

A REVIEW OF 28 YEARS OF BEACH-NESTING AUSTRALIAN PIED OYSTERCATCHER *Haematopus longirostris* CONSERVATION IN THE RICHMOND RIVER AREA, NEW SOUTH WALES

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SUMMARY

The Australian pied oystercatcher *Haematopus longirostris* is a shorebird that is Endangered in the state of New South Wales (NSW), Australia.

This study reviewed data from the Richmond River Area Pied Oystercatcher Protection Program 1997–2013 and Richmond River Area Shorebird Protection Program 2014–2024, on the north coast of NSW. The increase in breeding population size from an *estimated* 25 pairs in 2000 to observed 37 in 2024 masks underlying concerns.

Surplus production of fledglings indicates that population size is not limited by reproductive rate. Weak correlations indicate that population size is not sensitive to food resource. The step increase in population size at South Ballina associated with the closure of that beach to recreational vehicles on 31 March 2021 and the general ‘flight to safety’, where population size has increased largely at sites with fewer people, indicate that population size is sensitive to human recreation disturbance.

A scientific approach to beach-nesting bird conservation is needed.

INTRODUCTION

The Australian pied oystercatcher *Haematopus longirostris* is a shorebird that inhabits estuaries and sandy ocean beaches along the Australian coast, with major populations in the southern states South Australia, Tasmania and Victoria (Taylor *et al.* 2014). The species is listed as Endangered in Schedule 1 of the New South Wales (NSW) *Biodiversity Conservation Act 2016* (NSW Government 2016) because the estimated population size in NSW is low and projected or continuing to decline (NSW Scientific Committee 2010).

This study examined > two decades of beach-nesting oystercatcher results from the Richmond River area, on the north coast of NSW. The objectives were to:

- estimate breeding population size trends;
- estimate change points (also known as break points) in those trends that may be associated with management interventions and other historical events, and;
- investigate putative agents of decline.

The declining population paradigm in conservation biology deals with the causes of small population sizes and their treatment. An investigation into the causes of a decreasing population proceeds in four steps (Caughley 1994):

- 1 Study the natural history of the species to gain a knowledge of, and feel for, its ecology, context and status.

- 2 When confident that this background knowledge is adequate to avoid silly mistakes, list all *conceivable* agents of decline.
- 3 Measure their levels where the species now is and also where the species used to be. Test one set against the other. Any contrast in the right direction identifies a *putative* agent of decline.
- 4 Test the hypotheses so produced by experiment to confirm that the putative agent is *causally* linked to the decline and not simply associated with it.

1. Ecology, context and status

Relevant background knowledge from Taylor *et al.* (2014) is:

- The Australian pied oystercatcher is non-migratory. Breeding pairs are strongly territorial and commonly remain on their territories throughout the year.
- Individuals are sexually mature at 3–4 years but delayed age of first breeding to approximately 6–8 years is common.
- The existence of non-breeding adults within populations is a strong indication that population size is limited by territorial behaviour. Thus, counts of breeding pairs (breeding population size) are more relevant to conservation than are counts of all adult individuals.
- Adult mean annual survival rates are > 90% and the oldest known banded individual was 29.5 years old. Severe winter weather, as a source of mortality, does not occur in any part of the species range.
- Annual mean reproductive rate is usually low, 0.2–0.7 fledglings/pair, but within the range reported for other oystercatcher species worldwide.
- The species is adapted to nest predators. It is not clear that high depredation rates of eggs and chicks can cause population size to decrease.
- The foraging ecology of the species and the impact of commercial shellfishing are poorly understood.

Microtidal, high energy sandy ocean beaches ('surf beaches') are common along the SE Australian coast (Short 2007). With a narrow intertidal zone, these are essentially linear habitats.

The NSW oystercatcher beach-resident population size is small. The NSW Wader Study Group counted 232 oystercatchers on NSW beaches in Oct 1998 (Owner & Rohweder 2003). 222 were counted in 2015–2018 (Totterman 2020b). It was estimated from NSW Government information that there were < 54 oystercatcher breeding pairs on NSW beaches in around 2019 (Totterman 2020b). 40 territories were counted in 2015–2018 (Totterman 2020b).

Eastern Australian oystercatcher populations are not demographically closed, with birds moving up and down the coast as well as interstate (NSW Scientific Committee 2010, Totterman 2018).

There have not been sufficient counts to estimate the NSW oystercatcher population size trend. The NSW Scientific Committee (2010) relied upon counts from the north coast from 1998, 2000, 2003 and 2005 (Harrison 2009) to infer that the NSW population size was

decreasing. That small sample and short time span was insufficient to estimate the rate of decrease.

Commonly known as the 'pipi', *Donax deltoides* is a large surf clam (mean maximum shell lengths 37–55 mm on the north coast of NSW; Totterman 2019) that inhabits the intertidal and shallow subtidal zones on sandy ocean beaches from Fraser Island, Queensland, to the Murray River, South Australia (Ferguson *et al.* 2018). Oystercatchers are specialist predators of bivalves and pipis are the major food resource for beach-resident oystercatchers in NSW (Owner & Rohweder 2003, Harrison 2009, Totterman 2018). Owner and Rohweder (2003) and Harrison (2009) reported that oystercatcher count density (birds per km of beach length) is positively correlated with pipi count density (pipis/m²) and mean maximum shell length. The pipi data quality in those studies was poor (Totterman 2020c). A later study reported that pipi abundance was a poor predictor for oystercatcher count density on other beaches in NSW (Totterman 2020b).

2. Conceivable agents of decline

The NSW Scientific Committee (2010) recognised five conceivable agents of population size decline for the Australian pied oystercatcher in NSW:

- habitat degradation and loss (resulting from coastal development and engineering works);
- human recreation disturbance (including vehicles on beaches);
- decline of food resources (including harvesting of pipis);
- hydrological changes to estuaries and lagoons (which is not applicable to beach-resident populations), and;
- egg or chick predation by foxes, feral pigs and artificially high silver gull *Larus novaehollandiae* populations.

3. Putative agents of decline

Reproductive rate hypothesis

The reproductive rate hypothesis is that low reproductive rate, insufficient to replace losses due to mortality and emigration, will cause population size to decrease. Reproductive rate (also known as productivity) = (count of fledglings) / (count of breeding pairs) = fledglings per pair. It is commonly calculated for some defined population (*e.g.* a single site) and a single breeding season (year).

The reproductive rate hypothesis is popular in beach-nesting bird conservation. It is easily understood by laypersons and poor breeding success can appeal to sympathy and compassion emotions. Positive results can be achieved quickly (*i.e.* counts of young fledged) and annual reporting of those results suits annual funding cycles.

Since 1997, the primary conservation action for oystercatchers in the Richmond River area has been control of European red foxes *Vulpes vulpes* to reduce depredation of eggs and chicks (Wellman *et al.* 2000). This action has legislative support from the NSW *Biodiversity Conservation Act 2016* (NSW Government 2016), Schedule 4 of which lists depredation by the red fox as a Key Threatening Process (in general and not only for oystercatchers). This action has received financial and management support in successive Threat Abatement Plans

for depredation by the red fox (NSW National Parks & Wildlife Service 2001, NSW Office of Environment & Heritage 2010).

A sum of \$830830 was spent on fox baiting and monitoring breeding oystercatchers in the Richmond River area from 2000–2024 (NSW Department of Climate Change, Energy, the Environment & Water 2025). No reproductive rate threshold, above which population size is predicted to increase, has been proposed.

Population Viability Analysis (PVA) is commonly used for modelling demographic processes for small populations and estimating extinction risk. Caughley (1994) criticised PVAs for not investigating the causes of population decline. For example, Plissner and Haig (2000) carried out a thorough PVA for piping plover *Charadrius melodus* meta-populations in North America. They estimated mean reproductive rate to exceed a 95% probability of persistence over 100 years. Reproductive rate was targeted because it is a directly-manageable factor influencing extinction risk. Habitat degradation and loss was not modelled and they acknowledged that shortcoming in:

For species such as piping plovers, whose principal habitats are subject to high potential and actual human disturbance [as are beach-resident Australian pied oystercatchers], attempts to simulate population growth trajectories over long time periods [e.g. 100 years] are particularly problematic, given the uncertainties of future environmental policy and human pressures on demographic processes as well as on habitat availability and suitability.

The reproductive rate hypothesis for beach-nesting oystercatchers has not been experimentally tested.

The subordinate fox baiting hypothesis, *i.e.* that depredation of eggs and chicks by foxes is a major cause of low breeding success, has also not been experimentally tested (Taylor *et al.* 2014). Wellman *et al.* (2000) relied upon an informal comparison of 1994–1996 breeding results from South Ballina with those after fox baiting commenced in 1997. The former sample is too small for a statistical comparison and there were no control sites. Anyhow, the fox baiting hypothesis misses the important question that is whether or not losses to foxes will cause population size to decrease (Taylor *et al.* 2014).

Food resource hypothesis

The food resource hypothesis is that low food abundance will cause population size to decrease via emigration (a numerical predator-prey response) and, in extreme cases, increased mortality rates.

Pipi stocks on north coast beaches of NSW 'crashed' between c. 2003–2009 and the associated decrease in oystercatcher counts was interpreted by Harrison (2009) and then the NSW Scientific Committee (2010) as a decreasing trend. Harrison (2009) also thought that low pipi abundance can result in low reproductive success.

Oystercatcher counts did increase when pipi stocks recovered, but territory counts less so (Totterman 2018). No pipi abundance threshold (below which oystercatchers experience food stress) has been proposed and pipi stocks are not amenable to experimental manipulation.

Intake rate for shorebirds eating macro-invertebrates commonly reaches an asymptote at low prey count density (Goss-Custard *et al.* 2006, Totterman 2018). Other drivers of food stress

can be prey availability (e.g. tides, weather) and feeding time (e.g. disturbance) (Taylor *et al.* 2014).

Human recreation disturbance hypothesis

The human recreation disturbance hypothesis is that frequent disturbance can discourage use and occupation of otherwise suitable habitat, causing a decrease in population size. Disturbance is also commonly associated with low reproductive rate (e.g. Harrison 2009). Disturbance is closely linked to coastal development and other human demographic trends.

Habitat variables can be useful proxies for human recreation disturbance. A previous study reported negative associations between beach-resident oystercatcher count density and urban proportion of beach length and pedestrian beach access track density. With the exception of one beach, zero oystercatcher territories were observed when pedestrian beach access density exceeded 1.3 tracks per km of beach length (Totterman 2020b).

Vehicles are a vector for human recreation disturbance, transporting people along long stretches of beach. Thus, vehicle-based recreation is a subset of human recreation disturbance. Vehicle strikes are also an additional source of mortality for shorebirds (Harrison 2009, Totterman 2020c).

Human recreation disturbance is amenable to experimental manipulation via restrictions on access. Williams *et al.* (2004) reported that African Black Oystercatcher *Haematopus moquini* total counts, breeding pair counts and reproductive rate increased after recreational vehicles were banned on South African beaches.

However, protection of habitat via restricting access to and use of public beaches and restricting private beach front development is politically toxic. Legislative support is lacking. Schedule 4 of the NSW *Biodiversity Conservation Act 2016* (NSW Government 2016) does not recognise human recreation disturbance as a Key Threatening Process (not for any species).

4. Experimental tests for agents of decline

The oystercatcher conservation program in the Richmond River area has been guided by intuition and anecdote (Wellman *et al.* 2000) and carried out in a haphazard manner. There has been no experimental design (with replicated control and impact sites) and there are missing years in the time series data. No data on pipi abundance and human recreation disturbance have been collected within the program. Thus, necessary statistical tests to identify which of the three agents of decline above are *causally* linked to oystercatcher population size are not possible.

METHODS

Study sites

The five sites in this study were defined by natural breaks in the beach habitat (e.g. headlands and rivers) and different management histories (Figure 1).

South Ballina Beach is defined as the 21.3 km long rural beach from the Richmond River south to the Coffee Rocks. The northern 2.5 km of South Ballina is part of Richmond River Nature Reserve, the southern 2.4 km adjoins Broadwater National Park (NP) and the remaining 16.4 km adjoins a narrow Crown Reserve, backed by private properties.

The 3.9 km long Coffee Rocks (indurated sand) interval separates South Ballina from Airforce Beach. Short patches of sand are located within the eroding 'coffee rock'. Coffee Rocks adjoins Broadwater NP.

South Ballina and Coffee Rocks were closed to recreational vehicles on 31 March 2021. There were several reasons for the closure including conflict with non-driving beach users and local residents, damage to the beach and dune environment and impacts on the local oystercatcher population.

Airforce Beach is defined as the 5.8 km long beach from the Evans River north to Coffee Rocks. The northern 4.2 km of Airforce adjoins Broadwater NP and Salty Lake, an intermittently closed and open lake. Unrestricted beach driving continues on Airforce.

Bombing Range Beach is defined as the 10.3 km long beach from Goanna Headland south to Jerusalem Creek. Public access to Bombing Range is prohibited because it adjoins the Royal Australian Air Force Evans Head Air Weapons Range.

The 5.0 km long Black Rocks (indurated sand) interval separates Bombing Range from Ten Mile Beach. There is a campground at the southern end of Black Rocks.

Ten Mile Beach is defined as the 13.2 km long beach from Shark Bay north to Black Rocks. Unrestricted beach driving continues on Ten Mile. Bombing Range, Black Rocks and Ten Mile adjoin Bundjalung NP.

The Coffee Rocks and Black Rocks experience low human recreation disturbance compared to other public sites because the eroded shoreline makes those sites less accessible.

Data sources

This study reviewed data from the Richmond River Area Pied Oystercatcher Protection Program 1997–2013 and Richmond River Area Shorebird Protection Program 2014–2024 (Appendix). Included are South Ballina data from 1994–1996, preceding the start of conservation management, and from 1997–2000, when there were no formal reports (Wellman *et al.* 2000). The conservation program was extended to include Airforce from 2001 and Bombing Range from 2002. The Coffee Rocks and Black Rocks were included in the program from around 2016 and 2005 respectively when it was realised that there oystercatcher breeding pairs in those intervals. Ten Mile was included from 2005.

Following the c. 2003–2009 'pipi crash', the oystercatcher conservation program was cancelled in 2011 and did not completely resume until 2016 (Table 1). South Ballina, Coffee Rocks and Airforce were effectively cancelled in 2020 resulting from the late 3 November start.

The oystercatcher laying season in the Richmond River area is August–December and late attempts generally are replacement clutches (Wellman *et al.* 2000). Breeding pairs were located by informal territory mapping (*i.e.* identification of territorial pairs usually precedes detection of nesting attempts) and prior knowledge (known breeding territories from preceding years). The frequency for oystercatcher monitoring was mostly between 1–3 days per week.

Territory mapping and pipi counts were performed independently of the conservation program for South Ballina every two years from 2016 to 2024. Territory and pipi counts for other years and/or sites were obtained from previous studies (Totterman 2018, 2019).

Territory mapping can easily be applied to beach-resident oystercatchers because they are large, conspicuous birds and territories on sandy ocean beaches are practically one-dimensional. The effort was 3–5 counts during the breeding season (mode three) and the sampling interval Inter-Quartile Range (IQR) was 2–7 days (full range 2–43). Territories were identified as adult ‘pairs’ that were present on the same stretch of beach on $\geq 50\%$ of counts. These are rapid estimates and not all pairs that occupy a stretch of beach may breed. Territory counts are compared with breeding pair counts in the Results.

Pipis typically aggregate in discontinuous ‘bands’ that run parallel to the shoreline and, to be consistent with the early data where quadrat-based sampling was used, abundances are reported as pipi band mean density (pipis/m²). Biennial counts from 2016 on have used a rapid ‘feet digging’ swash zone sampling method. Mean feet digging pipi counts were converted to ‘pseudo pipi band densities’ (Totterman 2019; density = $1.7 + 0.8 \times \text{mean count}$).

Statistical analysis

Breeding population size trends were estimated for each site by two methods:

- 1 Bayesian Estimator of Abrupt change, Seasonal change, and Trend (BEAST) was used to detect trends and change points in a time series.
- 2 Bayesian regression was used to estimate trend and change point effect sizes. Model structure was based on the BEAST results.

BEAST is an ensemble algorithm that combines many models using Bayesian model averaging (Zhao *et al.* 2019). Key results from BEAST were:

- the mean number of trend change points;
- probability, location and credible intervals for each change point, and;
- slope probability (zero, positive and negative) and slope order (0 = flat, 1 = linear, 2 = quadratic, etc.) for segments between the change points.

BEAST can also detect outlier change points (points that are far off the trend).

Bayesian computational methods provide a way forward for analytically intractable change point regression models (Lindeløv 2020). Key results from change point regression models were:

- intercept and/or slope estimates for segments between change points, and;
- change point effect sizes (*i.e.* contrasts in intercepts and/or slopes across change points).

Breeding pair counts are under-dispersed (they usually vary little from year to year) and the Gaussian error distribution (constant variance) was a reasonable approximation for both BEAST (which did not allow count distributions) and change point regression models (which did not allow for under dispersion).

An ANOVA-like reproductive rate regression model was fitted to annual fledgling counts by site with breeding pair counts as an offset. A negative binomial error distribution was assumed to account for over dispersion (*i.e.* negative binomial variance $\sigma^2 = \mu + d \cdot \mu^2$ where $d = 0$ corresponds to a Poisson distribution).

Regression model priors were generally ‘weakly informative’ with zero (neutral) effect sizes (Goodrich *et al.* 2025; the Supplementary Material provides details). Exceptions are:

- Uniform population size change point priors with limits based on the corresponding credible interval from BEAST. The uniform distribution was selected to preserve the unusual shape of the data likelihood (*i.e.* in simple terms: *posterior probability density* = *prior density* × *data likelihood density*). The narrow scale makes the prior ‘informative’.
- Normal reproductive priors with 98% upper limit = 3 (on the count scale). The strong containment follows from clutch size is rarely > two (Marchant & Higgins 1993) and makes the prior informative.

Bayesian regression models were run with five chains, 3000 burn-in samples and no thinning. The number of draws was increased to achieve one or two decimal places precision for estimates and contrasts.

All statistical analyses were performed using the software R v. 4.5.1 (R Core Team 2025). BEAST models were fitted with the R package *Rbeast* v. 1.0.1 (Zhao *et al.* 2019). Bayesian regression models were fitted with *mcp* v. 0.3.4 (Lindeløv 2020) and *rjags* v. 4.17 (Plummer 2025). Diagnostics and summaries for Bayesian regression models were computed using *bayesplot* 1.11.1 (Gabry *et al.* 2019), *HDInterval* v. 0.2.4 (Meridith & Kruschke 2022) and *posterior* v. 1.6.1 (Bürkner *et al.* 2025). Graphics were prepared using *ggplot2* v. 3.5.1 (Wickham 2016), *ggtext* v. 0.1.2 (Wilke & Wiernik 2002) and *patchwork* v. 1.3.0 (Pedersen 2024).

RESULTS

Population size results

Independent territory counts agreed closely with breeding pair counts (Figure 2; median difference = -0.5, IQR = -1.0 to 0).

South Ballina

The South Ballina breeding pair count time series spanned 1994–2024 with a gap at 2011–2015 and a missing year at 2020 (Table 1). Territory counts were used to fill some missing years (Figure 3a).

BEAST estimated mean three trend change points: four approximately flat segments (median slope order = 0.34, 0.53, 0.38, 0.43 and corresponding median zero slope Pr = 0.66, 0.46, 0.60, 0.57 respectively) with three step changes. Mean 1.5 outlier change points were estimated.

Change point regression estimates (Table 2) were:

- A 14.9 to 15.9 step increase in median fitted counts (Pr > 0 = 0.85) at 1999.5, *i.e.* at the start of formal conservation management in 2000.
- A 15.9 to 8.5 step decrease in median fitted counts (Pr < 0 = 1) at 2006.5, during the pipi crash.
- A 8.5 to 11.1 step increase in median fitted counts (Pr > 0 = 0.97) at 2021.5, after South Ballina was closed to recreational driving on 31 March 2021.

The -3.5 pairs outlier at 2009 (change point $Pr = 0.91$; BEAST estimates) occurred at the end of the pipi crash (Figure 3a). The subsequent +1.3 pairs outlier at 2010 (change point occurrence $Pr = 0.57$; BEAST estimates) occurred during an unusual 'moon pipi' *Mactra contraria* mortality event that was noted in the 2011 Pied Oystercatcher Protection Program report. Moon pipis live in the subtidal depths and are usually not available to oystercatchers. Neither of those outliers influenced the change point regression trend estimates.

The southern five km of South Ballina, south of Boundary Creek, was generally ignored prior to 2000 (B. Totterman, pers. comm.). The +4 step increase in observed counts at 2000 (Figure 3a) is close to the observed three pairs south of Boundary Creek from 2001–2006 (nest locations prior to 2001 were not available) and the median three pairs (IQR = 2–4) for 2001–2024. Thus, counts for 1994–1999 are negatively biased.

The 31 March 2021 closure of South Ballina to recreational vehicles was close to the August–December laying season and population size didn't increase until the following 2022 breeding season (Figure 3a). Newcomers are assumed to start near the bottom of the local social order and are not expected to immediately occupy and vacant breeding space (Ens *et al.* 1996).

The net change in median fitted counts was -4.8 pairs ($Pr < 0 = 1$) between 1999.5–2006.5 and 2021.5–2024 (Table 2).

South Ballina beach front dwelling counts have increased from 19 in 2006 to 28 in 2024 (Figure 4). Eight of those nine additional dwellings are dispersed along the beach and six of those eight are unauthorised (pers. obs.). 10 of the 13 isolated dwellings have unauthorised direct beach access tracks across public foreshore land (pers. obs.).

Coffee Rocks

The conservation program largely ignored the Coffee Rocks interval prior to c. 2016. The breeding pair count time series spanned 2016–2024 with a missing year at 2020. Territory counts extended that time series back to 2012 (Figure 3b).

BEAST estimated mean one trend slope change point: an approximately flat segment (median slope order = 0.49, median slope zero $Pr = 0.51$) followed by a positive linear trend (median order = 0.99, median positive $Pr = 0.99$) (Figure 3b). Mean 0.6 outlier change points were estimated.

Change point regression estimates (Table 3) were a 0.4 median fitted count ($Pr > 0 = 1$) to 2017.6 followed by a median +1 pairs/year slope ($Pr > 0 = 1$).

The -0.5 pairs outlier at 2018 (change point $Pr = 0.60$; BEAST estimates) corresponded to zero breeding pairs (Figure 3b). The zero count appears to be an error because the individually marked breeding pair (leg flags A3 and L3) was resident in the Coffee Rocks from 2013–2024 (pers. obs.). The outlier did not influence the change point regression trend estimates.

Territory counts were lower than breeding pair counts by two pairs in 2022 and one in 2024, because territory mapping covered only the northern half of the Coffee Rocks (which explains two of the negative errors in Figure 2).

Airforce

The Airforce time series spanned 2001–2024 with four missing years. Territory counts were used to fill the missing year 2015 (Figure 3c).

Count variance was essentially zero and no statistical analysis was required. A descriptive intercept + intercept model was fitted to mode counts: two pairs to 2017 and three pairs from 2018 (Figure 3c).

Bombing Range

The Bombing Range time series spanned 2002–2024 with three missing years (Figure 4a).

BEAST estimated mean 0.1 (practically zero) trend change points: a constant positive linear trend (median slope order = 1, median positive slope Pr = 0.99). Mean one outlier change point was estimated.

Regression estimates (Table 4) were a 4.6 median fitted count intercept (Pr > 0 = 1) and median +0.23 pairs/year (Pr > 0 = 1) trend.

The –3.4 pairs outlier at 2014 (change point Pr = 0.99; BEAST estimates) occurred during the period 2011–2015 when the contractors for the conservation program were changing (Appendix). 2014 was the first year for the contractor Reconeco Pty Ltd and it is assumed that they were unfamiliar with the sites, existing breeding pairs and methods. That outlier did not influence the change point regression trend estimates.

Black Rocks

The conservation program did not cover Black Rocks and Ten Mile before 2005. The Black Rocks time series spanned 2005–2024 with four missing years (Figure 5b).

BEAST estimated mean 0.9 trend change points: two approximately flat segments (median slope order = 0.28, 0.53; median zero Pr = 0.70, 0.46) with one step change. Mean 0.6 outlier change points were estimated.

The BEAST trend change point occurrence probability distribution is diffuse and bimodal (Figure 5b). The 2019 trend change point (occurrence Pr¹ = 0.33) is confused with an outlier change point (change point Pr = 0.57) while the 2022 trend change point (occurrence Pr = 0.15) is simultaneous with an apparent step change.

The 2019 outlier was set aside and the step change point regression model assumed a uniform change point prior at 2021 ± 3 years. Estimates (Table 5) were a 2.2 to 4.3 step increase in median fitted counts (Pr > 0 = 1) at 2021.6.

Ten Mile

The Ten Mile time series spanned 2005–2024 with a gap at 2010–2015 (Figure 5c).

Count variance was essentially zero and no statistical analysis was required. A descriptive intercept + intercept model was fitted to mode counts: zero pairs to 2017 and 2 pairs from 2018.

Richmond River area

Rarely were all sites surveyed in a given year and the Richmond River area breeding pair count time series was poor (Table 1). The trend was estimated as the sum of the individual sites.

1 Change point occurrence probability (cpOccPr) is the probability of having a change point in the trend at a single point in time. Change point probability (cpPr), as reported in the BEAST model summary, is calculated by applying sum filtering to the cpOccPr distribution (*i.e.* cpPr > cpOccPr). Sum-filtering is not applicable to outliers.

The conservation program formally commenced at South Ballina in 2000. Trends for other sites were back-cast to 2000 assuming no change in median fitted counts before conservation management commenced (Figures 3b, 3c and 5).

The median predicted count for the Richmond River area was 25.1 pairs (95% Equal-Tailed Interval (ETI) = 20.2–29.9) in 2000 (Figure 6). The observed count was 37 pairs in 2024. The estimated net change for that period was +11.9 pairs (95% ETI = 7.1–16.8).

The estimated +15.5 pairs (95% ETI = 12.4–18.4) recent increase in median fitted counts from 2017–2024 largely occurred at sites without beach driving and/or lower human recreation disturbance: South Ballina after 2021, Coffee Rocks, Bombing Range and Black Rocks (Table 6). Those spatial shifts are evident in nest location plots (Figure 7), especially the southwards shift to Broadwater NP in the South Ballina embayment.

Further, the absence of nests close to Patches Beach, where human recreation disturbance is frequent, was previously noted by Harrison (2009). Beach walkers (with or without dogs) commonly travel several hundred meters along shore (mean = 809 m, 95th percentile = 1990 m, $n = 99$ in Totterman 2020a), consistent with the observed gap in the distribution of oystercatcher breeding territories. Dogs are prohibited more than 500 m from Patches Beach (NSW Department of Lands 2007) however Ballina Shire Council has ignored requests to install relevant signage and they do not carry out enforcement patrols at Patches Beach (pers. obs.).

Reproductive rate results

Reproductive rate varied among sites (Figure 8, Table 7):

- The Bombing Range median 0.73 fledglings/pair, where there is no public access, was 1.6× larger than the South Ballina mean 0.46, where beach driving was unrestricted to 31 March 2021 ($Pr > 1 = 1$).
- The Coffee Rocks median 0.84 fledglings/pair was 1.8× larger than the South Ballina mean 0.46 ($Pr > 1 = 0.98$), although the Coffee Rocks sample size was small ($n = 7$).
- The Airforce median 0.71 fledglings/pair was 1.5× larger than the South Ballina mean 0.46 ($Pr > 1 = 0.97$), although the lower credible interval estimate was < 1 .

The large mean and high variance at Airforce appears to be an artefact of the small breeding population size. For example, with two pairs and a mean clutch size of two, there are only five possible mean fledglings per pair results ($0/2 = 0.0$, $1/2 = 0.5$, $2/2 = 1.0$, $3/2 = 1.5$ and $4/2 = 2.0$) and only in the worst case of zero fledglings is the mean < 0.5 . Similarly, the two fledglings/pair outliers in the Coffee Rocks sample (Figure 8) occurred when there was only one breeding pair (Table 1).

Median fitted counts (= median fitted reproductive rate × corresponding breeding pair count) overestimate low fledgling counts (reproductive rate $<$ median) and underestimate high fledgling counts (reproductive rate $>$ median). Additional variables for the regression model that might predict departures from the average reproductive rate were not available (e.g. the identity of individual breeding pairs) or were unusable (e.g. fox baiting was highly imbalanced among sites; Table 8).

Pipi results

Only South Ballina had frequent pipi counts, spanning 2009–2024 (Figure 9).

Small sample sizes and high variance resulted in low confidence for pre-2009 mean pipi densities. The unreal 131 pipis/m² ($n = 5$; 95% Confidence Interval = 38–312) reported for South Ballina in 1997 (Owner & Rohweder 2003) has never been seen again for any beach in NSW and southeast Queensland (Totterman 2018, 2019, 2020b, 2020c). Poor quality pre-2009 pipi data are ignored.

Scatter plots (Figure 10) indicated:

- A weak and imprecise $r_s = +0.4$ ($n = 13$; 95% Confidence Interval (CI) = -0.2 to 0.8) correlation between breeding pair count density and pipi count density.
- A zero, $r_s = 0.06$ ($n = 8$; 95% CI = -0.7 to 0.7) correlation between reproductive rate and pipi count density.

Wide confidence intervals for those correlations reflect both the high scatter and small sample sizes.

DISCUSSION

Reproductive rate hypothesis

The increase in Richmond River area oystercatcher breeding pair counts from the predicted median 25 in 2000 to 37 in 2024 seems to support the hypothesis that high production of fledglings will cause population size to increase.

Results that do not support the reproductive rate hypothesis are:

- the hundreds of surplus fledglings produced from 1997–2024;
- the bulk of the estimated increase was delayed to 2017–2024 (*i.e.* approximately four generation lengths after conservation management commenced) and;
- the decrease in South Ballina median fitted counts from 16 in 2000–2007 to 11 in 2022–2024.

Fledgling production has also been surplus to the NSW beach-resident oystercatcher population size, which did not increase from 232 individuals in 1998 (Owner & Rohweder 2003) to 222 in 2015–2018 (Totterman 2020b).

Low reproductive rate does not appear to be a putative agent of decline because the association with breeding population size has not been convincingly shown in any study to-date. Worse, the reproductive rate hypothesis could be a silly mistake because population size is usually limited by territorial behaviour (Taylor *et al.* 2014).

Recording of breeding outcomes for individual breeding pairs requires frequent nest visits and is expensive. If population size is not limited by reproductive rate then conservation management does not require fledgling counts. Territory mapping could replace intensive nest monitoring because:

- breeding population size is a relevant conservation outcome;
- comparisons in this study and previously (Totterman 2020b) show that territory mapping provides accurate estimates of breeding population size, and;

- territory mapping is fast and cheap.

Further, intensive fox baiting only to increase oystercatcher reproductive rate appears to be wasteful of resources.

Food resource hypothesis

The observed decrease in South Ballina breeding pair counts from 16 pairs in 2004 to five in 2009 simultaneous with the pipi crash appears to support the hypothesis that low food abundance will cause population size to decrease.

Results that do not support the food resource hypothesis are:

- South Ballina breeding pair counts did not recover strongly with pipi stocks after the pipi crash, which suggests that other processes were acting;
- the weak and inconclusive correlation between breeding pair count density and pipi count density, and;
- the zero correlation between reproductive rate and pipi count density across all sites.

Those weak correlations resulted despite the large variance in pipi count density.

Food resource does not appear to be a putative agent of decline because the association between breeding oystercatchers and pipis has not been shown. Further, if depredation of eggs and chicks is the major cause of low reproductive rate then a strong correlation between reproductive rate and pipi abundance cannot occur.

If oystercatcher breeding populations usually are insensitive to variance in pipi count density (the pipi crash was an unusual and temporary event) then conservation management does not require regular pipi counts.

Human recreation disturbance hypothesis

Results that support the hypothesis that human recreation disturbance can cause oystercatcher population size to decrease are:

- the net decrease in South Ballina median fitted counts from 16 in 2000–2006 to 11 in 2022–2024, simultaneous with increases in vehicle-based recreation (pers. obs.) and beach front dwellings;
- the increase in South Ballina median fitted counts from 9 in 2007–2021 to 11 in 2022–2024, following the closure of the beach to recreational vehicles on 31 March 2021, and;
- the general ‘flight to safety’ where breeding pair counts have increased largely at sites with fewer people.

Further, median reproductive rate at Bombing Range, which is closed to the public, was 1.5× higher than at South Ballina.

A result that does not support the human recreation disturbance hypothesis is the persistent two–three breeding pairs at Airforce where vehicle-based recreation is intense (pers. obs.). Airforce differs from South Ballina in that the northern 4.2 km of the beach is adjacent to Broadwater NP, with zero residential development and two of the oystercatcher territories are adjacent to Salty Lake, which may afford greater protection than the open beach. Further, mean reproductive rate for that small population is inflated (as explained in the Results).

Recent increases in breeding pair counts at Coffee Rocks and Black Rocks are surprising because those sites were not recognised as breeding habitat early in the conservation program. As Caughley (1994) proposed, a declining species may end up not in the habitat most favourable to it, but in the habitat least favourable to an agent of decline.

The closure of South Ballina to recreational vehicles on 31 March 2021 can be interpreted as an experimental manipulation without replication of intervention sites. No simultaneous trend change point was detected for any of the 'control' sites except, perhaps, Black Rocks (where the change point location was unclear). A weak conclusion is that the removal of vehicle-based recreation was the *cause* of the recent increase in the South Ballina breeding population size.

Human recreation disturbance is clearly a putative agent of decline. The next and necessary step is test that hypothesis with designed experiments.

The Richmond River area oystercatcher management strategy (NSW Department of Lands 2007) recognised several human recreation disturbance related issues including:

- beach-user awareness and behaviour;
- recreational four-wheel driving and unauthorised camping;
- horse riding;
- dog exercising, and;
- beach front residential development and associated informal pedestrian beach access tracks.

Other than the closure of South Ballina to recreational vehicles, there has been little or no progress towards managing the above issues. Thus, the long-term outlook for beach-nesting oystercatchers in NSW (e.g. to the year 2100) is for increasing human recreation disturbance to drive further fragmentation and decline of the population.

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SUPPLEMENTARY MATERIAL

Supplementary Material for this article includes data and code for the population size and reproductive rate models.

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APPENDIX

Richmond River Area Pied Oystercatcher Protection Program 2001–2013 and Shorebird Protection Program 2014–2024 reports were released in accordance with the NSW *Government Information (Public Access) Act* 2009 (references OEH 19-452, DPIE 20-1043, DPE 23-2807, DCCEEW-24-3366, DCCEEW 50193).

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Table 1. Oystercatcher breeding and pipi abundance results for the six sites in the Richmond River area: SB = South Ballina; CX = Coffee Rocks; AF = Airforce; BR = Bombing Range; BX = Black Rocks; TM = Ten Mile. Breeding data sources were: 1) Wellman *et al.* (2000) to 1999, and; 2) unpub. reports to the NSW DPIE thereafter (Appendix). Pipi data sources were: 1) Owner and Rohweder (2003); 2) Harrison (2009); 3) Totterman (2018); 4) Totterman (2019); 5) Totterman (2020b), and; 6) this study. Harrison (2009) did measure pipi abundance in 2005 but did not report those results for individual beaches. Blank results indicate no data including unsatisfactory oystercatcher monitoring in 2013 and 2015. There was some fox control for South Ballina in 2011–2016 but it did not cover the long Crown Reserve section. Fox baiting for Coffee Rocks and Ten Mile was mostly on fire trails parallel to and close to the beach.

Year	Breeding pair count						Fledgling count						Fox baiting						Pipi abundance					
	SB	CX	AF	BR	BX	TM	SB	CX	AF	BR	BX	TM	SB	CX	AF	BR	BX	TM	SB	AF	BR	TM	Source	
1994	16						1																	
1995	15						0																	
1996	15						0																	
1997	14						4						Y						131	12		17	1	
1998	14						11						Y											
1999	14						3						Y											
2000	18						8						Y											
2001	17		2				14		2				Y		Y									
2002	17		2	4			8		2	1			Y		Y	Y								
2003	16		2	5			11		0	6			Y		Y	Y			20			26	2	
2004	16		2	5			1		2	5			Y		Y	Y								
2005	15		2	5	3	0	4		1	3	2		Y		Y	Y	Y	Y						
2006	14		2	7	2	1	9		0	10	1	0	Y		Y	Y	Y	Y						
2007	10		2	6	2	0	1		1	6	0		Y		Y	Y	Y	Y						
2008	9		2	7	2	0	4		1	7	1		Y		Y	Y		Y						
2009	5		2	6	3	0	3		1	6	0		Y		Y	Y		Y	0.0				3	
2010	12		2	7	3		0		1	2	0		Y		Y	Y			1				3	
2011																			4					3
2012			2	7	3				2	2	1				Y	Y			7					3
2013															Y	Y			16					3

2014			3	4						1	3				Y	Y		20			4
2015															Y	Y	Y		4		4
2016	7	1	2	9	3	1	4	2	4	8	3	0		Y	Y	Y	Y	11		4	5
2017	9	1	2	8	2	0	7	2	2	6	1		Y	Y	Y	Y	Y				
2018	7	0	3	8	2	2	6		2	7	1	0	Y	Y	Y	Y	Y	4		6	5, 6
2019	7	2	4	8	0	1	2	2	2	3		1	Y	Y	Y	Y	Y				
2020				9	1	2				11	1	3	Y			Y		6			6
2021	8	4	3	9	2	2	2	1	4	5	5	0	Y	Y	Y	Y	Y				
2022	13	5	3	10	4	2	4	7	5	7	1	1	Y			Y	Y	11			6
2023	11	6	3	9	4	2	7	6	1	2	3	1	Y			Y	Y				
2024	10	6	3	11	5	2	3	2	1	6	5	1	Y			Y	Y	12			6

Table 2. South Ballina oystercatcher breeding population size change point regression results. Estimates are posterior median and 95% Highest Density Interval (HDI). The fitted trend is plotted in Figure 3a. Contrast probability (Pr) is one-tailed (*i.e.* Pr > 0 if the median is positive and vice versa). Markov Chain Monte Carlo (MCMC) sampling was run with 5 chains, 3000 burn-in samples, 25000 draws for each chain and no thinning (25000 × 5 = 125000 total draws). Convergence diagnostics are improved \hat{R} (rounded to three decimal places; $\hat{R} < 1.01$ indicates that the chains have converged to a common distribution) and bulk Effective Sample Size ($ESS > 0.1 \times n$ draws is acceptable sampling efficiency) (Vehtari *et al.* 2021). Outliers (B) are BEAST estimates.

Segment	Segment		Change point		Segment contrast	
	Median [95% HDI]	\hat{R} , ESS	Median [95% HDI]	\hat{R} , ESS	Contrast	Median [95% HDI], Pr
Intercept 1	14.9 [13.4, 16.3]	1, 82931	1999.5 [1998, 2001.6]	1, 49791		
Intercept 2	15.9 [14.6, 17.3]	1, 87271	2006.5 [2006, 2007.0]	1, 65469	Int. 2 – Int. 1	+1.1 [-1.0, 3.1] 0.85
Outlier 1 ^B	-3.5 [-5.4, 0]		2009			
Outlier 2 ^B	+1.3 [0, 3.4]		2010			
Intercept 3	8.5 [7.6, 9.5]	1, 83475	2021.5 [2020, 2022.9]	1, 43847	Int. 3 – Int. 2	-7.4 [-9.1, -5.8] 1
Intercept 4	11.1 [9.0, 13.3]	1, 72913			Int. 4 – Int. 3	+2.6 [0.2, 5] 0.97
					Int. 4 – Int. 1	-3.7 [-6.3, -1.1] 1
					Int. 4 – Int. 2	-4.8 [-7.4, -2.3] 1

Table 3. Coffee Rocks oystercatcher breeding population size change point regression results. The fitted trend is plotted in Figure 3b. Statistics are defined in the caption to Table 2. MCMC sampling was run with 5 chains, 3000 burn-in samples, 25000 draws for each chain and no thinning. Outliers (B) are BEAST estimates.

Segment	Segment		Change point		Segment contrast	
	Median [95% HDI]	\hat{R} , ESS	Median [95% HDI]	\hat{R} , ESS	Contrast	Pr
Intercept 1	0.4 [0.0, 0.8]	1, 34809	2017.6 [2016.3, 2018.5]	1, 11620	Int. 1 > 0	1
Slope 1	1.0 [0.7, 1.2]	1, 14392			Slp. 1 > 0	1
Outlier 1 ^B	-0.5 [-1.9, 0]		2018			

Table 4. Bombing Range oystercatcher breeding population size regression results. There was no trend change point (*i.e.* the Intercept and Slope are one linear trend segment). The fitted trend is plotted in Figure 5a. Statistics are defined in the caption to Table 2. MCMC sampling was run with 5 chains, 3000 burn-in samples, 25000 draws for each chain and no thinning. Outliers (B) are BEAST estimates.

Segment	Segment		Change point		Segment contrast	
	Median [95% HDI]	\hat{R} , ESS	Location	\hat{R} , ESS	Contrast	Pr
Intercept and Slope	4.6 [3.7, 5.6]	1, 21653			Int. > 0	1
	0.23 [0.16, 0.3]	1, 21636			Slp. > 0	1
Outlier 1 ^B	-3.4 [-3.7, -2.2]		2014			

Table 5. Black Rocks oystercatcher breeding population size change point regression results. The fitted trend is plotted in Figure 5b. Statistics are defined in the caption to Table 2. MCMC sampling was run with 5 chains, 3000 burn-in samples, 25000 draws for each chain and no thinning. Outliers (B) are BEAST estimates.

Segment	Segment		Change point		Segment contrast	
	Median [95% HDI]	\hat{R} , ESS	Median [95% HDI]	\hat{R} , ESS	Contrast	Median [95% HDI], Pr
Intercept 1	2.2 [1.6, 2.8]	1, 91849	2021.6 [2020.0, 2023.4]	1, 41944		
Intercept 2	4.3 [2.9, 5.6]	1, 78653			Int. 2 – Int. 1	2.1 [0.6, 3.5] 1
Outlier 1 ^B	-0.6 [-2.6, 0]		2019			

Table 6. Richmond River area oystercatcher breeding population size changes for 2017–2024 (Figures 3, 5 and 6). South Ballina and Coffee Rocks were closed to recreational vehicles on 31 March 2021 (before the 2021 breeding season). Median and 95% Equal Tailed Interval (ETI, *i.e.* the 2.5–97.5% quantiles interval) estimates were computed from 10000 samples from the posterior distribution of each change point regression model. These two-point estimates can differ slightly from parameter contrasts in Tables 2–5.

Site(s)	Change 2017–2021		Change 2021–2024		Change 2017–2024
	Median [95% ETI]	Beach driving	Median [95% ETI]	Beach driving	Median [95% ETI]
South Ballina	0 [0, 2.9]	Yes	+2.5 [0, 4.8]	No	+2.7 [0, 4.8]
Coffee Rocks	+3.3 [2.6, 3.8]	Yes	+2.9 [2.2, 3.6]	No	+6.2 [5.0, 7.2]
Airforce	+1	Yes	0	Yes	+1
Bombing Range	+0.9 [0.6, 1.2]	No	+0.7 [0.5, 0.9]	No	+1.6 [1.1, 2.1]
Black Rocks	0 [0, 1.8]	No	+2.1 [0, 3.4]	No	+2.1 [0.6, 3.4]
Ten Mile	+2	Yes	0	Yes	+2
Sum (Richmond R. area)					+15.5 [12.4, 18.4]

Table 7. Richmond River area oystercatcher reproductive rate regression model results and fledgling sums. Statistics are defined in the caption to Table 2. Ratio contrasts result from the log link function in the regression model. Pairwise contrasts are not adjusted for multiple comparisons and the probability (Pr) is one-tailed (*i.e.* Pr > 0 if the log-scale contrast is positive and vice versa). Only contrasts with Pr ≥ 0.90 are reported. The model assumed a negative binomial error distribution for annual fledgling counts with over dispersion parameter *d*. MCMC sampling was run with 5 chains, 3000 burn-in samples, 50000 draws per chain and no thinning (50000 × 5 = 250000 total draws)

Site / Parameter	Reproductive Rate (fledglings/pair)		Site contrasts		Sum of fledglings (year range)
	Median [95% HDI]	\hat{R} , ESS	Contrast	Median [95% HDI], Pr	
South Ballina (SB)	0.5 [0.4, 0.6]	1, 37133			116 (1997–2024)
Coffee Rocks (CX)	0.8 [0.5, 1.4]	1, 92459	CX / SB	1.8× [1.05, 3.1] 0.98	22 (2016–2024)
			CX / TM	1.9× [0.8, 4.5] 0.93	
Airforce (AF)	0.7 [0.5, 1.0]	1, 66916	AF / SB	1.5× [0.99, 2.3] 0.97	35 (2001–2024)
Bombing Range (BR)	0.6 [0.6, 0.9]	1, 47275	BR / SB	1.6× [1.1, 2.2] 1	106 (2002–2024)
			BR / TM	1.6× [0.8, 3.5] 0.91	
Black Rocks (BX)	0.6 [0.4, 0.9]	1, 79623			25 (2005–2024)
Ten Mile (TM)	0.5 [0.2, 0.9]	1, 121141			7 (2006–2024)
Dispersion (<i>d</i>)	0.13 [0.01, 0.27]	1, 117615			

Table 8. Fox baiting frequency for the six sites in the Richmond River area (counts of years; data in Table 2).

Site	Fox baiting frequency	
	Yes	No
South Ballina	21	1
Coffee Rocks	4	3
Airforce	17	3
Bombing Range	20	0
Black Rocks	3	12
Ten Mile	6	3



Figure 1. Map of the Richmond River area with beach-nesting oystercatcher sites delimited by tick marks and numbered 1–6. Green-shaded areas are National Parks (NP) and Nature Reserves (NR). Yellow-filled circles are towns and villages with > 100 people in the 2021 census (Australian Bureau of Statistics 2021). 2021 human population sizes were: Broadwater = 524; Wardell = 662, Woodburn = 678; Iluka = 1764; Evans Head = 2894; Yamba = 6342, and; Ballina = 18532. Patches Beach is a cluster of 14–15 dwellings (human population size = 45) close to the middle of South Ballina.

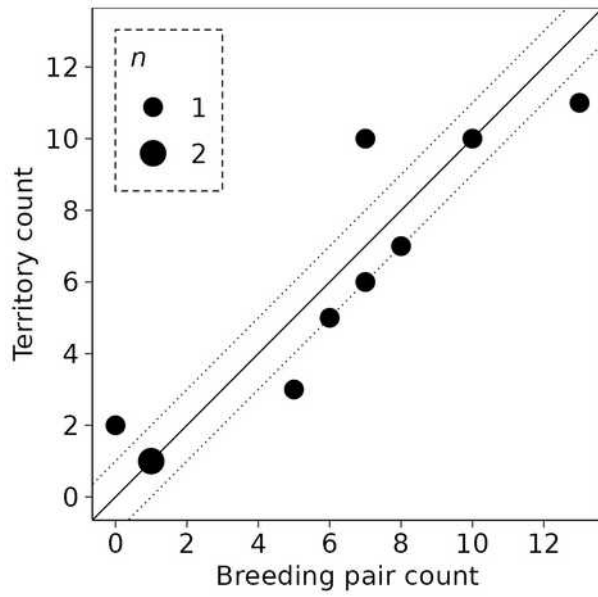


Figure 2. Oystercatcher independent territory counts compared to breeding pairs. Point sizes are proportional to the number of observations. 1:1 line of agreement (solid line) shown with ± 1 tolerance bands (dotted lines). A statistical comparison is not useful because the sample size ($n = 10$) is small and statistical power to detect differences is low.

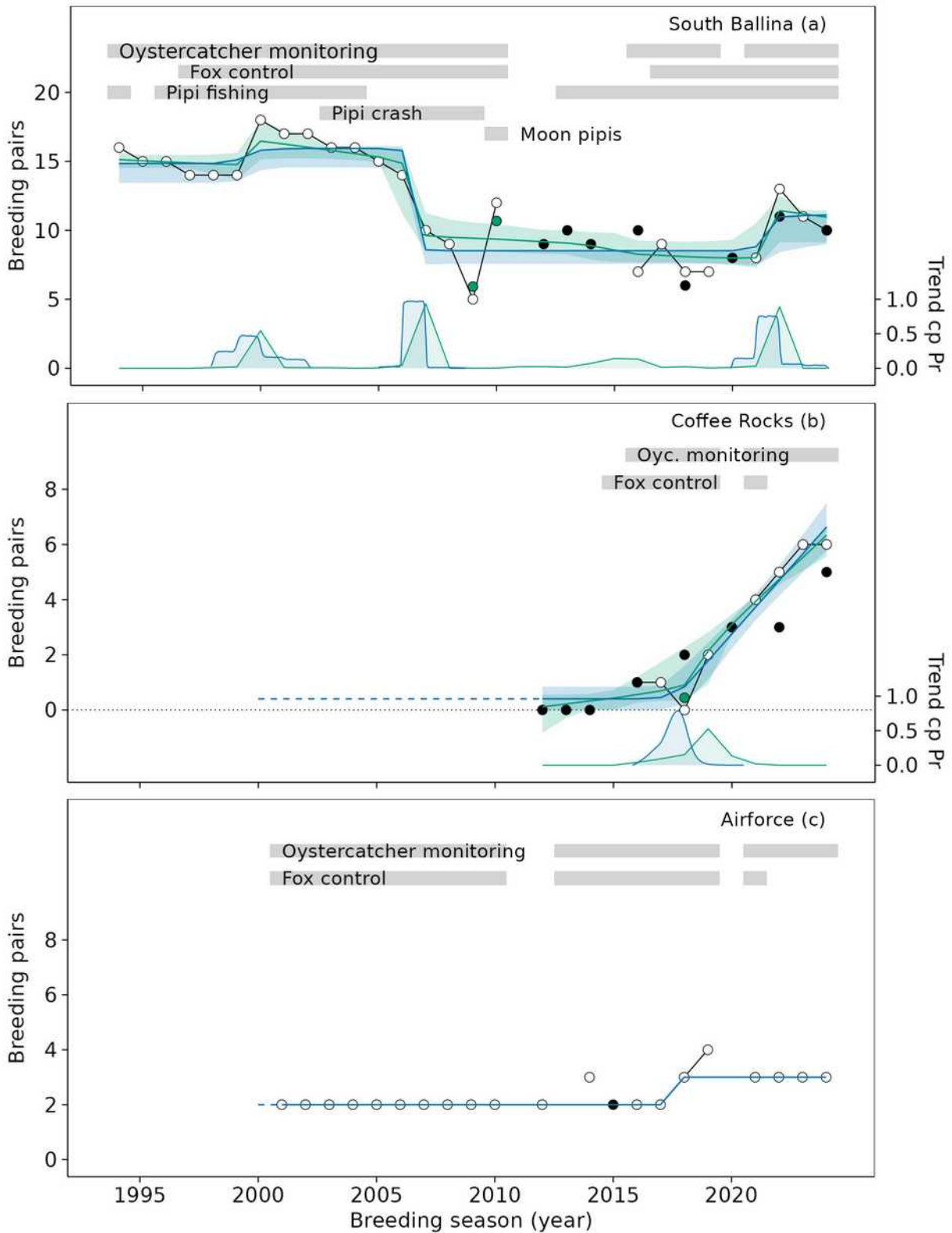


Figure 3. Oystercatcher breeding population size counts for the South Ballina embayment: breeding pairs (white-filled circles); territory counts (black-filled circles); BEAST estimates (green; green-filled circles are outliers), and; change point regression estimates (blue). Ribbons around median fitted trend lines are 95% credible intervals. Areas along the bottom are change point occurrence probabilities. Counts in (b) are shifted upwards to avoid over plotting (see the dotted line at zero). Dashed lines are back-cast estimates to the commencement of the conservation program at 2000. Variance was almost zero for Airforce (c) and no trend model was fitted for that site. The count scale in (a) is 2 × that in (b) and (c). Integer value BEAST change points are offset from continuous value regression model estimates.

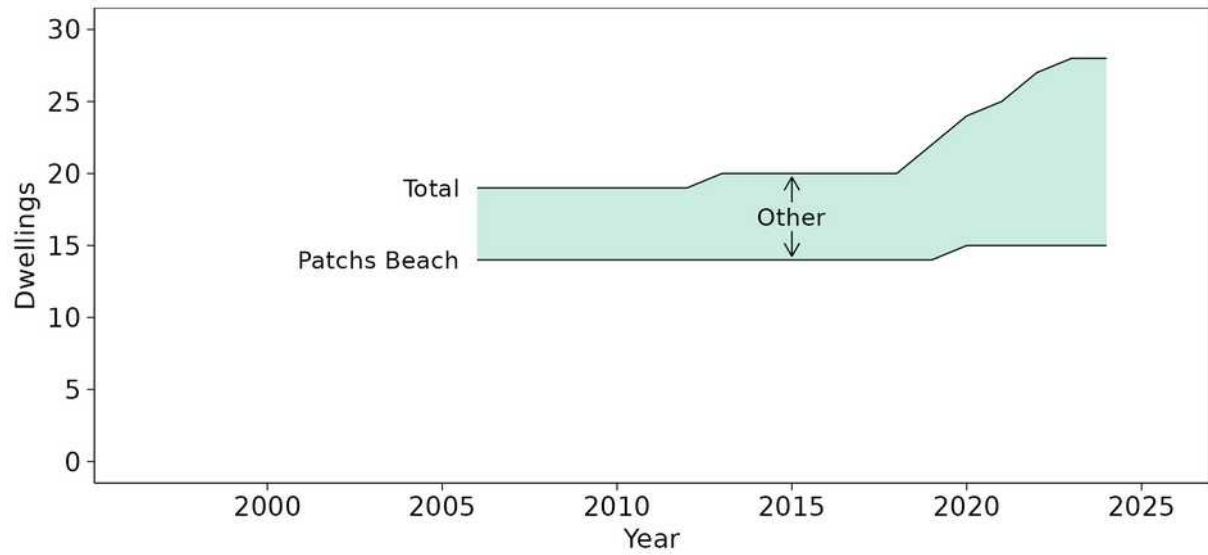


Figure 4. Beach front dwellings for South Ballina. Lines are Patches Beach (Figure 1) and Total counts. The difference (shaded area) represents Other dwellings that are dispersed along the beach. Dwelling counts are from Google Earth imagery dating back to 2006 (Google 2025).

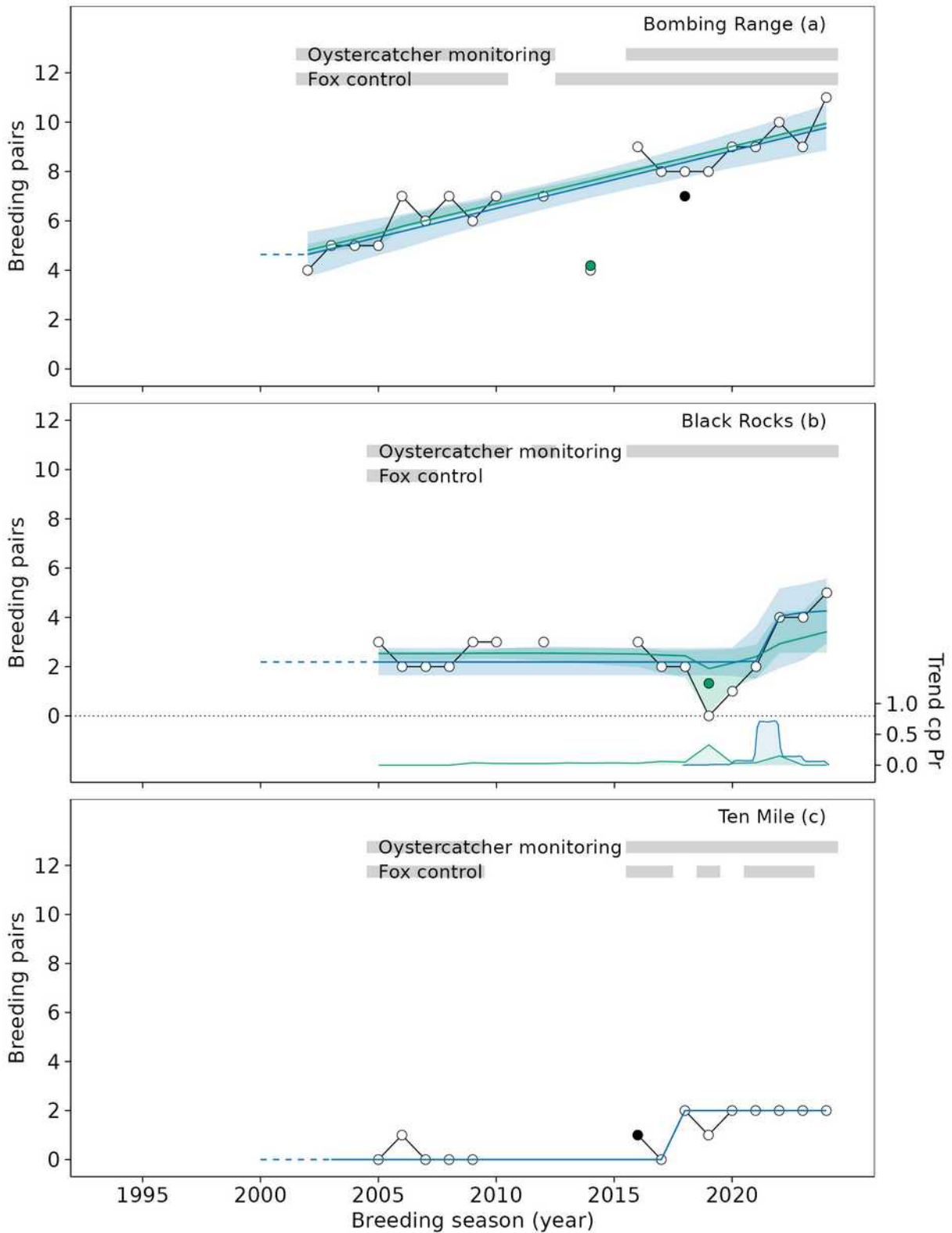


Figure 5. Oystercatcher Breeding population size counts for the Evans Head embayment. Plotting symbols are defined in the caption to Figure 3. Counts in (b) are shifted upwards to avoid over plotting (see the dotted line). Variance was almost zero for Ten Mile (c) and no trend model was fitted for that site. Pipi fishing is unknown for Ten Mile. The count scale in (a) is $2 \times$ that in (b) and (c).

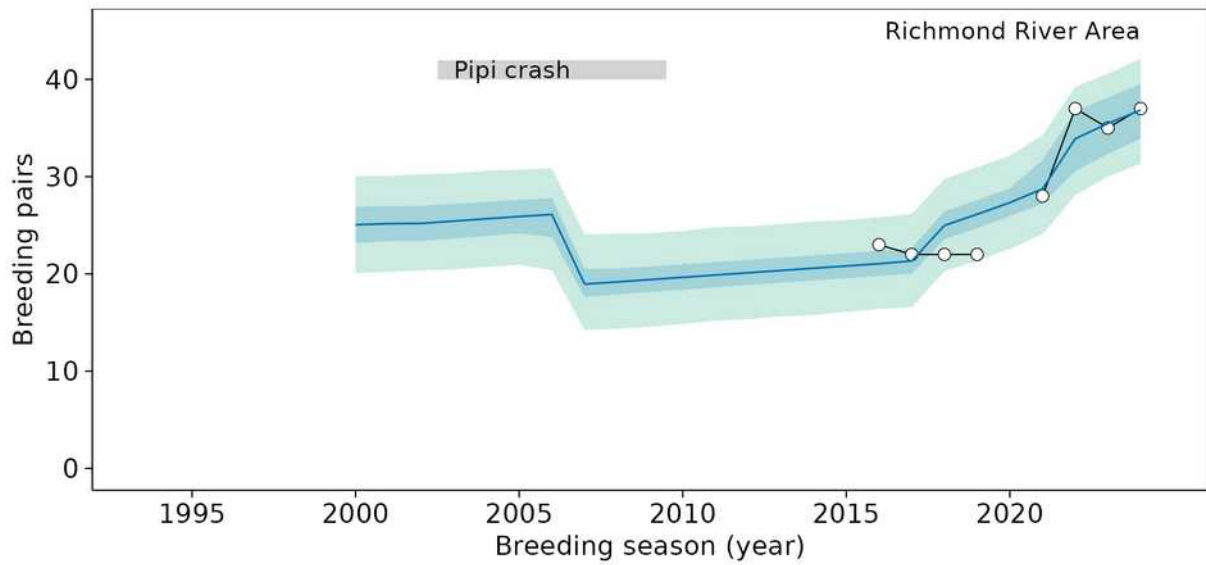


Figure 6. Oystercatcher breeding population size for the Richmond River Area (the sum of the six sites in Figures 3 and 5) from the commencement of the conservation program at South Ballina at 2000. White-filled circles are breeding pair counts. Ribbons around median fitted trend lines (blue) are 95% credible (blue) and prediction (green) intervals.

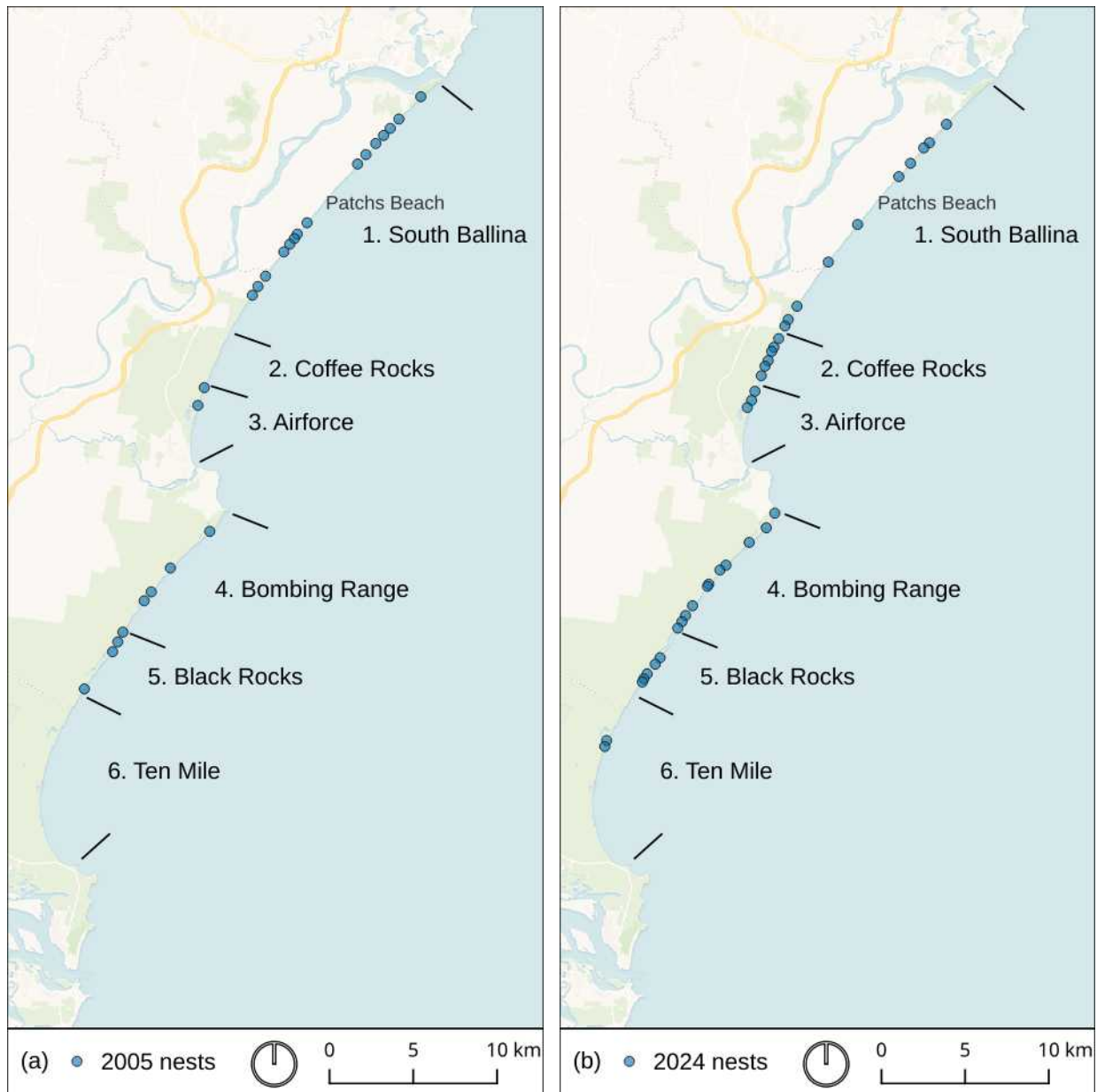


Figure 7. Oystercatcher nest location plots for 2005 (a; $n = 25$) and 2024 (b; $n = 37$). The first nest is plotted for pairs that made repeated attempts within a breeding season. Coffee Rocks was not monitored in 2005. Additional geographic information is provided in Figure 1.

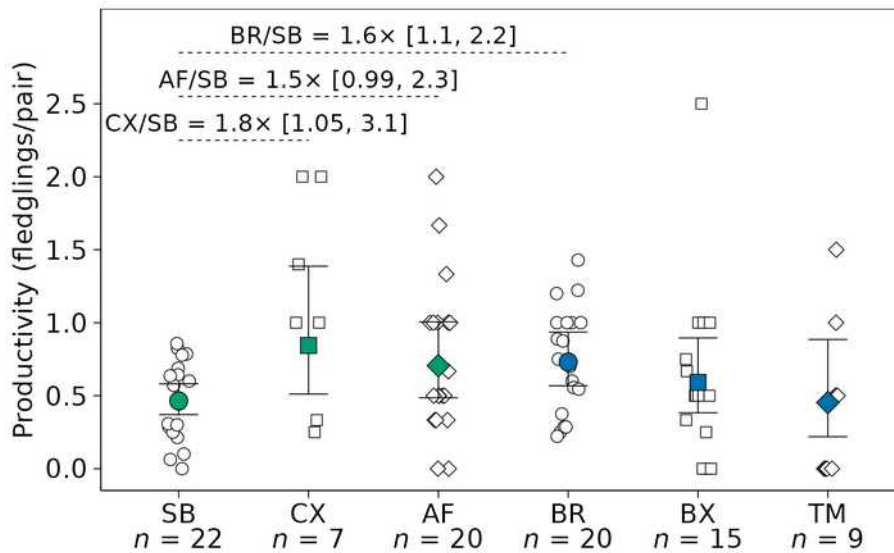


Figure 8. Oystercatcher reproductive rates for the six sites in the Richmond River area: SB = South Ballina; CX = Coffee Rocks; AF = Airforce; BR = Bombing Range; BX = Black Rocks, and; TM = Ten Mile. Plotted are: annual mean reproductive rates (white-filled points with a small amount of random horizontal 'jitter' to reduce over plotting); regression model fitted site medians (colour-filled points) and 95% Highest Density Intervals (error bars). Dashed, horizontal lines indicate pairwise comparisons (Table 7). The South Ballina sample excludes 1994–1996, before conservation management commenced.

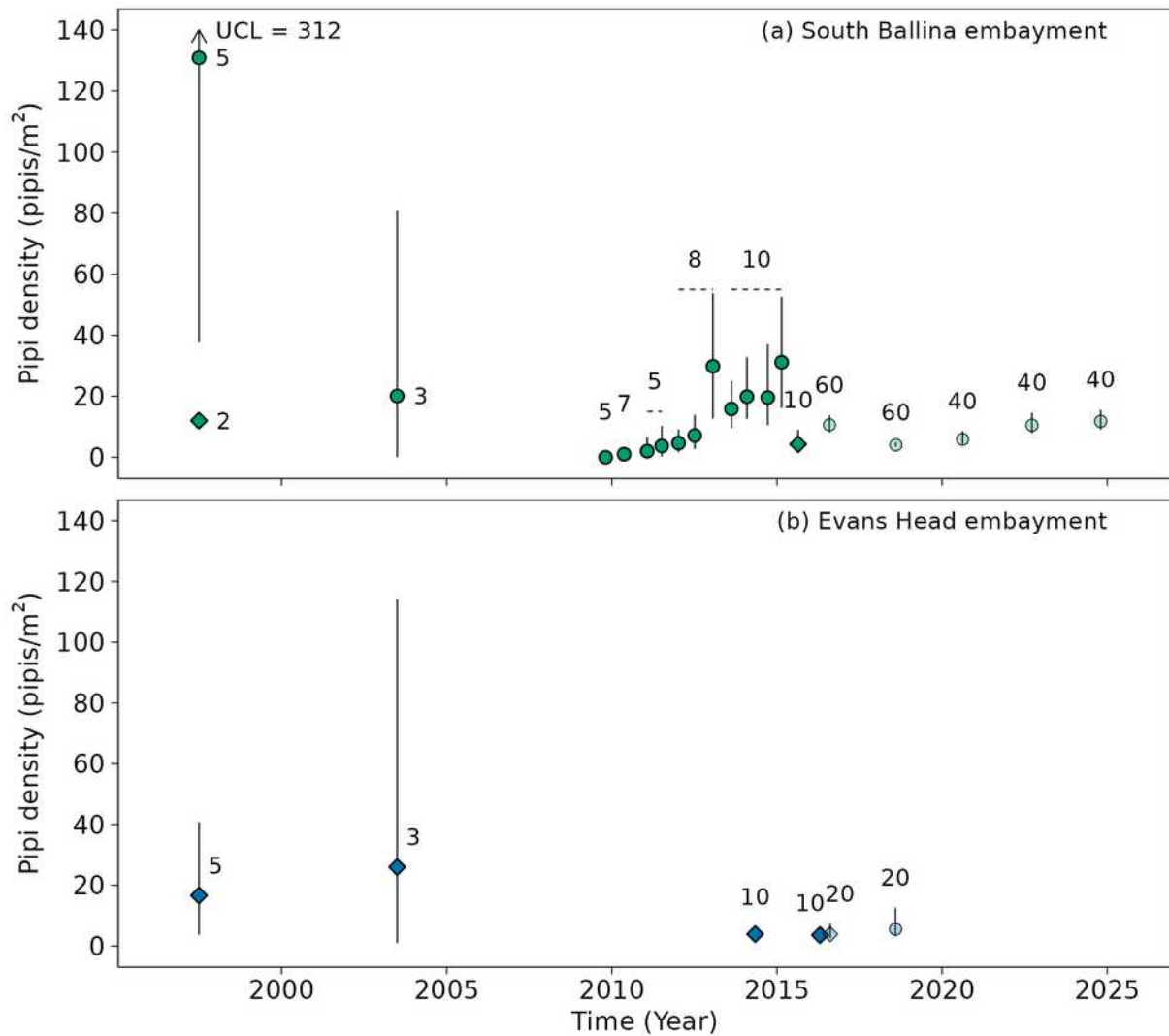


Figure 9. Pip abundance data for the Richmond River area: South Ballina (green-filled circles) and Airforce (green-filled diamonds) in (a); Bombing Range (blue-filled circles) and Ten Mile (blue-filled diamonds) in (b). Vertical lines are bootstrap 95% confidence intervals (Totterman 2020c). The Upper Confidence Limit (UCL = 312) for the 1997 South Ballina density (Owner & Rohweder 2003) is off the scale. Numbers above or beside points are sample sizes. Feet digging mean counts (semi-transparent points) were converted to equivalent pip band densities (Totterman 2019). Unlike other time series plots, the time scale is *continuous* with year axes ticks at 1 January and these plots include some oystercatcher non-breeding season results.

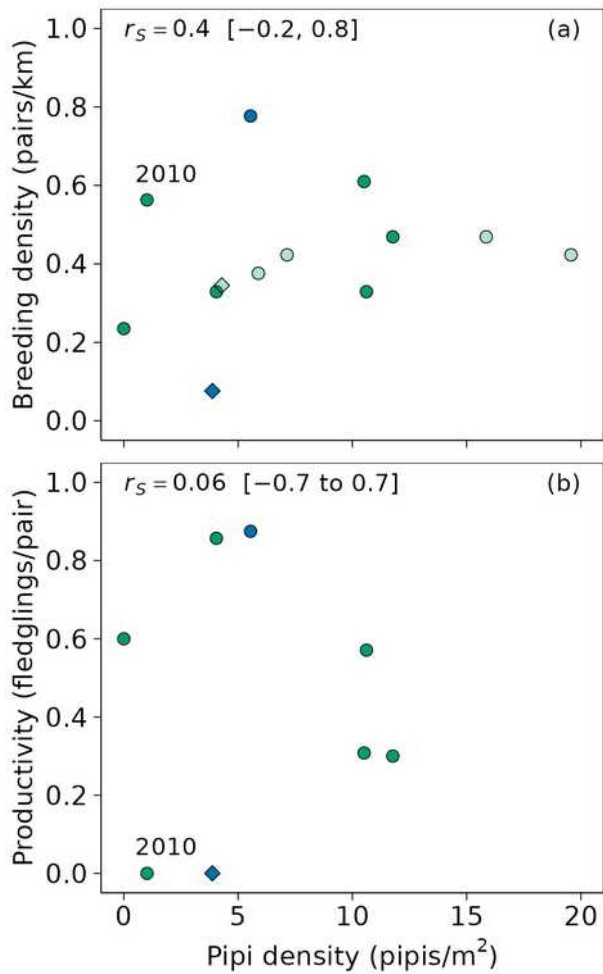


Figure 10. Breeding oystercatcher–pipi scatter plots: South Ballina (green-filled circles); Airforce (green-filled diamonds), Bombing Range (blue-filled circles), and; Ten Mile (blue-filled diamonds). Semi-transparent points in (a) are territory count densities. The unusual ‘moon pipi’ mortality event in 2010 resulted in a positive fluctuation in breeding pairs at South Ballina (Figure 3a). Spearman rank correlation coefficients are reported with 95% confidence intervals. The confidence interval estimates (that assume independent observations within the variables, *i.e.* zero autocorrelation) are useful as rough measures of precision but should not be relied upon for hypothesis testing.