TAKING FRIGHT: THE DECLINE OF AUSTRALIAN PIED OYSTERCATCHERS *Haematopus longirostris* AT SOUTH BALLINA BEACH, NEW SOUTH WALES

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ABSTRACT

This study reviewed data from the Richmond River Area Pied Oystercatcher Protection Program 1997–2013 and Richmond River Area Shorebird Protection Program 2014–2018, on the far north coast of New South Wales, Australia. The Australian Pied Oystercatcher *Haematopus longirostris* breeding population size for South Ballina Beach has decreased from 15–16 pairs in 1994–1996 to 7–9 pairs in 2016–2018 even though control of the European Red Fox *Vulpes vulpes* has been successful in reducing predation of eggs and chicks and > 208 oystercatchers have fledged from beaches in the Richmond River area between 1997–2018. The population for Bombing Range Beach has increased from 4–5 pairs in 2002–2004 to 8–9 pairs in 2016–2018. Vehicle-based recreation has increased strongly at South Ballina during the past decade versus Bombing Range is closed to the public. It is concluded that human recreation disturbance is preventing recruitment into the South Ballina oystercatcher breeding population. Without strong protection of habitat from such disturbance, the NSW oystercatcher breeding population size will continue to decrease in the next two decades.

INTRODUCTION

The Australian Pied Oystercatcher *Haematopus longirostris* inhabits estuaries and ocean beaches around the Australian coast, with major populations in the southern states of Tasmania, Victoria and South Australia (reviewed in Taylor *et al.* 2014). The species is non-migratory and many breeding pairs remain on their territories throughout the year. The species is long-lived and the breeding population size is usually limited by territorial behaviour and not by breeding success. Delayed age of first breeding is usual, resulting in a surplus of non-territorial adults in the population. This study examined breeding population size, *i.e.* the number of breeding individuals/pairs, and not the number of mature adults, which is a less focussed definition of population size (IUCN Standards and Petitions Committee 2019).

The Australian Pied Oystercatcher has been assessed as Endangered in New South Wales (NSW) because the estimated population size is low and projected or continuing to decline (NSW Scientific Committee 2010). The NSW Department of Planning, Industry and Environment (DPIE) (2019a, 2019b) has estimated that there are 200 oystercatcher breeding pairs in NSW and Totterman (2020) estimated that < 50 of those are 'beach residents', *i.e.* birds that forage, roost and spend most of the time on the beach. The Richmond River Area of the far north coast area (between the Richmond and Clarence Rivers) supports the largest numbers of beach resident

oystercatchers in NSW, with 22 or 23 breeding pairs between 2016–2018 (NSW DPIE unpub. data; this study).

Conservation threats for the Australian Pied Oystercatcher in NSW include habitat loss, human recreation disturbance, a decline of food resources and low breeding success, often because of predation of eggs and chicks by the European Red Fox *Vulpes vulpes* (NSW Scientific Committee 2010). Predation by the Red Fox was the first Key Threatening Process listed under the NSW Threatened Species Conservation Act 1995 (which has been replaced by the NSW Biodiversity Conservation Act 2016, NSW Government 2016). The first Threat Abatement Plan for predation by the Red Fox was published in 2001 (NSW National Parks & Wildlife Service 2001) with the primary objective to ensure that investment in fox control achieved maximum benefits for biodiversity. The number of priority sites for the Australian Pied Oystercatcher has increased from 13 in the first Threat Abatement Plan to 19 in the second and current plan (NSW Office of Environment & Heritage 2010).

This study gathered 20 years of breeding results for beach resident Australian Pied Oystercatchers in the Richmond River area and applied hypothesis testing to diagnose changes in the breeding population size (Sinclair *et al.* 2006). Conservation management for oystercatchers over this period has primarily involved fox control to reduce predation of eggs and chicks (Wellman *et al.* 2000). This breeding success strategy assumes that sufficient numbers of oystercatcher fledglings will: 1) survive to become adults; 2) stay in or return to NSW, and; 3) be successful in acquiring a breeding territory. The breeding success hypothesis tested in this study is that improved breeding success (*i.e.* reduced predation of eggs and chicks) will increase the oystercatcher breeding population size.

Oystercatchers are specialist predators of bivalves and Owner and Rohweder (2003) reported that Australian Pied Oystercatchers on the far north coast of NSW are associated with high abundance of the surf clam *Donax deltoides*, commonly known as the 'pipi'. Pipi stocks 'crashed' between 2003–2009 and the associated negative fluctuation in oystercatcher counts was misinterpreted by Harrison (2009) and then the NSW Scientific Committee (2010) as a reduction in the population size. Harrison (2009) further speculated that low pipi abundance can result in low oystercatcher breeding success. Totterman (2018) reported that oystercatcher counts on South Ballina Beach did increase when pipi stocks recovered (*i.e.* a numerical predator-prey response). However, Totterman (2020) reported that the far north coast oystercatcher-pipi association did not generalise to other regions. Two food resource hypotheses were tested in this study: 1) oystercatcher breeding population size is positively correlated with pipi abundance, and; 2) oystercatcher breeding success is positively correlated with pipi abundance.

Bird-habitat models for beach resident Australian Pied Oystercatchers in Totterman (2020) indicated a positive response to pipi abundance and a negative response to urbanisation, pedestrian access density and human recreation disturbance variables broadly. Birds typically respond to a perceived threat by moving away from the stimulus. Fisher *et al.* (1998) proposed that vehicle-based tourism on Fraser Island, Queensland, could negatively impact beach nesting bird populations via disturbance to their habitat. Harrison (2009) added that vehicle strikes are an additional source of mortality for shorebirds. Totterman (2020) noted that oystercatcher counts on

75 Mile Beach, Fraser Island, have decreased since Fisher *et al.* (1998) counted an average of 15 birds between Hook Point and Indian Head to a mean count of three in 2016. Two human recreation disturbance hypotheses are tested in this study is: 1) breeding population size will decrease at sites with increasing human recreation disturbance (*i.e.* effectively habitat loss), and; 2) breeding success is lower at sites with more intense human recreation disturbance.

METHODS

Study sites

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

South Ballina is the 23 km long continuous beach between the Richmond River and Broadwater Headland. A 2.3 km long 'coffee rock' interval separates South Ballina from Airforce Beach. The northern 2.5 km of South Ballina is part of Richmond River Nature Reserve, the southern 3.5 km adjoins Broadwater National Park and the remaining 17 km adjoins a narrow Crown Reserve, backed by private properties. Beach driving is freely allowed south of the Richmond River Nature Reserve.

Airforce is the 5.8 km long beach north of the Evans River. This site includes Salty Lagoon (29.077°S), an intermittently closed and open lake. The northern 4.3 km of Airforce adjoins Broadwater National Park. Beach driving is freely allowed north of the Evans Head township.

Bombing Range, Black Rocks and Ten Mile sites are contiguous sites adjoining Bundjalung National Park but are managed differently (Table 1). Bombing Range is the 10.5 km long beach between Goanna Headland and Jerusalem Creek. Public access to Bombing Range is prohibited because of the active Royal Australian Air Force Evans Head Air Weapons Range. The 4.8 km long Black Rocks' interval, where there is a campground, separates Bombing Range from the 12.5 km long southern end of Ten Mile Beach. Beach driving is freely allowed north of Shark Bay, ending at Black Rocks. South Ballina and Bombing Range are two Key Management Sites for the Australian Pied Oystercatcher in NSW (NSW DPIE 2019b).

To answer concerns that the results of this study are sensitive to these site definitions, analyses were repeated for the aggregate sites 'greater South Ballina Beach' (South Ballina and Airforce) and 'greater Ten Mile Beach' (Bombing Range, Black Rocks and Ten Mile).

Data sources

This study reviewed data from the Richmond River Area Pied Oystercatcher Protection Program 1997–2013 and Richmond River Area Shorebird Protection Program 2014–2018. Included are results from 1994–1996, preceding the start of fox control, and from 1997–2000, when there were no formal reports (Wellman *et al.* 2000). The conservation program was extended to include Airforce in 2001 and Bombing Range in 2002. Following the 2003–2009 pipi crash, the program was cancelled in 2011 and did not completely resume until 2016 (Table 2). The program was again cancelled in 2019, ostensibly because of the 2019 bushfire crisis (Anon., pers. comm.).

The oystercatcher breeding season in the Richmond River area is Aug–Dec and late attempts generally are replacement clutches (Wellman *et al.* 2000). Breeding pairs were located by territory mapping (*i.e* identification of 'pairs' usually precedes detection of nesting attempts) and prior knowledge (*i.e.* known breeding territories from preceding years). The frequency for oystercatcher monitoring was mostly between 1–3 days per week. Important results are counts of breeding pairs and young fledged. Breeding success (fecundity) was quantified as fledglings per breeding pair.

Pipi abundance data were obtained from several sources (Table 2). Pipis typically aggregate in discontinuous 'bands' that run parallel to the shoreline and, to be consistent with the majority of historical data, abundances are reported as pipi band mean density (pipis/m^2) in this study. The most recent pipi counts have used a 'feet digging' swash zone sampling method that is more efficient than quadrat sampling. Mean feet digging pipi counts were converted to 'pseudo pipi band densities' following Totterman (2019).

Statistical analysis

The statistical analysis applied linear regression and Analysis of Covariance (ANCOVA) to estimate and compare breeding population size trends, scatterplots to examine relationships between continuous variables and Analysis of Variance (ANOVA) to compare mean breeding productivities among sites.

Repeated data collection over time often leads to temporal autocorrelation, where a result at some point in time is related to, and not independent of, nearby values. The analysis strategy used was: 1) fit an ordinary least squares (OLS) model, 2) perform a Durbin-Watson test for first-order serial correlation in the OLS model residuals, 3) fit a generalised least squares (GLS) model with serial correlation in the residuals if required by 2). GLS adjusts the standard errors of OLS parameter estimates for more reliable statistical inference. A continuous autoregressive GLS correlation structure was used to accommodate temporally missing data.

The breeding population size ANCOVA included a site covariate:

$$y_{i,j} = a_j + b_j \times t_i$$

Where the response variable $y_{i,j}$ is the number of breeding pairs in year t_i at site j, a_j is the intercept for the linear trend for site j and b_j is the slope. This model effectively fits separate trend lines for each site and assumes a common serial autocorrelation structure for all sites.

With temporally missing data and multiple sites, Durbin-Watson tests were limited to contiguous time series within single sites. The threshold for Durbin-Watson tests was increased to $P < 0.1$ for greater statistical power for short time series. Other model assumptions were checked by plotting residuals versus fitted values, square-root absolute residuals versus fitted values and normal-probability plots.

All statistical analyses were performed using the software R version 3.5-2 (R Core Team 2018). Durbin-Watson tests for first-order autocorrelation were computed using the R package lmtest

version 0.9-36 (Zeileis & Hothorn 2002). GLS models were fitted using nlme version 3.1-137 (Pinheiro *et al.* 2018). Graphics were prepared using ggplot2 version 3.1.0 (Wickham 2016).

RESULTS

With seven Australian Pied Oystercatcher breeding pairs in 2018 (Table 2), South Ballina is estimated to hold > 14% (7/50) of NSW beach resident breeding oystercatchers or 3.5% (7/200) of total NSW breeding pairs. With eight breeding pairs, Bombing Range is estimated to hold > 16% (8/50) of beach resident breeding oystercatchers or 4% (8/200) of total breeding pairs. With only two or three breeding pairs for Airforce and Black Rocks and most often zero at Ten Mile Beach, these individual sites are too small for estimation of population size trends (*e.g.* a change of one is large relative to a breeding population size of two or three).

Oystercatcher breeding and pipi histories for South Ballina and Bombing Range are presented in Figure 2. Historical counts of 15–16 oystercatcher breeding pairs in 1994–1996 for South Ballina, before conservation management commenced, are nearly double the recent 7–9 pairs in 2016–2018 (Table 2, Figure 2a1). A negative fluctuation from 14 to 5 pairs between 2006–2009 occurred during the 2003–2009 pipi 'crash' (Figure 2a3). The temporary increase to 12 breeding pairs in 2010 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 foraging oystercatchers.

The maximum breeding pair density for South Ballina was 0.8 pairs/km in 2000 or 0.9 pairs/km for 18.7 km of available habitat (discounting 4.3 km of beach surrounding the hamlet of Patch's Beach where there have been zero recorded nests). The maximum density for Bombing Range was 0.9 pairs/km in 2016 and was most recently 0.8 pairs/km in 2018. This comparison suggests that the Bombing Range breeding population size could be limited by territorial behaviour at 8 or 9 pairs.

There have been > 208 oystercatchers fledged from beaches in the Richmond River area between 1997–2018. This number would increase if the 2011–2015 counts were complete. Despite this large fledgling result, the Richmond River area beach resident breeding population size has decreased from 25–26 pairs in 2005–2006 to 22–23 pairs in 2016–2018 (−0.04 pairs/year). There were insufficient count data for statistical evaluation of this near zero rate.

The breeding population size ANCOVA was a control-impact comparison. The major difference between the Bombing Range (control) and South Ballina (impact) is the absence of people at the former site. The South Ballina breeding population size has generally decreased at −0.4 pairs/year (95% CI = −0.6 to −0.2; Figure 2a1) versus Bombing Range has increased at +0.2 pairs/year (95% CI = −0.1 to +0.6; Figure 2b1). The year × site interaction $P = 0.005$ indicated different control-impact responses. Further evidence for this control-impact difference is the $r = -0.7$ correlation between South Ballina and Bombing Range breeding pairs (95% CI = −0.9 to −0.2; OLS residuals first-order autocorrelation = −0.5, Durbin-Watson statistic = 2.7, $P > 0.1$). Despite the response being a count variable, GLS model diagnostics were satisfactory (Figure 3).

For South Ballina and Airforce combined, the breeding population size has decreased at −0.5 oystercatcher pairs/year (95% CI = −0.9 to −0.2) while the combined Bombing Range, Black

Rocks and Ten Mile has increased at +0.3 pairs/year (95% CI = −0.2 to +0.8) (GLS first-order serial correlation = 0.5, 95% CI = 0.1–0.9). The year × site interaction result was $P = 0.01$. These results are similar to those for the individual sites South Ballina and Bombing Range above.

Pipi abundance at South Ballina has decreased from *c.* 20 to 4 pipis/m^2 between 2014−2018 (Figure 2a3) and Bombing Range pipi abundance in 2018 was similar with 6 pipis/m2 (Figure 2b3). Scatterplots suggests a moderate Spearman $r = 0.58$ correlation between breeding pair density (pairs/km) and pipi abundance (Figure 4a) but no substantial correlation between breeding productivity and pipi abundance (Spearman $r = 0.26$; Figure 4b). There was also no correlation between breeding productivity and breeding pairs density (Spearman $r = 0.02$; Figure 4c). There was insufficient pipi abundance data for a more detailed statistical analysis, taking into account temporal autocorrelation, and these correlations are reported without confidence intervals and/or *P*-values.

There were no apparent temporal trends in oystercatcher breeding productivity (Figures 3a2, 3b2) and an ANOVA-equivalent OLS linear model assuming constant mean productivity for each site was fitted (Figure 5). The highest mean productivity was 0.8 fledglings/pair for Bombing Range, with no public recreation, and the lowest were 0.5 fledglings/pair for South Ballina, with beach driving (−0.3 difference, 95% CI = −0.7 to +0.04), and 0.4 fledglings/pair for Black Rocks, adjoining a campground (−0.4 difference, 95% CI = −0.8 to +0.01). The Airforce productivity variance was high because of the small two to three pairs and discrete distribution. For example, for two pairs and an average clutch size of two, there are only five possible mean fledglings per pair results: 0.0, 0.5 (1/2), 1.0 (2/2), 1.5 (3/2), and 2.0 (4/2). The Airforce mean 0.7 fledglings/pair was also inflated because only in the worst case of zero fledglings is the mean < 0.5. For these two reasons, there is low confidence in the apparently high Airforce mean productivity.

For Bombing Range, Black Rocks and Ten Mile combined, mean productivity was 0.8 fledglings/pair and larger than 0.5 fledglings/pair for South Ballina and Airforce combined (Welch two-sample *t*-test difference −0.2 fledglings/pair, 95% CI = −0.5 to −0.03, OLS residuals first-order autocorrelation ≥ −0.6, Durbin-Watson statistic ≤ 2.8, $P > 0.1$). These results are similar to those for the individual sites South Ballina and Bombing Range above.

There have been six reported vehicle strike oystercatcher mortalities for South Ballina been 2001–2011 and 2015–2018. Assuming a typical six months of monitoring (Jul–Dec) at three days a week, the known 0.4 mortalities/year can be extrapolated to 2 mortalities/year, including 0.7 breeding adult mortalities/year. Over 1997–2018 it is estimated that 15 breeding adults have died from vehicle strikes at South Ballina. The mortality rate could actually be increasing with vehicle traffic. Resulting vacancies were rapidly filled by birds from the non-territorial surplus in the past (Harrison 2009), however the decreasing South Ballina breeding population size indicates that recruitment has more recently been poor.

DISCUSSION

Count data allow a definitive evaluation of the success or otherwise of conservation management. The Australian Pied Oystercatcher breeding population size for the Richmond River area has not

increased despite two decades of fox control and > 208 young fledged. This result does not support the hypothesis that improved breeding success will increase the oystercatcher breeding population size. In turn, this result does not support the idea that predation by the Red Fox is a Key Threatening Process for the Australian Pied Oystercatcher.

The majority of the > 208 oystercatcher fledglings from the Richmond River area are unaccounted for. Counts of oystercatchers on NSW ocean beaches have not increased in the past two decades (Totterman 2020). Stuart (2010, 2011) has reported counts of > 100 oystercatchers in the large estuary of Port Stephens, including a few individually-marked birds from the Richmond River area, but with inadequate historical data to determine if that population has increased in the past two decades as well as few breeding records. The NSW oystercatcher population, which is defined by political boundaries, apparently is not closed. Many oystercatcher fledglings have been individually marked with band and/or leg flags over the past two decades and a study of recapture and resighting data is recommended to better understand the survival and dispersion of these birds.

Proponents of the breeding success oystercatcher conservation strategy may deploy a 'smokescreen' in their defence and argue that conservation science is far more complex than is presented in this study and/or criticise the statistical significance. However, counts do not lie and a complete review of all oystercatcher breeding sites in NSW is recommended. It is expected that there will be little or no evidence for increasing oystercatcher breeding population sizes during conservation management and that the majority of fledglings produced cannot be accounted for.

The moderate Spearman's $r = 0.58$ correlation between breeding pair counts and pipi abundance in this study provides some support for the hypothesis that the oystercatcher breeding population size is positively correlated with pipi abundance. However, available data do not indicate that pipi abundance at Bombing Range, where the oystercatcher breeding population size has increased, is higher than that at South Ballina, where the population has decreased. Secondly, the weak Spearman's $r = 0.26$ correlation between breeding productivity and pipi abundance does not support the hypothesis that oystercatcher breeding productivity is positively correlated with pipi abundance.

Data quality concerns are noted for historical pipi abundance data in Owner and Rohweder (2003) and in Harrison (2009). Owner and Rohweder (2003) reported an exceptionally high mean pipi band density of 8.2 pipis per 25 × 25 cm (0.0625 m^2) quadrat (the correct quadrat size was reported in Owner 1997), equivalent to 131 pipis/m^2, from five across shore transects at four km intervals on South Ballina Beach in 1997 (which is not presented in Figure 2a3). The highest mean pipi band density for South Ballina in Totterman (2018) was 31 pipis/m^2 (which was recorded in the oystercatcher non-breeding season and is not presented in Figure 2a3). The high 131 pipis/m^2 in Owner and Rohweder (2003) is inflated by an unspecified proportion of pipi recruits < 20 mm (Owner 1997). More concerning is that while such very high densities of large pipis can occur (100 pipis/m^2 is equal to one pipi every 10 cm), pipis have a patchy alongshore distribution and unbiased sampling is unlikely to intersect dense pipi bands in every transect. A further concern is that mean pipi length was < 20 mm for five of 11 beaches sampled in Owner and Rohweder (2003) versus Totterman (2020) sampled 72 beaches and found large pipis on all beaches. Low pipi abundances and the absence of large pipis for some beaches suggests under

sampling in Owner and Rohweder (2003). Concerns are also raised over poorly described pipi sampling designs in Harrison (2009) and because the 2005 result in that study combined results from six beaches. Therefore, the suggestion of a decrease in pipi abundance in the Richmond River area over the past two decades (Figures 2a3, 2b3) is not reliable. Proponents of the hypothesis that low pipi abundance is a driver for decreasing oystercatcher populations should be counting pipis.

The approximately linear decrease in the South Ballina oystercatcher breeding population suggests habitat loss, *i.e.* for territorial species the population size will be reduced in proportion to the amount of habitat lost (IUCN Standards and Petitions Committee 2019). The concomitant increase in the Bombing Range population suggests a preference for that site. Anecdotal evidence indicates that vehicle-based recreation on South Ballina has greatly increased in the past decade. Recent shorebird protection program reports have recognised this conservation threat and explicitly recommended that the beach be closed to 4WDs during the oystercatcher breeding season. The preceding discussion suggested that Bombing Range does not have a higher pipi abundance than South Ballina. Although no quantitative analysis was attempted in this study, oystercatcher conservation program reports indicate that canid nest predators are common at both sites, although South Ballina, adjoining an agricultural landscape, may have a higher proportion of foxes compared to Bombing Range, which could have a higher proportion of wild dogs. On the other hand, domestic dogs amplify human recreation disturbance for shorebirds (Glover *et al.* 2011) and are common on South Ballina Beach, with both dog walkers and dogs brought to the beach in vehicles. Even though data on human recreation disturbance are not available, the contrasting South Ballina and Bombing Range breeding population size trends supports the hypothesis that beach resident oystercatcher population sizes are negatively impacted by human recreation disturbance.

South Ballina, with uncontrolled beach driving, and Black Rocks, adjoining a popular camp ground, had the lowest mean breeding productivities in this study. Bombing Range, with no public access, had the highest mean productivity. These results support the hypothesis that oystercatcher breeding success is lower at sites with more intense human recreation disturbance. However, low breeding success is quite normal for oystercatchers and mean productivity results of 0.4–0.8 fledglings/pair from the Richmond River area are within the range 0.2–0.9 fledglings/pair from other studies for Australian Pied Oystercatchers reviewed in Taylor *et al.* (2014).

Taylor *et al.* (2014) suggested that adult oystercatcher annual survival is likely to exceed 90%. The estimated 0.7 breeding adults lost to vehicle strikes for South Ballina each year in this study suggests annual survival < 92% (= 1 – 0.7/8; for the 2016–2018 mean breeding population size of 8 pairs). However, the continued surplus of non-territorial adults at South Ballina (Totterman 2018) indicates that mortality is not limiting the breeding population size. Nonetheless, this result provides further evidence of the negative impacts of vehicle-based recreation on beach resident oystercatchers.

It is concluded that human recreation disturbance and vehicle-based recreation in particular are driving the South Ballina oystercatcher breeding population lower. Frequent disturbance can make otherwise suitable breeding habitat less attractive, resulting in poor recruitment and a

reduction in breeding population size as old birds die and are not replaced. Banning 4WDs could be a reprieve but beach house developments are permanently increasing human recreation disturbance.

Conservation management for the Australian Pied Oystercatcher has focussed almost exclusively on fox control in the Richmond River area and breeding success generally in NSW. A primary reason for this bias must be that Fox Threat Abatement plans have directed funding and attention towards actions to improve breeding success. Fox control can increase breeding success, positive breeding success results can be achieved quickly (*e.g.* Wellman *et al.* 2000), annual reporting of breeding success results suits the annual funding cycle and the idea that poor breeding success reduces the population size is easily understood by the public. Moreover, European cultures have a negative opinion of the Red Fox, which is a convenient 'scapegoat' for the decreasing Australian Pied Oystercatcher population size in NSW. None of this reasons are evidence-based.

Birdlife Australia have stated that the greatest conservation threat to beach nesting birds is disturbance from people visiting the beach (Birdlife Australia 2018). However, Governments typically shy away from protecting beach nesting bird habitat because of social interests in maintaining the *status quo* for access and recreation and commercial interests in developing coastal land. Environment and planning legislation has been crafted to avoid this politically toxic issue, *e.g.* Schedule 4 of the NSW Biodiversity Conservation Act 2016 (NSW Government 2016) does not recognise human recreation disturbance as a Key Threatening Process. The decisions and actions of public authorities and the Courts (*e.g.* NSW Land and Environment Court 2007, *Ringtank v Ballina Shire Council and others* (2007) NSWLEC 580) assume that beach front development and uncontrolled beach driving and human recreation can continue because fox baiting alone will sustain oystercatcher populations. This assumption is not supported by two decades of oystercatcher breeding results from the Richmond River area. Without strong protection of habitat from human recreation disturbance, the NSW oystercatcher breeding population size will continue to decrease in the next two decades.

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Table 1. Beach-nesting oystercatcher sites in the Richmond River area of NSW (Figure 1). Sites are defined by breaks in the beach habitat and different management histories. Data are from Totterman (2020) except that Black Rocks was not sampled in that study. Protected length includes adjacent Nature Reserves and National Parks (NP). Four wheel drive (4WD) length includes zones where beach driving is allowed. Dogs length includes zones where domestic dogs are allowed. Only public pedestrian access tracks were counted. The Black Rocks site adjoins a campground and walking track along Jerusalem Creek with many informal tracks to the beach. There is one track across the Goanna Headland isthmus to the northern end of Bombing Range Beach even though signs advise that public access is prohibited. Beach front human population is the sum of Australian Population Grid 2011 (Australian Bureau of Statistics 2011) 1 × 1 km cells intersected by the coastline.

Site	N–S end	Total (km)	Protected (km)	Urban (km)	4WD (km)	Dogs (km)	Pedestrian access	Camping & c'van parks	Human pop.
1. South Ballina	Richmond R. to Broadwater Hd. (28.876–29.040°S)	23.0	6.0	0.2	23.0	1.0	17	1	63
2. Airforce Beach	'Coffee rocks' to Evans River (29.060–29.113°S)	5.8	4.3	1.3	4.6	1.3	4	1	173
3. Bombing Range	Goanna Hd. to Jerusalem Creek (29.140–29.216°S)	10.5	10.5	0.0	0.0	0.0	1	0	0
4. Black Rocks	Jerusalem Creek to Black Rocks (29.216–29.254°S)	4.8	4.8	0.0	0.0	0.0		1	0
5. Ten Mile	Black Rocks to Shark Bay (29.254–29.358°S)	12.5	12.5	0.0	12.5	0.0	2	0	0

Table 2. Oystercatcher breeding and pipi abundance results for South Ballina (SB), Airforce (AF), Bombing Range (BR), Black Rocks (BX) and Ten Mile (TM). Breeding data sources are: 1) Wellman *et al*. (2000) to 1999, and 2) unpub. reports to the NSW DPIE thereafter. Pipi data sources are: 1) Owner and Rohweder (2003), 2) Harrison (2009), 3) Totterman (2018), 4) Totterman (2019) 5) Totterman (2020), and 6) this study. Blank results indicate no data or, for 2013 and 2015, unsatisfactory data. There was some fox control on South Ballina between 2011–2016 but not including the long Crown Reserve section. There was some fox control inland from Ten Mile Beach between 2010–2018 but not on the beach.

	Breeding pairs						**Young fledged**						**Fox control**					**Pipi density (pipis/m^2)**				
Year	**SB**	**AF**	**BR**	**BX**	**TM**	**Total**	**SB**	**AF**	**BR**	**BX**	**TM**	**Total**	**SB**	**AF**	**BR**	**BX**	**TM**	**SB**	**AF**	**BR**	**TM**	**Source**
1994	16					16	1					1										
1995	15					15	0					0										
1996	15					15	0					0										
1997	14					14	4					4	Y					131	12		17	1
1998	14					14	11					11	Y									
1999	14					14	3					3	Y									
2000	18					18	8					8	Y									
2001	17	2				19	14	2				16	Y	Y								
2002	17	2	4			23	8	2	1			11	Y	Y	Y							
2003	16	2	5			23	11	0	6			17	Y	Y	Y			20			26	2
2004	16	2	5			23	1	2	5			8	Y	Y	Y							
2005	15	2	5	3	0	25	4	1	3	2		10	Y	Y	Y	Y	Y	6.0				2
2006	14	2	7	2	1	26	9	0	10	1	0	20	Y	Y	Y	Y	Y					
2007	10	2	6	2	0	20	1	1	6	0		8	Y	Y	Y	Y	Y					
2008	9	2	7	2	0	20	4	1	7	1		13	Y	Y	Y	Y	Y					
2009	5	2	6	3	0	16	3	1	6	0		10	Y	Y	Y	Y	Y	0.0				3
2010	12	2	7	3		24	0	1	2	0		3	Y	Y	Y	Y		1.0				3
2011																		3.7				3
2012		2	7	3		12		2	2	1		5		Y	Y	Y		7.1				3
2013														Y	Y			16				3
2014		3	5			8		1	4			5		Y	Y			20			4.0	4
2015														Y	Y				4.3			4
2016	8	2	9	3	1	23	6	4	8	3	0	21		Y	Y	Y		11			3.9	5
2017	9	3	8	2	0	22	7	4	6	1		18	Y	Y	Y	Y						
2018	7	3	8	2	2	22	6	2	7	1	0	16	Y	Y	Y	Y		4.1		5.5		5,6
Sum												208										

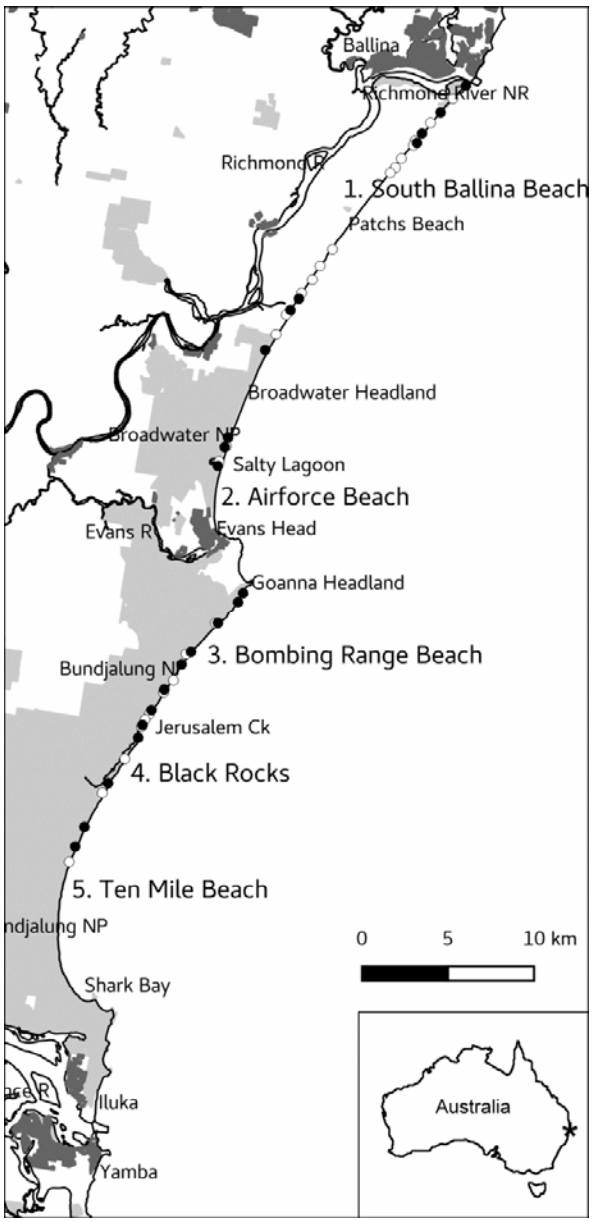

Figure 1. Map of the Richmond River area with beach-nesting oystercatcher sites numbered 1–5 (Table 1). Light grey areas are National Parks and Nature Reserves. Dark-grey areas are urban. White-filled circles are 2005 nest locations (which was the first year where all sites were counted; n = 26). Black-filled circles are 2018 nest locations (n = 22). The first nest is plotted for pairs that made repeated attempts within a breeding season. The inset map shows the location of the Richmond River area in Australia.

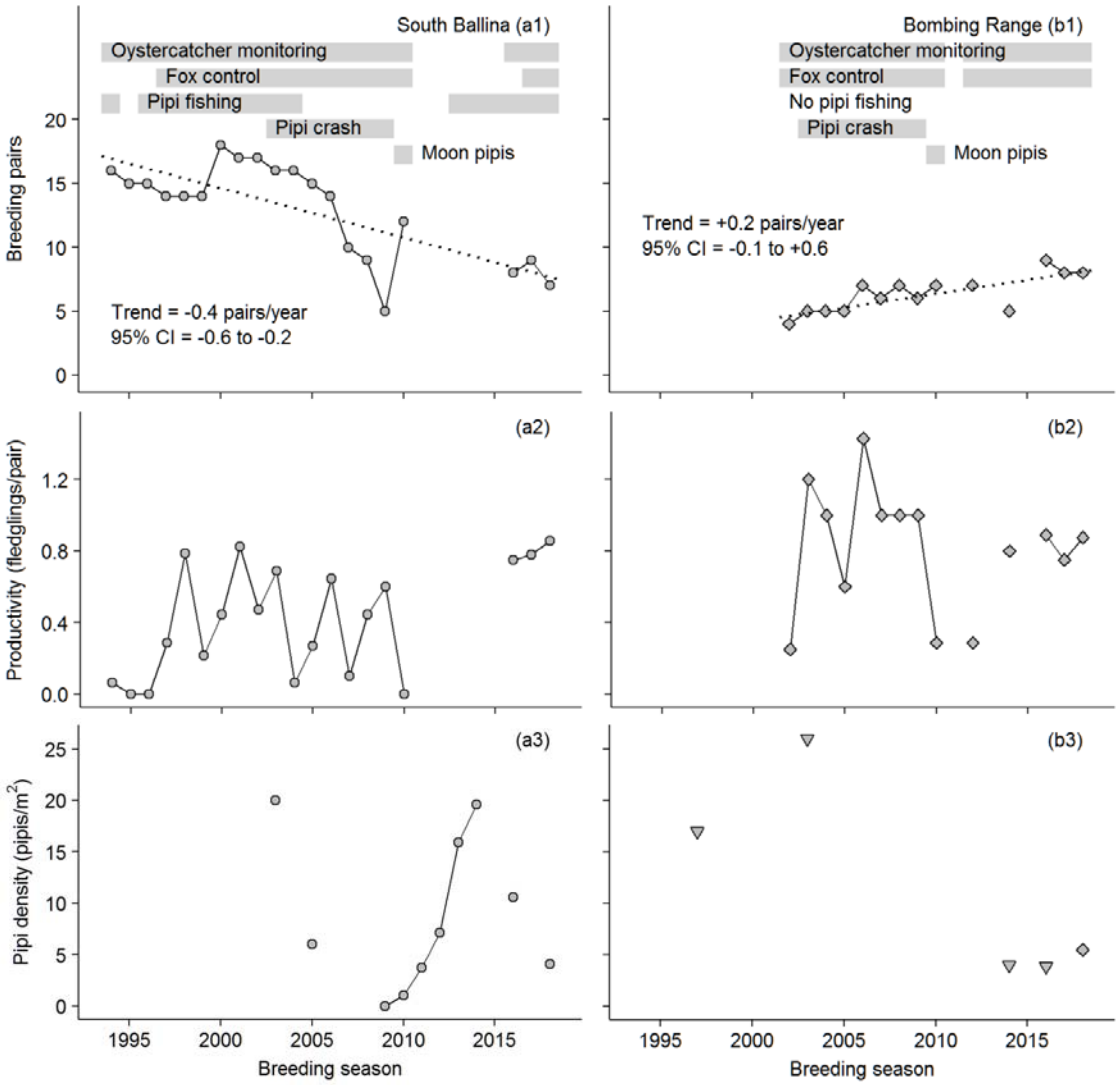

Figure 2. Oystercatcher breeding and pipi abundance histories for South Ballina (left, plotted as circles) and Bombing Range (right, plotted as diamonds). The exceptionally high 131 pipis/m^2 for South Ballina in 1997 (Owner & Rohweder 2003) is not plotted (see Discussion). Ten Mile pipi abundance results are plotted as inverted triangles in (b3). The linear trends in (a1) and (b1) are from a GLS ANCOVA model with beach as a covariate for year (GLS first-order serial correlation = 0.6, 95% CI = 0.3–0.9).

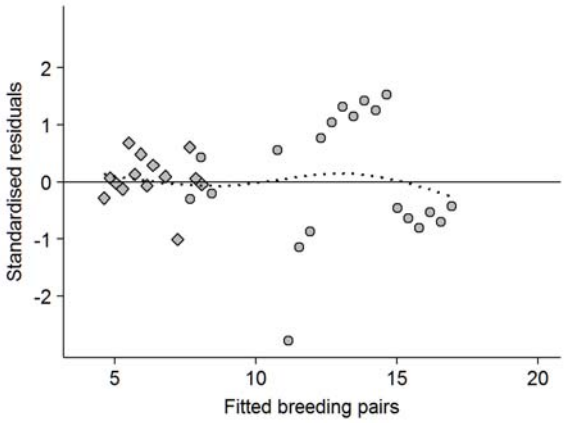

Figure 3. Standardised residuals for the breeding population size ANCOVA model in Figure 2 (South Ballina plotted as circles and Bombing Range as diamonds). The curve is a local polynomial regression smoother. The increase in scatter at higher fitted values resulted from a step increase in breeding pairs in 2000 (Figure 2a1).

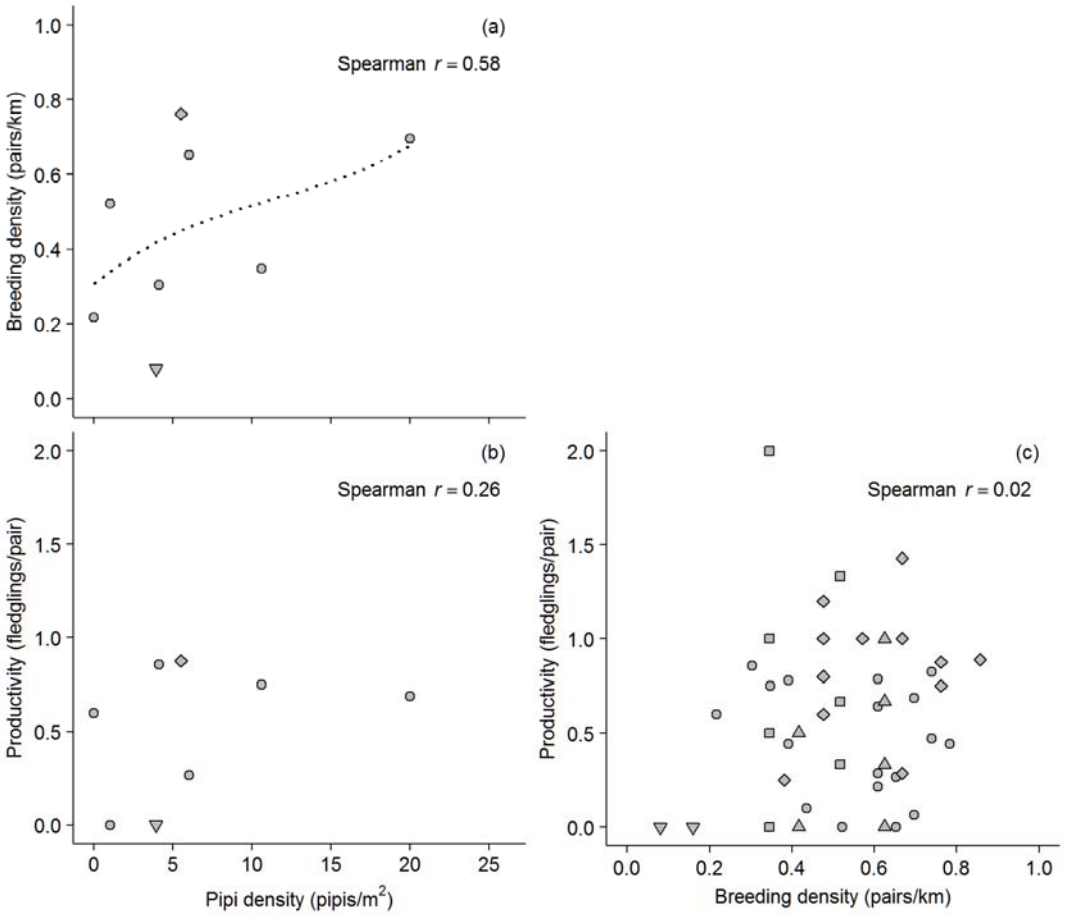

Figure 4. Scatterplots for oystercatcher breeding and pipi abundance results from all five sites (South Ballina plotted as circles, Bombing Range as diamonds, Airforce as squares, Black Rocks as triangles and Ten Mile as inverted triangles). The exceptionally high 131 pipis/m^2 for South Ballina in 1997 (Owner & Rohweder 2003) is not plotted (see Discussion). The curve in (a) is a local polynomial regression smoother. A more detailed statistical analysis for (a) and (b) was not appropriate because of small sample sizes. A more detailed analysis for (c) was not necessary because of the zero correlation.

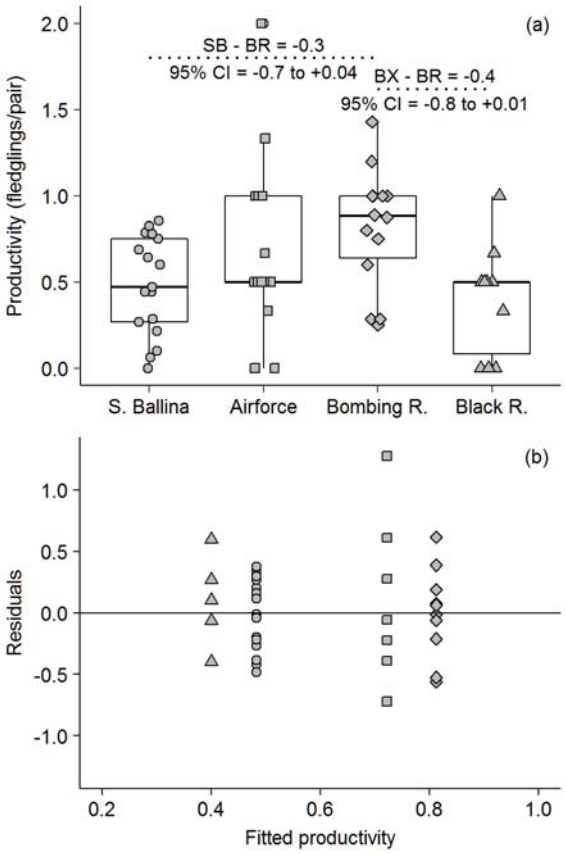

Figure 5. Box-and-whisker plots comparing oystercatcher breeding productivity for 1997–2018 (with fox control) at sites 1–4 (a) and ANOVA model residuals (b). Ten Mile (site 5) is not included because that sample size was very small (n = 3 seasons with breeding oystercatchers). Box-and-whiskers are overlaid with scatterplots, including a small amount of random horizontal jitter (South Ballina plotted as circles, Airforce as squares, Bombing Range as diamonds, Black Rocks as triangles). Horizontal lines and associated statistics are Tukey pairwise mean comparisons (SB = South Ballina, BR = Bombing Range, BX = Black Rocks). The overall ANOVA statistic was $F_{3,52} = 3.3$ (P = 0.03) and there was no serial correlation in the residuals (first-order autocorrelation ≥ –0.3, Durbin-Watson statistic ≤ 2.5, P > 0.1). Boxes show the first quartile, median and third quartile, whiskers extend to a maximum of 1.5 times the interquartile range and data outside the whiskers are plotted as individual points (white-filled circles).