t,Su,J} \tag{1}$$

$$N_{t,B,J} = N_{t,NB,J} \times s_{t,Wi,J} \tag{2}$$

Juvenile non-breeding season abundance ($N_{t,NB,J}$) is the product of the number of breeding pairs during the previous breeding season ($\frac{N_{t-1,B,A}}{2}$), 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:

$$N_{t,NB,S_i} = N_{t-1,B,S_{i-1}} \times s_{t,Su,S} \tag{3}$$

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 ($s_{t,Su,S}$).

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 ($s_{t,Wi,S}$):

$$N_{t,B,S_i} = N_{t,NB,S_i} \times s_{t,Wi,S} \tag{4}$$

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:

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

where $1 - \alpha_t$ is the proportion of subadults that remains as non-breeding subadults ($N_{t,B,S_i}$) and $s_{t,Wi,S}$ is subadult winter survival (Figure 1). The proportion of subadults maturing to breeders ($\alpha_t$) 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:

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

where $N_{t-1,B,A}$ 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 $s_{t,Su,S}$ are adult and subadult summer survival, respectively.

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

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

It includes adults ($N_{t,B,A}$) and 3.5- and 4.5-year-old subadults that recruited into the breeders $\left(\alpha_t \times (N_{t,NB,S_4} + N_{t,NB,S_5})\right)$ 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:

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

### 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

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

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

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

where $\bar{f}$ is mean productivity and $\sigma_f^2$ 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):

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

$$y_{i,j} \sim \text{Bernoulli}\left(z_{i,j} \times p_{j,c}\right) \tag{12}$$

where $z_{i,j}$ 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 $j$, $y_{i,j}$ 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

$$\text{logit}\left(s_{t,k,c}\right) \sim \text{normal}\left(\overline{s_{k,c}}, \sigma^2_{s_{k,c}}\right) \tag{13}$$

$$\text{logit}\left(p_{t,k,c}\right) \sim \text{normal}\left(\overline{p_{k,c}}, \sigma^2_{d_{k,c}}\right) \tag{14}$$

where $\overline{s_{k,c}}$ is mean seasonal ($k$) survival probability of stage $c$, $\sigma^2_{s_{k,c}}$ is the temporal variation of seasonal survival, $\overline{p_{k,c}}$ is the mean seasonal detection probability, and $\sigma^2_{d_{k,c}}$ 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 formation 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:

$$s_{t,An,c} = s_{t,Su,c} \times s_{t,Wi,c} \tag{15}$$

*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

$$C_{t,NB,m} \sim \text{normal}\left(I_{t,NB}, \sigma^2_{t,NB}\right) \tag{16}$$

where $\sigma^2_{t,NB}$ 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:

$$\frac{I_{t+1,NB}}{I_{t,NB}} = \frac{N_{t+1,NB}}{N_{t,NB}} = \lambda_{t,NB} \tag{17}$$

*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 ($n$):

$$C_{t,B,n} \sim \text{normal}\left(I_{t,B}, \sigma^2_{t,B}\right) \tag{18}$$

where $\sigma^2_{t,B}$ 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:

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

### 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 tōrea 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).

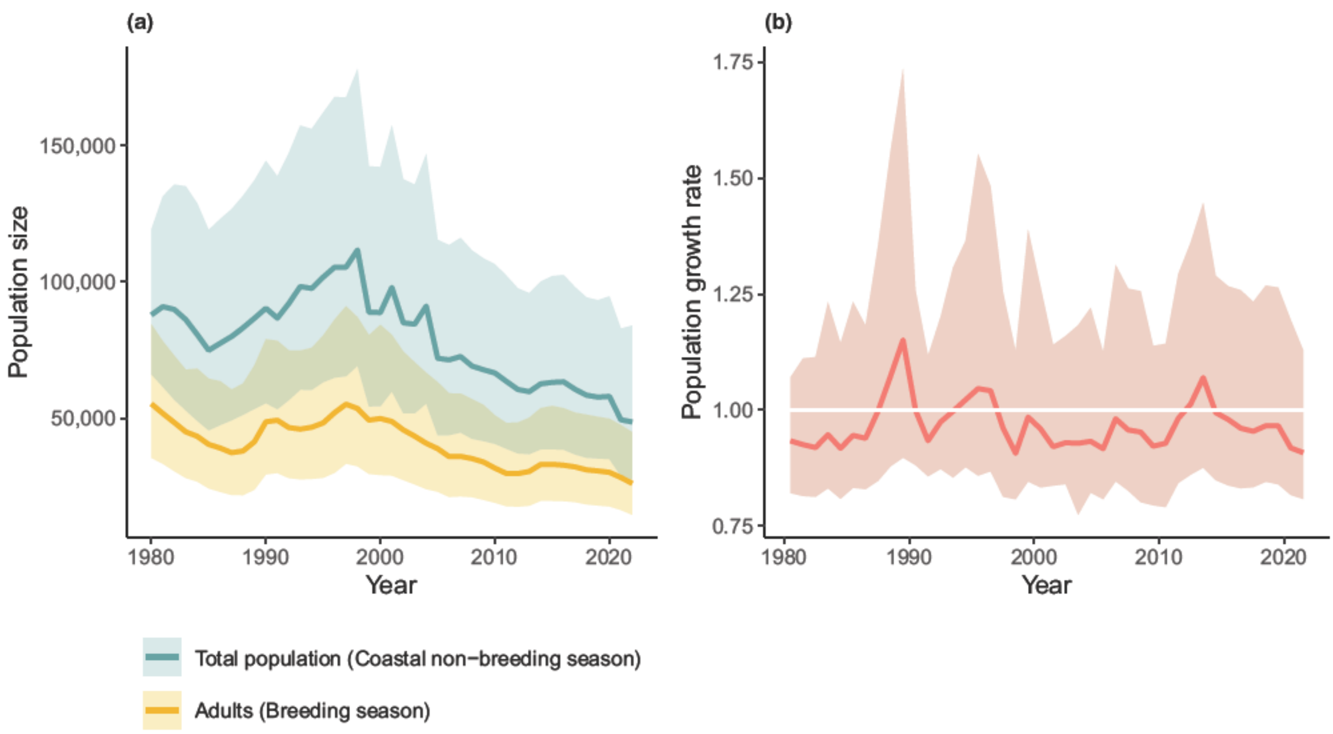


FIGURE 2 (a) Estimates of the total (combined adults, subadults and juveniles) during the non-breeding season and breeding adult population size; and (b) annual population growth rates of the breeding adult population of tōrea ($\lambda_{t,B,A}$) in New Zealand, 1980–2022. Shading represents 95% credible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

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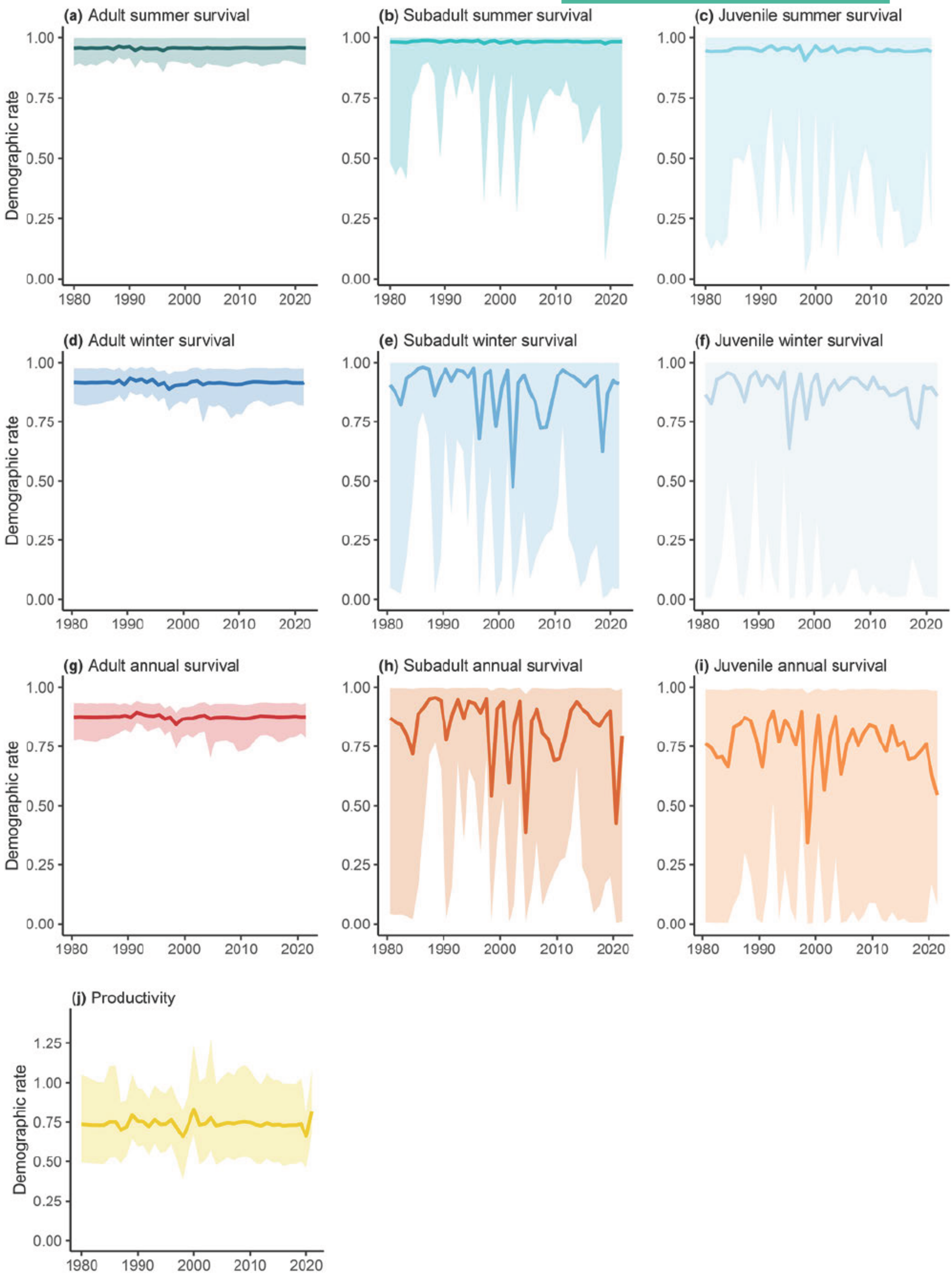

TABLE 2 Estimated medians and 95% Bayesian credible intervals of demographic parameters of tōrea.

| 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 | $\overline{f}$ | 0.74 | 0.64 | 0.83 |
| Proportion of first-time breeders | $\overline{\alpha}$ | 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 3j; 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

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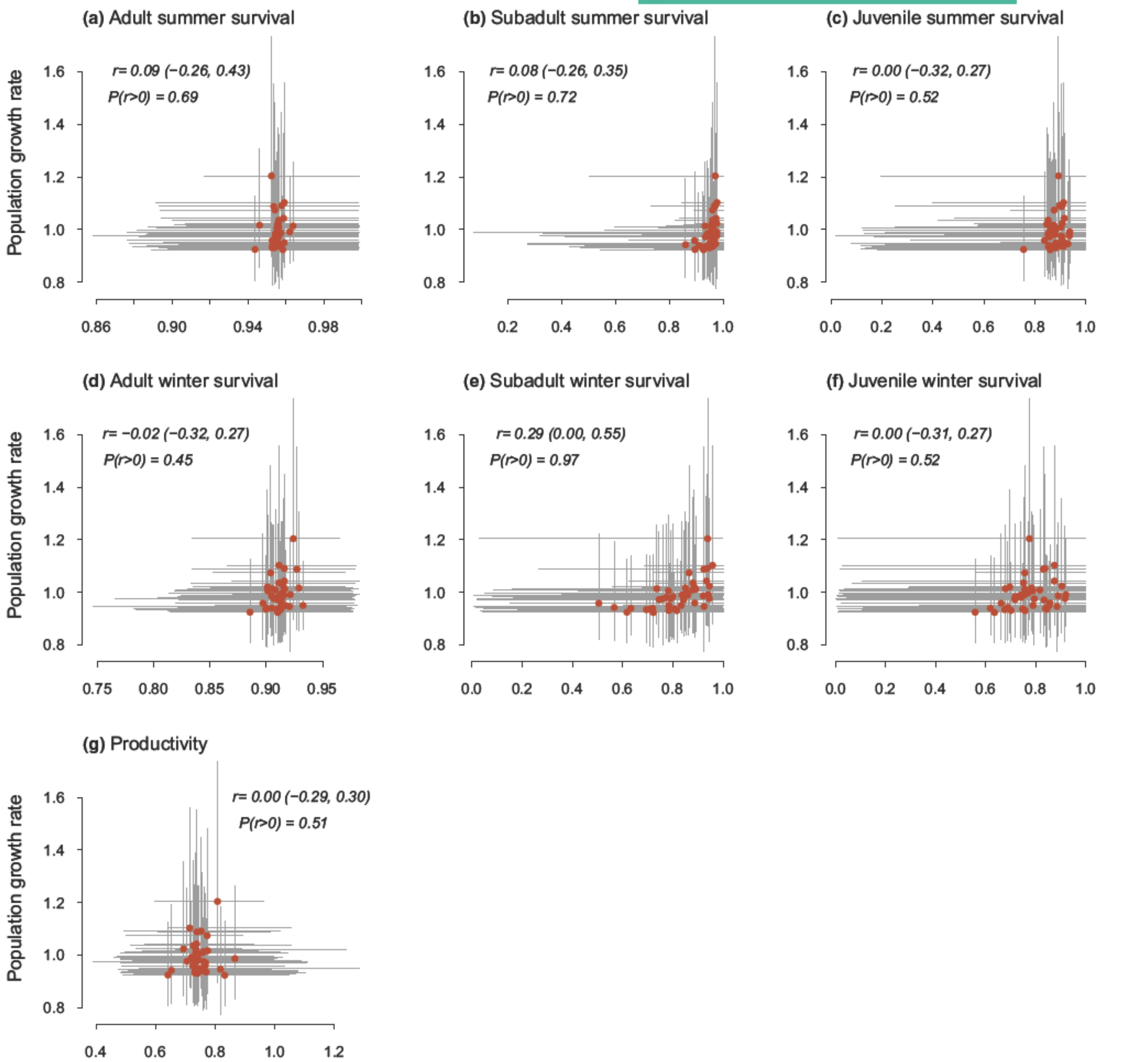


FIGURE 4 Annual posterior means of the estimated population growth rates ($\lambda_{t,B,A}$) plotted against posterior means of the estimates for (a) adult summer survival, (b) subadult summer survival, (c) juvenile summer survival, (d) adult winter survival, (e) subadult winter survival, (f) juvenile 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 ≥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

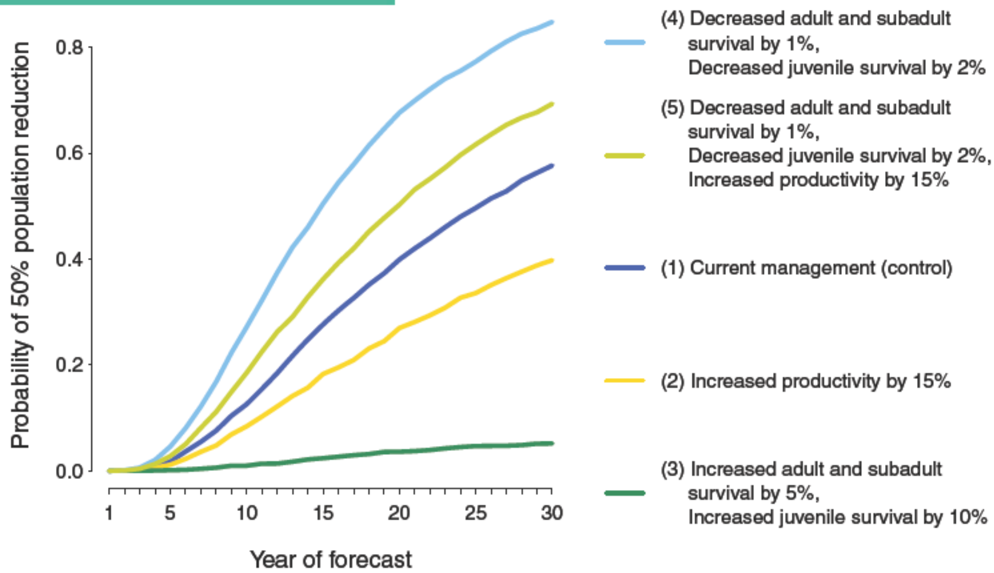


FIGURE 5 Comparison of the cumulative probabilities of a 50% population reduction of the tōrea population 30 years into the future under five different intervention and threat scenarios as described in Table 1.

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.

## ACKNOWLEDGEMENTS

Thank you to the many people who have contributed resightings of banded birds and counted birds over the decades; to Nikki McArthur, David Thomas, Holly Thompson, George Hobson, Morgan Coleman, Luke Sutton and Archie MacFarlane for assistance with field work; to Nicolette Faville for assistance with Figure 1; to Mandy Barron, Brett Sandercock, an anonymous reviewer and editor for helpful comments; and to Ray Prebble for editing this manuscript. This research was funded through the Ministry of Business, Innovation and Employment (MBIE), the Regional Braided River Initiative from Environment Canterbury and the Department of Conservation's Mobile Terrestrial Threatened Species Workstream.

## 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).

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## 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.

**How to cite this article:** Schlesselmann, A.-K., Monks, A., Walker, S., Sagar, P., Melville, D. S., Schuckard, R., Williams, E., Krouse, S., O'Donnell, C. F. J., & Schaub, M. (2025). A range-wide full-annual-cycle model informs conservation of a declining migratory shorebird. *Journal of Applied Ecology*, *00*, e70228. https://doi.org/10.1111/1365-2664.70228

**Attachment 3**

**Attachment 3. Locations (pink points) of the Threatened – Nationally Critical vascular plant species *Lepidium solandri* in the terrace edge zone adjacent to the Project site in 2016.**

[redacted]