Avifauna Population Viability Analysis for Nova Energy Solar Farm at Twizel, Canterbury

Contract Report No. 7414c.iii

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Wildlands

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Project Team:

Joshua Thoresen – Report author and biostatistician
Samantha Beckman – Report author
Della Bennet – Report author and technical advice

Prepared for:

Nova Energy
95 Customhouse Quay
PO Box 3142
Wellington 6141

Reviewed and approved for release by:

[redacted]

Justyna Giejsztowt
Principal Ecologist – Biodiversity and Modelling Specialist
Wildland Consultants Ltd
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Auckland Office
12 Nixon Street, Grey Lynn, P.O. Box 46299, Herne Bay, Ph 09 360 6083

Head Office
99 Sala Street, P.O. Box 7137, Te Ngae, Rotorua, Ph 07 343 9017, Email rotorua@wildlands.co.nz

www.wildlands.co.nz

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1.0 Introduction

Wildland Consultants Ltd (Wildlands) has prepared a Population Viability Analysis (PVA) for Nova Energy Ltd on a range of avifauna species that may be affected by the proposed Twizel Solar Farm. The proposed site is located within the Pukaki Ecological District and covers *c.*866 hectares of farmland, with a proposed panel area of 565 hectares. Wildlands has prepared an Avifauna Management Plan (AMP) for the proposed solar farm (Wildland Consultants 2026), which identifies the avifauna present or likely to be present at the site, the ecological values of the site, and the measures recommended to minimise impacts on indigenous fauna.

A PVA model was used to project how each species' population is likely to change over the next few decades, to understand the risk of the proposed solar farm to local bird populations. The models were used to explore how populations of species respond to extra bird deaths each year over time. These additional deaths represent birds that may collide with infrastructure at the proposed solar farm, as well as deaths from any other new sources. The PVA's purpose is to test whether the trigger points[1] proposed in the AMP and in proposed conditions of consent are set at a sufficiently conservative level. It is not to assess the potential for collisions. The PVA is not able to calculate how many bird collisions with the solar farm could be permitted, because it cannot differentiate among sources of additional deaths. The PVA is also not intended to illustrate the number of collisions with solar farm infrastructure the population can withstand before concern is raised. Instead, the PVA can illustrate whether the trigger points are low enough that, if they were ever reached, the population would not be at imminent risk of population collapse.

This report includes a PVA for five Threatened and four At Risk bird species, which may be affected by the proposed solar farm.

2.0 Analysis Background

Population viability analysis is a set of methods for evaluating threats faced by populations or species, their risk of decline or extinction, and their chances of recovery (Keedwell 2004). It allows assessment of different management options for a population or species, helps assess how to use limited resources, and identifies future research needs. PVA models can assess the effects of habitat quality, habitat patches, fragmented populations, migration rates between subpopulations, and genetic effects, such as inbreeding depression, on population viability (Keedwell 2004). They can also incorporate demographic parameters such as size and age, as well as birth, death and migration rates.

Depending on the type of PVA run, the required demographic parameters can vary. Three critical values are needed: the starting population size, survival rates (either across life stages or as one value), and fecundity. Population size is the number of individuals from which the projection begins, and is generally expressed as mature breeding individuals (as it is here), or sometimes as breeding females. Survival is the probability that an individual alive at the start of a year is still alive a year later, and is usually estimated by marking individuals and recording how many are subsequently re-sighted or recaptured. Survival, by nature of how it is measured, incorporates any potential factor that could result in the death of an individual, including predation, weather events, and starvation. Survival usually differs between age classes, with young birds typically surviving less well than adults, so a PVA should include survivability across life stages where data is available, or there is reason to do so. When not modelled as an entire population, migration (both emigration and immigration) can influence survival values. If migration is not factored into a mark-recapture study, an individual that leaves a site

[1] A trigger point is an agreed number of bird deaths that, once reached, requires changes in how the site is managed and, where necessary, to provide compensation.

and does not return but also does not die may be recorded as a death. Migration rates can therefore be included in a PVA to account for this where needed. Fecundity is the number of offspring each individual contributes to the population each year. Here it is calculated as a per-capita rate, and incorporates information from breeding success, hatching success and fledging success. Fecundity is usually derived from measured productivity, expressed as fledglings per pair or per nest, adjusted for the proportion of young surviving to breeding age. Age at first breeding determines how long individuals spend in pre-breeding stages and, therefore, how long a delay occurs between adult loss and any consequent shortfall in recruitment. Two further quantities can be calculated rather than estimated or measured. The population growth rate (lambda; λ) is the factor by which the population is multiplied each year. This value is calculated from survival and fecundity, providing a key check: if a model's growth rate does not match observed trends, the model is misparameterised, no matter how reliable its input data. Generation time, calculated from age at first breeding and adult survival, sets the period over which population change is conventionally assessed.

As populations do not experience constant conditions, PVA models represent chance variation explicitly and this is generally split into two forms: demographic and environmental stochasticity (Lande et al. 2003). Demographic stochasticity is chance variation arising from individuals living or dying and breeding or failing as discrete events. For example, a population of 100 birds with a survival rate of 0.8 does not lose exactly 20 birds each year, every year. This variation is negligible in large populations but becomes a substantial driver of extinction risk in small ones, because the relative risk posed by individual events increases as population size falls. Environmental stochasticity is year-to-year variation in population growth rates, arising from weather, food supply, predator abundance and similar external influences, and affects all individuals simultaneously. Models may also represent catastrophes, infrequent events, such as storms, floods or predator incursions, that cause severe reproductive or survival failure in a single year. Ordinary environmental variation cannot reproduce this, but catastrophes are often a dominant influence on extinction risk in small populations (Mangel & Tier 1994). These processes are random by nature, and a PVA can compound this randomness to create substantial variability across multiple population growth trajectories for any given species. Therefore, it is important to run a PVA across thousands of iterations and report the results as a distribution of outcomes rather than a single trajectory.

Where a population is not closed, migration should also be represented. Movement of individuals between subpopulations, or between a modelled population and areas outside it, can substantially alter viability. Immigration can sustain a subpopulation whose own births and deaths would not support it. Emigration removes individuals as effectively as mortality does. Survival estimates derived from marked individuals usually estimate apparent survival, because a marked bird that has emigrated and one that has died are indistinguishable to an observer, so emigration can inflate apparent mortality. Migration also determines whether the effects of a localised impact remain local: where movement between subpopulations is substantial, losses at one site may be partly replaced from elsewhere, but the effect is then displaced to the wider population rather than avoided. Correlated environmental variation and catastrophes acting simultaneously across subpopulations further limit the extent to which movement can buffer local losses (Hanski 1998). Where migration rates are unknown, as is commonly the case, a population may be modelled as closed provided the assumption is stated, since this will overstate the vulnerability of a subpopulation receiving immigrants and understate the wider consequences of local losses.

All required parameters are rarely all available for any single species. Where one cannot be estimated directly, it may sometimes be constrained by calibration; that is, by requiring the model to reproduce an independently observed population trend, which fixes the value of a single unknown parameter. This is a legitimate but limited approach. A model calibrated in this way reproduces the trend to which it is fitted by construction, and therefore provides no independent test of its own assumptions. This type of model may be overfitted and underperform when predicting into novel conditions. Where parameters are almost entirely absent, calibration will not help, as it can only be used to estimate a single parameter based on the fixed nature of others. Estimates and best guesses based on conspecifics

can be used for missing parameters. However, these estimates need to be made explicit. Although PVA is a useful technique, a quality PVA cannot be performed without sufficient data on the target species (Keedwell 2004).

3.0 Methods

3.1 Literature review

A literature review was conducted to collect credible information on the parameters needed for nine bird species in the PVA. This was undertaken by searching the literature using Google Scholar. We estimated the abundance data, survival rates across life stages, and fecundity parameters for each species. We estimated the carrying capacity, density-dependence, and the effects of catastrophic events and incorporated them where possible. However, the data allowed reasonable estimates for only a few species. Where these variables could not be estimated for a species, they were calculated internally within the model (as discussed in the results sections for each species). We also documented risk factors, such as flight behaviours, recorded proximity to the site, and migratory patterns. The following list provides the bird species included in the PVA and their conservation status according to Robertson *et al*. (2021):

- Kakī/black stilt (*Himantopus novaezelandiae*, Threatened – Nationally Critical).
- Matuku-hūrepo/Australasian bittern (*Botaurus poiciloptilus*, Threatened – Nationally Critical).
- Kōtuku/white heron (*Ardea alba modesta*, Threatened – Nationally Critical).
- Tarapirohe/black-fronted tern (*Childonias albostriatus*, Threatened – Nationally Endangered).
- Pūteketeke/Australasian crested grebe (*Podiceps cristatus australis*, Threatened – Nationally Vulnerable).
- Māpunga/black shag (*Phalacrocorax carbo novaehollandiae*, At Risk – Relict).
- Pohowera/banded dotterel (*Charadrius bicinctus bicinctus*, At Risk – Declining).
- Tarāpuka/black-billed gull (*Chroicocephalus bulleri*, At Risk – Declining).
- Tōrea/South Island pied oystercatcher (*Haematopus finschi*, At Risk – Declining).

3.2 Population Viability Analysis model

A series of species-specific PVA models was developed as a stage-structured, stochastic matrix projection, implemented in R version 4.4.3 and run with 2000 Monte Carlo simulations. The models were custom-built using base R functions. Each species was represented either as a vector of stage-specific abundances across three life stages or, where age-class data were unavailable, as a single pooled class. Annual transitions between stages were governed by a Lefkovitch (stage-structured) projection matrix constructed from three parameters per stage: annual survival, the proportion of survivors remaining in their current stage rather than advancing, and per-capita recruitment into the first stage. The terminal stage was specified as a self-loop, such that surviving adults remain in the adult class indefinitely. This structure allows stage duration to be determined by the available data, so species that differ in age at first breeding are accommodated within a single framework by varying the stasis parameter alone. Parameter values and their sources are given in Table 1.

Rather than multiplying stage abundances by expected rates, we draw integer outcomes at each time step. We partitioned each stage cohort into those that died, survived-and-remained, and survived-and-advanced by a multinomial draw, and recruitment was drawn from a Poisson distribution. These draws use the corresponding matrix entries as their expectations and therefore introduce no bias, but they introduce variance that scales inversely with the square root of abundance. This demographic stochasticity is a source of extinction risk in small populations and is negligible in large ones. Population viability (but not necessarily collision risk; see below) was assessed for breeding adult populations of each species rather than on total population size. Adult survival accounts for most of the flexibility on

population growth rate in the model, and adults are resident breeding birds that commute daily to forage, giving them the highest exposure to site infrastructure.

Table 1 - Parameters used in the PVAs for the nine avifauna species, including survival, fecundity and risk factor parameters required.

Parameters	Species								
	Kakī/black stilt	Matuku-hūrepo/Australasian bittern	Kōtuku/white heron	Tarapirohe/black-fronted tern	Pūteketeke/Australasian crested grebe	Māpunga/black shag	Pohowera/banded dotterel	Tarāpuka/black-billed gull	Tōrea/South Island pied oystercatcher
Survival									
Current population size	< 200 individuals (141-169 adults 2023-2025) (DOC, 2025)	<1000 individuals (Williams & Brady 2014). Of this, there are estimated to be around 500 breeding adults across their entire range.	Around 100 breeding adults (*c.*50 nests/season), population stable since 1984 (Miller 2001). Total population, including immatures, estimated at 150-200 individuals (Adams, 2013). Colony held 54 breeding pairs in 2025.	Population modelled is from the Upper Ōhau 'Tern Island' colony: 718 breeding adults, from a peak of 359 active nests on 23 Oct 2024 (colony size = 2 x max active nests; Nelson et al. 2025). Largest black-fronted tern colony in NZ.	1,047 individuals - minimum count, January 2024 national census of South Island lakes and lagoons (Wanaka Grebe Project). Trend (used for calibration) is independently corroborated by O'Donnell (2013): *c.*200 birds in the 1980s, *c.*400 in 2004, *c.*600 in 2012.	5,000-10,000 nationally; New Zealand Threat Classification System (NZTCS) confirms this is mature individuals. Model uses an estimate of 7,500 mature individuals (Powlesland, 2025; Robertson et al. 2021).	12,730 birds - sum of the most recent counts across 119 braided and shingle river reaches nationally (O'Donnell & Monks 2020). The model uses this figure as it is the same population on which the -3.7%/year trend used for calibration was measured. The wider national population across all habitats is around 50,000, but has different dynamics and no matching trend estimate.	Model uses Canterbury population size (estimated by 20,675 nests x 2 = 41,350 mature individuals). Total population is 120,512 mature individuals = 60,256 nests x 2 (from Mischler, 2018a).	An estimated 77,000 birds (Sagar et al. 1999; Riegen & Sagar, 2020). Model uses *c.*38,000 mature individuals (estimated by 77,000 x 49% adult fraction). An estimate supported by the fact that it falls within the NZTCS band of 20,000-100,000 mature individuals.
Chick survival rates and estimated population	20% managed, <1% unmanaged. Estimated at 140 individuals. Could not find source for population size or releases from recovery programme. Several sources were used including Facebook Kakī recovery programme page, DOC 2025, North-South magazine, and media releases (Stuff NZ; RNZ). Model uniquely parameterises this see Section 4.4.1.	Unknown. However, sexual maturity occurs at around one year, so the model cannot use three life stages.	Model used an estimate of 0.24 from Kahl (1963) who estimate a 76% first-year mortality (i.e. 24% survivability) in great egrets (*Ardea alba*). Estimated population size = 55.	Estimated at 0.49, based on tata teo/little tern (*Sternula albifrons*) (Schekler et al. 2019). No NZ estimate. Estimated population size = 250.	Not used. No NZ estimate of first-year survival exists.	Model uses an estimate based on a non-New Zealand congener: 0.35 - midpoint of first-year survival estimates for British/Irish great cormorant (*Phalacrocorax carbo)* 24% (Scotland), 45% (Wernham & Peach, 1999). Estimated population size = 5,408.	Not used - sexual maturity occurs at around one year, so the model cannot use three life stages.	Not used. No NZ estimate of first-year survival exists.	Estimated at 0.51 from Schlesselmann et al. (2026). Estimated population size = 14,445. (Sagar et al. 2000)
Fledgling survival rates and estimated population	30% managed, or 57% under optimal conditions (DOC 2025). Estimated population size = 45 individuals from survival data showing only *c.*30% survive to this stage from those released each year.	Unknown. However, sexual maturity occurs at around one year, so the model cannot use three life stages.	No estimate exists; Miller (2001) explicitly states that there are no data on post-fledging survival or subadult return rates. Here we estimate this at 0.837 from preliminary modelling. For example, with the other two survival rates, fecundity, and maturation rate fixed, and requiring population growth to = 1.00 (i.e. as per the reported population stability) then survival must = 0.837. Estimated population size = 23.	Estimated at 0.80 based on preliminary modelling, but no NZ estimate found. Estimated population size = 160.	No estimate. Combined with the absence of a first-year estimate, this leaves two unknown pre-adult rates, which preliminary modelling cannot resolve. A three-life stage model is therefore not supportable for this species.	No estimate exists. Model uses 0.61 from preliminary modelling. Falls plausibly between juvenile (0.35) and adult (0.89). Estimated population size = 2,721.	Not used - sexual maturity occurs at around one year, so the model cannot use three life stages.	No estimate. Combined with the absence of a first-year estimate, this leaves two unknown pre-adult rates, which preliminary modelling cannot resolve. A three-life stage model is therefore not supportable for this species.	Estimated at 0.77 from Schlesselmann et al. (2026). Estimated population size = 24,226.

Parameters	Species								
	Kakī/black stilt	Matuku-hūrepo/Australasian bittern	Kōtuku/white heron	Tarapirohe/black-fronted tern	Pūteketeke/Australasian crested grebe	Māpunga/black shag	Pohowera/banded dotterel	Tarāpuka/black-billed gull	Tōrea/South Island pied oystercatcher
Adult survival rates	Three estimates modelled: 0.56 – the survivability reported by Bulling (2008) for the non-Tasman release cohort, 0.78 – the survivability reported for the Tasman release cohort and a central value of 0.71. See Section 4.1.1.	Unknown. For model estimated across three bands (0.72, 0.73 and 0.75) based on extrapolation from the NZTCS criteria for a Threatened – Nationally Critical species (Townsend *et al.* 2008).	Estimated at 0.905 derived from Miller (2001) who noted that NZ mortality must be lower than the Kahl (1963) figure of 26% because the colony grew. An estimate of 0.905 reconciles the recorded fecundity with the reported stable population growth.	0.88 measured on the Ōhau River from 72 colour-banded adults (Keedwell, 2005). Range 0.88-0.92, CI 0.57-0.99.	0.75 measured for European great crested grebe (*Podiceps cristatus*) from 57 years of European ring recoveries (Abt & Konter 2009). The authors state two biases that make 0.75 an underestimate for NZ: (i) recoveries were biased toward young birds, shown by an atypical rise in apparent survival after age three; (ii) 45% of great crested grebe recoveries were anthropogenic, mainly drowning in fishing nets and hunting, mortality sources that are largely absent in NZ. Model uses 0.85, which also reconciles the model with recorded population growth.	Estimated at 0.89 taken from Wernham & Peach (1999) study on great cormorants. Model uses the Northern England estimate, not the 0.79 Scottish estimate which gives a population growth rate of 0.906 (i.e. a 9.4%/year decline). This contradicts the species' Relict classification (Townsend et al. 2008).	Model uses 0.85, extrapolated from calculating population growth as 0.99 and from measures of population size, generational time, and the decline rate as per the NZTCS criteria for an At Risk – Declining species (Townsend et al. 2008)	No estimate could be found for tarāpuka/black-billed gull, so estimated as 0.86 based on a tarāpunga/red-billed gull proxy (Mills et al. 2008).	Estimated at 0.87 (Sagar 2013, and Schlesselmann et al. 2026).
Fecundity									
Typical lifespan (years)	15 (Pierce 2013a).	Unknown, but estimated 5-10 years (O'Donnell & Robertson, 2016).	22 years maximum (Adams, 2025).	26 (Keedwell, 2002, Bell, 2013).	19.2 years maximum from banding (Fransson et al. 2010).	20 years (Powlesland, 2025).	6-7 years (Robertson et al. 2013); 12 years max (Pierce, 2013b); 20 years max (Keedwell, 2004). Model uses a generation time of 6.7 years (derived from O'Donnell & Monks, 2020).	29 years maximum (McClellan & Habraken, 2023).	25 years maximum (Sagar 2013). With adult survival 0.89 and maturity at ~5 yr, generation time = 5 + 0.89/0.11 = 13.1 years. The longest of any species assessed, so three generations is *c.*39 years and a 25-year projection substantially understates decline against NZTCSs criteria.
Survival variance (across life stages or years)	No published estimate. Model used Standard Deviation (SD) 0.05.	No published estimate. Model used SD 0.05.	No published estimate. Model used SD of 0.06 (juvenile, subadult) and 0.04 (adult).	Estimated annual adult survival 0.88-0.92 (Keedwell 2005; Nelson et al. 2025). Model used SD of 0.05 adult, 0.06 subadult, 0.08 juvenile.	No published estimate. Model used SD of 0.05.	No published estimate. Model used SD of 0.05 across all stages.	No published estimate. Model used SD of 0.05.	No published estimate. Model used of SD 0.05.	No published estimate. Model used SD of 0.05.
Average clutch size per year	4 eggs (Pierce, 2013a).	4.3 eggs (O'Donnell, 2011).	4 eggs (Adams, 2025)	2 eggs (Keedwell, 2002; Bell 2013)	3.1 eggs (Marchant & Higgins, 1990).	4.1 eggs (Powlesland 2025).	3 eggs (Pierce, 2013b)	1.85 eggs (McClellan & Habraken, 2023)	2.3 eggs (Sagar 2013)
Average clutch frequency per year	2-3	One (Williams, 2022)	One (Adams, 2025)	1-2 (Bell, 2013)	If clutches are lost, broods may be replaced. Long breeding season (September to March).	One (Powlesland & Reese, 1999). The authors note that the *c.*5-month breeding cycle plus moult makes two broods per year unlikely.	Capacity for more than one clutch per season, if threat factors are removed (Keedwell, 2004).	One, but able to re-nest if eggs or chicks are lost (Higgins & Davies, 1996).	One brood per season, but up to two or three replacement clutches if earlier attempts fail early (Sagar et al. 2000).
Breeding success rate	Extremely low (near zero); unknown, estimated at 5%. Model	Unknown	91% (Marchant & Higgins, 1990).	0.427 = the proportion of nests hatching at least one egg (Bell, 2013 and	Low; no quantified NZ estimate. Fecundity was therefore not estimated	83% = 153 successful nesting attempts out of	Hatching success 56-68%, fledging success 48% (Keedwell, 2002).	0-88% per nest; mean 32% (McClellan 2009).	28% overall; 47% hatching x 59% fledging (Sagar et al. 2000).

Parameters	Species								
	Kakī/black stilt	Matuku-hūrepo/Australasian bittern	Kōtuku/white heron	Tarapirohe/black-fronted tern	Pūteketeke/Australasian crested grebe	Māpunga/black shag	Pohowera/banded dotterel	Tarāpuka/black-billed gull	Tōrea/South Island pied oystercatcher
	adapted to include an 'unmanaged' scenario and an 'augmented' scenario to take into account the 'artificial' fecundity of the breeding program.			2017), upper Clarence, n = 1,510 nests). Productivity from the same study was 0.13 fledglings/nest.	from breeding data - it was estimated at 0.20 and survival calibrated instead.	185 clutches (Powlesland & Reese, 1999).			However, note that 52% of pairs that laid failed to rear any fledglings.
Age of first breeding	2-3 years (Pierce, 2013). Giving a 'stasis'[1] of 0.3 for fledglings in the model.	Unknown - suggested 1-2 years (Williams 2022). Taken as one year and so the model is unstructured (i.e. only one life stage).	Three years, stasis = 0.5. (Adams 2025)	Two years, stasis = 0.3. (Bell 2013)	Two years, but model is unstructured as no estimates on chick/fledgling population or survivability could be found.	3 years, stasis = 0.5	One year. There is no subadult class, so an unstructured model is correct rather than a simplification (as for Australasian bittern).	Two to three years, but model is unstructured as no estimates on chick/fledgling population or survivability could be found.	4-6 years (Sagar & Veitch, 2014). The latest maturation of any species assessed, which drives the long generation time and requires a three stage structure with a four year subadult stage. Requiring stasis to = 0.75.
Fecundity	Unknown but very low. Model estimated unmanaged fecundity at 0.05. The breeding programme and chick releases each year increase this. However, this addition is modelled separately (See Section 4.1.1).	Unknown, but for model estimated at a low rate of 0.2.	0.55 per adult per year taken from Miller (2001) who estimates 1.1 fledglings per nest per 2 adults.	Fecundity unmanaged = 0.1, managed fecundity = 0.55 (Keedwell, 2005). Model runs both scenarios.	0.20 per adult per year. No NZ source. Held fixed because low fecundity is biologically plausible for this species. The calibration was applied to survival instead.	0.72 per adult per year = 1.44 fledged young per brood across all 185 clutches (Powlesland & Reese, 1999) divided by two adults per nest. The same study gives 1.75 fledglings per brood from successful attempts only; 1.44 is used as it includes failed attempts. Seasonal range 1.1-1.7 fledglings/nest.	0.113 recruits per adult per year. Model calibrated so that λ = 0.963, matching the measured decline of -3.7%/yr (O'Donnell & Monks 2020). Consistent with the breeding data: 0.40-0.49 fledglings per adult x first-year survival of 0.23-0.28.	0.14 recruits per mature individual per year. Model calibrated to λ = 1.00. Consistent with the breeding data: 0.30 fledglings per adult x ~47% survival to first breeding at age three.	Estimated as 0.74 +/- 0.13 chicks fledged per female, as per Schlesselmann et al. 2026).
Fecundity variance	Estimated SD of 0.03.	Unknown. Estimated at 0.08.	0.15, taken from Miller (2001) range of 0.4-2.0 fledglings/nest.	0.05 estimated.	0.10 estimated.	SD 0.10 (assumed; observed seasonal range 1.1-1.7 fledglings/nest).	SD 0.05 estimated.	SD 0.06 estimated.	SD 0.13 (Sagar et al. 2000, Table 3: 0.76 +/- 0.13 chicks fledged per pair across 10 seasons).
Risk Factors									
Flight behaviours	Aerobatic displays and dive bombing if threatened near chicks.	Fly between wetlands (up to 140-550 kilometres) in response to flooding, drought or seasonal increase in food availability.	Gliding, foraging flights, and feeds at night. All NZ birds fly to Waitangiroto to breed, then disperse nationally. The most common recorded cause of kotuku/white heron mortality in NZ is vehicle strike, because kōtuku/white heron fly low and slow when first becoming airborne (Adams, 2013).	Breed only on South Island braided rivers, where they will mob predators. Forage on braided rivers and farmland during the breeding season, and in harbours, estuaries, and lagoons during the non-breeding season. Fly faster when moving between foraging and colony areas compared to when they are foraging.	Completely aquatic – physiologically unable to walk on land, nests float, rarely ashore. Collision relevance is the strongest of any species assessed: (i) birds are present at the site – nine observed on Lake Ruataniwha (Miskelly, Te Papa, Feb 2024) and recorded on Lake Benmore; (ii) grebes routinely fly between lakes, with post-breeding movements within lake groups including *c*.100 kilometres to Lake Alexandrina. Sagar	Southland: breeds on inland riverbeds. including braided rivers, and breeds in coastal aeras. Forage alone, usually in waters under *c*.3 metres. Commute between roosts and foraging waters. Recognised anthropogenic mortality: capture in set nets and craypots, and historic/occasional shooting by fishermen and hunters.	Wing-clicking courtship displays and territorial chase behaviour by males. Nests in shallow scrapes in gravel, sand or soil, riverbeds, outwash fans, herbfields, beaches and farmland. Partially migratory: South Island high-country and inland braided river breeders mostly migrate to Australia after breeding, so birds undertake long-distance movements; coastal and lowland breeders mostly fly north to harbours and	Gliding and soaring; defensive mobbing; foraging flight over braided rivers and paddocks. Prefer to forage on ploughed farmland for worms and insects (Higgins & Davies 1996). Colonies are abandoned readily if nearby food supplies are insufficient, and birds minimise time at colonies, so foraging flights are frequent (Evans 1982a and 1982b). Most South Island birds move to the coast after breeding.	Will dive-bomb intruders when nests or chicks are present. Breeding was formerly confined to gravel riverbeds but, over the past 60-70 years, has spread onto farmland and, most recently, coastal beaches (Sagar & Veitch, 2014). Strongly migratory in NZ: most breeding adults leave inland territories from December for coastal estuaries, mostly in the northern South Island and northern North Island, returning from early June. Birds

[1] The probability of an individual not progressing to the next life stage in any given year given that it has survived the year.

Parameters	Species								
	Kakī/black stilt	Matuku-hūrepo/Australasian bittern	Kōtuku/white heron	Tarapirohe/black-fronted tern	Pūteketeke/Australasian crested grebe	Māpunga/black shag	Pohowera/banded dotterel	Tarāpuka/black-billed gull	Tōrea/South Island pied oystercatcher
					(1982) concludes all lakes in a group matter because birds redistribute among them. The proposed project footprint lies between two occupied lakes in such a group; (iii) hydroelectric development is already a recognised threat to this species (O'Donnell 2013).		estuaries within the North Island.		therefore undertake long-distance internal movements twice a year in addition to local foraging flights.

Environmental stochasticity was incorporated as a separate process. Demographic variation is independent among individuals, whereas environmental variation is shared across the population. Vital rates were therefore redrawn each year, prior to the application of transitions, from distributions centred on the input means and scaled by the input standard deviations. Survival was drawn from a Beta distribution and fecundity from a Gamma distribution, so that draws remain within the natural bounds of each quantity (zero to one for survival, non-negative for fecundity). As the entire cohort experiences a common annual draw, this component of variance does not diminish as population size increases. Environmental variance also depresses long-run population growth, which is governed by the geometric rather than the arithmetic mean of annual growth rates. A population with a nominally positive mean growth rate may therefore decline once inter-annual variability is accounted for.

Environmental stochasticity generates moderate variation between years but cannot generate complete reproductive failure, since a fecundity distribution centred on a viable mean assigns negligible probability to a zero-fledgling season. As such, catastrophic breeding failure was modelled explicitly for species where there was a defensible reason to do so (i.e. a single small breeding colony or documented past catastrophic failures). This comprised an annual probability of near-total reproductive failure, with a persistence term allowing failures to cluster across consecutive seasons rather than occurring independently. Clustered failures have a greater cumulative effect than the same number of isolated failures, as the population lacks an intervening recovery period. Adult survival was left unaffected, on the basis that colony abandonment prevents breeding without necessarily increasing adult mortality.

3.3 Collision mortality

Collision losses were applied as an additional mortality layer acting on the population prior to each annual demographic step. Losses could be specified in the model as a fixed number of individuals per year, as a per-capita annual probability, or as an explicit year-by-year schedule. Stage-specific vulnerability weights were applied to restrict losses to the age classes exposed to collision risk (i.e. actively flying birds), so chicks were never included as additional losses from collision. Removals were capped at the number of individuals available in each stage.

No site-specific collision data were available for the project or for solar farms in New Zealand in general. Rather than adopt a single assumed collision rate, the model was run across a range of annual loss levels for each species. These loss levels depended on each species' estimated risk and population size (see the per-species methods) to produce a relationship between the number of birds lost each year and the predicted outcome for the population. This range-based approach serves two purposes: it avoids reliance on a single uncertain collision estimate, and it allows the loss level at which a population-level effect first emerges to be identified and compared against the trigger level set for that species. Risk factors, such as flight behaviours and population size were used to manually alter the range of additional losses. For species that were considered more likely to be present or collide with a solar panel, the range of additional losses was increased.

The point at which an effect emerges was assessed using up to three complementary measures (See Section 5.2), these included; an observable reduction in population size relative to the no-collision projection, an increase in the probability of quasi-extinction by 1%, and a suppression of the population growth rate by 1%, with the most conservative measure adopted for each species. No single measure performs equally well across species that differ widely in abundance and trend; in particular, the quasi-extinction measure is uninformative for species whose baseline probability of quasi-extinction is already high. Reporting all three, and taking the most conservative available, ensures a more robust assessment by not depending on any one metric. The loss levels reported are effects thresholds, not allowable levels of mortality. They describe the point at which the modelling begins to detect a consequence for the population, and their role here is solely to confirm that the trigger levels sit below that point.

3.4 Managed population augmentation

For species subject to active breeding management, including kakī/black stilt, this assessment included an augmentation layer. Managed additions, comprising captive-reared birds released to the wild, were applied following the annual demographic step, so that released birds join the wild population and are then subject to the survival rate of their stage in the subsequent year. This mechanism captures high post-release mortality. Releases were allocated across stages according to the age at which birds are released. Modelling releases explicitly, rather than incorporating them into elevated vital rates, is important for interpreting results. An annual input of individuals converts the projection from a purely multiplicative process into one capable of maintaining a population at a non-zero level even when the intrinsic growth rate is below replacement. This describes the function performed by a captive-rearing programme, and it allows the contribution of management to be distinguished from the intrinsic performance of the wild population.

Release output was modelled as dependent on the wild adult population rather than as a fixed annual input. This is necessary as the captive-rearing programme is fed mostly by eggs collected from wild nests, so a declining wild population reduces the number of eggs available and therefore the number of birds available for release. Modelled as a fixed input, releases hold the projected population at an artificially stable level even after it has fallen below the abundance capable of sustaining the programme, and any apparent stability below that level is an artefact of the assumption rather than a property of the population. Output was accordingly scaled in proportion to the number of wild breeding adults, subject to a floor representing the component produced by captive breeding pairs, which continues irrespective of wild numbers.

3.5 Simulation and reporting

For each species modelled, each parameter set was run for 2000 independent stochastic monte-carlo simulations over 25 years. However, 2000 simulations over 40 years were reported for tōrea/South Island pied oystercatcher as their generation length is substantially longer than all other species modelled (for example, for pohowera/banded dotterel generation time is shorter, at 6-7 years). Results were interpreted from the resulting distribution across these 2000 simulations rather than from individual trajectories. The following metrics were reported across a range of collision scenarios per species:

- Median adults remaining after 25-years and the 95% confidence intervals. It should be noted that for this metric, the confidence intervals can be extremely wide. This reflects the uncertainty of any specific species' future under any scenario across the 2000 trajectories. They are not a measure of uncertainty about the effect of collision, and their overlap between collision scenarios should not be read as indicating an absence of effect.
- Per cent reduction against baseline (i.e. no collisions).
- Ratio of losses relative to baseline (i.e. no collisions) and the 95% confidence intervals.
- Probability of quasi-extinction (population falling below set thresholds per species).

Collision scenarios were compared using a set seed, ensuring that paired runs shared the same identical sequence of environmental and demographic events. This approach means that the differences in the ratio of losses relative to the baseline (i.e. no collision) isolate the effect of the parameter being tested, rather than reflecting simulation variance.

3.6 Verification and limitations

Matrix construction in R was verified against manual calculation, and the deterministic growth rate implied by the fitted parameters was compared with independently published population trends as an external check on plausibility, where available.

Species-specific model limitations are listed for each species. However, several general model limitations should be noted in interpreting the results:

- The model is largely density-independent. This is appropriate for populations well below carrying capacity over 25-years, but not appropriate for longer projections or for recovering populations approaching habitat limits. For species where there are recorded significant density-dependence effects (e.g. kōtuku/white heron), density-dependence terms were included in the model.

- Where species were modelled as discrete colonies or sub-populations, the models do not incorporate immigration and emigration rates, and so these open populations were modelled as closed. These models therefore overstate the vulnerability of the subpopulation as arriving immigrants will buffer the collision losses, and also understate the wider consequences of local losses, as losses to the wider population through emigration are not represented.

- Allee effects (whereby the growth of a population reduces more quickly at low densities) are not represented, so projections at very low abundance should be regarded as optimistic. However, as most bird populations with very low numbers tend to collapse quickly regardless (e.g. kakī/black stilt), an additional Allee effect term was considered unnecessary.

- Flight path and movement data were not available for this assessment. It is therefore not possible to estimate what proportion of any species' population transits the project area and to convert a collision rate into an annual mortality estimate. Results are reported as the population response to a range of annual losses per species rather than as a prediction of effect.

- Although not a general limitation, matuku-hūrepo/Australasian bittern has extremely limited available data and no recorded local population or colony. The model results for this species should be interpreted with caution and treated only as broadly indicative and subject to verification. The model for this species should not be interpreted with the same weight as the other species in this report.

4.0 Results

4.1 Kakī/black stilt

4.1.1 Species specific model methods

Kakī/black stilt were modelled as a single managed population over a 25-year projection, divided into juvenile, subadult (15 months to two years), and adult classes. Starting abundance was set at 141 adults, following the most recent DOC estimate (DOC 2025), with juvenile and subadult classes derived from the recent release cohorts rather than from the stable stage structure, as the age structure of this population is determined by the captive-rearing programme rather than by wild recruitment. Bulling (2008) estimated the adult survival separately for kakī/black stilt released in the Tasman River (0.78) and elsewhere in the Mackenzie Basin (0.56). However, this study assigned individuals that were all part of the captive-rearing programme to these different groups by their initial release site. As such, the groups are release cohorts within a single managed population rather than demographically independent populations. The apparent difference in survivability between areas also confounds true

mortality with emigration, as van Heezik et al. (2009) found that 32% of released kakī/black stilt reaching breeding age ended up away from their release site. The Bulling (2008) study also predates the expansion of predator control across the Tasman, Godley and Cass valleys. Therefore, the reported survivability differences were presented as a sensitivity of adult survival for a single population and run at 0.56, 0.71 and 0.78. The central value of 0.71 is intermediate between the two release-site estimates, consistent with a mixed population, and broadly compatible with recorded adult counts of between 129 and 170 birds since 2019.

As kakī/black stilt cannot currently sustain themselves in the wild, in-situ reproduction was modelled at a low rate reflecting the small proportion of eggs left in wild nests, and the release programme was represented explicitly as an annual addition of birds allocated across the juvenile and subadult classes. The release output for this model was set at up to 148 birds per annum from current adult numbers, and decreasing in proportion as wild adults numbers decline, but not dropping below a floor of 50 birds per year. The floor represents the captive-bred component: DOC collects eggs from both wild and captive breeding pairs, with captive pairs held at the Twizel facility and the Isaac Conservation and Wildlife Trust, so release capacity would not fall to zero even if the wild population were lost. The proportion of releases derived from captive-laid rather than wild-collected eggs is not published, and the value adopted here should be confirmed with DOC. Sensitivity analysis across floors of zero to 148 birds per annum indicated that a floor of 50 individuals produces the most plausible result, and that the parameter principally determines the level at which the population stabilises rather than the proportional effect of collision mortality. This model addition therefore introduces a positive feedback loop: collision mortality reduces the number of wild adults, which reduces the number of eggs available for collection, which reduces release output and so further reduces adult numbers. Two management scenarios were run for the zero-collision model to demonstrate the contribution of the breeding programme: releases continuing at current levels, and releases ceasing entirely (Figure 1). Where releases cease, the population declines to zero in every simulation. The comparison is shown for the zero-collision panel only, as the effect of ceasing releases is essentially independent of the collision level (Figure 1).

Collision losses were applied to subadult and adult birds, as both subadult and adult birds are at risk of collision. Juveniles were excluded as first-year birds remain largely dependent until the age of 6-8 months and most chicks are released by DOC at nine months. Once released, these birds may be exposed to site infrastructure as they move around the Mackenzie area foraging and will move into the subadult class at 15 months. Losses are allocated between the two exposed classes in proportion to their abundance, so approximately a quarter of modelled mortality falls on subadults. This moderates the effect on breeding numbers relative to an adults-only formulation, because a proportion of subadults would not have survived to breed in any case. Subadult survival is 0.57, so some subadult mortality is compensatory rather than additive.

Viability was assessed on the breeding adult population rather than on total population size, as total abundance is dominated by the annually released individuals from the breeding programme. The viability threshold was set at 80 breeding adults, approximately the population required to supply the egg-pulling programme with sufficient clutches to maintain releases at its current output, based on approximately 41 productive breeding pairs recorded in the 2024 season. The resulting metric is the probability that the wild population falls below the level needed to sustain its own recovery programme, rather than a conventional extinction threshold.

4.1.2 Results and interpretation

Under continuing releases and in the absence of collision mortality, the breeding population is projected to decline from 141 adults to a median of 53 adults over 25 years under the central survival case (95% interval 34-77), 94 adults at the upper survival value and only 27 at the lower. The probability of the breeding population falling below the 80 adults required to sustain the captive-rearing

programme is 0.98 centrally, 0.24 at the upper value and one at the lower (Table 2). In addition, quasi-extinction is predicted to be all but certain for every model iteration other than the lowest losses for the high survivability estimate, which are still high, ranging from 0.24 to 0.63 (Table 2). The relatively stable documented historic population growth rate for kakī/black stilt, and the fact that the species is not yet extinct, suggests that the lower adult survivability estimate of 0.56 is likely to be pessimistic, and that the higher estimates are more likely to reflect a truer adult survivability rate. Where releases from the breeding programme cease, the breeding population reaches zero within 5-10 years without collision losses (Figure 1), confirming that the population is not self-sustaining and that its persistence relies on continued management.

An annual loss of one individual results in a reduction relative to the baseline population decline of 3.7% for the lowest survivability estimate, 5.6% for the central estimate, and 7.4% for the higher estimate (Table 2). Losses of more than one bird per year result in substantial relative changes in the population decline rate, with two losses per year at the highest survivability estimate resulting in a 12% relative reduction and a probability of quasi-extinction near 50% (Table 2, Figure 1).

Table 2 - Population Viability Analysis (PVA) results for kakī/black stilt breeding adults across 1-15 bird losses per year after 25 years while under active breeding management. Three different adult survivability values were modelled: 0.56 (non-Tasman survivability reported by Bulling, 2009), 0.78 (Tasman survivability reported), and 0.71 a central value between these two extremes. The probability of quasi-extinction is the probability that the adult population falls below 80 total birds, or 40 breeding pairs, which would significantly impact the breeding programme. A probability of one represents near-certain quasi-extinction, and 0 represents no likelihood of quasi-extinction.

Adult Losses per Year	Median Adults Remaining After 25 years [95% CI]	Reduction Against Baseline (%)	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
Adult survivability: Low estimate: 0.56				
0	27 [16-40]	0	1.00 [1.00-1.00]	1
1	26 [15-39]	3.704	0.96 [0.54-1.67]	1
2	24 [14-36]	11.111	0.88 [0.49-1.52]	1
3	22 [12-34]	18.519	0.82 [0.44-1.44]	1
5	19 [10-31]	29.63	0.70 [0.37-1.27]	1
10	11 [4-20]	59.259	0.41 [0.17-0.80]	1
15	4 [0-11]	85.185	0.14 [0.00-0.40]	1
Adult survivability: central estimate: 0.71				
0	53 [34-77]	0	1.00 [1.00-1.00]	0.983
1	50 [32-72]	5.66	0.93 [0.59-1.48]	0.996
2	46 [29-69]	13.208	0.87 [0.54-1.39]	0.999
3	43 [26-63]	18.868	0.80 [0.49-1.26]	0.999
5	35 [21-54]	33.962	0.66 [0.40-1.08]	1
10	19 [8-32]	64.151	0.35 [0.16-0.63]	1
15	5 [0-14]	90.566	0.10 [0.00-0.27]	1
Adult survivability: high estimate: 0.78				
0	94 [62-136]	0	1.00 [1.00-1.00]	0.241
1	87 [56-129]	7.447	0.93 [0.64-1.35]	0.358
2	82 [50-121]	12.766	0.87 [0.57-1.29]	0.474
3	75 [47-116]	20.213	0.80 [0.53-1.19]	0.629
5	62 [38-98]	34.043	0.67 [0.43-1.01]	0.86
10	33 [16-59]	64.894	0.35 [0.19-0.59]	0.999
15	9 [1-23]	90.426	0.09 [0.01-0.23]	1

4.1.3 Model limitations

The model has several kakī/black stilt specific limitations. First, and most significantly, kakī/black stilt were not recorded in the Ōhau River system in any round of Project River Recovery's walk-through surveys, including the 2023 and 2024 surveys of the Upper and Lower Ōhau Rivers (Nelson et al. 2025). The species' stronghold within the basin is the Tasman River, where 17 birds were recorded in 2024 (Nelson et al. 2025). The population modelled in this report is therefore the managed Mackenzie Basin population rather than a population resident at or adjacent to the project footprint, and results should be read as an assessment of effects on that wider population conditional on birds using the site.

Second, the model is density-independent and does not represent an Allee effect. This matters more for kakī/black stilt than for the other species assessed, because reduced mate availability and hybridisation with poaka/pied stilt (*Himantopus himantopus leucocephalus*, Not Threatened) are documented mechanisms by which reproductive performance declines at low abundance. Projections in the lower-abundance scenarios, and in particular the releases-ceasing scenario, should therefore be regarded as optimistic relative to the likely outcome. However, as the model already predicts rapid declines, including an Allee effect term was considered unnecessary.

Third, the results are conditional on the Kakī Recovery Programme maintaining at least a 50 individual annual output for the duration of the projection period. The wild fecundity rate is also derived rather than a measured value, estimated from reported per-pair productivity and the proportion of adults in productive pairs. No published per-capita recruitment rate for wild kakī/black stilt was located.

4.2 Matuku-hūrepo/Australasian bittern

4.2.1 Species specific model methods

Matuku-hūrepo/Australasian bittern was modelled as a single unstructured population of mature individuals over a 25-year projection. An unstructured model is appropriate for this species rather than being a limitation of available data as Matuku-hūrepo/Australasian bittern reach sexual maturity at approximately one year, so there is no subadult class to represent in the model, and the stage structure collapses to a single reproductive class. It should be noted that neither adult survival nor fecundity has ever been measured for matuku-hūrepo/Australasian bittern, and O'Donnell (2011) states explicitly that there are no quantifiable data on nesting success. Documented information is limited to breeding biology: a clutch of three to five eggs (mean four), a single brood per season, incubation of approximately 25 days, and chicks remaining in the nest for approximately seven weeks. The literature suggests nest success and chick survival are particularly low but has not been quantified. In the absence of measured rates, the model was calibrated to the New Zealand Threat Classification System (NZTCS) 2021 criterion, under which the species qualifies as Threatened – Nationally Critical on the basis of 250-1,000 mature individuals undergoing a decline of 50-70% over three generations (Robertson et al. 2021). Generation time is approximately 4.5 years, so the assessment period is 13.6 years. As such, extrapolating the decline rate from the NZTCS and the generation time, implies an annual growth rate between 0.916 and 0.950. Fecundity was set at 0.20 recruits per mature individual per year, and survival varied across 0.72, 0.73 and 0.75 to give growth rates of 0.92, 0.93 and 0.95, respectively. Results are presented across all three models as calibrated parameter bands rather than a single central projection (Table 4, Figure 3). Collision losses were applied as additional annual mortality across 0-10 birds per year, and demographic and environmental stochasticity were represented as for the other species, with no additional catastrophe components as per tarapirohe/black-fronted tern or additions as per kakī/black stilt.

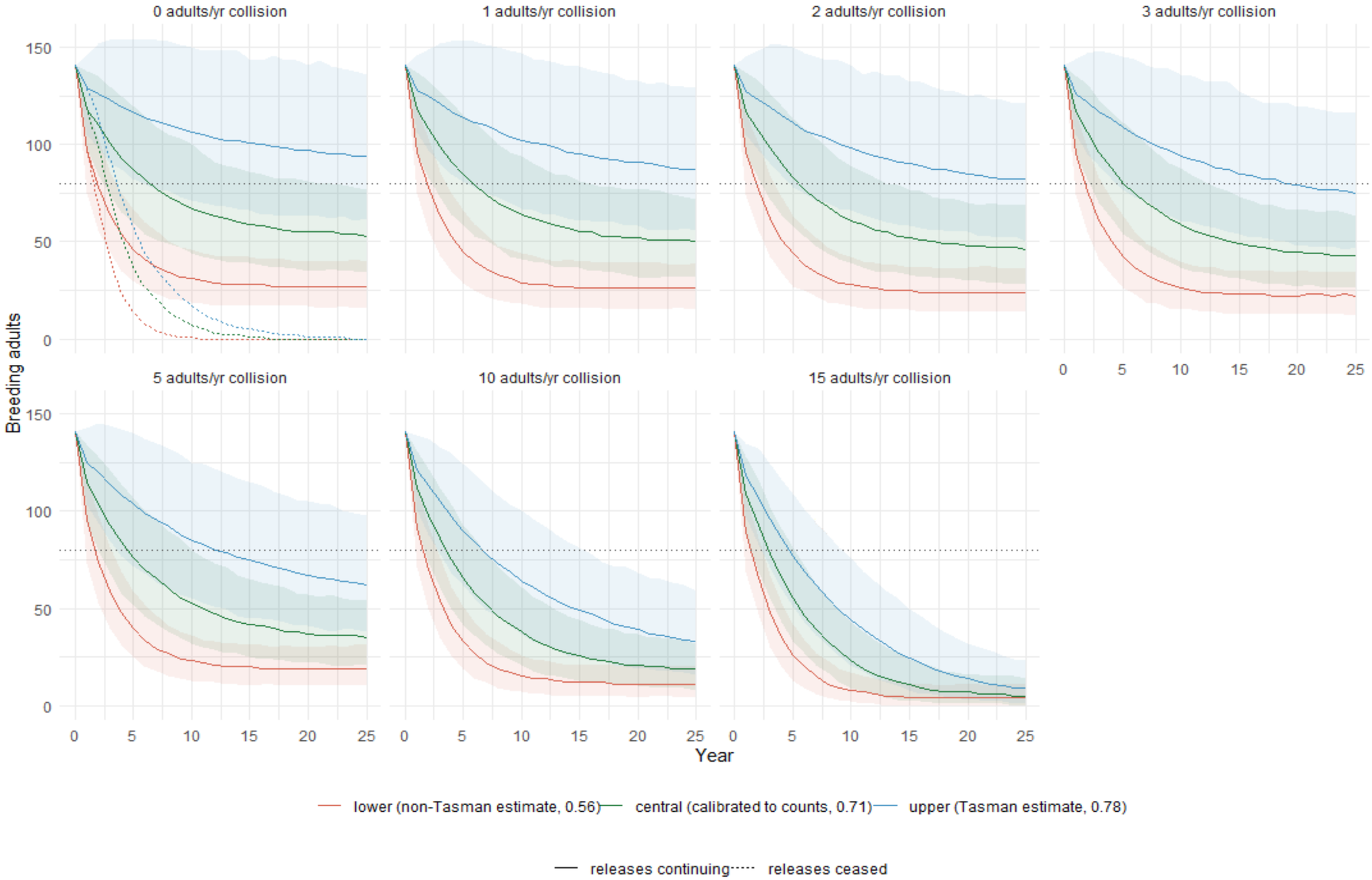

Figure 1. Projected national population of adult kakī/black stilt over 25 years including the breeding program (i.e. releases) and non-breeding program (i.e. no releases), for the zero collision model as well as modelled collisions (between 1 and 15 adult losses per year). Three different adult survivability values were modelled: 0.56 (non-Tasman survivability reported by Bulling, 2009), 0.78 (Tasman survivability reported by Bulling, 2009), and 0.71 a central value between these two extremes. The median and 95% confidence intervals are presented for each scenario. The dashed line represents quasi-extinction, the population at which adult kakī/black stilt could no longer support the breeding programme.

4.2.2 Results and interpretation

Across the calibrated parameter band, and in the absence of any collision mortality, the national matuku-hūrepo/Australasian bittern population is projected to decline from 500 to a median of 70 mature individuals over 25 years, falling below the 250 mature individual threshold in year nine and reaching that threshold with a probability of 0.991 (Table 3, Figure 2). The species is therefore projected to decline below the Nationally Critical population criterion, irrespective of any predicted collisions from the solar farm.

Additional losses to the population speed up this decline rate substantially. An annual loss of one mature individual reduces the projected population by between 12% and 18%. A loss of three by between 34% and 55%, a loss of five by between 55% and 92%, and a loss of 10 by 100% across all growth rate scenarios (Table 3, Figure 2). Based on the outputs of this model, matuku-hūrepo/Australasian bittern populations are already declining, and even very minor additional losses will increase the speed of decline. However, extremely limited data are available on this species, and so the uncertainty of this model is very high.

Table 3 - PVA results for matuku-hūrepo/Australasian bittern across 1-10 bird losses per year after 25-years under three different growth rate scenarios (0.95, 0.93 and 0.92). Probability of quasi-extinction is the probability of the national adult population falling below 250 individuals.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
Upper decline: 50% decline, λ 0.95				
0	120 [35-336]	0	1.00 [1.00-1.00]	0.925
1	105 [25-320]	12.5	0.92 [0.50-1.56]	0.941
2	91 [17-292]	24.16	0.84 [0.43-1.41]	0.955
3	78 [9-283]	34.58	0.75 [0.32-1.29]	0.961
5	54 [0-224]	55	0.59 [0.09-1.07]	0.984
10	0 [0-128]	100	0.18 [0.00-0.68]	0.995
Central decline: 60% decline, λ 0.93				
0	71 [19-198]	0	1.00 [1.00-1.00]	0.992
1	60 [11-183]	15.49	0.93 [0.56-1.51]	0.993
2	48 [3-171]	32.39	0.86 [0.48-1.40]	0.997
3	37 [0-154]	47.88	0.78 [0.39-1.30]	0.998
5	14 [0-118]	80.28	0.64 [0.19-1.12]	0.998
10	0 [0-44]	100	0.28 [0.00-0.73]	1
Lower decline: 70% decline, λ 0.92				
0	52 [12-157]	0	1.00 [1.00-1.00]	0.998
1	43 [6-141]	17.30	0.94 [0.58-1.49]	0.999
2	33 [0-128]	36.53	0.87 [0.54-1.37]	0.999
3	23 [0-115]	55.76	0.81 [0.44-1.28]	0.999
5	4 [0-84]	92.30	0.69 [0.26-1.13]	1
10	0 [0-22]	100	0.38 [0.00-0.80]	1

4.2.3 Model limitations

The limitations for matuku-hūrepo/Australasian bittern are significantly greater than for most other reported species, and these results should be given less comparable weight in the assessment.

The major constraint is that no vital rate governing the population trajectory of this genus has been measured, in New Zealand or internationally. Survival and fecundity were consequently calibrated to reproduce the observed historical decline, which means the model reproduces that decline by construction and offers no independent check on its own parameters. A related consequence is that parameter uncertainty is comparable in magnitude to the effect being assessed. At low collision levels, the separation between the three parameter sets exceeds the difference attributable to collision

mortality, as is apparent in Figure 2, so interpretations resting on small differences between collision scenarios are not supported.

The matuku-hūrepo/Australasian bittern population is modelled at the national level rather than at the site level. Project River Recovery recorded only one individual in the Lower Ōhau River in 2024, none in 2023, and none during the 2008-2010 surveys. No records exist for the Upper Ōhau in any survey (Nelson et al. 2025). There is no site population that can be modelled, and the results therefore describe effects on the national population conditional on birds using the site. The results cannot be converted into site-specific predictions without data on site use, and such data cannot be reliably derived from the available surveys. Matuku-hūrepo/Australasian bittern flight activity occurs throughout 24-hours and peaks before dawn and after sunset, much of it undetected by standard survey methods. Hence, absence from daytime walk-through counts (as per the methods in Nelson et al. 2025) is weak evidence of absence. Establishing site absence would require acoustic surveys of booming males during the breeding season, supported by radio-tracking data.

The starting abundance adopted here predates the October 2024 fire at Whangamarino Wetland, which destroyed much of the habitat at a site supporting approximately 20% of the national matuku-hūrepo/Australasian bittern population, and is likely to be optimistic as a result. The model also omits density dependence and Allee effects, so projections at low abundance should be regarded as optimistic. This consideration is particularly important for a species numbering fewer than 1,000 mature individuals across fragmented wetland habitat, where mate-finding failure is a plausible mechanism at low density. However, because the model was built on multiple different estimation already with no clear available data, adding another layer of estimates for Allee effects was considered unsupportable.

It should also be noted that juvenile matuku-hūrepo/Australasian bittern were not explicitly modelled here. However, they may be especially vulnerable as they are at higher risk of collision with vehicles and other structures. The loss of juveniles would decrease recruitment into the adult population, reducing future reproductive output for this species.

4.3 Kōtuku/white heron

4.3.1 Species specific model methods

Kōtuku/white heron is one of the best-documented species in this assessment. Miller (2001) reports a continuous monitoring record of the Waitangiroto colony extending from 1944, comprising annual counts of nests, chicks and fledglings. As all New Zealand breeding occurs at this single colony, that record describes the entire national breeding population rather than a sample. The species was modelled with a three-stage structure, including measured density dependence and documented catastrophic events. In addition, as the entire population is found in this one managed area, and there are no significant absences in the data, no alternative management scenarios were modelled, nor were any alternative models run across a plausible range of survival values (i.e. as per matuku-hūrepo/ Australasian bittern).

The population was modelled at the national scale across juvenile (0-1 year), subadult (1-3 years) and adult (3-years and older) classes, projected over 25 years, with viability assessed on breeding adults. Starting abundance was set at 100 breeding adults, following Miller's (2001) finding that the colony has been stable at approximately 100 birds and around 50 nests per season since 1984, with juvenile and subadult classes derived from the stable stage distribution of the projection matrix. The quasi-extinction threshold was set at 50 mature individuals, approximately half the current breeding population. Three of the four vital rates were derived from published sources. Fecundity was taken

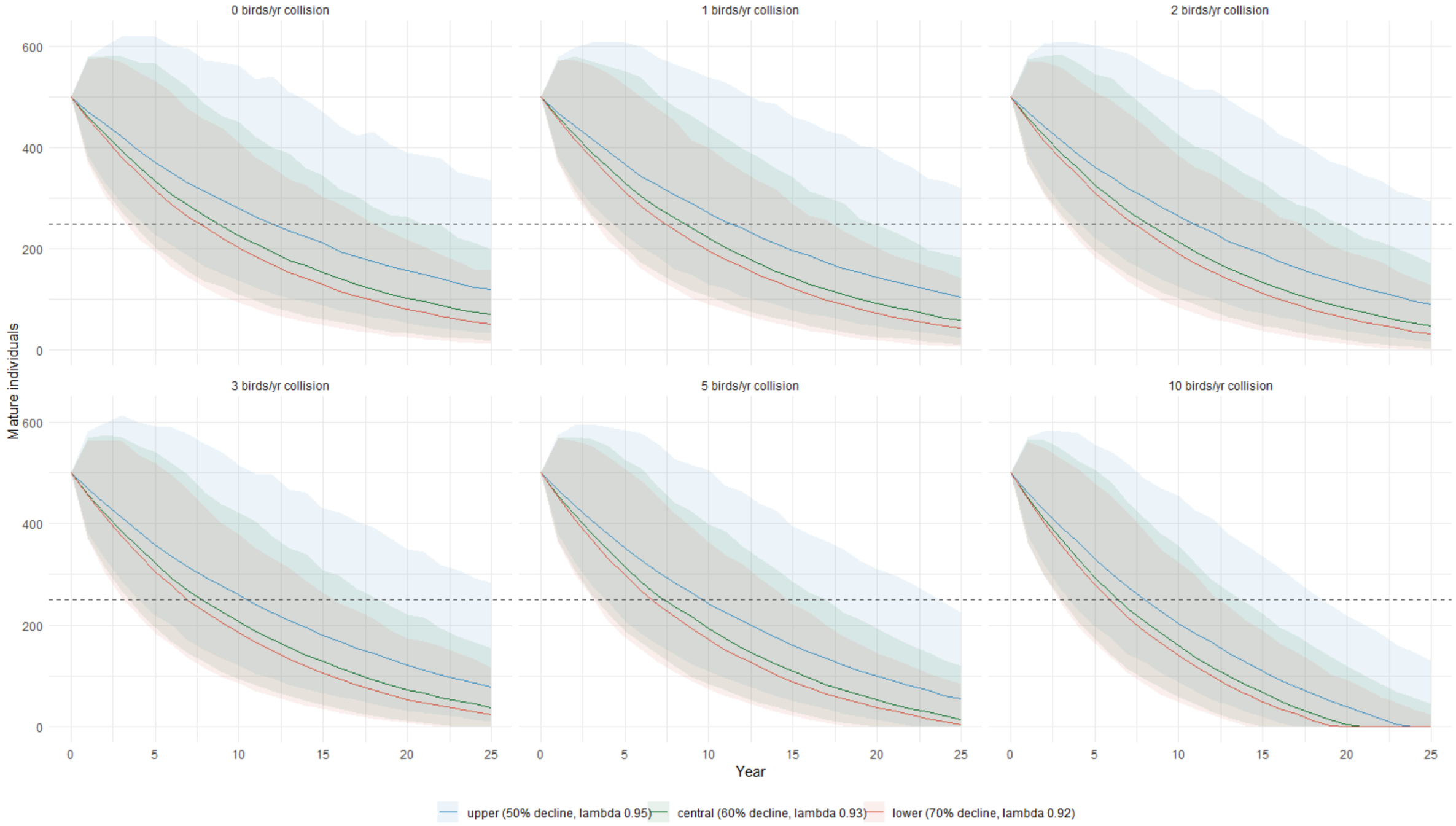

Figure 2. Projected national population of matuku-hūrepo/Australasian bittern nationally over 25 years under six annual collision loss levels (0-10 birds per year), showing the median and 95% interval for each of three adult survivability estimates (λ). The dashed line marks the quasi-extinction level at 250 mature individuals, and the level at which a national population is estimated to collapse with further losses.

directly from Miller (2001), whose mean of 1.1 fledglings per nest across the monitoring record equates to 0.55 recruits per adult per year. The first-year survival of 0.24 was taken from Kahl (1963), who reported 76% first-year mortality in common egrets. This is a non-New Zealand congener and is the weakest parameters. Adult survival was set at 0.905, following Miller's observation that New Zealand mortality must be lower than Kahl's (1963) reported 26% because the colony grew over the monitoring period. The 0.905 value reconciles the measured fecundity with an approximately stable population from preliminary modelling.

Subadult survival has never been estimated. Miller (2001) notes explicitly that no data exist on post-fledging survival or subadult return rates to the colony. However, this is not a free assumption in this model. By fixing the other three rates and the maturation time, and requiring the population growth rate to equal 1.00, consistent with the observed stability recorded since 1984, subadult survival is set at 0.837, a plausible value falling between juvenile and adult survival. The resulting stage structure provides an independent check unavailable for the other species. A stable distribution, of 31% juveniles, 13% subadults, and 56% adults, implies a total population of approximately 178 birds for 100 breeding adults, compared with a published estimate of 150-200 individuals (Adams, 2013).

Negative density dependence was included for kotuku/white heron alone. Miller (2001) found that the mean number of fledglings per nest declines significantly as nest numbers increase, across an observed range of approximately 2.0 fledglings per nest at 20 nests to 0.4 fledglings per nest at 65 nests. This shows that density dependence for this species operates at current population levels rather than at an inferred distant carrying capacity, and its direction is important to the assessment. Reduced adult numbers produce higher per-nest productivity, so density dependence partially compensates for additional mortality. Omitting it would render the model overly pessimistic.

Catastrophic breeding failure was also included. Miller (2001) identifies the timing of westerly and south-westerly storms during October and November as the single most critical determinant of chick survival, and notes that westerly intensity over New Zealand has increased since 1977. Two events with measured impacts were documented. A storm in 1980 caused nest numbers to drop from 38 to 23, and another storm between 19 and 24 November 1984 reduced chick numbers from 100 to 35. These were represented as a 20% annual probability of a storm year reducing fecundity to 45% of its normal value, the approximate mean of the two documented events. The magnitudes of these storm events are measured, but the frequency is assumed. The frequency of storm events may increase over time with climate change, but this was not included in the model.

Collision losses were applied to subadults and adults across a range of annual levels (0-10 birds lost per year). The species is a stronger candidate for collision risk than most in this assessment. Their most common recorded cause of mortality in New Zealand is vehicle strike, with kotuku/white heron flying low and slowly when first becoming airborne (Adams 2013). However, it should be noted that the Waitangiroto colony is far from the project site, and as there are no available flight records for this species, we cannot say whether this species will ever be subject to losses from this solar farm. No records of kotuku/white heron strike at solar farms or wind farms in New Zealand have been detected. However, large-scale solar farms are an emerging land-use and kotuku/white heron strike risk is not well understood.

4.3.2 Results and interpretation

With no additional losses, while including the effects of both catastrophic storm events and density dependence, the projected population after 25 years is a median of 92 breeding adults with a confidence interval of 65 to 119, and a probability of 0.001 of falling below 50 mature individuals (Table 4, Figure 3). The population is therefore projected to remain approximately stable in the absence of collision mortality, consistent with the observed trajectory since 1984. However, one additional annual loss is predicted to transition the population growth from relatively stable to a gradual decline. An annual loss of three mature individuals reduces the projected breeding population by approximately

27% and raises the probability of falling below 50 mature individuals from 0.001 to 0.151 (Table 4). At five losses per year, the population is more than halved, and the probability of falling below the threshold reaches 0.669 (Table 4, Figure 3). At 10 additional losses per year, the population falls below 50 mature individuals after around seven years and is completely lost after 15 years (Figure 3). It is unlikely that the kōtuku/white heron colony at Waitangiroto can absorb more than 2-3 additional annual losses and remain viable. Further, one bird collision per year is predicted to result in a lower population of this species over 25 years.

Table 4 - PVA results for kōtuku/white heron across 1-10 bird losses per year after 25-years. Probability of quasi-extinction is the probability of the adult population falling below 50 total adults.

Adult Losses per Year	Median Adults Remaining After 25 Years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	92 [65-119]	0	1.00 [1.00-1.00]	0.001
1	84 [56-113]	8.6	0.92 [0.68-1.20]	0.007
2	77 [48-105]	16.3	0.84 [0.59-1.12]	0.038
3	67 [38-97]	27.1	0.74 [0.45-1.02]	0.151
5	42 [6-77]	54.3	0.47 [0.07-0.79]	0.669
10	0 [0-0]	100	0.00 [0.00-0.00]	1

4.3.3 Model limitations

Although kōtuku/white heron is one of the best-documented species in this assessment, there are still limitations in the model that should be noted.

Subadult survival has never been measured, and the value adopted was obtained by pre-modelling, in which the model was required to reproduce the observed stability of the colony. The model, therefore, reproduces that stability by construction and cannot independently validate it. Some independent support is nevertheless available. The stage structure implied by the calibrated rates predicts a total population consistent with published counts, providing a check that the calibration was not fitted to the data. First-year survival is likewise taken from a non-New Zealand congener and remains an assumption, and adult survival is derived rather than a measured quantity.

The frequency of storm-driven breeding failure is assumed rather than estimated. The magnitudes of such events are documented, but the monitoring record does not support a frequency estimate. Miller (2001) notes that westerly intensity has increased since 1977, and a frequency derived from the historical record may understate current conditions. The density dependence function is fitted within Miller's observed range of approximately 20 to 65 nests and extrapolates poorly above that range, reaching zero fecundity at approximately 80 nests. Model projections above approximately 130 breeding adults should be treated with caution, although this exceeds any level yet recorded at the colony.

The most significant limitation is not one of model parameterisation. All New Zealand breeding occurs at a single site, and Miller (2001) reports high natal site fidelity. The recorded invasions of Australian birds in 1952 and 1957 failed to establish breeding populations. External recruitment is low at best, and the entire national breeding effort is exposed to any single catastrophic event at one location. The model represents storm-driven breeding failure but not the loss of the colony itself, whether through habitat change, predator incursion, or a severe weather event, and as such is optimistic.

Finally, the modelled population is national rather than site-based. Kōtuku/white herons disperse widely across New Zealand outside the breeding season, and occurrence at the project site is plausible, but no resident population exists to model. Model results, therefore, describe the effect on the national population conditional on birds using the site. Similar to the other species modelled at a national level or in colonies far from the project site, the results should be interpreted as a threshold

against which any subsequent obtained mortality estimate can be compared, rather than as a prediction of effect.

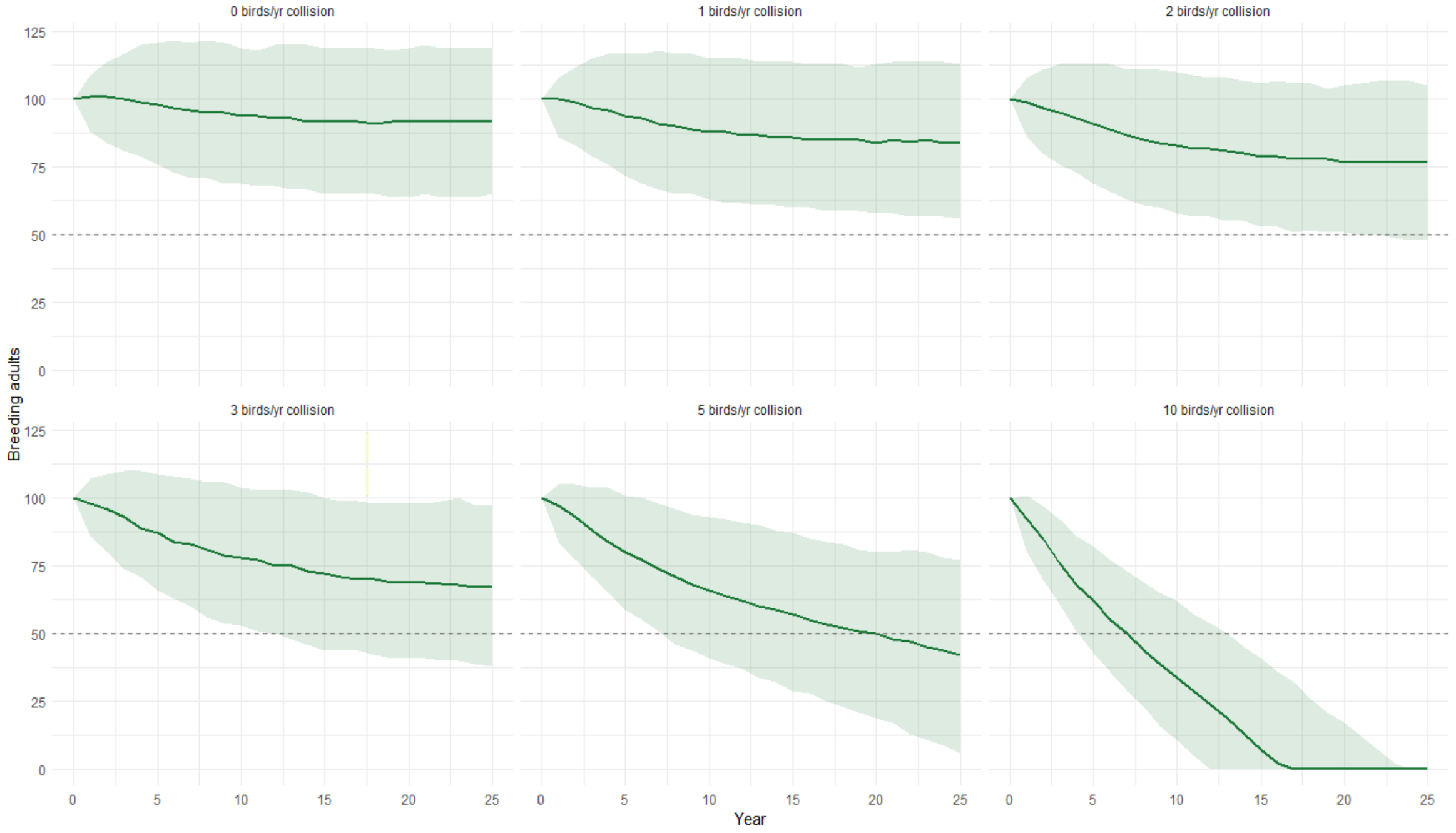

Figure 3. Projected breeding adult population of kōtuku/white heron over 25-years under six annual collision loss levels (0-10 birds per year), showing the median and 95% confidence interval. The model includes measured density dependence and documented storm-driven breeding failure (Miller 2001). The dashed line marks the quasi-extinction level of 50 mature individuals, a population level higher than recorded in the past but set at a conservative level.

4.4 Tarapirohe/black-fronted tern

4.4.1 Species specific model methods

Tarapirohe/black-fronted tern was modelled at the scale of the Upper Ōhau River colony ("Tern Island"). This colony was selected as the demographic unit at risk as it is the nearest breeding population to the project footprint and a plausible source of any birds using the site. Starting abundance was estimated at 718 individuals, taken from Project River Recovery's 2025 monitoring, which recorded a peak of 359 active nests on 23 October 2024 (Nelson et al. 2025). Since colony size is estimated by doubling the maximum active nest count, the 718 birds represent breeding adults. Pre-adult age classes were then derived from the stable stage structure of the projection matrix. Adult survival was set to 0.88, following Keedwell (2005), which remains the only mark-resight estimate for the species and was also derived from the Ōhau River. Productivity was modelled under two regimes: with the predator control that Project River Recovery has maintained at this colony since 2009 (i.e. managed) and without it (i.e. unmanaged). Fecundity under management was calibrated against the observed increase in managed populations within the basin, where tarapirohe/black-fronted tern counts on the Tasman River have risen from approximately 137 birds (2004–2014 mean) to 539–598 birds (2022–2023) (Nelson et al. 2025). The unmanaged rate was set to reproduce the 5.5–15.8% annual declines observed on unmanaged rivers nationwide (O'Donnell & Hoare 2011). Unlike kakī/black stilt, no augmentation term was applied, as this species is not subject to captive rearing or release.

Two distinct forms of catastrophic breeding failure were modelled separately, reflecting the causes recorded at this colony. Predator incursion produces total failure and may be localised to a single colony. Four consecutive seasons of zero fledging occurred within the 21-year monitoring record (Nelson et al. 2025), and these coincided with the most productive seasons recorded on the Tasman River, indicating that such events are not synchronised between colonies. Severe weather produces partial failures but operates at a regional scale. The snowfall on 26 October 2024 buried all nests on Tern Island, accounting for 70% of nest failures that season, and simultaneously destroyed 11 of the 24 colonies monitored on the Tasman River (Nelson et al. 2025). The distinction is important, as localised failures may be offset by movement between colonies, whereas synchronised failures cannot. Collision losses were then applied to the flying age classes across a range of annual mortality levels, and results are reported as the relationship between annual loss and population outcome.

4.4.2 Results and interpretation

Under continuing predator control and in the absence of collision mortality, the Upper Ōhau River colony is projected to remain broadly stable to slowly increasing from 718 to a median of 917 mature individuals over 25 years, with a probability of 0.002 of falling below the threshold of 100 mature individuals (Figure 4, Table 5). Without predator control, the same colony declines to a median of 83 mature individuals, with a probability of 0.708 of falling below the threshold even when no collision mortality is applied (Figure 4). Under continued management, an annual loss of five birds reduces the projected colony by approximately 13%, rising to approximately 24% at 10 birds per year, 47% at 20, and 68% at 30 birds per year (Figure 4, Table 5). The probability of the colony falling below 100 adult individuals rises from 0.002 in the absence of collision mortality to 0.024 at 10 birds per year, 0.054 at 15, and 0.105 at 20 birds per year (Figure 4, Table 5).

In the absence of mammalian predator control (i.e. the unmanaged scenario), the colony is already declining steeply. Any additional mortality accelerates that decline substantially. An annual loss of five birds halves the projected colony and raises the probability of falling below 100 mature individuals to 0.973, while at 10 birds per year, the colony is effectively lost within 25 years (Figure 4). This scenario should be interpreted as an indication that the colony's persistence is highly dependent on continued

predator control, and that collision mortality and any future reduction in the predator control programme would greatly compound on each other rather than act independently.

Table 5 - PVA results for tarapirohe/black-fronted tern across 2-35 bird losses per year under active mammalian predator control after 25-years. The model also includes the effects of catastrophic events (such as predator incursions and severe snowstorms). Probability of quasi-extinction is the probability of the adult population falling below 100 total, or 50 breeding pairs.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	917 [227-2582]	0	1.00 [1.00-1.00]	0.002
2	870 [206-2516]	5.1%	0.95 [0.76-1.14]	0.002
5	796 [165-2417]	13.2%	0.88 [0.66-1.06]	0.006
10	700 [101-2224]	23.6%	0.77 [0.42-0.96]	0.024
15	598 [53-2084]	34.7%	0.65 [0.21-0.86]	0.054
20	489 [0-1850]	46.7%	0.54 [0.00-0.78]	0.105
25	384 [0-1673]	58.1%	0.43 [0.00-0.70]	0.186
30	289 [0-1461]	68.5%	0.31 [0.00-0.62]	0.287
35	180 [0-1290]	80.4%	0.20 [0.00-0.55]	0.4

4.4.3 Model limitations

There is one main limitation to the modelling approach taken for tarapirohe/black-fronted tern. First, the project footprint lies on the Lower Ōhau River, downstream of Lake Ruataniwha, while the modelled colony is located on the Upper Ōhau River upstream of the lake. Walk-through surveys of the Lower Ōhau recorded 15 birds in 2024 and nine in 2023, compared with 40-108 birds in the 2008-2010 surveys (Nelson et al. 2025). This reach therefore does not support a population of sufficient size to model as a discrete demographic unit, and any assessment at this scale would be dominated by sampling variation rather than population processes.

4.5 Pūteketeke/Australasian crested grebe

4.5.1 Species specific model methods

Pūteketeke/Australasian crested grebe is one of the only species for which presence at the project location is demonstrated rather than inferred. Nine pūteketeke/Australasian crested grebe were counted on Lake Ruataniwha in February 2024 during a systematic count (Miskelly 2024), and the species is recorded on Lake Benmore and other Waitaki hydro lakes. The project footprint lies between two occupied waterbodies. Sagar (1982) documents post-breeding movements from lake to lake within lake groups, including approximately 100 kilometre movements to Lake Alexandrina, and concludes that all lakes within a group are important because birds redistribute among them. Hydroelectric development is already recognised among the threats to this species (O'Donnell 2013). Accordingly, the collision loss range was extended to 50 birds per year for this species, despite the low population size, because risk exposure is better documented here.

The species was modelled as a single unstructured population of mature individuals at a national scale over 25 years. Starting abundance was set at 630 mature individuals. The January 2024 census recorded a minimum of 1,047 birds (Miskelly 2024). As the assessment is based on mature individuals for consistency with the other species, an adult fraction was estimated. The stable age distribution under the calibrated rates gives 51-63% adults across a plausible range of first-year survival, and 60% was adopted. This is appropriate to a January count, taken mid-breeding season with young of the year present. The earlier national survey of Sagar (1981), which counted 170 adults out of approximately

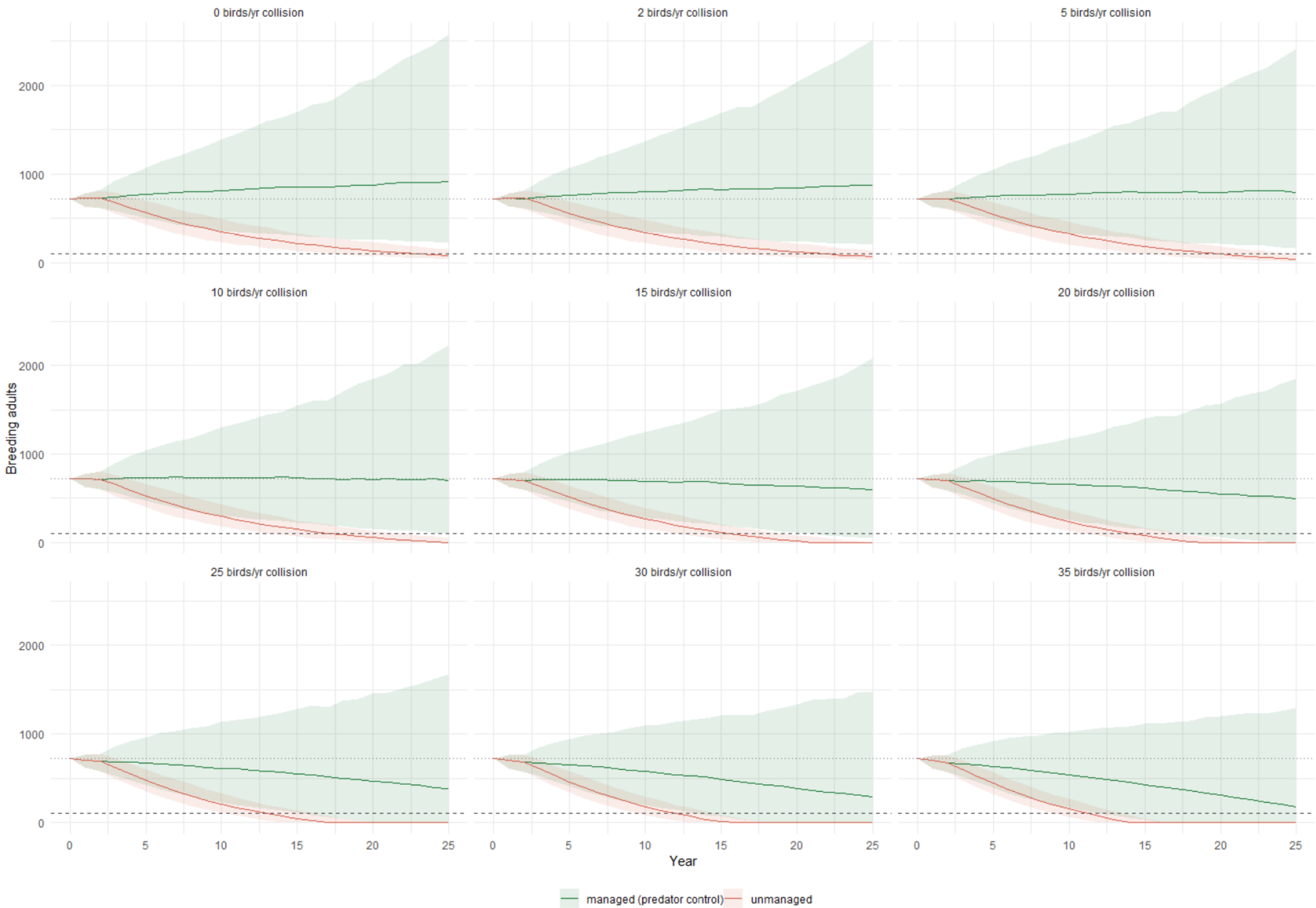

Figure 4. Projected breeding adult population of tarapirohe/black-fronted tern over 25-years under nine annual collision loss levels (0-35 birds per year) for both managed and unmanaged scenarios, showing the median and 95% confidence interval. The dotted line shows the population starting point and the dashed line represents quasi-extinction, the population at which adult tarapirohe/black-fronted tern numbers are estimated to no longer support a viable colony at Upper Ōhau.

200, suggests a higher adult fraction but was conducted in November and December, before chicks had fledged. If the 2024 census counted adults alone, the starting abundance would be 1,047, and the model presented here would be conservative. Such an unstructured model is a simplification for this species rather than an exact representation. Maturity occurs at approximately two years, indicating a maturation lag that the model does not represent and making it mildly optimistic. A three-stage model was considered and rejected as both first-year and subadult survival could not be estimated.

The only measured estimate for the species adult survival is 0.75, from Abt and Konter's (2009) analysis of 57-years of European ring recoveries for the European great crested grebe. The authors identify two features of that dataset that make it likely to underestimate the New Zealand context. Recoveries were biased toward young birds, evidenced by an untypical increase in apparent survival after age three, and 45% of great crested grebe recoveries were attributable to anthropogenic causes, principally drowning in fishing nets and hunting, neither of which is a significant mortality source for New Zealand grebes. Based on these findings, adult survival was set at 0.85. This value also reconciles the model with the observed historic population trajectory. Fecundity was estimated at 0.20 recruits per mature individual per year. No New Zealand estimate exists, and the parameter was held fixed rather than calibrated on the grounds that low fecundity is biologically plausible for this species, whereas the survival estimate had documented reasons to be adjusted The growth rate of 1.05 obtained from preliminary modelling and shown in Figure 5 (0 bird collions/year) is consistent with the observed national increase from approximately 400 birds in 2004 to approximately 600 in 2012 (+5.2% per year; O'Donnell 2013), and from 701 in 2014 to 1,047 in 2024 (+4.1% per year). The viability threshold for quasi-extinction was set at 250 mature individuals, the NZTCS population criterion associated with Nationally Critical status, as adopted for matuku-hūrepo/Australasian bittern.

4.5.2 Results and interpretation

In the absence of additional mortality, the national population is projected to increase from 630 to a median of 1,812 mature individuals over 25-years, with an annual rate of increase of approximately 4.3%, consistent with the observed trend. Pūteketeke/Australasian crested grebe is one of the only species assessed with a growing baseline. As such, any collision mortality would suppress population recovery rather than accelerate population decline. An annual loss of five mature individuals reduces the projected population by approximately 13%, and a loss of 10 by approximately 25% (Table 6, Figure 5). In both cases, the population is still predicted to increase, but attains only 87% and 75%, respectively, of the size it would otherwise have reached (Table 6, Figure 5). The population stops increasing at an annual loss of approximately 25 mature individuals. Below this level, the national increase continues at a reduced rate, while above it the population declines. At an annual loss of 50 pūteketeke/Australasian crested grebe, the population is predicted to be lost within 10-15 years (Table 6, Figure 5).

As with tarapirohe/black-fronted tern, the confidence intervals for the projected population size are wide, ranging from 384 to 4,139 mature individuals at losses of 10 birds per year (Table 6, Figure 5). This is because a growing population compounds its differences in the model over 25-years. These intervals describe uncertainty about the future trajectory, not about the effect of a collision, and their overlap between scenarios should not be interpreted as indicating an absence of effect. The ratio of losses relative to the baseline isolates this effect. At 10 birds per year, the population reaches 75% (CI = 0.48-0.98) of its no-collision size (Table 6) with an upper confidence interval that does not reach one (i.e. no different from the no collision scenario).

Table 6 - PVA results for pūteketeke/Australasian crested grebe across 1-50 bird losses per year after 25-years. Probability of quasi-extinction is the probability of the adult population falling below 250 total adults.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	1812 [671-5082]	0	1.00 [1.00-1.00]	0
1	1778 [603-5002]	1.8	0.97 [0.76-1.24]	0
2	1710 [606-4941]	5.6	0.95 [0.74-1.21]	0
3	1688 [564-4788]	6.8	0.92 [0.71-1.18]	0.001
5	1582 [504-4722]	12.6	0.87 [0.65-1.11]	0.002
10	1356 [384-4139]	25.1	0.75 [0.48-0.98]	0.007
20	890 [99-3403]	50.8	0.49 [0.12-0.76]	0.098
30	431 [0-2480]	76.2	0.24 [0.00-0.56]	0.365
50	0 [0-973]	100	0.00 [0.00-0.22]	0.905

4.5.3 Model limitations

The starting population is based on a census undertaken by the Wānaka Grebe Project, with an estimated 1,047 birds and an earlier count of 701 in 2014, reported by the community conservation organisation on their website. Unfortunately, it does not appear that a primary census report was produced. This would have clarified whether the total count included all birds or only adults, a question that determines whether the starting abundance is 630 (as modelled) or if it should be updated to 1,047 mature individuals. Adult survival was adjusted from the only measured value to reproduce the observed growth rate, so the model reproduces that growth rate by construction and offers no independent check on its own parameters. Fecundity has no New Zealand source and was set to an assumed value. The model form without an age structure omits the two-year maturation lag, so it does not account for the delay between the loss of adults and the resulting shortfall in recruitment, making projections mildly optimistic. Density dependence is likewise omitted. For a population projected to approach 1,800 mature individuals from a base of 630, some density-related constraint on growth is expected, and its absence makes the higher baseline projections optimistic. Finally, as with other species assessed, the modelled population is national in scale. Unlike matuku-hūrepo/Australasian bittern and kōtuku/white heron, a local population demonstrably exists. Still, nine birds on Lake Ruataniwha is too small a number to model as a discrete demographic unit, and the proportion of the national population flying over the project is unknown. Results describe the effect on the national population conditional on the modelled losses occurring. At the higher loss levels tested, those losses would necessarily draw on birds from the wider Waitaki lakes group rather than from Lake Ruataniwha alone. Sagar's (1982) movement data support this as plausible, but it has not been quantified for this location.

4.6 Māpunga/black shag

4.6.1 Species specific model methods

Māpunga/black shag was modelled as a three-stage population with juveniles, subadults and adults at the national scale over 25-years, with viability assessed for mature individuals. The three-stage structure is supported by the fact that age at first breeding is three years, so the subadult class spans years one to three. Starting abundance was set at 7,500 mature individuals, with juvenile and subadult classes derived from the stable stage structure at 5,408 and 2,721, respectively, giving a total modelled population of 15,629 birds. Published estimates place the national population at 5,000-10,000 mature individuals (Powlesland 2025; Heather and Robertson 2005), supported by the current threat classification where Robertson et al. (2021) assess the taxon as At Risk – Relict, which requires 5,000-20,000 mature individuals with a stable population and the midpoint of 7,500 was adopted.

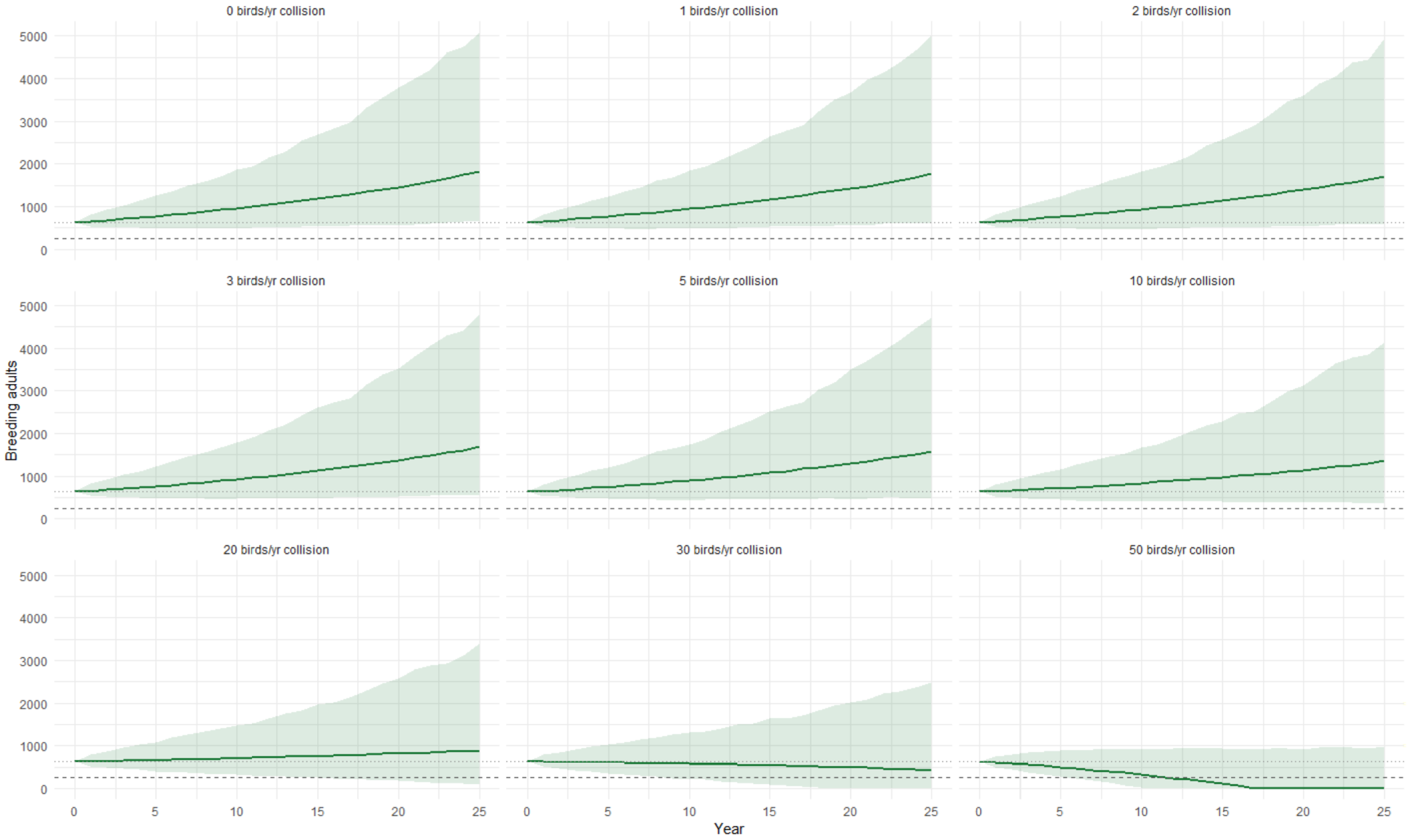

Figure 5 - Projected breeding adult population of pūteketeke/Australasian crested grebe over 25-years under nine annual collision loss levels (0-50 birds per year), showing the median and 95% confidence interval. The dotted lines show the starting point for the population, and the dashed line marks the quasi-extinction level of 250 mature individuals.

The viability threshold was set at 5,000 mature individuals, which is a defined regulatory boundary. At 7,500 mature individuals, the taxon sits in the lower part of the 5,000-20,000 band, and a fall below 5,000 moves it into the 1,000-5,000 band. Within that band, a decline of 10-50% over three generations meets the criteria for Threatened – Nationally Vulnerable, and crossing the threshold represents a defined deterioration in threat classification. It should nonetheless be noted that this is not a quasi-extinction threshold in the conventional sense. At this population size and under the parameters adopted, collision losses at levels plausibly arising from a single site would not be expected to reduce the population below 5,000 within 25-years. The threshold is more appropriately read as a level at which concern should arise.

The decline rate is a more sensitive criterion for this species, and should be reported alongside the threshold. Māpunga/black shag is currently classified as At Risk rather than Threatened, so entry into a Threatened category is triggered by rate of decline rather than by population size. Within the 5,000-20,000 band, a decline of 10-30% over three generations meets the criteria for At Risk – Declining, and a decline of 30-70% meets the criteria for Threatened – Nationally Vulnerable. Generation time for this species is approximately 11.1 years, so the assessment window is 33 years. The 25-year projections reported here, therefore, understate the decline over the full assessment period and are conservative relative to this criterion.

Fecundity was taken directly from Powlesland and Reese (1999), who monitored 185 clutches at Lake Kohangatera near Wellington and recorded a mean of 1.44 fledged young per brood across all clutches, equating to 0.72 recruits per adult per year. Māpunga/black shags rear a single brood each season, and the approximately five-month breeding cycle, combined with the moult, makes two broods unlikely. Adult survival was set at 0.89 and first-year survival at 0.35, both derived from Wernham and Peach's (1999) analysis of British and Irish ring recoveries for great cormorant (*Phalacrocorax carbo*). These are non-New Zealand estimates for the same species but a different subspecies, and represent the only available survival data. The value of 0.89 is the Northern England and Wales estimate rather than the Scottish figure of 0.79, on the basis that Scottish cormorants are subject to heavier persecution at fisheries than is likely in New Zealand.

Subadult survival has never been estimated, and as for kōtuku/white heron, it is not a free assumption in this model. Holding the other three rates and the maturation schedule fixed and requiring a population growth rate of 1.00 determines subadult survival at 0.61, a value falling plausibly between juvenile and adult survival. The calibration target of stability is supported in two directions. Powlesland and Reese (1999) assessed their study population as at or near the carrying capacity of its habitat, and the Relict criterion under which the taxon is classified requires a population that is stable within ±10%.

4.6.2 Results and interpretation

In the absence of collision mortality, the national population remains approximately stable, projecting a slight increase from 7,500 to a median of 7,522 mature individuals over 25-years with a probability of 0.05 of falling below the threshold of 5,000 (Table 7, Figure 6). Māpunga/black shag is less sensitive to collision mortality than most other species assessed. An annual loss of 10 birds reduces the projected population by approximately 2%, against reductions of approximately 25% for pūteketeke/crested grebe at the same loss level. This reflects the relative differences in population size between these species and the associated differences in demographic resilience. It should be noted that the paired intervals for this species (for example, 0.96 [0.90–1.01] at 20 birds per year) are considerably tighter than for several other species. As the population is stable rather than growing, the projected populations across multiple simulations are less variable, and the paired comparison consequently resolves smaller effects.

With the parameters set, no reasonable collision losses are likely to significantly affect the population. However, an annual loss of one bird is predicted to result in a reduction in the population growth rate by 1%. An annual loss of more than 50 birds produces an 11.0% decline over 25 years and so would

meet the criterion for Threatened – Nationally Vulnerable even without a population below the quasi-extinction threshold. Higher losses increase this margin with a 23.2% decline at 100 adult losses per year, but the projected medians remain below the 50% threshold at which Nationally Endangered would be triggered. The loss level at which a change in threat classification becomes likely due to the change in decline rate is therefore substantially lower than that at which the quasi-extinction threshold is approached.

Table 7 - PVA results for māpunga/black shag across 1-100 bird losses per year after 25-years. Probability of quasi-extinction is the probability of the adult population falling below 5,000 total adults, the threshold at which the species transitions to a Threatened threat classification.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	7522 [4711-11351]	0	1.00 [1.00-1.00]	0.049
1	7512 [4667-11322]	0.14	1.00 [0.94-1.06]	0.05
2	7478 [4659-11364]	0.59	0.99 [0.94-1.06]	0.051
5	7442 [4664-11272]	1.07	0.99 [0.93-1.05]	0.054
10	7358 [4564-11211]	2.19	0.98 [0.92-1.03]	0.062
20	7188 [4394-10879]	4.44	0.96 [0.90-1.01]	0.076
50	6658 [4020-10306]	11.48	0.89 [0.82-0.94]	0.13
100	5788 [3362-9265]	23.06	0.77 [0.69-0.84]	0.28

4.6.3 Model limitations

Both survival estimates are derived from British populations of the same species, but a different subspecies, and neither has been measured in New Zealand. Survival may differ substantially between these two geographic regions. Subadult survival was obtained by requiring the model to reproduce an approximately stable population, so the model reproduces that stability by construction and provides no independent check on its own parameters. Unlike kōtuku/white heron, no independent validation of the resulting stage structure was available, as no published estimate of the age composition of the New Zealand population exists. Density-dependence was not represented. This warrants an explicit note for māpunga/black shag, as Powlesland and Reese (1999) describe their study population as at or close to the carrying capacity of its habitat, suggesting that some density-related constraint operates. Its omission makes the higher projections appear more optimistic, although it does not affect the proportional collision-loss effects.

Two further limitations concern the applicability of the results. The modelled population is national. Project River Recovery recorded two māpunga/black shags in the Lower Ōhau and two in the Upper Ōhau in 2024 (Nelson et al. 2025), approximately four birds or 0.05% of the modelled adult population. As such, there is no site population that can be modelled, and results describe effects on the national population conditional on birds using the site. Additionally, the evidence that this species is prone to collision is weaker than for others assessed. Recognised anthropogenic mortality for māpunga/black shag comprises capture in set-nets and craypots and occasional shooting, rather than aerial collision. Birds do commute between roosting and foraging waters, and flights over land occur. Still, there is no documented collision-mortality pathway comparable to the vehicle-strike record for kotuku/white heron or the inter-lake movement documented for pūteketeke/Australasian crested grebe.

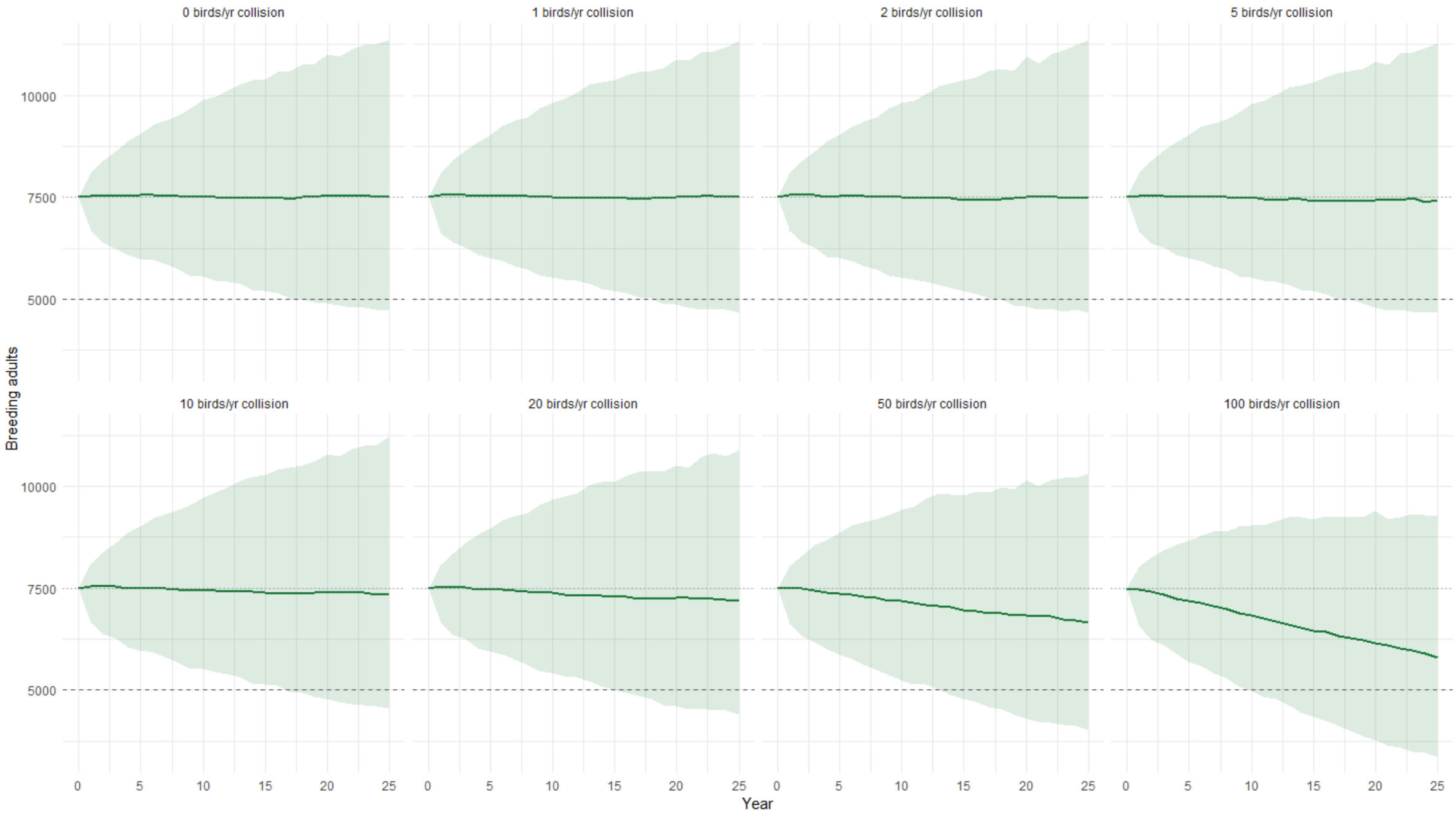

Figure 6. Projected breeding adult population of māpunga/black shag over 25-years under eight annual collision loss levels (0-100 birds per year), showing the median and 95% confidence interval. The dotted lines show the starting point for the population, and the dashed line marks the quasi-extinction level of 5,000 mature individuals (the threshold for which the species would transition from At – Risk to Threatened).

4.7 Pohowera/banded dotterel

4.7.1 Species specific model methods

Pohowera/banded dotterel is among the best-parametrised species in this assessment, being one of the few for which directly measured rates of population change are available from more than one source. O'Donnell and Monks (2020) analysed counts from 119 braided and shingle river reaches spanning 1962 to 2017 and reported a weighted mean rate of change of –3.7% per year across 33 South Island rivers, equating to a decline of 52.3% over 20 years, or approximately three generations. National wader counts covering 1983 to 2019 (Riegen & Sagar 2020) subsequently indicated a slower rate of decline. Based on the latter, the 2021 threat classification panel improved the taxon's status from Threatened – Nationally Vulnerable to At Risk – Declining (Robertson et al. 2021). The model has, therefore, been calibrated to the current threat classification. Under the At Risk – Declining threat category, pohowera/banded dotterel is assessed as having 5,000-20,000 mature individuals undergoing a decline of 10-30% over three generations. This implies an annual growth rate of between 0.982 and 0.995, and a value of 0.99 was adopted.

The species was modelled as a single unstructured population of mature individuals over 25-years. An unstructured model is correct for pohowera/banded dotterel rather than a simplification. Age at first breeding is typically one year (Pierce 2013), and so no subadult class can be modelled similar to matuku-hūrepo/Australasian bittern. Starting abundance was set at 12,730 mature individuals, being the sum of the most recent counts across the 119 river reaches surveyed by O'Donnell and Monks (2020). This population was adopted in preference to the wider national estimate of approximately 50,000 birds, as it is the population on which the trend data were collected, and because braided river birds are more likely to constitute the population at risk in this location. The national figure includes coastal, estuarine, and farmland birds whose dynamics differ from those of riverbed breeders and for which no matching trend estimate exists. River counts are made during the breeding season and comprise predominantly territorial adults. The estimate of 12,730 is consistent with the NZTCS population band of 5,000-20,000 mature individuals, which provides an independent check on the starting abundance.

Adult survival has never been measured for this species. The braided river vital rates synthesis records that no information at all was available on adult survival of pohowera/banded dotterel. Adult survival was therefore derived by two independent routes that converge. O'Donnell and Monks (2020) describe the 20-year assessment window as approximately three generations, implying a generation time of about 6.7 years. With an age at first breeding of one year and the standard relationship between generation time and adult survival, this gives 0.851. Independently, a typical lifespan of six to seven years implies 0.833 to 0.85. As such, a value of 0.85 was adopted. Fecundity was then calibrated to reproduce the growth rate of 0.99, giving 0.14 recruits per mature individual per year. This is consistent with the available breeding data, with a clutch of three, hatching success of 56-68% and fledging success of 48% (Keedwell, 2002) giving 0.81 to 0.98 fledglings per nest, or 0.40 to 0.49 per adult, requiring survival to first breeding of approximately 29-35%, a plausible value for a small plover.

The viability threshold was set at 5,000 mature individuals, as the species currently sits in the 5,000-20,000 mature individual band, within which a decline of 10-30% over three generations gives its present classification of At Risk – Declining. A fall below 5,000 mature individuals moves it into the 1,000-5,000 band, within which the same rate of decline meets the criteria for Threatened – Nationally Vulnerable. Crossing the threshold, therefore, represents a defined deterioration in threat classification.

4.7.2 Results and interpretation

The modelled population declines under all scenarios, including the absence of any additional collision mortality, as the model reproduces the measured national trend (Table 8, Figure 7). Under the no-

collision scenario the median population at 25-years is 9,295, and the probability of falling below 5,000 mature individuals is 0.05 (Table 8), meaning a 5% probability of 'quasi-extinction'. At additional losses of one to two birds per year, the ratio relative to the baseline decline rate does not statistically change, although at five losses per year the reduction in the decline rate does change by nearly 1%.

An annual loss of 10 birds reduces the projected population relative to the baseline rate of decline by approximately 2%, and 50 birds by approximately 12% (Table 8, Figure 7). An annual loss of 50 birds raises the probability of the species meeting the criteria for Threatened – Nationally Endangered within 25-years from 0.05 to 0.1, and 100 birds raises it to 0.21 (Table 8).

Table 8 - PVA results for pohowera/banded dotterel across 1-100 bird losses per year after 25-years. Probability of quasi-extinction is the probability of the adult population falling below 5,000 total adults, the threshold for which the species would transition from At Risk to a Threatened status.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	9295 [4508-18062]	0	1.00 [1.00-1.00]	0.049
1	9281 [4510-18084]	0.15	1.00 [0.94-1.06]	0.047
2	9272 [4442-18103]	0.24	1.00 [0.94-1.06]	0.046
5	9210 [4475-17960]	0.92	0.99 [0.92-1.05]	0.051
10	9108 [4332-17702]	2.01	0.98 [0.92-1.03]	0.054
20	8878 [4212-17413]	4.48	0.95 [0.89-1.02]	0.06
50	8194 [3802-16582]	11.84	0.88 [0.81-0.94]	0.102
100	7147 [2929-15048]	23.10	0.77 [0.64-0.86]	0.217

4.7.3 Model limitations

Adult survival has not been measured for this species anywhere in its range, and the value used was derived from generation time and typical lifespan rather than from demographic data. Fecundity was then calibrated to reproduce the observed decline, so the model reproduces that decline by construction and offers no independent validation of its own parameters. Two features nevertheless provide some support. The two independent derivations of adult survival agree closely, and the calibrated fecundity is consistent with the measured breeding parameters of Keedwell (2002) under a plausible assumption about first-year survival.

The population modelled comprises braided and shingle river birds rather than the national population across all habitats. This was a deliberate choice so that the population and the calibration trend describe the same birds. However, the results do not describe effects on the national population, which is approximately four times larger and includes coastal and farmland breeders. Given the project's location near several braided rivers, the braided-river population is considered more relevant.

The model omits density dependence and catastrophic events. Flood and predation events are known to affect nesting success on braided rivers (Rebergen et al. 1998; Sanders & Maloney 2002), but no frequency data adequate to parameterise a catastrophe term were located. Their omission makes the projections optimistic to an unquantified degree.

Pohowera/banded dotterel were observed during site surveys and are present in the vicinity of the site. Project River Recovery recorded 10 birds in the Lower Ōhau River and 13 in the Upper Ōhau in 2024, with 518 on the Tasman River (Nelson et al. 2025). This is a stronger local presence than for matuku-hūrepo/Australasian bittern, kōtuku/white heron or māpunga/black shag. It is notable that

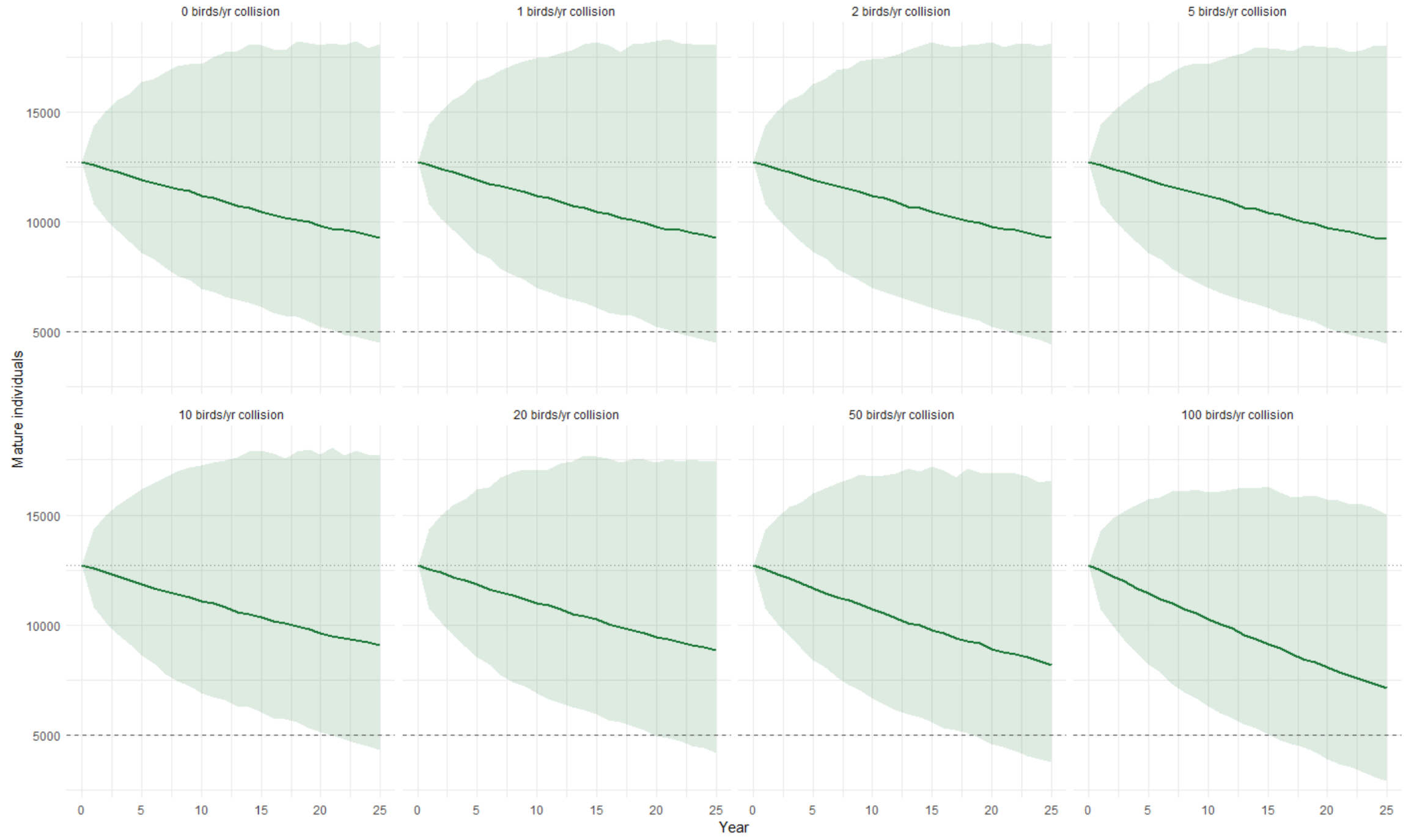

Figure 7. Projected breeding adult population of pohowera/banded dotterel over 25-years under eight annual collision loss levels (0-100 birds per year), showing the median and 95% confidence interval. The dotted lines show the starting point for the population, and the dashed line marks the quasi-extinction level of 5000 mature individuals (the threshold at which the species would transition to Threatened).

the Lower Ōhau count is well below the 51 to 91 birds recorded in the 2008-2010 surveys, consistent with the national decline.

4.8 Tarāpuka/black-billed gull

4.8.1 Species specific model methods

The tarāpuka/black-billed gull was modelled as a single unstructured population of mature individuals at the scale of the Canterbury region, projected over 25-years. Two calibrations were run, and both are reported. The reasons for this are set out below and are critical to the interpretation of the results.

The species was modelled at regional rather than national scale. The project lies within the Waitaki basin in Canterbury, and losses of the order of a few hundred birds per year are not meaningfully assessed against a national population of approximately 120,500 mature individuals (Mischler, 2018a). Canterbury is around the finest scale at which this species behaves as a demographic unit. Mischler (2018a) recorded a mean annual variability of 42.2% in nest counts within regions between consecutive seasons, and colonies relocate between rivers and between years (Mischler, 2018b). Scoping further, to the upper Waitaki rivers alone (Waitaki, Ahuriri and Hopkins, together 4,091 nests or 8,182 mature individuals in 2016/17), would mean that a single relocating colony could obscure any collision signal entirely.

Starting abundance was set at 41,350 mature individuals, based on 20,675 Canterbury nests recorded in Mischler's (2018a) 2016/17 national census, assuming two adults per nest. That census was the first to achieve full coverage of New Zealand and remains the most reliable estimate available. It should be noted that Mischler's counts are of nests rather than birds. The equivalent figure for Canterbury and Otago combined would be 23,435 nests, or 46,870 mature individuals, which is an equally defensible scope if birds are considered to move between the two regions. Canterbury alone was adopted as the more conservative option, as it is the project region. Adult survival was set at 0.86, derived from Mills et al. (2008), who found that 86% of 12,792 re-sighted female tarāpunga/red-billed gulls (*Chroicocephalus novaehollandiae scopulinus*, At Risk – Declining) were re-sighted a second time. This is a re-sighting rate which approximates, but is not identical to, apparent survival, and it is drawn from a congeneric species. No estimate exists for tarāpuka/black-billed gull. Fecundity was varied to set the population growth rate rather than fixed independently, because the division between the two rates is not separately known. The resulting values of 0.13 and 0.16 recruits per mature individual per year are consistent with the available breeding data. A clutch of 1.85 with a mean breeding success of 32% (McClellan, 2009) gives approximately 0.59 fledglings per nest, or 0.30 per adult, requiring survival to first breeding at age three of approximately 45-55%. Due to a lack of population or survivability data for chicks and fledglings, an unstructured model was adopted. Age at first breeding is approximately three years, so a maturation lag exists which the model does not represent. This results in the model projections being mildly optimistic.

The national threat classification and the regional population data point in opposite directions, and this assessment models both rather than selecting between them. Robertson et al. (2021) classify the species as At Risk – Declining (more than 100,000 mature individuals with a decline of 10-70% forecast over three generations). Generation time is approximately 9.1 years, so three generations is 27 years, and the threat ranking implies an annual growth rate of between 0.957 and 0.996. The Canterbury data, however, show the population increasing. Mischler's (2018a) regional series records 12,719 nests in 1995/96, 13,589 in 1996/97, 18,801 in 2015/16 and 20,675 in 2016/17. This is an increase of 2.12% per year over the twenty years to 2016/17, the strongest of any region. Calibrating a Canterbury model to the national declining band would impose a decline on a population that the regional data show to be growing. Calibrating solely to the regional increase would disregard the current classification. Both were therefore modelled: a precautionary calibration at a growth rate of 0.99, the shallow end of the national NZTCS band, and a regional calibration at 1.02, the observed Canterbury trend. These are not

scenarios of management or of impact, as elsewhere in this assessment, but two competing readings of the same population trajectory.

The quasi-extinction threshold was set at 20,000 mature individuals, approximately half the regional population, which also corresponds to an NZTCS population band boundary. Since it is applied to a regional population, it does not itself trigger a change in national threat status, which is assessed nationally, and it should be read as a level at which regional concern should arise.

4.8.2 Results and interpretation

Under the precautionary model, the Canterbury population of tarāpuka/black-billed gull is projected to decline from 41,350 to a median of 29,712 mature individuals over 25-years, with a probability of 0.15 of falling below 20,000. Under the regional model, it increases to a median of 62,844, with a probability of 0.002 of falling below 20,000. The difference between these two baselines is substantially larger than the effect of any collision loss modelled, and illustrates the major uncertainty for this species.

The effect of collision proportional to the baseline growth rate is broadly similar under both population growth models. An annual loss of 50 birds reduces the projected population by 2.6-3.4%, a loss of 100 by 4.9-7.1%, and a loss of 500 by 24.7-36.0%. The effect is larger under the precautionary calibration, as would be expected where losses are imposed on a declining rather than a growing population, but the difference is minimal relative to the difference in the baselines themselves.

Under the precautionary model, the probability of the regional population falling below 20,000 mature individuals rises from 0.15 with no collision to 0.22 at 100 birds per year and 0.54 at 500. Under the regional model the same probabilities are 0.002, 0.002 and 0.035. At an annual loss of 500 birds, the two models therefore differ by a factor of approximately fifteen. Taken together, the results indicate that collision losses at levels plausibly arising from a single site would produce a small but observable proportional reduction in the Canterbury population.

Table 9 - PVA results for tarāpuka/black-billed gulls within the Canterbury Region across 1-500 bird losses per year after 25-years. Probability of quasi-extinction is the probability of the adult population falling below 20,000 total adults, approximately half of the estimated current regional mature population.

Adult losses per year	Median adults remaining after 25-years [95% CI]	Reduction against baseline %	Ratio of losses relative to baseline [95% CI]	Probability of quasi-extinction
Precautionary population growth scenario: NZTCS national band, λ 0.99				
0	29712 [13513-62378]	0	1.00 [1.00-1.00]	0.15
10	29510 [13380-61938]	0.68	0.99 [0.97-1.02]	0.16
25	29208 [13158-61664]	1.69	0.98 [0.96-1.00]	0.17
50	28689 [12720-60681]	3.44	0.96 [0.93-0.99]	0.19
100	27598 [12091-59718]	7.11	0.93 [0.88-0.96]	0.22
200	25403 [10490-56094]	14.50	0.86 [0.76-0.91]	0.28
350	22140 [8149-51417]	25.48	0.75 [0.60-0.84]	0.40
500	19003 [5851-46910]	36.04	0.64 [0.43-0.77]	0.54
Regional trend: Canterbury observed, λ 1.02				
0	62844 [29556-131889]	0	1.00 [1.00-1.00]	0.002
10	62527 [29331-131523]	0.50	1.00 [0.98-1.01]	0.002
25	62083 [29038-130145]	1.21	0.99 [0.96-1.01]	0.002
50	61184 [28394-128873]	2.64	0.97 [0.95-1.00]	0.002
100	59793 [27273-126355]	4.85	0.95 [0.91-0.98]	0.002
200	56526 [25046-121631]	10.05	0.90 [0.84-0.94]	0.007
350	52021 [21486-113787]	17.22	0.82 [0.73-0.88]	0.018
500	47322 [18301-106434]	24.69	0.75 [0.62-0.83]	0.035

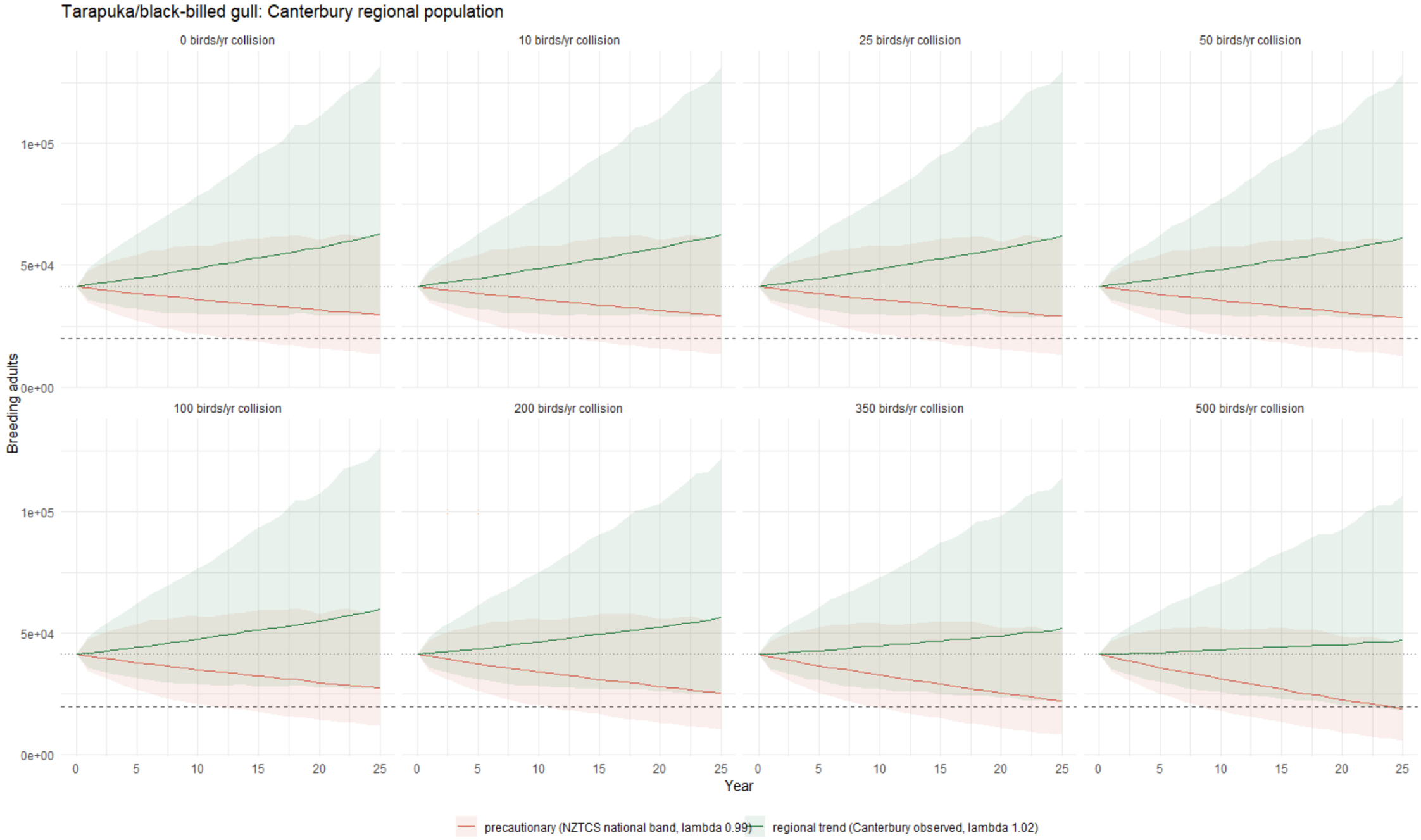

Figure 8. Projected breeding adult population tarāpuka/black-billed gulls within the Canterbury Region across 1-500 bird losses per year after 25-years, showing the median and 95% confidence interval. Two different growth rate scenarios are shown: the national NZTCS At Risk – Declining band (λ = 0.99) and the observed Canterbury trend of +2.12%/yr (λ = 1.02). The dotted lines show the starting point for the population, and the dashed line marks the quasi-extinction level of 20,000 mature individuals (half the current estimated mature regional population).

4.8.3 Model limitations

The most significant limitation is the unresolved conflict between the national classification and the regional trend, which has been addressed by modelling both. The national assessment forecasts a decline of 10-70% over three generations. The Canterbury data records a 2.12% annual increase over the twenty years to 2016/17. Mischler (2018a) concluded that tarāpuka/black-billed gull numbers were likely stable but urged a further census within 10 years to confirm this, and the 2021 panel adopted a precautionary position pending that confirmation. A repeat national census would reduce the uncertainty in this assessment and would provide the single most valuable additional information for this species.

The population estimate, while the most reliable available, relies on a single census season. Mischler (2018a) documented substantial methodological problems with all earlier surveys, noting that historical Southland counts using nest-density and gulls-per-nest extrapolations "probably vastly over-estimated the population between the 1960s and 1980s", and that data gaps in the 1995-98 census likely caused it to underestimate the breeding population. Regional counts are also subject to a mean annual variability of 42.2% between consecutive seasons, and a single-season figure carries appreciable uncertainty even where the method is sound. In addition, no ground correction factor could be applied to the Canterbury aerial counts, as different observers counted those photographs from the rest of the country. The Canterbury figure therefore carries greater uncertainty than the national total from which it is drawn.

Neither vital rate has been measured for this species. Adult survival is taken from a congeneric tarāpunga/red-billed gull re-sighting rate, and fecundity was set by calibration rather than estimated, so the model reproduces its target growth rate by construction and provides no independent check on its own parameters.

The unstructured model form omits the approximately three-year maturation lag, so the delay between a loss of adults and the resulting shortfall in recruitment is not represented, making the projections mildly optimistic. A three-stage model would be preferable and would become feasible if a first-year survival estimate could be obtained.

Finally, no data on site use were available, and local counts are highly variable. Project River Recovery recorded 13 birds in the Upper Ōhau in 2024 against none in 2023, none in the Lower Ōhau in any survey round, and 104 on the Tasman River in 2024 against 505 in 2023. As the species is colonial and colonies relocate between years, local presence in any given season is a poor guide to exposure over 25-years. Separately, tarāpuka/black-billed gull is both a subject of this assessment and a predator and active competitor of another. Project River Recovery recorded a tarāpuka/black-billed gull colony on Tern Island reaching 150 individuals by January 2025 and predation and harassment of tarapirohe/ black-fronted tern eggs and chicks (Nelson et al. 2025). No single-species population model can represent that interaction, and it should be considered in the wider assessment of effects.

4.9 Tōrea/South Island pied oystercatcher

4.9.1 Species specific model methods

Tōrea/South Island pied oystercatcher is one of the best-parameterised species in this assessment. Adult survival is a directly reported New Zealand estimate rather than a congeneric proxy, fecundity has been measured directly across 10 breeding seasons, and the population trend is derived from two national wader surveys. The projection period for this species is 40-years, rather than the 25-years used for all other species in this assessment. The NZTCS assesses population change over 10-years or three generations, whichever is longer (Townsend et al. 2008). Tōrea/South Island pied oystercatcher have an age at first breeding of 4-6 years and an adult survival rate of 0.87, giving a generation time of approximately 13.1 years, the longest of any species assessed. Therefore, three generations is

approximately 39 years. A 25-year projection would report against a window the classification system does not use and would understate decline relative to the criteria against which the species is assessed. The projection period was extended to 40 years to align with the assessment window. All other species have generation times short enough that a 25-year projection covers or exceeds the relevant window.

The species was modelled as a three-stage population of juveniles, subadults and adults at national scale, with viability assessed on mature individuals. The three-stage structure is necessary here: with first breeding at 4-6 years, an unstructured model would omit a 4-5 year maturation lag between the loss of adults and any resulting shortfall in recruitment. Starting abundance was set at 38,330 mature individuals, derived from the approximately 77,000 birds counted in national wader surveys covering 2005-2019 (Riegen & Sagar, 2020) and an adult fraction of 49.8% from the stable stage structure. Juvenile and subadult classes were derived on the same basis at 14,445 and 24,226, respectively. The resulting figure sits within the NZTCS population band of 20,000-100,000 mature individuals.

Fecundity was taken directly from Sagar et al. (2000), who monitored breeding on eleven farms in mid-Canterbury between 1987 and 1997 and reported a mean of 0.76 ± 0.13 chicks fledged annually across the 1988-97 seasons. A recent modelling exercise by Schlesselmann et al. (2026) also reported productivity per female as 0.74 (CI: 0.64-0.83). Adult survival was set at 0.87, while fledgling survival was set at 0.77 and juveniles at 0.51, following Schlesselmann et al. (2026). The viability threshold was set at 20,000 mature individuals, the NZTCS band boundary between At Risk – Declining (20,000-100,000 mature individuals) and Threatened – Nationally Vulnerable (covering 5,000-20,000 mature individuals), assuming decline rates become worse. As the population decline rate, as set in the model, takes the population across this threshold within the projection period regardless of collision mortality, results are reported as the year in which the threshold is crossed and the ratio of additional losses relative to the baseline rather than as a probability of crossing it (i.e. quasi-extinction).

4.9.2 Results and interpretation

In the absence of additional collision mortality, the national population is projected to decline from 38,330 to a median of 9,900 mature individuals over 40 years (Table 10, Figure 9), a reduction of 75%, with the median crossing the threshold of 20,000 mature individuals around year 20 (Figure 9). This decline is a result of the population trend at which the model is calibrated, and further results should be interpreted in relation to this predicted decline. The probability of quasi-extinction is therefore not an informative metric for this species. With no collision mortality the probability of quasi-extinction is 99% and rises to 100% at an annual loss of 500 birds (Table 10). The threshold is crossed under every simulation (as can be seen from the 95% confidence intervals) across all loss levels (Figure 9). Collision

Table 10 - PVA results for tōrea/South Island pied oystercatcher across 10-500 bird losses per year after 40-years. Probability of quasi-extinction is the probability of the adult population falling below 20,00 total adults, the threshold for which the species would transition from At Risk to Threatened status.

Adult Losses per Year	Median Adults Remaining After 25-years [95% CI]	Reduction Against Baseline %	Ratio of Losses Relative to Baseline [95% CI]	Probability of Quasi-extinction
0	9900 [5330-17764]	0	1.00 [1.00-1.00]	0.992
10	9772 [5173-17729]	1.29	0.99 [0.94-1.04]	0.993
25	9574 [5072-17437]	3.28	0.97 [0.92-1.01]	0.994
50	9216 [4820-17046]	6.90	0.93 [0.88-0.98]	0.995
100	8540 [4298-16350]	13.74	0.87 [0.79-0.93]	0.996
200	7252 [3207-14577]	26.74	0.73 [0.58-0.83]	0.997
350	5280 [1584-12021]	46.66	0.54 [0.29-0.68]	0.999
500	3355 [0-9536]	66.11	0.34 [0.00-0.54]	1

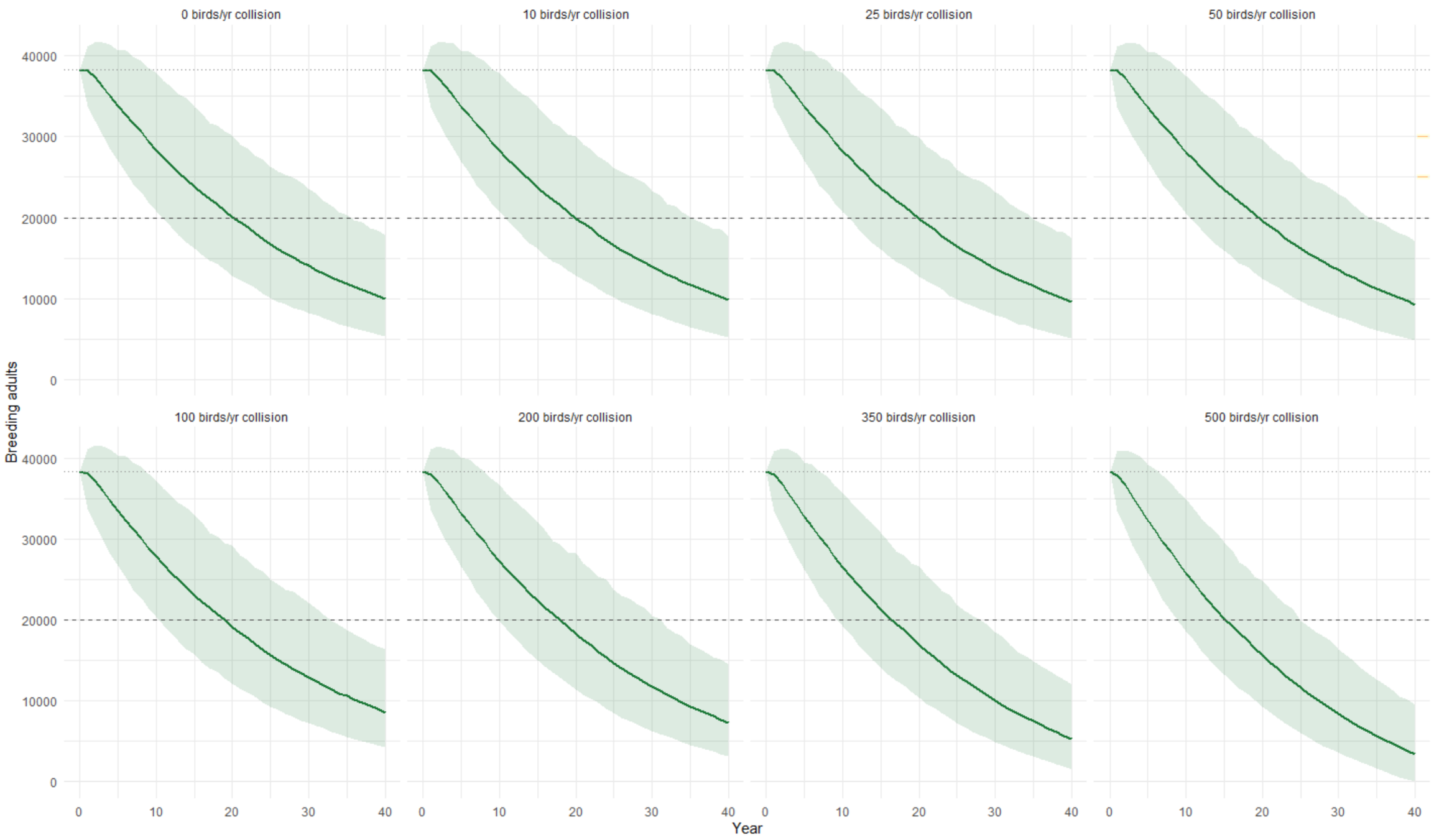

Figure 9. Projected breeding adult population of tōrea/South Island pied oystercatcher over 40-years under eight annual collision loss levels (0-500 birds per year), showing the median and 95% confidence interval. The dotted lines show the starting point for the population, and the dashed line marks the quasi-extinction level of 20,000 mature individuals.

mortality advances the point at which the population enters the 5,000-20,000 mature individual threshold, within which the same rate of decline would meet the criteria for Threatened – Nationally Vulnerable. With only 10 annual losses, the population growth rate is reduced by 1.3%; at 25 annual losses this increases to 3.3%; at 50 annual losses it reaches a near 7% reduction relative to the baseline decline rate (Table 10). At 50 annual losses the reduction in the decline rate relative to the baseline decline is also significantly different from the baseline (Table 10).

The wide intervals on the projected population medians, spanning 4298 to 16,350 mature individuals at 100 birds per year, reflect the variance accumulated over a 40-year projection and describe uncertainty about the population's future rather than uncertainty about the effect of collision.

4.9.3 Model limitations

Two tōrea/South Island pied oystercatcher were observed using the site during surveys. Project River Recovery recorded one bird in the Lower Ōhau River in 2024, one in 2023, and 3-11 in the 2008-2010 round, with none recorded in the Upper Ōhau in any round and 95 recorded on the Tasman River in 2024 (Nelson et al. 2025). Tōrea/South Island pied oystercatcher is nonetheless strongly migratory within New Zealand. Starting in December, most breeding adults leave inland territories for coastal estuaries, largely in the northern South Island and northern North Island, before returning from early June (Sagar & Veitch, 2014). Exposure is therefore not confined to the breeding season, and the proportion of the national population passing through the project area at any point in the annual cycle is unknown.

5.0 Model Outcomes and Recommendations

5.1 Interpreting models to set trigger levels

There are numerous considerations in how PVAs have been put together, as well as inherent limitations of this technique, which influence the way in which they should be interpreted. These major considerations are outlined below.

5.1.1 Data scarcity

The model results need to be interpreted considering the inherent limitations of population modelling. A PVA is a simplified representation of a complex biological system, and its projections are only ever as reliable as the demographic data used to build it. For several species those data are sparse, and for matuku-hūrepo/Australasian bittern many key data are absent. Modelled population predictions for the following species have the greatest associated uncertainty:

- **Matuku-hūrepo/Australasian bittern:** virtually no data available.
- **Kōtuku/white heron:** Adult and chick survivability estimated from congeneric species, fledgling survivability estimated based on taking these values as true.
- **Pūteketeke/Australasian crested grebe:** no reliable data for chick or fledgling survivability could be found, so population was modelled as a single life stage; also fecundity was estimated.
- **Tarāpuka/black-billed gull:** no reliable data for chick or fledgling survivability could be found, so population was modelled as a single life stage; adult survivability estimated from congeneric species.

5.1.2 Sources of mortality

The models incorporate natural year-to-year variability in survival and breeding, but the true extent of this variability is not well known. Additional pressures such as climate change, which may progressively

change habitats, food availability and breeding success over the lifespan of the project, cannot be reliably forecast and are not included in any model. The analysis also cannot account for cumulative effects, where losses at this site combine with other mortality sources and with ongoing land-use change, climate change risk and development across the wider landscape. By design, the models only factor in the potential additional mortality in a general sense. These models cannot separate collisions at this proposed solar farm from other emerging sources of mortality. Appropriate trigger levels within this project must consider the risk to species within the context of any known or cumulative pressures within the Mackenzie Basin.

5.1.3 Influence of management

Several of the species assessed are also not self-sustaining under current conditions and persist only because of ongoing conservation management. The modelled population trajectories for these species are therefore dependent on this management continuing at the current level of effectiveness. Kakī/black stilt were modelled on the basis that the captive breeding and release programme currently supporting the wild population would continue for the duration of the project. The modelled population depends on these releases and would not be expected to sustain itself without them. Tarapirohe/black-fronted tern were modelled under two scenarios, with intensive predator control maintained at the breeding colony, and without it. The contrast is stark: with predator control the population is broadly stable, whereas without predator control the population is projected to decline steeply and to face a high probability of quasi-extinction even in the complete absence of collisions.

5.1.4 True mortality versus monitoring

Finally, species-specific trigger levels (see below) depend on carcass monitoring, which is known to systematically underestimate true mortality. Carcasses are removed by scavengers, missed by searchers, or fall beyond the searched area, so the number of birds recorded will represent only a fraction of those actually killed.

5.1.5 Interpretation

Each of the factors outlined above means the real risk to a population is likely to be greater than the modelled figures alone would suggest. Setting the trigger levels well below the point at which the modelling first detects an effect is the appropriate response to these combined uncertainties: it builds in space to absorb what the model cannot predict, and ensures that a management response is initiated on a precautionary basis.

5.2 Assessment of species specific trigger levels following modelling

The PVAs are used below to confirm the appropriate trigger levels for each species. Table 11 shows several metrics, including the annual collision loss at which a population-level effect first emerges from the modelling, set against trigger levels as specified in the AMP. The model results and the trigger levels are distinct and should not be conflated. The trigger level is a management threshold and should be set conservatively so that action is taken well before a demographically meaningful loss has occurred. The modelled loss levels reported are biological effects thresholds (Table 11). These are the points at which the analysis can detect a change in the population's trajectory, but must be interpreted within the limitations of the modelling process to set trigger levels (see section 5.1). The species-specific trigger levels therefore should sit below the effects threshold, as shown by the models, and the purpose of this comparison is to confirm that they do.

As no single metric can capture a population-level effect equally well across species that differ substantially in abundance and trend, we report three complementary criteria, each expressed as the lowest annual loss at which it is met:

1. ***C1, detectable population shift*** (i.e. paired-ratio resolution) is the lowest annual loss at which excess mortality produces a population reduction large enough to be distinguished from chance. For each simulation, the final population size with collision applied is divided by the size from the exact same simulation run without collision, giving a ratio in which a value of one indicates no effect and values below one indicates a reduction. The statistical uncertainty is reflected in the 95% confidence intervals for the ratio of losses relative to the baseline. If the upper interval is < 1, the effect is statistically resolvable.
2. ***C2, extinction risk*** (i.e. achieving quasi-extinction) is the lowest annual loss at which the probability of the population falling below its quasi-extinction threshold rises by at least 1% above the baseline probability. This criterion is most sensitive for small populations, where extinction risk responds sharply to added mortality. Still, it is uninformative where the baseline probability of quasi-extinction is already close to one due to sharp natural decline rates.
3. ***C3, growth-rate suppression*** is the lowest annual loss at which the mean population growth rate is depressed by at least 1% (in log units). This equates to approximately a one percent additional annual decline (i.e. on top of the natural decline rate). This is the least sensitive criterion for large or stable populations, which can absorb substantial losses before their growth rate shifts by a full percentage point (e.g. for tōrea/South Island pied oystercatcher).

Which of these three criteria is most conservative varies with a species' starting abundance and population trend, so all three are presented rather than a single value. The relevant comparison for each species is between its trigger level and the most conservative criterion.

Table 11 - Three population loss level criteria as predicted by the species-specific population viability models and the species-specific trigger loss levels. Each trigger value should be below or equal to the most conservative predicted loss level.

Species (operative scenario)	C1 Detectable Population Shift	C2 Extinction Risk	C3 Growth-Rate Suppression	Trigger Loss level (p.a.)	Is Trigger < Most Conservative Risk
Kakī/black stilt (at 0.78)	10	1	5	1	Equals
Matuku-hūrepo/Australasian bittern (all decline rates)	5-10	N/A	2	1	Yes
Kōtuku/white heron	5	2	5	1	Yes
Tarapirohe/black-fronted tern (managed)	10	10	10	1	Yes
Pūteketeke/Australasian crested grebe	10	20	10	3	Yes
Māpunga/black shag	50	10	100	5	Yes
Pohowera/banded dotterel	50	20	100	5	Yes
Tarāpuka/black-billed gull (at 0.99)	50	25	350	5	Yes
Tōrea/South Island pied oystercatcher	50	N/A	350	5	Yes

Kakī/black stilt

For kakī/black stilt, the quasi-extinction risk thresholds are met for only a single individual lost annually, whereas a 1% decrease in the growth rate is met once five individuals are lost annually. The most conservative annual loss-level prediction is therefore one individual, which equals the proposed trigger level for this species. It should be noted that for such a threatened species a trigger value of one is

likely not conservative based on the model results. If management actions triggered after one loss do not result in fewer losses (i.e. less than one loss per year) then the population of kakī/black stilt is likely to be impacted. It is also important to note that these predictions assume that the captive breeding programme continues at the current scale.

Matuku-hūrepo/Australasian bittern

As the population of matuku-hūrepo/Australasian bittern is already modelled to be in sharp decline, the quasi-extinction risk thresholds are met before any loss levels are applied, so the change to the growth rate is used instead. The population growth rate is predicted to decrease by 1%, with an annual loss of two birds per year, which is more than the suggested trigger level of one bird. However, the model for this species was based on extremely limited data and so the uncertainty is very high. Two annual losses are predicted to affect the population and so an annual loss of one individual may pose a risk to the population.

Kōtuku/white heron

For kōtuku/white heron, the quasi-extinction risk thresholds are met at annual losses of two individuals, while a 1% decrease in the growth rate is not reached until five individuals are lost annually. The most conservative annual loss level prediction is therefore two individuals, which is greater than the set trigger value. However, there is high uncertainty in the model for this species as mentioned above.

Tarapirohe/black-fronted tern

Under the managed (predator-control) scenario for tarapirohe/black-fronted tern, the extinction risk threshold (at the 1% probability level) and a 1% decrease in the growth rate are both met at an annual loss of 10 individuals. The most conservative annual loss level prediction is therefore 10 individuals, above the trigger level of one bird. This result is dependent on the continuation of active predator control in the breeding colony at the current level. If this were to stop or the effort be reduced, this trigger value would no longer avoid exceeding risk thresholds.

Pūteketeke/Australasian crested grebe

For pūteketeke/Australasian crested grebe, the extinction risk thresholds are not met until an annual loss of 20 individuals, but because the population is increasing, the change in the growth rate provides a more sensitive measure. It is reached at an annual loss of 10 individuals. The most conservative annual loss level prediction is therefore 10 individuals, which sits above the trigger level of two birds within a survey (or three within a 12-month period).

Māpunga/black shag

The -extinction risk thresholds are met for māpunga/black shag at an annual loss of 10 individuals, whereas a 1% decrease in the growth rate is not reached until 100 individuals are lost annually. The most conservative annual loss level prediction is therefore 10 individuals, which sits above the trigger level of five birds within a 12-month period.

Pohowera/banded dotterels

The extinction risk thresholds are met at an annual loss of 20 pohowera/banded dotterels, whereas a 1% decrease in the growth rate is not reached until 100 individuals are lost annually. The most conservative annual loss level prediction is therefore 20 individuals, which sits above the trigger level of five birds within a 12-month period.

Tarāpuka/black-billed gull

Using the precautionary growth rate scenario (national growth rate of 0.99) for tarāpuka/black-billed gull, the quasi-extinction thresholds are met at an annual loss of 25 individuals and a 1% decrease in the growth rate at 350. The most conservative annual loss level prediction is therefore 25 individuals, which sits above the trigger level of five birds within a 12-month period.

Tōrea/South Island pied oystercatcher

As tōrea/South Island pied oystercatcher are modelled to be in sharp decline irrespective of additional losses, the quasi-extinction thresholds are met before any loss levels are applied. Additionally, the change to the growth rate is very insensitive with such a large starting population. As such, the statistically detectable population shift is used. This appears at an annual loss of 50 individuals, well before a 1% increase in the decline rate at 350 individuals lost. The most conservative annual loss level prediction is therefore 50 individuals, which sits above the trigger level of five within a 12-month period.

6.0 Conclusion

This report provides PVA models for five Threatened and four At Risk bird species that may be affected by the proposed solar farm to assess whether the proposed trigger levels set out in the AMP and conditions of consent which, if breached, require adaptive management and potentially compensation, are sufficiently conservative to ensure ongoing, viable populations of these species. Across all species assessed, the trigger levels sit at or below the most conservative annual loss level identified by the modelling, which was estimated using several techniques. For the At-Risk species, the margin is substantial, with triggers set many times below the modelled effects thresholds. The trigger levels therefore provide the space required for a management response to be initiated and to take effect well before collision losses could reach a level detectable as a population-level effect.

The adequacy of the trigger levels set for Nationally Critical species (kakī/black stilt, kōtuku/white heron and matuku-hūrepo/Australasian bittern) is more complex to assess. The mortality level at which an effect first emerges from the modelling is as low as one to two individuals annually. A trigger level cannot be set below a single bird, and so for these species the numerical margin between the trigger and the effects threshold is very low. In the case of kakī/black stilt, the trigger coincides exactly with the point at which the model first registers an effect. Because the trigger cannot be lowered further, the additional protection these species require cannot be delivered through the trigger value alone and must instead be built into the broader management response, including initial mitigation measures and compensation payments. Key to this is the recognition that a trigger level is not an allowable annual take. It does not follow from a trigger of one bird that the loss of one bird each year is acceptable. The trigger is the point at which management must change.

Where a trigger level is reached, adaptive management responses should be structured as a tiered escalation rather than a single fixed threshold. Under such a structure, the first recorded trigger breach would require an immediate change to site management and a review of its cause. Should a further trigger breach occur despite those measures, additional and more substantial management changes would be required. Should mortalities continue beyond that point, the option to cease operation of the affected part of the site, up to and including decommissioning, must be explicitly available. Escalation of this kind makes the severity of the response proportionate to the accumulating risk.

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