The Whistler

Features

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Observations of Australian Pied Oystercatcher
Tracking Ruddy Turnstones
Common Mynas versus native parrots

Bird populations of:
Belmont Wetlands
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The *Whistler* Editorial

During the last decade we have worked with numerous authors and referees to produce over 600 pages of information about the birds of the Hunter Region. Throughout that time we have been supported by Liz Crawford who has tidied up many things that we missed and helped achieve a consistent standard. Chris Herbert and Rob Kyte have played important roles in the final layout and production phase. We thank the many experts who have acted as referees for their constructive comments. Numerous photographers have helped produce a vibrant product. We extend our thanks to all these people and hope they will share with us the overwhelming satisfaction of a job well done.

When *The Whistler* was first published in 2007 it was unclear whether we could attract sufficient copy to sustain an annual publication. The publication of this, the twelfth issue, resolves that question. Not only is annual production possible, but most years available copy exceeds the 64 pages which is our size limit.

Writing, editing and producing articles for *The Whistler* is at times difficult and frustrating for all concerned. However, those who have survived the ordeal will know the pleasure and sense of achievement in seeing their contribution circulating and their observations permanently recorded. As Editors perhaps our greatest satisfaction is seeing people who would not have dreamed they were capable of writing a technical paper making useful contributions. They too should be proud that they have documented the outcomes of their field studies and opportunistic observations to advance our knowledge of the behaviour of Australian birds and to document the bird communities of the Hunter Region.

The Hunter Region itself, with its diversity of habitats that support so many avian species, many of them threatened to a greater or lesser extent, will continue to inspire the efforts of residents and visitors alike to understand these species and their distinctive needs better, and to share their enjoyment not only of their observations but of what these observations might teach us. There is no doubt that our understanding of the Hunter Region's birds needs to keep growing if our efforts to conserve them are to have maximum effect. As we celebrate what all that collaboration has achieved through *The Whistler*, let us celebrate equally our source of inspiration, the birds of the Hunter Region, which have offered us so much and must continue to do so.

In finishing our term we wish the incoming Joint Editors, Neil Fraser and Alan Stuart, every success, in the knowledge that *The Whistler* is in good hands.

Mike Newman and **Harold Tarrant**
Joint Editors

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- ***To encourage bird observing as a leisure-time activity***

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Front cover: Australian Pied Oystercatcher *Haematopus longirostris* feeding on pipis at Stockton Beach, Worimi Conservation Lands - Photo: Ann Lindsey

Back cover: Galah *Eolophus roseicapilla* adult female preening unfledged juvenile - Photo: Kim Pryor

Fledging of Galahs nesting in a suburban environment near Newcastle, NSW

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Wild Galahs *Eolophus roseicapilla* were observed breeding in nest boxes in a suburban garden between 2002 and 2017. The outcome of 12 nesting attempts made in four different nest boxes was similar to the outcome of nesting attempts made in natural tree hollows in the wild. Galah chicks often fledge in the afternoon, as well as in the morning. Most fledge less than two hours after sunrise or less than two hours before sunset.

INTRODUCTION

Galahs *Eolophus roseicapilla* nest in natural tree hollows, which are destroyed during land clearing. In rural areas, hollows are lost when large old trees, in which hollows form, are cleared. In urban areas, hollows are lost when living or dead hollow-bearing trees are cut down and when dead branches are cut off living trees for public safety.

Although Galahs are one of the most common backyard birds (BirdLife Australia 2017), they may lack sufficient resources in urban areas. Domestic gardens provide valuable bird habitat when they offer food, water, shelter and nesting sites. Nest boxes that are designed to copy the characteristics of natural tree hollows can be useful nesting sites (Parsons 2007). They can be installed in trees, or on poles or other structures in domestic gardens.

This short note provides insights into the breeding of Galahs in urban NSW at Thornton (32°24'S 150°38'E) near Newcastle based on observations made of nest boxes in a suburban garden between 2002 and 2017.

METHODS

Nest boxes were built to attract Galahs, Eastern Rosellas *Platycercus eximius* and Common Brushtail Possums *Trichosurus vulpecula* (Dengate 1997; **Table 1**). They were installed 7 m above the ground in two eucalypts (approximately 21 m tall, species unknown) and/or on two steel poles, 2, 5.5 or 6.5 m above the ground (**Table 2**) in the back right corner of our residential property, area 765 m². Each year, one to three nest boxes were installed. Different nest boxes were available in different years and some were mounted in different places in different years. Below them, the mid-storey and understorey vegetation consisted of native shrubs, including Acacia sp., Callistemon sp., Banksia sp. and Grevillea sp.

The nest boxes were made of plywood to provide insulation and painted with Dulux Weathershield to prevent moisture from penetrating and providing a suitable environment for mould to grow. Internal ladders were fitted to allow the chicks to climb up to the entrance holes (**Figure 1**).

Figure 1. A wide internal ladder allowed siblings, such as Chick 2/2016 and Chick 3/2016 shown here, to look out of the nest box at the same time.

The roofs were hinged and sloped downwards to the front. The nest boxes originally had one to three external perches, but these were removed in some years to prevent Laughing Kookaburras *Dacelo novaeguineae* using them to predate eggs and chicks.

Several handfuls of green eucalypt leaves treated with bird lice and mite spray or powder were put inside the nest boxes at the beginning of each breeding season. The nest boxes were installed using brackets and straps and faced to the east or north-east. They were left alone during the breeding season unless the chicks were at risk from lice or mites or the eucalypts were at risk of being ringbarked because the adults were removing patches of bark (scarring).

We occasionally provided wild bird seed in a bird feeder in our backyard. We also provided clean water in a pedestal bird bath.

Most of the clutch sizes (with five exceptions) were confirmed in the Galah Box mounted at 2 m, because of ease of access. Access to boxes mounted at 5.5 m and higher required a ladder and climbing equipment.

As soon as the first chick began to lean out of the nest box and make a distinct 'quack-quack' call to its parents, we spent each day monitoring the nest box. We sat outside for several hours each morning from before sunrise and for several hours each afternoon until after sunset. We went outside every time we heard the parents visit the nest box during the day. We defined fledging as the first time a chick left the nest box.

Table 1. Dimensions of nest boxes used by nesting Galahs in our backyard from 2002 to 2017

Nest box	Entrance hole diameter (mm)	Base (mm)	Front wall height (mm)	Back wall height (mm)	Ladder (attached to inside of front wall)	Orientation of nest box
Galah	120	260 x 260	700	770	parallel wooden strips	Vertical
Rosella	80	230 x 230	470	500	parallel wooden strips	Vertical
Possum	110	300 x 300	570	600	diagonal eucalypt branch	Vertical
Galah 2	100	250 x 250	700	750	wooden panel with cut outs	Vertical

Table 2. Clutch size, hatching success and fledging success for Galahs in four different nest boxes

Year	Nest box	Location	Height (m)	Eggs laid	Chicks hatched	Chicks fledged
2002	Galah	Right eucalypt	7			ND (3?)[1]
2003	Galah	Pole	2			ND (3?)
2004	Galah	Pole	2	NK[2]	NK	3
2005	Galah	Pole	2	NK	NK	2
2006	Galah	Pole	2	3	NK	1
2007	Galah	Pole	2	3, 3[3]	3	2
2008	Galah	Pole	2	3	2	1
2009	Galah	Pole	2	NK	1	0
2010	Galah	Pole	2	0	0	0
2011	Galah	Pole	2	0	0	0
2012	Rosella	Right eucalypt	7	NK	3	3
2013	Rosella	Right eucalypt then pole	7 then 5	4	3	3
2014	Possum	Pole	6.5	NK	3	3
2015	Galah 2	Pole	5.5	NK	NK	2
2016	Galah 2	Pole	5.5	NK	3	3
2017	Galah 2	Pole	5.5	NK	NK	1

[1] Not documented (successful nesting attempt; we believe 3 chicks fledged)
[2] Not known
[3] Replacement clutch of 3 eggs

RESULTS

Breeding statistics

The breeding season was July to December. However, Galahs visited the nest box intermittently during the non-breeding season. They placed sprays of green eucalypt leaves in the nest box to line it in the breeding and non-breeding seasons. Galahs used the sides of their faces to rub preen oil on the nest boxes and poles in the breeding and non-breeding seasons (**Figure 2**). Preen oil is produced by a gland near the base of a bird's tail and keeps feathers in good condition. It makes surfaces near the nest entrance smooth and slippery and gives them a musky odour. It may be difficult for nest predators such as monitor lizards to climb to the nest and the odour may be a repellent (Young 2014).

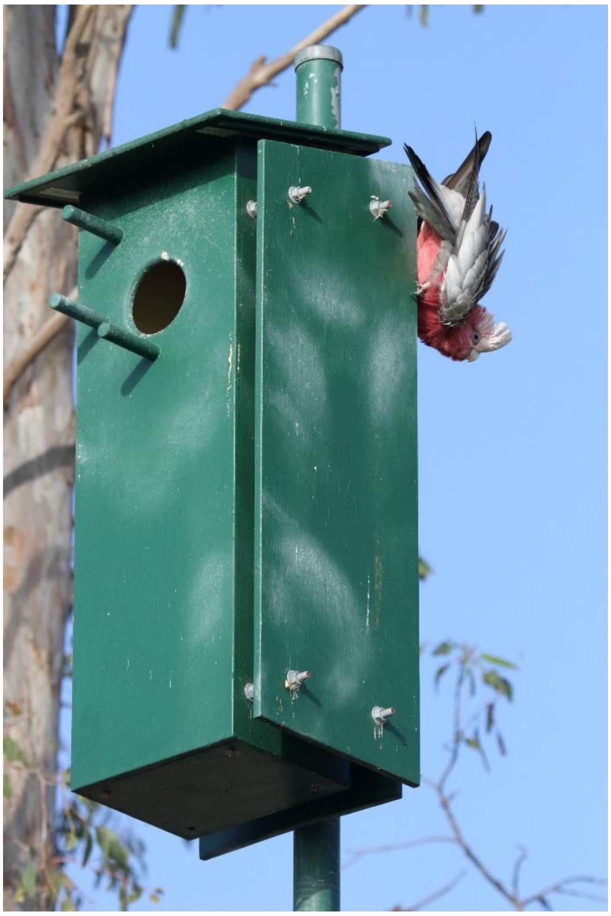

Figure 2. When Galahs rubbed preen oil on the nest box, they left white patches of 'feather dust', a very fine powder shed by their powder-down feathers.

Galahs have made 13 successful nesting attempts (when at least one chick fledged) in four nest boxes. They have made 10 successful nesting attempts in the longer Galah Box and Galah 2 Box and three successful nesting attempts in the shorter Rosella Box and Possum Box (**Table 2**). The Rosella Box was available in all years. In 2003, they tried to enlarge the hole of the Rosella Box before nesting successfully in the Galah Box. In 2013, they nested in the Rosella Box when the larger Possum Box was available.

For three nesting attempts in which the numbers of eggs and chicks were known, 13 eggs were laid, 8 chicks hatched (61.5%) and 6 chicks fledged (46.2%).

For four nesting attempts in which the clutch size was known, a total of 16 eggs were laid. This included a double clutch in 2007, involving initial and replacement clutches. The average clutch size was 3.2 (n=5).

For seven nesting attempts involving eight clutches (i.e. one replacement clutch) in which the numbers of chicks hatched were known, 18 chicks hatched, involving a maximum of three and an average of 2.6 chicks hatched per pair/annum, equivalent to 2.3 chicks per clutch.

For 12 nesting attempts in which the numbers of chicks fledged were known, 24 chicks fledged. A maximum of 3 and an average of 2 chicks fledged per pair/annum.

For four nesting attempts in which three young fledged, the fledging period measured as the time from the first chick(s) being heard to the third chick fledging was 47 to 69 days/7 to10 weeks (an average of 56 days/8 weeks).

Chick behaviour

Six to eight and a half weeks after hatching, a chick began to lean out of the nest box entrance hole and make a distinct 'quack-quack' call to the parents. The parents responded with a similar call from branches near the nest box. The chick and parents called intermittently for up to three hours (a calling session) in the early morning and late afternoon for up to three days before the chick fledged (the calling period). The male parent often preened the chick for several minutes the day before it fledged.

The parents did short demonstration flights from branch to branch in the eucalypts and in slow circles in front of the nest box during the calling session in which the chick fledged, and often during the previous calling session. They sometimes flew over several residential properties or flew to the roof of a nearby house, landed then flew back in less than a minute. Just before the chick left the nest box, it stood on the entrance hole with its claws visible and leaned even further out. The chick and parents called more loudly and insistently. As soon as the chick left the nest box, the parents flew to it then flew one to two metres on either side of it (**Figure 3**). In 2015, Chick 1 returned to our yard with the parents the afternoon before and the morning Chick 2 fledged. Chick 1 did short demonstration flights in front of the nest box and flew closely beside Chick 2 and the parents when Chick 2 fledged.

Figure 3. As soon as Chick 1/2017 left the nest box, the male parent left the roof and the female parent left the tree to fly away with their young.

While the parents were calling a chick to leave the nest box, they either did not feed it or gave it a very short feed. They gave it a long feed if it had not left by later in the morning or by sunset. After this, they flew away (morning) or went to sleep near the nest box (evening). They began calling again the next morning or afternoon. The parents did not call a chick to leave the nest box during the middle of the day or during wet or windy weather, even if the calling period had begun.

Only one chick called at a time. The next chick did not begin calling until the previous chick had fledged, even if it looked out of the entrance hole during a calling session.

If a chick left the nest box prematurely and only flapped to the ground, the parents cared for it in our backyard while caring for older siblings that had fledged successfully and were capable of sustained flight. They visited the chick on the ground many times during the day to feed, shelter with and preen it. They also left it for intervals that ranged from ten minutes to more than four hours. They sometimes slept in our backyard at night and sometimes slept elsewhere (presumably near the older sibling(s) that had fledged successfully). They cared for Chick 2/2007 (second of two) for 10 days (**Figure 4**) and Chick 3/2014 (third of three) for 2.5 days (**Figure 5**). Both chicks flew out of our backyard with their parents.

Figure 4. The parents preened and cared for Chick 2/2007 on the grass and in the gardens in our backyard for 10 days.

Figure 5. The parents fed and cared for Chick 3/2014 on the grass and in a low banksia shrub for 2.5 days.

Timing of fledging

The timing of fledging relative to sunrise and sunset for seven nesting events involving one to three chicks is shown in **Table 3**. Fourteen out of 15 chicks (93%) left the nest box less than two hours after sunrise or before sunset. Nine out of 15 chicks (60%) left an average of 54 minutes after sunrise and six out of 15 chicks (40%) left an average of 80 minutes before sunset. When the exception was removed, five out of 15 chicks (33%) left an average of 43 minutes before sunset. In 2012, 2013 and 2016, siblings left in the morning and the afternoon (**Table 3**).

Table 3. Timing of fledging of Galah chicks relative to sunrise and sunset (Australian Eastern Standard Time)

Chick hatch number / year	Date chick left the box	Time chick left the box (h)	Sunrise [1] (h)	Time after sunrise (minutes)	Sunset [1] (h)	Time before sunset (minutes)
Chick 1/2008	10/12/08	0500	0444	16		
Chick 1/2012	2/11/12	0641	0458	103		
Chick 2/2012	4/11/12	1650			1826	96
Chick 3/2012	9/11/12	1410			1830	260
Chick 1/2013	19/10/13	1730			1812	42
Chick 2/2013	20/10/13	0618	0512	66		
Chick 3/2013	21/10/13	1758			1814	16
Chick 1/2014	23/10/14	0600	0509	51		
Chick 2/2014	3/11/14	0535	0458	37		
Chick 3/2014	7/11/14	0615	0454	81		
Chick 1/2015	no data					
Chick 2/2015	13/11/15	0530	0450	40		
Chick 1/2016	22/10/16	0545	0509	36		
Chick 2/2016	26/10/16	1735			1818	43
Chick 3/2016	27/10/16	1759			1819	20
Chick 1/2017	16/10/17	0610	0516	54		

[1] (Geoscience Australia 2010).

Chick 2/2007, which flapped to the ground on the morning of 22 November 2007, left the backyard on 2 December 2007 at 0630 h, 107 minutes after sunrise, which was at 0443 h. Chick 3/2014, which flapped to the ground at 0615 h on 7 November 2014, left the backyard at 1830 h on 9 November 2014, exactly at sunset, which was at 1830 h.

For seven nesting attempts for which the fledging date was known, the first chick left the nest box between 16 October and 10 December. Four of the seven first chicks left between 16 and 23 October.

For four nesting attempts for which the fledging date was known, the third and last chick left the nest box between 21 October and 9 November.

For four nesting attempts in which three chicks fledged, the interval from the first chick fledging to the third chick fledging ranged from 2 to 15 days (an average of 7.3 days). On average, Chick 2 left the box 4.5 days after Chick 1 and Chick 3 left the box 2.8 days after Chick 2.

DISCUSSION

In a comprehensive study between 1970 and 1977 in the wheatbelt of Western Australia, the typical number of siblings which finally left nest hollows was three to four (Rowley 1990), slightly higher than the mean value of 2.3 in this study.

It has been suggested that chicks fledge in the morning to decrease their chance of being preyed upon in the nest (Chiavacci *et al*. 2015). We observed that 40% of Galah chicks left the nest box in the afternoon. This may mean that nest predation risk was low in our backyard. During the nestling period, the parents did not react to our dog. However, they performed the heraldic display (stood upright with crest raised, wings spread and tail fanned and gave the screech call (Pidgeon 1970)) when they saw a cat or an Eastern Blue-tongue Lizard *Tiliqua scincoides*. Neither cats nor lizards attempted to climb to the nest boxes.

It has also been suggested that chicks in riskier nests fledge over a shorter period of time than chicks in safer nests (Chiavacci *et al*. 2015). We observed siblings leaving the nest box over a period of 2 to 15 days. We were not aware of any additional risks in 2013, when they left over 2 days, compared with 2014, when they left over 15 days.

In our study, siblings that fledged in the morning did not necessarily also fledge over a shorter period. In 2013, three chicks fledged in the afternoon, morning and afternoon but over a shorter time (two days) while in 2014, all three chicks fledged in the morning but over 15 days. This suggests that other factors may influence fledging, for example parental care, chick development, nest height, nest concealment and proximity to other birds and animals.

Previous studies have found that Galah chicks usually fledge in the morning and fly strongly to a preferred fledgling habitat, or crèche (Higgins 1999). We observed the parents calling to each chick every morning and every afternoon for up to three days until it fledged. Furthermore, the calling in the afternoon appeared to be as prolonged and urgent as the calling in the morning. This suggests that the parents are able to quickly escort the chick to the safety of the crèche.

Fourteen out of 15 Galah chicks in our nest boxes (93%) fledged shortly after sunrise or shortly before sunset. In addition, the two chicks that had flapped to the ground finally left our backyard less than two hours after sunrise or at sunset. This may be because their parents are most active at these times. Galahs spend most of the day sheltering from the heat in trees (BirdLife Australia 2012a). Chicks may also fledge at these times to avoid natural predators, such as Peregrine Falcons *Falco peregrinus*, which hunt mainly during the day (BirdLife Australia 2012b).

In the Western Australian study (Rowley 1990), large differences were noted in the number of days between the first and last Galah chicks leaving the nest (range of 1 to 12 days), which is consistent with our observations. Rowley suggests that delayed fledging may be a consequence of a number of factors including difference in the timing of eggs hatching, the availability of food, particularly when provisioning clutches containing a runt. Within the confines of natural nest hollows the larger siblings may be preferentially fed and have increased opportunity to exercise and develop their wings. Galahs are surprisingly powerful fliers on fledging, allowing them to be moved to juvenile crèches some distance from the nest site where there is readily available food. Hence, sibling Galahs may be contemporaneously provisioned at nest hollows and at the juvenile crèche.

CONCLUSIONS

Our study conducted in a suburban environment using artificial nest sites gave generally similar results to those found in more comprehensive studies of birds using natural nest hollows. In this study involving Galahs habituated to the observer's passive presence, it was possible to get more intensive data on the timing of fledging than has been possible in other studies (Rowley 1990).

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Winners and losers – Changes in the bird population on removing cattle from woodland near Paterson NSW

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The start of monthly bird monitoring in woodland at Green Wattle Creek in the Hunter Valley of New South Wales in April 1996 coincided with the cessation of cattle grazing. This paper describes the subsequent changes in the bird population at a 2-ha survey site over an 18-year period.

In the absence of grazing the understorey vegetation increased, changing the foraging opportunities for many bird species. This resulted in a temporary increase in the number of types of birds recorded, but after a mid-study peak the number of species recorded decreased to its initial level. However, there were substantial differences between the bird assemblages at the start and end of the study. These differences were explained in terms of differences in the foraging behaviour of individual species (e.g. birds which foraged on the ground were disadvantaged by dense understorey vegetation).

The "winners" were species that benefitted from increased shrub layer vegetation. These included Lewin's Honeyeater *Meliphaga lewinii*, Brown Thornbill *Acanthiza pusilla*, White-browed Scrubwren *Sericornis frontalis* and Eastern Whipbird *Psophodes olivaceus*. However, as the shrub layer vegetation became denser some of these species decreased from their mid-study peak levels, presumably because they were no longer able to forage effectively.

The "losers" were predominantly species which like open habitat, particularly those which forage on the ground, such as the Speckled Warbler *Pyrrholaemus sagittatus*, Double-barred Finch *Taeniopygia bichenovii* and two species of Fairy-wren *Malurus* spp. Foliage-feeding honeyeaters were indirectly affected with the Fuscous Honeyeater *Ptilotula fuscus* and the White-naped Honeyeater *Melithreptus lunatus* decreasing. In their absence the Yellow-faced Honeyeater *Caligavis chrysops* increased, becoming the most frequently recorded species.

INTRODUCTION

It is widely acknowledged that grazing impacts adversely on bird populations, and considerable emphasis has been placed on the need for woodland restoration (Lindenmayer 2011). However, within woodland historical grazing may have opened up unique opportunities for some bird species as shown in this paper, which tracks the loss of a number of species in an area of woodland after grazing ceased.

Green Wattle Creek is a 90-ha area of woodland near Paterson in the Hunter Region of New South Wales (NSW). It is now known as the Butterwick Crown Lands Reserve and is managed by Crown Lands, a division of the NSW Department of Trade and Investment. Green Wattle Creek, formerly a travelling stock route, was grazed under lease when this study commenced in 1996. However, during that year the cattle were removed providing an opportunity to monitor changes in the bird population in response to the change in land management.

The results presented in this paper are from a single survey site at Green Wattle Creek. A previous paper provided general background to these studies at Green Wattle Creek, including an overview of the bird populations and a description of the habitat (Newman 2009). This paper provides an in depth evaluation of the results for one of the four 2-ha survey sites and updates the records until the end of 2013 when the study finished.

METHODS AND ANALYSIS

Monthly surveys were conducted at Green Wattle Creek between April 1996 and December 2013 using BirdLife Australia's (BLA) standard 2-ha survey, which involves recording all species present in a 2-ha area during a period of 20 minutes. All birds seen and heard were recorded. Surveys were conducted in the morning at four 2-ha survey sites and the observations were submitted to BLA's Birdata archive. The results reported in this paper were recorded at survey Site 3 (BLA site identification 273038; 32.660°S, 151.653°E). Numbers of each species were recorded, but were not used in this analysis, which was based on presence-absence. Reporting rates (RR) expressed as a percent value were used to compare the frequency at which each species was recorded (e.g. a species with a RR of 50% was recorded in half of the surveys).

Using a survey method which samples a small area (2 ha) for a short time (20 min) has the advantage of detecting differences in the RRs of frequently observed species (e.g. with RRs > 20%), but has limited statistical power for less common species with low RRs.

All the surveys were conducted by the same person (MN). While this provided consistency, the data set was potentially subject to systematic errors associated with the decreasing detectability of some species as the density of the understorey vegetation increased. This placed increased reliance on vocal detection.

Trend analysis and statistical tests were performed using a method which was developed for the analysis of Birdata survey results (Cunningham & Olsen 2009).

RESULTS

Seventy-seven species were recorded in 208 surveys conducted between April 1996 and December 2013 at monthly intervals. The results, as summarised in **Table 1**, have been divided into six intervals, each of three years' duration. The number of species recorded peaked in 2002–2004, when the mean number of species/survey was also highest. There was a similar increase in the number of frequently recorded species with RRs exceeding 20% (**Figure 1**).

The results were divided into six three-year periods for evaluation of temporal changes in RR. The results for 33 species which had RR>10% in at least one three-year period are shown in the **Appendix**, which contains the scientific names of all species not discussed in the text.

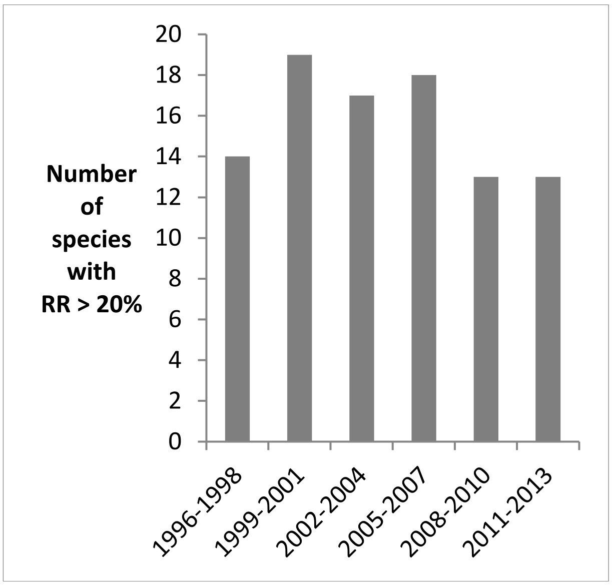

Figure 1. Temporal variation in the number of species which were regularly recorded (RR > 20%) at Green Wattle Creek Survey Site 3 over the period 1996 to 2013.

Although there was relatively little change in the status of species like the Grey Fantail *Rhipidura fuliginosa* and the Yellow-faced Honeyeater *Caligavis chrysops*, there were dramatic changes in the occurrence of other species. For instance, three species, Speckled Warbler *Pyrrholaemus sagittatus*, Double-barred Finch *Taeniopygia bichenovii* and Jacky Winter *Microeca fascinans,* which were present on a number of occasions in the first six years, were not recorded during the last six years of the study (**Figure 2**). The RR of the Superb Fairy-wren *Malurus cyaneus*, the second most frequently recorded species in 1996-1998, decreased by 75% (**Figure 3**). Although the Variegated Fairy-wren *Malurus lamberti* RR increased initially, it subsequently followed the decrease of the Superb Fairy-wren RR (**Figure 3**).

Table 1. Summary of statistics for surveys at Green Wattle Creek Site 3 between April 1996 and December 2013.

	1996-1998	1999-2001	2002-2004	2005-2007	2008-2010	2011-2013
Number of species	35	47	53	50	37	43
Number of surveys	33	34	36	34	35	36
Mean species/survey	7.8	10.2	11.4	8.1	9.3	8.1
Standard Deviation	3.3	4.3	3.9	4.3	3.0	3.2

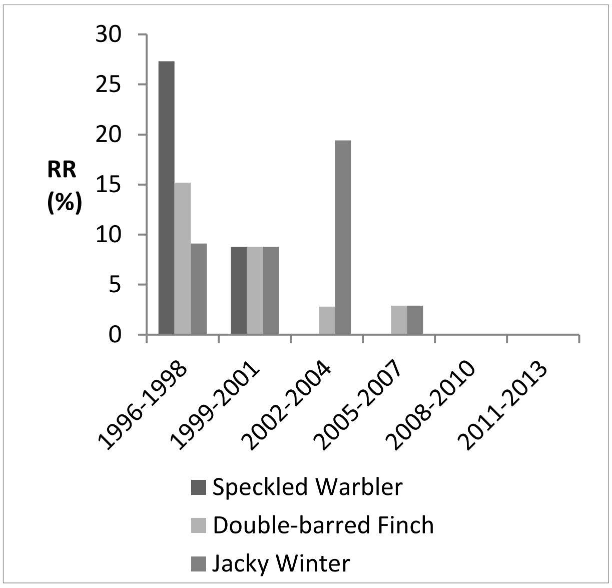

Figure 2. Reporting rates of Speckled Warbler, Double-barred Finch and Jacky Winter, which were not recorded at Green Wattle Creek Survey Site 3 after 2005-2007.

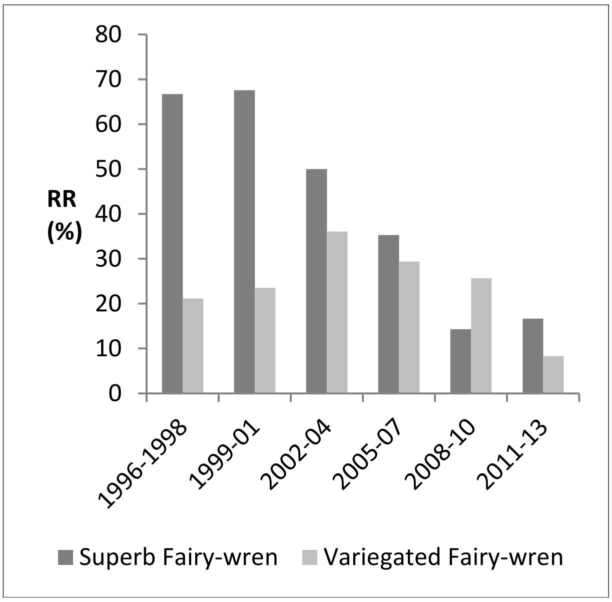

Figure 3. Reporting rates of the Superb and Variegated Fairy-wrens; examples of species which decreased at Green Wattle Creek Survey Site 3 between 1996 and 2013.

The Fuscous Honeyeater *Ptilotula fuscus* and White-naped Honeyeater *Melithreptus lunatus*, also decreased, and were not recorded in the last six years of the study. In contrast, there was a sustained increase in the occurrence of the Yellow-faced Honeyeater (**Figure 4**). Three other species which increased were Lewin's Honeyeater *Meliphaga lewinii*, White-browed Scrubwren *Sericornis frontalis* and Brown Thornbill *Acanthiza pusilla*, although the latter two species were recorded less frequently in the last three years (**Figure 5**).

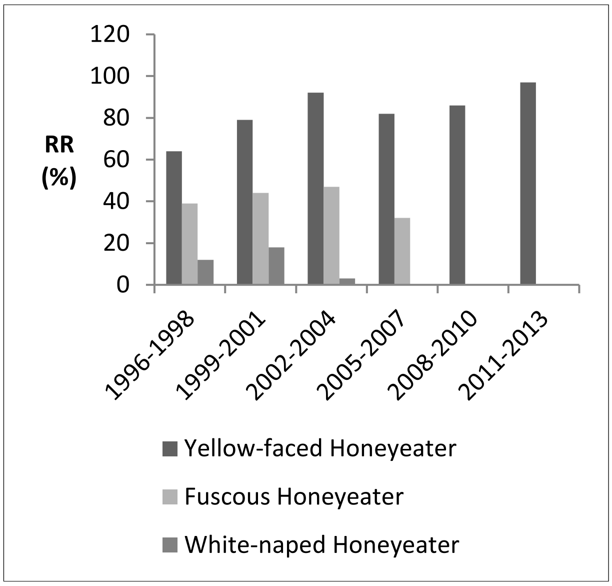

Figure 4. Changes in the status of three honeyeater species, with one increasing and two unrecorded during the last six years of the study at Green Wattle Creek Site 3.

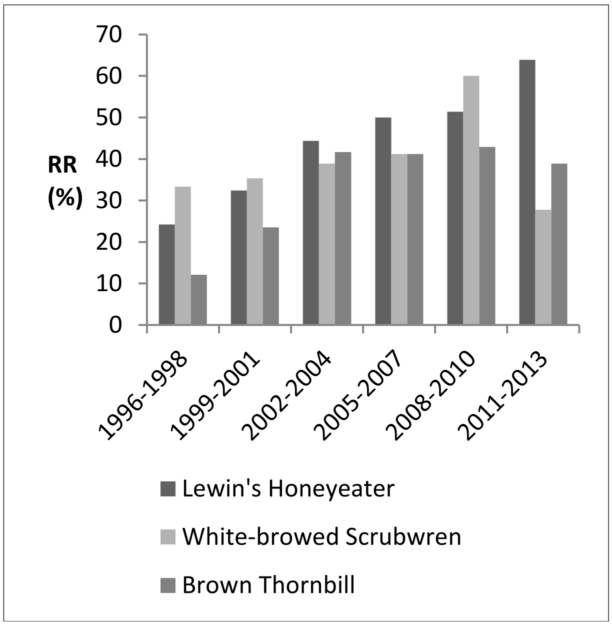

Figure 5. Temporal variations in the reporting rates of Lewin's Honeyeater, White-browed Scrubwren and Brown Thornbill, which increased at Green Wattle Creek Survey Site 3 between 1996 and 2013.

Statistical Modelling

Examples of increasing (Lewin's Honeyeater) and decreasing trends (Superb Fairy-wren) are shown in **Figures 6a** and **6b** respectively. In both cases there was a statistically significant change in the status of the species as indicated by the linear trend line. However, shorter term fluctuations are also apparent as indicated by the smooth trend line. In contrast, the Eastern Whipbird *Psophodes olivaceus* increased initially before decreasing from a mid-study peak value (**Figure 6c**). Although there was an overall increase in the RR, the linear trend was not statistically significant (p=0.28). Similar, but less well-defined mid-study peaks, were observed for a number of other species including the Rufous Whistler *Pachycephala rufiventris.*

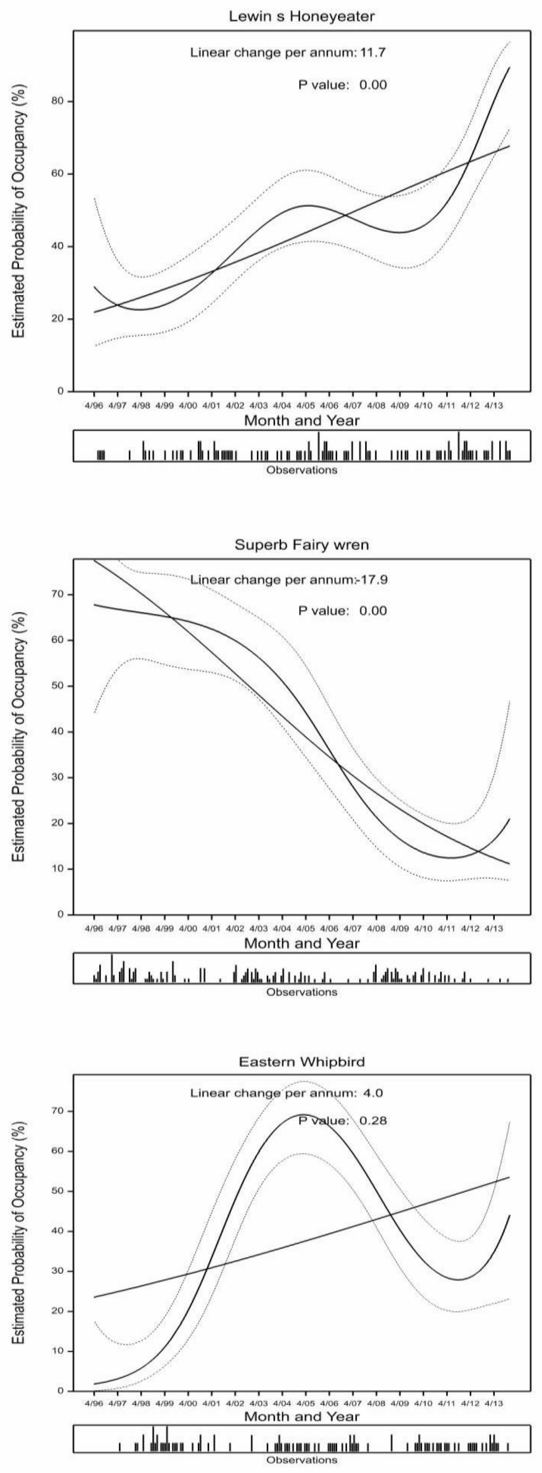

Figure 6. Smooth trends showing (a) the increase of Lewin's Honeyeater, (b) the decrease of the Superb Fairy-wren and (c) the mid-study peak occurrence of the Eastern Whipbird. Solid smooth lines show the estimated temporal trends in the probability of occurrence. The dotted lines show the 95% confidence limits of the smooth trend estimate. The linear trend lines, after back transforming the probability function, are also shown. The linear trends for (a) and (b) were highly significant statistically (p<0.001). The lower rug plot shows when records occurred.

The linear trend lines for 12 species indicated statistically significant population changes (p=0.05 or less), four increasing and eight decreasing (**Table 2**). Fuscous Honeyeater (p=0.06) was close to statistical significance. The trend rate is a measure of annual rate of change of the population of each species.

Table 2. Species for which there were statistically significant linear trends in status at Green Wattle Creek Survey Site 3 (1996-2013).

Species	Trend Rate[1]	*P* [2]
Australian Raven	15.4	0.01
Brown Thornbill	8.9	0.01
Double-barred Finch	-24.1	0.01
Eastern Rosella	-11.0	0.01
Fuscous Honeyeater	-9.4	0.06
Grey Fantail	-7.9	0.03
Lewin's Honeyeater	11.7	0
Speckled Warbler	-45.0	0.02
Striated Pardalote	-14.2	0.01
Superb Fairy-wren	-17.9	0
White-naped Honeyeater	-15.0	0.03
Yellow-faced Honeyeater	12.4	0.01
Yellow Thornbill	-7.3	0.05

[1] Positive and negative values indicate increasing and decreasing species respectively.
[2] All species were statistically significant (p not greater than 0.05) except the Fuscous Honeyeater.

Changes in habitat

The following description of the habitat at Survey Site 3 was completed for the New Atlas of Australian Birds project (Barrett *et al.* 2003) in 1999 during the early stages of regeneration following the removal of cattle. There were many trees of varying age and size (2 to 8 m height). Two or three species dominated. The shrub layer comprised a mix of many small, mainly native shrubs and a few taller shrubs dominated by two or three species. There were a small number (<6) of fallen trees/large branches.

Ten years later the understorey was described as follows (Newman 2009): 'There is an extensive dense understorey growth of Blackthorn *Bursaria* *spinosa* to a height of 3m with amounts of Kurrajong *Brachychiton populneus,* wattle *Acacia* spp. and the introduced species Lantana *Lantana camara* and Wild Olive *Olea europaea africana.* There are minor patches of the small native shrub Prickly Beard-heath *Leucopogon juniperinus*.'

The survey site and surrounding area had not been burnt for at least five years prior to the study. In spring 2010 fuel reduction burns were conducted in patches of the reserve (Newman 2014a), including areas adjacent to, but not at, Survey Site 3.

The extent of the densification of understorey vegetation required the undergrowth to be cut back to allow access during the period 2011-2013.

More comprehensive details of the habitat at and surrounding Survey Site 3 are provided in previous publications (Newman 2009, 2010 and 2014a).

DISCUSSION

The extent of the change in the bird population at Green Wattle Creek Survey Site 3 is dramatically demonstrated by a comparison of the ten most frequently recorded species in the first and last three years of this study (**Table 3**). Only three species, Grey Fantail, Yellow-faced Honeyeater and Lewin's Honeyeater sustained their top-ten ranking. Fuscous Honeyeater and Speckled Warbler, ranked 5 and 9 respectively, were not recorded during the last six years of the study.

Birds are mobile species which complicates discussion of trends at a single survey site. For instance, the observed trends may be influenced by changes in conditions at the survey site or by external factors. The following sections seek possible explanations for the observed changes in the status of species by evaluating the impact of changes in habitat that occurred after cattle grazing ceased.

Changes in habitat

Following the removal of cattle in 1996 the understorey vegetation progressively increased. This had implications for birds using the area, including both changes in food availability/type and foraging opportunities. For instance, the niche for species which feed in areas of open ground largely disappeared and there was less opportunity for birds to forage at sub-canopy levels. However, the dense shrub layer vegetation increased cover for small species seeking shelter from predators, as well as benefitting species that forage and nest in dense cover.

Decreasing species

There were statistically significant decreases in eight species and a ninth species decreased at near-significant level (p=0.06) based on linear trend analysis (**Table 2**). Feeding on or close to the ground is an important component of the foraging activities of Double-barred Finch, Jacky Winter and Speckled Warbler, three of the five species which were unrecorded in the last six years of the

Table 3. Comparison of the ten most frequently recorded species at Green Wattle Creek Survey Site 3 for the periods 1996-1998 and 2011-2013.

Rank	1996-1998	RR (%)	Rank	2011-2013	RR (%)
1	Grey Fantail	81.8	1	Yellow-faced Honeyeater	97.2
2	Superb Fairy-wren	66.7	2	Grey Fantail	75.0
3	Yellow-faced Honeyeater	63.6	3	Lewin's Honeyeater	63.9
4	Eastern Yellow Robin	45.5	4	Spotted Pardalote	47.2
5	Fuscous Honeyeater	39.4	5	Eastern Yellow Robin	41.7
6	Spotted Pardalote	36.4	6	Brown Thornbill	38.9
7	White-browed Scrubwren	33.3	7	Golden Whistler	38.9
8	Striated Thornbill	27.3	8	White-throated Treecreeper	33.3
9	Speckled Warbler	27.3	9	Eastern Whipbird	30.6
10	Lewin's Honeyeater	24.2	10	Silvereye	30.6

study. The development of dense shrub layer vegetation eliminated this opportunity, rendering Site 3 unsuitable. In the case of the Speckled Warbler this decrease occurred throughout the woodland at Green Wattle Creek (Newman 2010). In spring 2011 weed removal and controlled burns temporarily restored patches of habitat to a condition similar to that when grazed. The Speckled Warblers returned (Newman 2014a) suggesting that the decrease at Survey Site 3 was caused by lack of grazing. While a similar explanation can be proposed for the decrease of the Double-barred Finch, there may have been other contributing causes as there was a contemporaneous decrease throughout much of the Paterson area (Newman 2014b). Jacky Winter have more foraging options than the previous two species. They hunt by spotting prey while perched and taking food from a variety of substrates as well as hawking aerially (Keast 1985). Development of tall dense shrub vegetation is counterproductive to some of these foraging techniques.

It was around six years before Superb Fairy-wrens decreased (**Figures 3** and **6b**). In contrast, the Variegated Fairy-wren RRs increased initially, before decreasing (**Figure 3**). Both species were present throughout the study. It is suggested that in the initial stages increased shrub layer vegetation provided improved shelter, and that this was particularly advantageous to the Variegated Fairy-wren. However, as the vegetation became denser, foraging became more difficult. However, some of the apparent decrease may have been a consequence of decreased detectability as discussed in the methods section.

It is less obvious why the Fuscous and White-naped Honeyeaters should abandon the survey site towards the end of the study (**Table 3**) as both forage in the canopy and changes in the shrub layer vegetation would have less impact. Chan (1990) indicates that Fuscous Honeyeaters favour eucalypt woodland away from stream beds and with a poorly developed shrub layer. This supports the proposition that a dense shrub layer would render the habitat unsuitable. The same logic presumably applies to the sympatric White-naped Honeyeater. Chan (1990) indicates that both species are highly aggressive and generally avoid each other. The decrease of the Fuscous Honeyeater coincided with a statistically significant increase in the Yellow-faced Honeyeater (**Figure 4**), which may have benefitted from the decreased competition with Fuscous Honeyeaters.

The other species which decreased to a statistically significant extent were the Eastern Rosella *Platycercus eximius*, Grey Fantail, Striated Pardalote *Pardalotus striatus* and Yellow Thornbill *Acanthiza nana*. Most of the Eastern Rosella records occurred in the first half of the study. The lack of records after dense shrub layer vegetation developed reflects their preference for similar habitat to Noisy Miners *Manorina melanocephala* with both species generally avoiding areas with dense understorey vegetation (Higgins *et al.* 2001; Newman 2013). Despite experiencing a statistically significant decrease in RR the Grey Fantail remained the second most frequently recorded species (**Table 3**). In a previous analysis (Newman 2012) fluctuations in Grey Fantails were attributed to a combination of the impacts of removal of cattle and rainfall. Grey Fantails use most strata when foraging, but concentrate on the periphery of trees and shrubs, as well as hawking in the open air adjacent to them (Cameron 1985). Dense understorey vegetation would progressively limit the opportunities for Grey Fantails to forage other than in the canopy. It is not obvious why changes in the understorey vegetation should have affected the Striated Pardalote and Yellow Thornbill adversely as both are predominantly canopy-feeding species.

Increasing species

Lewin's Honeyeater increased from the 10^{th} to the 3^{rd} most frequently recorded species (**Table 3**). The increase, which was highly significant statistically, was sustained throughout the study (**Figure 6a**). Lewin's Honeyeater, an arboreal species, feeds at all levels, in shrubs and trees and occasionally on the ground (Higgins *et al.* 2001) and would have benefitted from the establishment of understorey vegetation. It is a vocal species which is easily detected, even when foraging in dense cover.

Brown Thornbill, which glean from the foliage (Bell 1985), was another species which benefitted from the increase in shrub layer vegetation. Despite a slightly lower RR in the final three years (**Figure 5**) the overall increase was highly significant statistically. A similar increase was observed for the White-browed Scrubwren, a species which forages mainly in undergrowth (McDonald 2007), but in this instance the drop in the RR in the final three years was more pronounced, and the overall increase was not statistically significant. A possible explanation is that this species has an optimal shrub layer structure for foraging and the vegetation density eventually exceeded this limit.

The increased occurrence of the Australian Raven *Corvus coronoides* was unexpected and no explanation is offered.

Species which peaked mid-study

In the previous section it was suggested that the shrub layer vegetation can become so dense that it inhibits optimal foraging. The smooth trend for the Eastern Whipbird, a species which habitually lives in dense ground level vegetation (**Figures 3** and **6c**) supports this hypothesis. The timing of the peak occurrence of the whipbird, approximately six years before the peak for White-browed Scrubwren, may be a consequence of its larger size restricting its ability to forage as the vegetation became increasingly dense.

For species such as the Rufous Whistler, which forage predominantly in or close to the canopy, it is less obvious how there would be a stage in understorey vegetation which provided optimal foraging opportunities. However, Rufous Whistlers do spend some time foraging in the upper levels of shrub layer vegetation and on trunks (Keast 1985), and this may be a contributing factor.

Cattle as bird habitat managers

This opportunistic study demonstrates how the exclusion of cattle can transform an area of woodland and its bird population. In such situations cattle are usually cast as the villains, but it is worth reflecting that at the start of this study they contributed to maintaining habitat which supported an assemblage of birds that were locally unusual, perhaps unique in the Paterson area of the Hunter Valley. In addition to the Speckled Warbler and Double-barred Finch other species like the Brown Treecreeper *Climacteris picumnus*, Buff-rumped Thornbill *Acanthiza reguloides* and Painted Button-Quail *Turnix varius*, which all favour open woodland, decreased elsewhere in the Green Wattle Creek Reserve (Newman 2009). The core range of many of these species lies well to the west of Green Wattle Creek. Some of these species have become scarce and one, the Brown Treecreeper, is no longer found in the area surrounding Green Wattle Creek (Stuart 2017).

It is possible to speculate that removing cattle from the woodland at Green Wattle Creek might allow it to revert to a state broadly similar to that which existed before European settlement. Attempts to find locations in the Paterson area with similar bird assemblages to those existing at the start of this study suggests that Green Wattle Creek may have been the last bastion of extensively grazed woodland there. This suggests that local changes in land use and management, such as acreage subdivisions, have resulted in a change in avian biodiversity including changes in the status of threatened species (e.g. Speckled Warbler).

Unfortunately, when an area which has been grazed, reverts to its natural state, weeds, such as Lantana, will often dominate the shrub layer vegetation, as occurred at Green Wattle Creek. Consequently, the vegetation may revert to a condition involving floristic and structural attributes that are different from those that existed previously. Therefore the area may no longer be capable of supporting the bird assemblages present before grazing commenced.

At Green Wattle Creek a program of burning and weed removal has been implemented to restore the vegetation to a state which may provide habitat for species which require more open shrub layer vegetation (Newman 2014a).

Limitations of a single survey site analysis

Survey Site 3 sampled approximately 2% of the Green Wattle Creek Reserve in which there was considerable variation in habitat (Newman 2009). As anticipated, the results of monthly surveys provided reliable trends for species that were regularly recorded (see 95% confidence bounds in **Figure 6**) and it was possible to determine whether statistically significant changes in status had occurred over the 18-year duration of the study. However, there are two caveats to the conclusions drawn from this study; they concern the extent to which Survey Site 3 was representative of the whole reserve and the lack of statistical power for the less common species with low RRs (e.g. Speckled Warbler and Double-barred Finch). To address these problems surveys were conducted at four 2-ha sites selected to sample differences in habitat, which provides the future opportunity to address both these issues. For instance, the occurrence of species with high RRs, such as the Yellow-faced Honeyeater, can be compared between survey sites and for species with low RRs the statistical power can be increased by pooling the results across the survey sites. However, the complexity of the analysis is much greater, as exemplified by results published for the Grey Fantail (Newman 2012).

CONCLUSIONS

When cattle were removed from woodland at Green Wattle Creek there was a rapid increase in the number of bird species, including those which were recorded regularly (2-ha survey RRs exceeding 20%). This increase was attributed to the development of understorey vegetation. For many species these increases were not sustained, and after 15 years the avian diversity had decreased to its initial level. However, there were substantial differences between the bird assemblages at the start and end of the 18-year study.

The "winners" were species that benefitted from increased shrub layer vegetation, such as the Brown Thornbill, White-browed Scrubwren and Eastern Whipbird. However, as the shrub layer vegetation became denser some of these species decreased from their mid-study peak levels, consistent with the proposition that they were no longer able to move through the shrub layer and forage effectively.

The "losers" were predominantly species which prefer open habitat, particularly those which forage on or near the ground, such as the Speckled Warbler, Double-barred Finch and two species of fairy-wren. Foliage-feeding honeyeaters were indirectly affected, with Fuscous and White-naped Honeyeaters both decreasing. These aggressive species avoid areas of woodland with dense understorey vegetation. In their absence the Yellow-faced Honeyeater increased, becoming the most frequently recorded species.

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APPENDIX

Species recorded at Green Wattle Creek Survey Site 3 between April 1996 and December 2013 which had a Reporting Rate exceeding 10% in at least one three-year period.

Species		**1996-98 RR (%)**	**1999-01 RR (%)**	**2002-04 RR (%)**	**2005-07 RR (%)**	**2008-10 RR (%)**	**2011-13 RR (%)**
Australian Raven	*Corvus coronoides*	3.0	2.9	13.9	11.8	28.6	19.4
Bar-shouldered Dove	*Geopelia humeralis*	0.0	2.9	2.8	11.8	2.9	11.1
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	12.1	8.8	13.9	20.6	8.6	11.1
Brown Gerygone	*Gerygone mouki*	12.1	0.0	8.3	2.9	0.0	2.8
Brown Thornbill	*Acanthiza pusilla*	12.1	23.5	41.7	41.2	42.9	38.9
Double-barred Finch	*Taeniopygia bichenovii*	15.2	8.8	2.8	2.9	0.0	0.0
Eastern Rosella	*Platycercus eximius*	18.2	32.4	13.9	11.8	0.0	11.1
Eastern Spinebill	*Acanthorhynchus tenuirostris*	18.2	38.2	36.1	41.2	45.7	27.8
Eastern Whipbird	*Psophodes olivaceus*	6.1	26.5	63.9	61.8	42.9	30.6
Eastern Yellow Robin	*Eopsaltria australis*	45.5	50.0	47.2	32.4	28.6	41.7
Fuscous Honeyeater	*Ptilotula fusca*	39.4	44.1	47.2	32.4	0.0	0.0
Golden Whistler	*Pachycephala pectoralis*	24.2	41.2	44.4	20.6	34.3	38.9
Grey Fantail	*Rhipidura fuliginosa*	81.8	88.2	75.0	76.5	57.1	75.0
Grey Shrike-thrush	*Colluricincla harmonica*	18.2	14.7	19.4	14.7	17.1	8.3
Jacky Winter	*Microeca fascinans*	9.1	8.8	19.4	2.9	0.0	0.0
Laughing Kookaburra	*Dacelo novaeguineae*	0.0	8.8	8.3	5.9	0.0	13.9
Leaden Flycatcher	*Myiagra rubecula*	12.1	2.9	5.6	5.9	5.7	5.6
Lewin's Honeyeater	*Meliphaga lewinii*	24.2	32.4	44.4	50.0	51.4	63.9
Red-browed Finch	*Neochmia temporalis*	9.1	32.4	19.4	20.6	8.6	16.7
Rufous Whistler	*Pachycephala rufiventris*	18.2	17.6	30.6	11.8	11.4	13.9
Scarlet Honeyeater	*Myzomela sanguinolenta*	0.0	23.5	11.1	23.5	11.4	13.9
Silvereye	*Zosterops lateralis*	12.1	29.4	27.8	17.6	11.4	30.6
Speckled Warbler	*Pyrrholaemus sagittatus*	27.3	8.8	0.0	0.0	0.0	0.0
Spotted Pardalote	*Pardalotus punctatus*	36.4	38.2	52.8	73.5	40.0	47.2
Striated Pardalote	*Pardalotus striatus*	21.2	14.7	27.8	5.9	0.0	2.8
Striated Thornbill	*Acanthiza lineata*	27.3	32.4	47.2	32.4	48.6	27.8
Superb Fairy-wren	*Malurus cyaneus*	66.7	67.6	50.0	35.3	14.3	16.7
Variegated Fairy-wren	*Malurus lamberti*	21.2	23.5	36.1	29.4	25.7	8.3
White-browed Scrubwren	*Sericornis frontalis*	33.3	35.3	38.9	41.2	60.0	27.8
White-naped Honeyeater	*Melithreptus lunatus*	12.1	17.6	2.8	0.0	0.0	0.0
White-throated Treecreeper	*Cormobates leucophaea*	12.1	17.6	16.7	35.3	11.4	33.3
Yellow Thornbill	*Acanthiza nana*	21.2	26.5	8.3	14.7	8.6	11.1
Yellow-faced Honeyeater	*Caligavis chrysops*	63.6	79.4	91.7	82.4	85.7	97.2

Geolocators track Ruddy Turnstone to Newcastle, NSW en route to King Island (Tasmania)

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The tracking of birds using light-level geolocators has become a relatively frequent technique in the study of migratory shorebirds. The geolocator program, commenced in Australia by the Victorian Wader Studies Group in 2009, has provided insights into many of the strategies and outcomes of the species studied. The most numerous of these studies have been on the Ruddy Turnstone *Arenaria interpres*. While an increasing number of these are multiple tracks for the same bird, there are relatively few with field sightings to enable supporting calibration and confirmation of computed locations, hence the value of the sightings in Newcastle of Ruddy Turnstone with leg flag WMA described in this paper. The migrations of this bird, described over three consecutive years, show southward tracks over the Pacific Ocean and stopovers in Newcastle on its return journeys to King Island (Tasmania). Information regarding breeding locations and incubation characteristics are also described.

INTRODUCTION

While light-level geolocators have long been deployed to provide an understanding of a range of animal movements, it is over the last 10 years that they have been used on a number of shorebird species to track migratory movements and identify breeding, stopover, and wintering areas (Bridge *et al*. 2011, Tomkovich 2016, Lisovski *et al*. 2016a). These devices measure and store ambient light levels which can be used to determine latitude and longitude when the data are downloaded. They have become a frequently used tool in migration research. Australia was one of the first countries to utilise these loggers for tracking the movements of migratory shorebirds. Since 2009 the Victorian Wader Study Group (VWSG) and the Australasian Wader Studies Group have deployed geolocators on a range of species at non-breeding locations around the country, including coastal Victoria, King Island (Tasmania), SE South Australia, NW Western Australia, and SE Queensland. This extensive program has gathered a wealth of information on the movements of eight of Australia's long-distance migratory species, mostly with high retrieval rates, and, after some initial technical issues, high success of the units deployed. The migratory tracks obtained, including several multi-year tracks, allow us to detail routes and strategies used along the East Asian-Australasian Flyway. Critically, this information has allowed the assessment of the relative importance of stopover sites along the Flyway - fundamental to developing conservation strategies. More recent geolocator units have also enabled assessment of breeding locations and incubation strategies, many of which were unknown given the remote, low density breeding sites used by these species. These insights have informed conservation measures Flyway-wide (including the development of initiatives for the Yellow Sea) and on a local scale.

The Ruddy Turnstone *Arenaria interpres* was chosen for the initial studies using geolocators due to its relative abundance at selected sites and for its site faithfulness, having in mind the need to recapture the bird to retrieve the instrument. Over the period 2009 to early 2017, a total of 485 geolocators were deployed on this species, of which 206 have been retrieved (52%). The VWSG has undertaken studies on other species in southern Australia including Eastern Curlew *Numenius madagascariensis,* Sanderling *Calidris alba* and Red-necked Stint *Calidris ruficollis*; geolocators were deployed on Curlew Sandpiper *Calidris ferruginea* for the first time this year. These programs have only been possible through the dedicated volunteers of the VWSG and the close collaboration of Professor Marcel Klaassen and his team at Deakin University.

The following provides a snapshot of the nature and extent of the program and highlights the Ruddy Turnstone with leg flag WMA which appears to have regularly utilised the Newcastle region as part of its migration strategy on its southward journey back to King Island where it spends at least part of its non-breeding period.

METHODOLOGY

In common with all species on which geolocators have been deployed, Ruddy Turnstones were captured in cannon nets at high-tide roosts. Over the last 4 years, geolocators identified as Intigeo-W65 supplied by Migrate Technology Ltd, Cambridge, UK, have been adopted (see http://www.migratetech.co.uk/IntigeoSummary.pdf). The units were mounted on plastic leg-flags (made from a Darvic PVC sheet) using Kevlar thread reinforced with Araldite resin cement. The geolocator unit weighed 0.65g and when mounted vertically on a flag, the combined weight was 1.2g (1.2% of fat-free body weight). The geolocator was placed on the left tibia of each bird (**Figure 1**). All units were deployed on adult birds considered to be in their second year of life (or older). During the subsequent non-breeding periods following deployment, Ruddy Turnstones carrying geolocators were specifically targeted in cannon net catches.

Figure 1. Geolocator deployed on a Ruddy Turnstone

The bird, WMA, a male (sexed by plumage), was initially captured and banded in a cohort of 39 Ruddy Turnstones at Burgess Bay on King Island on 13 February 2015. The initial geolocator was deployed on WMA at this time. It was recaptured at Manuka South on King Island on 30 March 2017 and the geolocator removed and a new one fitted. This was subsequently retrieved at Burgess Bay on 9 December 2017.

Migration Pathway

Data were downloaded from the retrieved geolocators and the initial analysis undertaken using the threshold method embodied in the manufacturer's software (Lisovski & Hahn 2012). This enabled locations to be computed from the record of sunrise and sunset events. This method fails to produce position estimates under 24-hour daylight conditions as the sun does not fall below the horizon and sunrise/sunset times cannot be detected (Lisovski *et al.* 2012). To further improve the location estimation accuracy and estimates of uncertainty, an analysis using the R package FlightR (Rakhimberdiev *et al.* 2017) was subsequently undertaken. The location accuracy is estimated to be between 70 and 200 km depending on weather and latitude. The record of conductivity was used to supplement these analyses as this indicates when the bird was in flight and when it was feeding in saline water. These results enabled maps to be drawn to show the best estimate of the routes taken by each bird together with major stopover locations. Dates were also extracted for all important elements of each bird's movement.

Breeding Sites

We used the template-fit method described in Lisovski *et al.* (2016a) to estimate the positions of the breeding sites. The level of accuracy of the estimated breeding sites is 100-300 km Lisovski *et al.* (2016a).

Incubation Pattern

Both the light intensity recordings and the recorded temperature patterns over time can be used to make inferences of the incubation and chick-rearing behaviour on the breeding grounds. Gosbell *et al.* (2012) describes how the occurrence of alternating "light" and "dark" signals in the geolocator output recorded in the breeding area was interpreted as an indication for shading associated with nesting activities including, especially, incubation and brooding (see also Lisovski *et al.* (2016b)).

RESULTS

The data collected from the two geolocators fitted to WMA recorded 3 consecutive years of migrations viz 2015, 2016 (partial) and 2017. The migration tracks are shown in **Figure 2**.

2015 Track

Following deployment of the geolocator in February 2015, the bird remained on King Island until departure on 18 April. It flew nonstop to

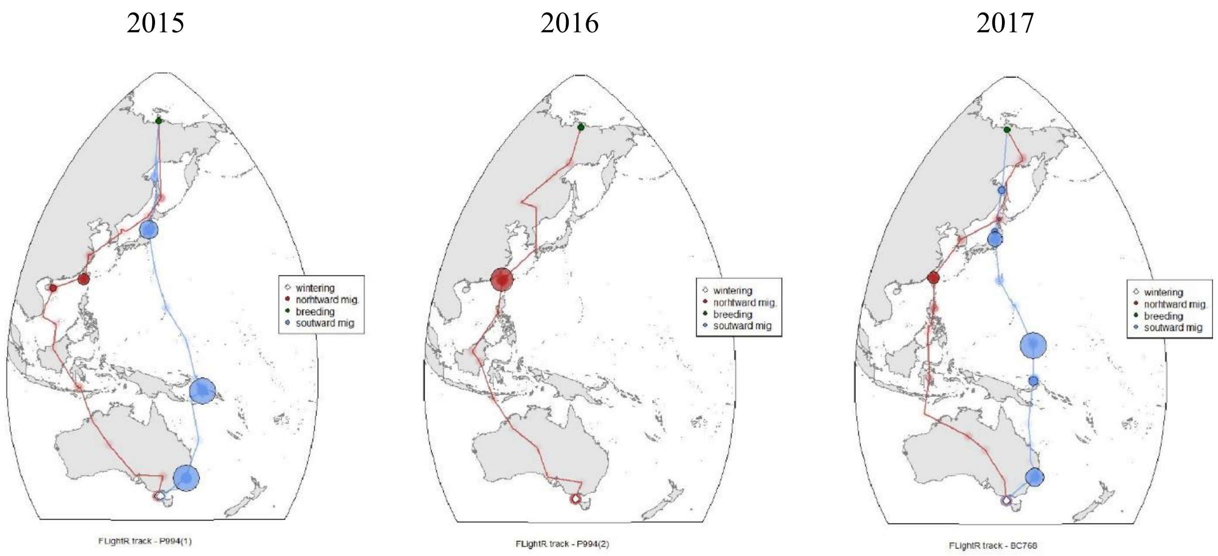

Figure 2. Migration tracks for Ruddy Turnstone WMA for years 2015, 2016, 2017.

Hainan where it remained until 6 May when it went to Taiwan and stayed 6 to 23 May before flying to Sakhalin Island and on to the breeding grounds (estimated latitude 72°N longitude 142°E), arriving 2 June. It stayed on the breeding grounds until 23 July when it travelled south to the west side of the Sea of Okhotsk and on to Japan where it stopped over from 1 to 29 August. It then made a long non-stop flight of 5,500 km across the Pacific to Bougainville Island arriving 8 September and staying until 20 October. On this date it flew south to the Newcastle region of NSW. It was observed at Stony Point and Newcastle Beach on 23 October, 28 December and was last photographed there on 16 January 2016 (pers. comm. J. Thomas). The bird returned to King Island on 13 February 2016.

2016 Track

The bird was not recaptured before it departed again on 12 April 2016 and flew non-stop to Taiwan (7,500 km) in 6 days at an estimated average ground speed of 55 kph. It stopped over in Taiwan until 24 May when it headed for the breeding grounds with only a short stop on the western side of the Sea of Okhotsk. It reached the breeding grounds, estimated to be latitude 70°N longitude 146°E, on 31 May. It departed the breeding grounds on 23 July; however, after this time the geolocator failed to record any further data. The track south is therefore not available, but the bird was observed again at Newcastle Beach on 4 November 2016. The bird subsequently returned to King Island where it was recaptured on 30 March 2017 and a new geolocator attached.

2017 Track

WMA again departed on its northward migration on 23 April 2017 and flew to Taiwan arriving 6 May after a short stop in the Philippines. On 24 May it flew to Sakhalin Island, and then on to the breeding grounds arriving 6 June (latitude 70°N longitude 142°E). It departed the breeding grounds 9 August, flew to the west coast of the Sea of Okhotsk and on to Japan where it arrived 16 August. On 7 September it flew south across the Pacific, 3,500 km to the Chuuk Islands, a series of atolls in Micronesia. It stayed from 15 September to 12 October when it flew south to East New Britain. On 24 October it flew to Newcastle arriving 28 October. It was observed there on 28 October (pers. comm. J. Thomas). On 24 November it undertook its final leg back to King Island where it was captured 9 December and the geolocator removed (see **Figure 3**).

Refer to **Table 1** for key dates by year.

Incubation

A review of the minimum temperatures and light intensity recorded on the geolocator indicated that in 2015 there was no sign of incubation, in 2016 there were signs of a start of incubation, but it ceased on 6 July. In 2017 there were clear signs of incubation from 24 June to 11 July; as this was only 17 days it was probably not successful. (See **Figure 4**).

Table 1. Key dates for Ruddy Turnstone WMA by year

Year	Depart King Is.	Arrive breeding grounds	Depart breeding grounds	Arrive Newcastle	Depart Newcastle	Arrive King Is.
2015	18 Apr	2 Jun	23 Jul	23 Oct	?	13 Feb 16
2016	12 Apr	31 May	23 Jul	Seen 4 Nov[1]	?	Geo retrieved 30 Mar 17
2017	23 Apr	6 Jun	9 Aug	28 Oct	24 Nov	Geo retrieved 9 Dec 17

[1] Geolocator failed July 2016.

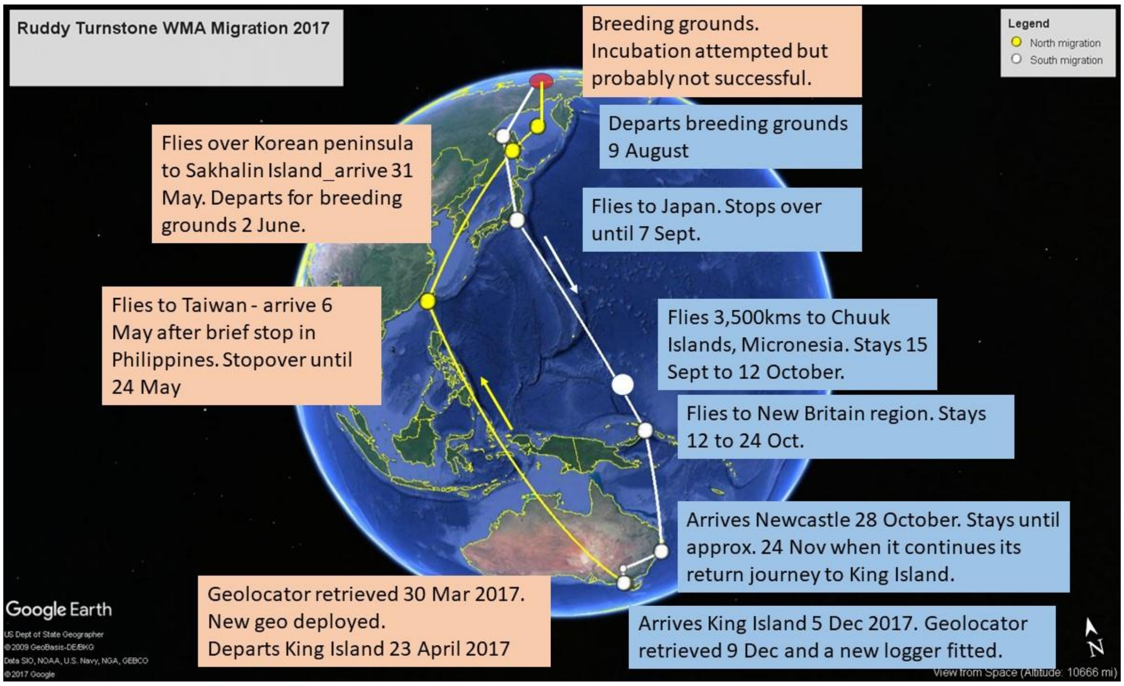

Figure 3. Track of Ruddy Turnstone WMA in 2017.

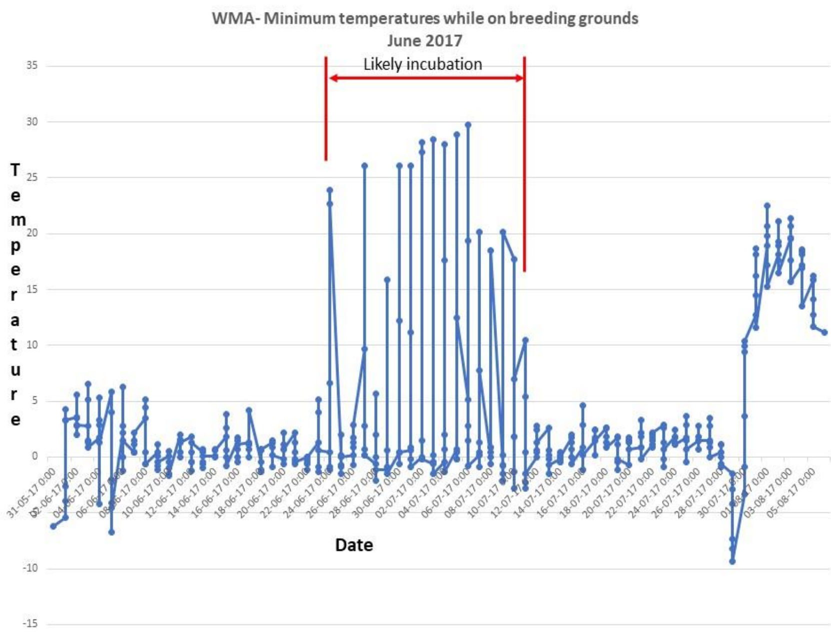

Figure 4. Chart showing minimum temperatures recorded by geolocator for the period the bird was on the breeding grounds in June 2017. The chart indicates the likely incubation period.

DISCUSSION

The tracking of birds using light-level geolocators has become a relatively frequent technique in the study of migratory shorebirds. The geolocator program, commenced in Australia by the VWSG in 2009, has provided insights into many of the strategies and outcomes of the species studied. There are currently 197 tracks now available as a result of studies on Ruddy Turnstone at three locations in southern Australia, namely Flinders (Victoria), SE South Australia and King Island (Tasmania). An increasing number of these are multiple tracks for the same bird. However, there are relatively few with field sightings to enable supporting calibration and confirmation of computed locations, hence the value of the field sightings in Newcastle.

There are several features of the tracks described for WMA that are of particular interest. Firstly, while this bird undertook its northern migration through Taiwan and the coasts of China and the Sea of Okhotsk to the breeding grounds in northern Siberia, its track south was via Japan and the Pacific. While the majority of pathways south to the non-breeding areas undertaken by Ruddy Turnstone are generally similar, but not identical, to the pathways followed on northern migration, there have been only a limited number of examples where the bird has travelled southeast from the breeding grounds and crossed the Pacific on its return to Australia. The earliest example was bird 9Y which returned to Flinders (Victoria) with a stopover in the Marshall Islands (Minton *et al.* 2010). The subject bird, WMA, flew from Japan to Bougainville Island in 2015 and an atoll in Micronesia in 2017 (the logger failed in 2016 so no track available). At this stage there are no cogent explanations why this is done, but it would seem that having established this strategy, the bird repeats the same or a similar route in subsequent years.

Secondly, it made a major stopover in the Newcastle region before returning to King Island. Once again, this is relatively unusual based on the tracks available at this stage. Furthermore, it appears to have made this a major stopover and even refuelling site, judging by the fact that it returned to King Island in 2016 only 7 weeks before it departed on its northern migration. Again, it adopted the same stopover location for 3 consecutive years. The value of the sightings in Newcastle are obvious not only for confirming the computed tracks but establishing the importance of this area as a refuelling location.
The breeding locations derived from the geolocator data for WMA are in the high Arctic on the northern slopes of Yakutia. Analyses of other Ruddy Turnstone data have indicated the breeding areas of this species from southern Australia to cover a range from Yakutia to the New Siberian Islands (unpublished). The 3 breeding locations identified from the analyses are within a 90 km radius, well within the accuracy for this methodology. Although incubation was attempted for the 3 years covered by these data, WMA did not achieve the full term of incubation, reported to be in the range 20.5 - 24 days for this species (Cramp & Simmons 1983). The variability of incubation success has been shown in Gosbell *et al*. (2012), and subsequent unpublished results.

The departure dates from King Island were 18, 12, and 23 April in 2015, 2016 and 2017 respectively. In common with the majority of other Ruddy Turnstones, the first leg is either a non-stop flight to Taiwan or an initial stop in Hainan and then on to Taiwan. Taiwan has been shown to be a critical stopover and refuelling location for northward migrating Ruddy Turnstones (Minton *et al*. 2013 and unpublished data).

CONCLUSION

The study of this one bird over three years has provided a lot of useful information relating to migration strategies, timings and major stopover locations including the value of appropriate Australian sites. In addition, breeding locations were derived to be in the high Arctic regions of Siberia in common with other birds of this species. An insight into incubation behaviour on the breeding grounds was also outlined.

This study has also reinforced the value of sighting and reporting birds equipped with a geolocator anywhere in the Flyway. With another replacement geolocator deployed on WMA in December 2017 it will be interesting to retrieve it and follow another year of migration.

ACKNOWLEDGEMENTS

Thanks are due to the many people who have contributed to the fieldwork including the deployment and retrieval of geolocators on King Island. To the members of the VWSG who so generously travel to King Island as part of this program and to the many local people involved, we thank you. Of particular significance has been the collaboration with Professor Marcel Klaassen and his team at Deakin University who have contributed expertise as well as physical and financial resources to the program. Special thanks are also given to the members of the Hunter Bird Observers Club who so diligently recorded WMA when it was present in Newcastle.

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Do nest boxes facilitate breeding success in the Hunter Valley? Common Mynas versus native parrots

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Urban environments rich in native wildlife play an essential role in fostering broader public appreciation of natural areas on a global scale. On the New South Wales eastern seaboard, a few native secondary cavity-nesting bird species are successfully colonizing our cities, but their population growth is likely limited by an overall paucity of tree nesting cavities in urban habitats. Here, we sought to determine whether urban nest boxes support breeding by native parrots, or whether competition with non-native secondary cavity-nesters, particularly the invasive Common (Indian) Myna *Acridotheres tristis*, offsets the benefits. We installed and monitored 126 nest boxes in three different locations of the Hunter Valley (Greater Newcastle area, Gloucester and Krambach) across three breeding seasons. Overall, across all locations and all three seasons, native parrots were more common occupants of our nest boxes than mynas. While mynas were the first to breed successfully in the nest boxes, over time native parrots fledged more and more chicks. We discuss the possibility that nest boxes, provided they are maintained across several years, might facilitate colonization of cities by native parrots and their population expansion, while leaving populations of mynas relatively unaffected despite some mynas choosing to nest in boxes. If future research supports this hypothesis, then urban nest boxes could provide a viable management tool for enhancing the presence of native cavity-nesting birds in our cities.

INTRODUCTION

Wildlife in urban areas raises public awareness of biodiversity and the natural world, strengthening citizen support for global environmental protection efforts. Consequently, it is imperative that we design wildlife-friendly cities. In response to rampant urban expansion, a few native Australian secondary cavity-nesting bird species are successfully colonizing our cities. One factor limiting their population growth in these new environments, however, is the overall paucity of tree nesting cavities in urban habitats (Harper *et al.* 2005; Morton 2013) . Australia exhibits the highest number of cavity-nesting birds in the world, but all of them are secondary cavity-nesters (they use existing hollows). No excavating vertebrate species, such as the woodpeckers (primary cavity-nesters), are present in Australia to regularly create new cavities (Gibbons & Lindenmayer 2002). Installing nest boxes is a potential means of supplementing the natural supply of nesting opportunities to facilitate population growth (Griffith *et al.* 2008; Newton 1994). Unfortunately, invasive secondary cavity-nesting bird species, like the Common Myna *Acridotheres tristis*, and mammalian egg and chick predators, like the Brush-tail Possum *Trichosurus vulpecula*, are also common occupants of nest boxes (Grarock *et al.* 2013; Harper *et al.* 2005). There is a risk that the benefits of providing additional nesting resources to native secondary cavity-nesting bird species might end up being off-set by the presence of these competing avian and mammalian cavity users. Even worse, nest boxes might lead to undesirable increases in invasive cavity-nesting birds. Despite these caveats, without careful, quantitative, long-term research, the costs and benefits of urban nest-box programs, and their relative effectiveness, will remain a matter of conjecture. The need to collect quantitative data on the use and effectiveness of urban nest boxes by native and invasive species provided the impetus for the present study. After building an array of 126 nest boxes distributed in several locations across the Hunter Region (New South Wales), we sought to record occupancy and breeding success of native parrots and the introduced Common Myna across three successive breeding seasons.

METHODS

To explore the breeding success of introduced Common Mynas and native secondary cavity-nesting parrots, we installed 126 nest boxes in seven study sites in the Hunter Valley. Seventy-eight boxes were located in the

Newcastle area (New Lambton, Glendale, Waratah, Jesmond and Broadmeadow racecourse) and a further twenty-four were placed in each of Gloucester and Krambach. These boxes were installed within a gradient of urbanisation divided into three sub-environments (egde of the bush, park, and urban) (**Figure 1**). We used vertical nest boxes made of plywood (Nest Boxes Australia, Loganholme 4129 Australia) of internal dimensions 400 (H) x 170 (W) x 170 (D) mm, and equipped with a hole size of 65 mm suitable for native cavity-nesting birds the size of the Psittacidae family and the Common Myna.

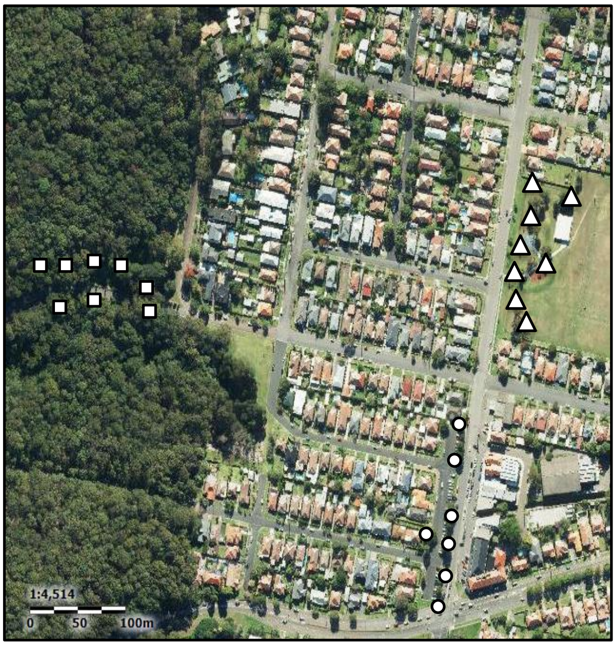

Figure 1. Map of the three different sub-environments of one study site (New Lambton, Newcastle). White dots (urban sub-environment), triangles (park sub-environment) and squares (edge of the bush sub-environment). Each symbol refers to one nest box.

We monitored all nest boxes weekly using a gooseneck camera set up on a long pole for three entire breeding seasons (September 2014 to April 2017). We noted which species was in each box, as well as the number of eggs, nestlings and fledglings. For each nest box, we recorded the number of times a species attempted to breed (at least one egg laid). In cases where no egg in a clutch hatched, or none of the nestlings fledged, we recorded a nest failure. Each time we found a box to be empty the week after it had been recorded as containing nestlings very close to fledgling age, we recorded a successful nest. To estimate the breeding success, the number of fledglings per individual was calculated by dividing the number of fledglings produced by a pair of parents over each entire breeding season by two (i.e. number of chicks surviving /2).

RESULTS

Nest box occupancy

Overall, 23.02% of the 126 nest boxes were occupied by bird species during the first (2014-2015) breeding season. Occupancy increased to 30.16% during the second (2015-2016) breeding season and to 32.54% during the third (2016-2017) breeding season. We found native parrots in approximately 55% of occupied boxes, whereas Common Mynas occurred only in approximately 45% of occupied boxes (**Table 1**). The first breeding season, 11 boxes were occupied by Eastern Rosellas *Platycercus eximius*, 3 by Crimson Rosellas *Platycercus elegans* and 1 by Rainbow Lorikeets *Trichoglossus moluccanus*. The second breading season, 20 boxes were occupied by Eastern Rosellas, 1 by Crimson Rosellas and 2 by Rainbow Lorikeets. The third breeding season, 21 boxes were occupied by Eastern Rosellas, 1 by Crimson Rosellas and 1 by Rainbow Lorikeets. A few boxes were occasionally occupied by Brushtail Possums. Six boxes were regularly occupied by this species but essentially in boxes set on the edge of the bush (3 boxes in Glendale and 3 in Gloucester) where no birds were found nesting in this sub-environment.

Breeding success

Common Mynas fledged on average 1.06 ± 0.15 SE chicks per individual over the three seasons. This rate was twice that of native parrots, which fledged on average 0.59 ± 0.87 SE chicks per individual (**Table 1**, **Figure 2**). Myna breeding success increased from Year 1 to Year 2, but showed a slight decline in Year 3. In contrast, breeding success of native parrots increased consistently across the three years of the study (**Figure 2**).

Causes of failure

Across the three breeding seasons, we recorded 39 nest failures in native parrots for 71 nesting attempts (57%), whereas we recorded only 27 nest failures in Common Mynas for 68 attempts (38%). The percentage of nest failures varied across the breeding seasons. Both native parrots and mynas experienced high levels of nest failures in the first breeding season (75% and 59% respectively). Percentage failure decreased the following year to around 30% in mynas before remaining stable. Native parrot nest failures progressively decreased across the three breeding seasons to reach levels comparable to mynas in the third year of monitoring. Unfortunately, it is impossible to tease apart the causes of failure. For this reason, we split the causes of failures into only two categories for which we could be relatively certain: hatch failure and chick death. Using this categorization, the cause of nest failure differed between mynas and native parrots. Native parrot clutch failure was mostly attributable to hatch failure (~85%), whereas chick death was the most common cause of nest failure in Common Mynas (70%).

Table 1. Nest occupancy and breeding success in Common Mynas and native parrots across three breeding seasons in the Hunter Valley.

Site	Breeding season	Species	% of occupied boxes	Number of eggs	Number of fledglings
Newcastle	1	mynas	20.37	52	11
		parrots	16.66	33	4
	2	mynas	25.92	89	40
		parrots	37.03	115	22
	3	mynas	31.48	108	36
		parrots	31.48	77	26
Gloucester	1	mynas	0	0	0
		parrots	16.66	20	0
	2	mynas	0	0	0
		parrots	8.33	9	0
	3	mynas	0	0	0
		parrots	12.5	17	7
Krambach	1	mynas	12.5	16	4
		parrots	8.33	10	6
	2	mynas	8.33	9	3
		parrots	4.16	6	3
	3	mynas	4.16	5	2
		parrots	12.5	11	5

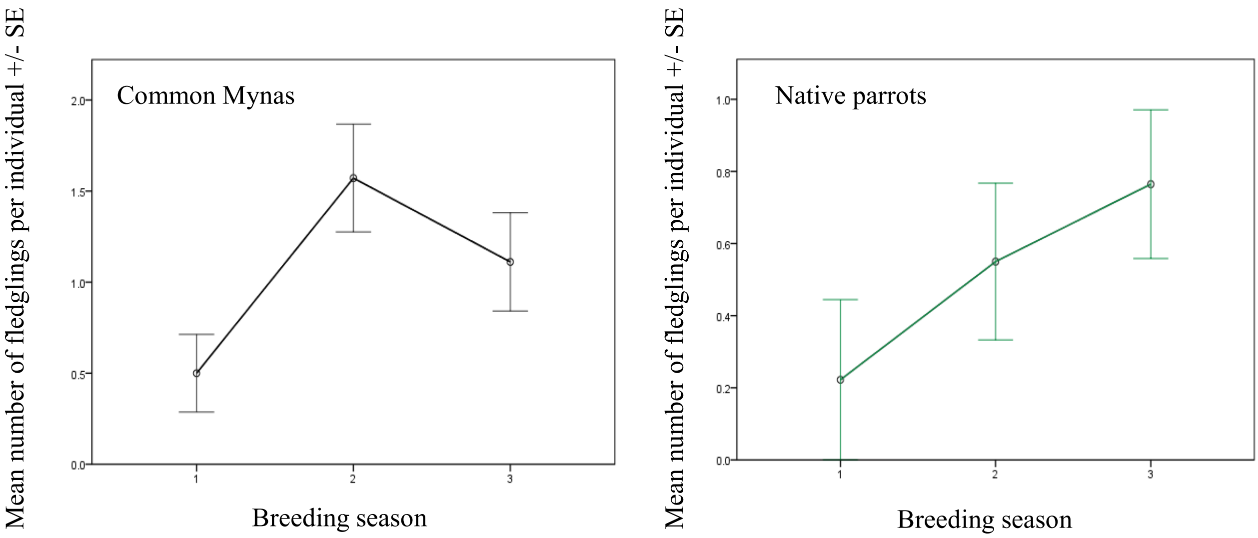

Figure 2. Evolution of breeding success (mean number of fledglings per individual +/-SE) in Common Mynas and native parrots across three successive breeding seasons in the Hunter Valley.

DISCUSSION

We undertook to study nest box occupancy and breeding success of native and invasive secondary cavity-nesting birds in the Hunter Region over the course of three successive breeding seasons. Our aim was to evaluate the effectiveness of urban nest boxes as a wildlife intervention strategy to support colonization of urban areas by native parrots. We found that occupancy increased across years, but native parrots, particularly Eastern Rosellas, were consistently more common occupants of the boxes than Common Mynas. Even though possums occupied a few boxes and many boxes remained empty, these patterns of occupancy by native parrots suggest that supplementing natural nesting cavities has the potential to assist the reproduction of native secondary cavity-nesting parrots even in the presence of the introduced secondary cavity-nesting Common Myna.

Nest box uptake is only the first step, however. Birds also need to be able to breed successfully in them. Although we found that native parrots displayed initially a lower reproductive success than the invasive Common Myna, this disadvantage decreased to almost zero in the third breeding season. Across the three years, the percentage of nest failures in native parrots (mostly Eastern Rosellas) dropped significantly and breeding success almost doubled. At this stage, we do not know why the principal cause of nest failure appears to be hatch failure in native parrots, and chick death in mynas.

At this stage of our research, we can only speculate about whether our boxes provided opportunities to individuals that would not have otherwise reproduced. However, based on our general observations, we suggest that native parrots do not (at this stage of urban colonization) commonly nest on man-made structures, such as the eaves and gutters of houses. In contrast, it is well known that Common Mynas are capable of nesting in a very large range of man-made structures. Hence, we suggest that our nest boxes are likely to have provided opportunities for native parrots that would not have otherwise nested elsewhere, whereas in contrast, Common Myna occupants, given the high nesting flexibility of this species, would have found alternative nesting locations had they not nested in our experimental nest boxes. This idea might be consistent with the heavy hatch failure in native parrots. We speculate that our nest boxes might have been taken up in Year 1 by young native parrot pairs looking for a territory to breed for the first time, and perhaps incapable of competing for natural tree cavities with older established pairs. These inexperienced breeders showed low breeding success initially, but returning each year to the same nest box, gradually gained breeding success and increased their breeding success across the three years of the study. At the present time, this idea remains pure conjecture. More research involving long-term monitoring and individual identification of native parrots, must be undertaken to test this hypothesis. Determining why so many eggs of native parrots do not hatch and whether this phenomenon is limited to the early years of nest box colonization will be particularly important for informing management practices.

CONCLUSION

Providing additional nesting sites (e.g. nest boxes) for native secondary cavity-nesting birds in urban areas could help support their colonization and population expansion in our cities. However, our new study indicates that these artificial nesting opportunities must remain in place for several years for native parrots to breed in them successfully. Our hypothesis that nest boxes provide supplemental nesting opportunities for native parrots that would not otherwise have reproduced, but only alternative nesting opportunities for mynas, which would have otherwise bred elsewhere, will require further investigation. If supported, this scenario will have the important implication that urban nest boxes can enhance population growth in native parrots while leaving Common Myna populations unaffected despite their use of them. Increasing the number of native parrots in urban areas might ultimately provide a competitive barrier to the Common Mynas and place downward pressure on the urban populations of this invasive bird.

ACKNOWLEDGEMENTS

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Early Hunter Region avian records

Part 4. 1951-1980 Articles in *The Emu*

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Articles about 1951-1980 Hunter Region birdlife appearing in *The Emu* have been reviewed. A bibliography of all the articles is included. Possible changes to the distribution range for some species since the 1950s have been discussed.

INTRODUCTION

The Hunter Region has long been known for its importance for Australian birds (Morris 1975; Cooper *et al.* 2014, 2016; Stuart 2009, 2013). A useful source of information about birds of the Region has been the journal *The Emu*, published since 1901 by BirdLife Australia (BLA) and its predecessor organisations e.g. the Royal Australasian Ornithologists Union (Robin 2001). For almost eight decades, *The Emu* contained many papers and short notes dealing with aspects of Hunter Region ornithology. However, since the mid 1970s it had relatively few regionally focussed articles and for several years prior, its importance for documenting regional birdlife had diminished considerably (for example, see Morris 1975; Cooper *et al.* 2014, 2016).

The objective of this present paper is to provide an overview of information about Hunter Region birdlife reported in *The Emu* for the period 1951-1980. Parts 1 and 2 in this series (Stuart 2009, 2013) reviewed *Emu* articles appertaining to the Hunter Region for the periods 1901-1925 and 1926-1950 respectively. Historical records of shorebirds, sourced from a range of publications, were reviewed in Part 3 (Stuart 2014).

It should be noted that articles in *The Emu*, taken in isolation, potentially create a misleading view of Hunter Region birdlife in the review period. A more complete perspective requires a review of all applicable literature and is outside the scope of the present paper.

METHODS

Approach taken

In reviews of 1901-1925 and 1926-1950 *Emu* articles (Stuart 2009, 2013), the Hunter Region was considered as six sub-regions; papers relating to each sub-region were discussed separately. That approach seemed less suitable for the present review. Although there was a steady stream of reports with a local focus, several significant papers presented information for a species or selection of species from a regional or national perspective. Discussion would have been repetitive if these were analysed at a sub-regional level, hence a different approach was taken. Firstly, reports relating to the various guilds (waterbirds, bush birds, etc) have been reviewed. Other sections deal with apparent changes in range or abundance, and with reports of rare or uncommon birds. A bibliography has been provided which lists every article relevant to the Hunter Region that appeared in *The Emu* between 1951 and 1980.

Nomenclature

This paper uses the taxonomy of BLA's Working List of Australian Birds V2.1 (BirdLife Australia 2018a). All the 1951-1980 articles in *The Emu* used then-current versions of the taxonomic list; amendments have been made wherever necessary.

THE 1951-1980 LITERATURE

Waterbirds

Heavy rain in 1950-51 created huge swamps in the Kurri Kurri / Maitland area and around Hexham; the latter was regularly surveyed during train journeys through the area (D'Ombrain 1951). Very large numbers of Pacific Black Duck *Anas superciliosa*, Grey Teal *Anas gracilis*, Purple Swamphen *Porphyrio porphyrio*, Eurasian Coot *Fulica atra*, and White-faced Heron *Egretta* *novaehollandiae* were present, and also lesser but substantial numbers of many other waterfowl species (D'Ombrain 1951). Noteworthy records included small numbers of Musk Duck *Biziura lobata* and Great Crested Grebe *Podiceps cristatus*, and many small parties of Hoary-headed Grebe *Poliocephalus poliocephalus*, Banded Lapwing *Vanellus tricolor* and Black-tailed Native-hen *Tribonyx ventralis*, Also, there were many sightings of Nankeen Night-Heron *Nycticorax caledonicus*, which originated from a large colony near Beresfield (D'Ombrain 1951). Intriguingly, D'Ombrain reported many breeding records around Hexham for the Hoary-headed Grebe, which now is considered a bird of passage with no modern evidence of any breeding attempts (Stuart 2017).

Arguably the most interesting Hexham record was of a pair of Australasian Bittern *Botaurus poiciloptilus* which habituated to the train's passage and often could be observed very close to the track (D'Ombrain 1951).

A pair of Black Bittern *Ixobrychus flavicollis* nested on the Paterson River near Tocal in 1954-55; three advanced young were in the nest in early January 1955 (D'Ombrain 1955). In December 1959 there was a very large influx of Whiskered Tern *Chlidonias hybrida* at drying swamps in the Hinton, Seaham and Paterson areas. Flocks of over 1,000 birds were often seen; prior to that it had seldom been recorded in the Hunter Valley (Hobbs & Kavaney 1962).

The Black-necked Stork *Ephippiorhynchus asiaticus* was first recorded in the lower Hunter Valley in 1930; seven birds were estimated to be present in the mid-1960s (Bell 1963; Salmon 1965). To the north, another 5-8 birds were believed to be in the Tea Gardens - Myall River area (Salmon 1965). A pair successfully bred at Tea Gardens in 1959 (Hobbs & Kavaney 1962). Salmon (1965) was uncertain if two birds recorded at Wallis Lake were additional or part of the Tea Gardens population. However, there were other reports of birds at Tuncurry near the mouth of the lake (Hobbs & Kavaney 1962).

Hobbs & Kavaney (1962) reported an influx of Great Crested Grebe to Smiths Lake near Bungwahl beginning in early 1960. The peak count was 23 birds in July. They commented that it was an uncommon species near the coast. Strangely, they did not mention D'Ombrain's report of small numbers of them present in the swamps around Hexham in 1951.

Shorebirds

Large numbers of shorebirds are known to have been present (in the Hunter Estuary in particular) in the 1960s and 1970s (Stuart 2014). However, there were very few reports of Hunter Region shorebirds in *The Emu*. It seems that birdwatchers mostly were using other forums to report their shorebird counts (see Stuart 2014 for details).

Over-wintering Whimbrel *Numenius phaeopus* in the Hunter Estuary and at Forster in the 1960s were mentioned briefly, and the presence of small numbers of Australian Pied Oystercatcher *Haematopus longirostris* in Port Stephens was recorded (Hobbs & Kavaney 1962).

Surveys in 1972 around Myall Lakes were focussed on terrestrial habitats and did not assess areas where shorebirds might occur (Recher 1975). However, an Appendix to the report included many shorebirds amongst a list of all birds known to have been recorded near Myall Lakes and Forster. The list was based upon numerous unpublished sources and some published ones, and spanned many decades (Recher 1975). More than 20 shorebirds were listed, including rarities such as Oriental Plover *Charadrius veredus*, Little Curlew *Numenius minutus* and Wandering Tattler *Tringa incana*. No additional details were supplied for any of the historical records.

Recher (1975) listed Latham's Snipe *Gallinago hardwickii* in a table of breeding residents of the Myall Lakes. This may just have been a case of clumsy wording. Latham's Snipe is a non-breeding visitor to Australia; it breeds in Japan and eastern mainland Asia (Marchant & Higgins 1993; BirdLife Australia 2018b).

Woodland and rainforest birds

The most comprehensive report of the era documented surveys by RAOU members in 1972 of areas under consideration for inclusion into the proposed Myall Lakes National Park (Recher 1975). Eight terrestrial habitat types were surveyed, producing a list of 81 species. One surprising record was Red-tailed Black-Cockatoo *Calyptorhynchus banksii*, reported to have been present in two of the eight habitat types (and listed as breeding). This species does not appear on the modern Hunter Region checklist (Stuart 2017). All the other listed species fit with modern understandings; some noteworthy omissions will be discussed in a later section.

An Appendix to Recher's report listed all species known to have been recorded near Myall Lakes and Forster, spanning many decades and based on published and unpublished sources (Recher 1975). Included were Masked *Artamus personatus* and White-browed Woodswallow *A. superciliosus*, Flame *Petroica phoenicea* and Scarlet Robin *P. multicolor* and Hooded Robin *Melanodryas cucullata*. All are interesting easterly records from a modern perspective. Also, there were some surprising records, such as Marbled Frogmouth *Podargus ocellatus* and Large-tailed Nightjar *Caprimulgus macrurus*, both now known only from northern Queensland and surely these were mis-identifications.

A review of the status of many woodland and rainforest birds in the 1960s noted that the Emu *Dromaius novaehollandiae* was still present in the Myall Lakes area (Hobbs & Kavaney 1962). The 1962 review also provided many insights into apparent changes in distribution and/or abundance that have occurred since the 1960s. These will be discussed in later sections.

In 1958, the Region experienced an influx of Swift Parrot *Lathamus discolor* and the Tasmanian sub-species of Striated Pardalote (the Yellow-tipped Pardalote) *Pardalotus striatus striatus* (Hobbs & Kavaney 1962; Hindwood & Sharland 1964). These influxes were associated with an extremely heavy infestation of lerp psyllids (Hindwood & Sharland 1964).

The Sulphur-crested Cockatoo *Cacatua galerita* was exterminated in the Williams River area (presumably by agriculturists), and then was absent for several decades until the late 1950s when a flock of *c.* 100 birds reappeared (Hobbs & Kavaney 1962).

Although there were several other articles in *The Emu* about bush birds in the Hunter Region, these usually were anecdotal in their nature and provided very little insight into the overall state of species in the Region (for example, Hyem 1953, 1956; Lane 1966). An exception was a report of a Rufous Songlark at Muswellbrook in winter 1949, which also noted the frequent summer records for that area (Doyle 1953).

Seabirds

Hindwood & D'Ombrain (1960) visited islands of the Broughton Group in December 1959. They confirmed that Wedge-tailed *Ardenna pacifica* and Short-tailed Shearwater *A. tenuirostris*, Silver Gull *Chroicocephalus novaehollandiae* and Crested Tern *Thalasseus bergii* were breeding. They also found two Sooty Shearwater *A. grisea* in burrows on Little Broughton Island; however breeding was not confirmed during their visit[1]. It is interesting that they reported Little Broughton Island had the larger concentration of breeding birds and that shearwaters bred in limited numbers on Broughton Island. The authors stated that Little Penguin *Eudyptula minor* bred in considerable numbers on Cabbage Tree Island. They found some in burrows on Broughton Island but could not confirm breeding. Hindwood & D'Ombrain also searched without success for White-faced Storm-petrel *Pelagodroma marina* burrows. However, a local fisherman told them that some years earlier he had found some at Bassett Hull's 1910 site of "hundreds of burrows" (Bassett Hull 1911).

One year later, a non-breeding Buller's Shearwater *A. bulleri* was found in a burrow on Cabbage Tree Island[2] (D'Ombrain & Gwynne 1961). Thus, four species of shearwater were using islands off Port Stephens in 1959-1960 (also see Lane 1962). Also, the Gould's Petrel *Pterodroma leucoptera* continued to breed on Cabbage Tree Island although the size of the colony was thought to be small (Gibson & Sefton 1957).

A pair of Kelp Gull *Larus dominicanus* bred on Moon Island off Swansea in 1958-59 (Gwynne & Gray 1959). This was unusual as there were no prior Hunter Region breeding records (McGill 1955c). Records of what now are considered rare or uncommon seabirds are detailed in **Table 1**. Mostly these were near-coastal records e.g. occurring after storms. There did not seem to have been any pelagic surveys. Supporting this conclusion, a review of Grey-faced Petrel *Pterodroma gouldi* records mentioned no Hunter Region sightings (Hindwood 1957b); now it is often recorded at the continental shelf off Port Stephens (Stuart 2017).

[1] Sooty Shearwater was first reported breeding on Broughton Island in 1914 (Rohu 1914). There now are known breeding colonies on Boondelbah, Broughton and Little Broughton Islands (Cooper *et al.* 2014).

[2] At the time this was only the second live record for NSW (see Cooper *et al.* 2014 for details).

Table 1. Records of uncommon Hunter Region seabirds reported in *The Emu* during 1951-1980

Species	Date	Details	Reference
Pacific Gull *Larus pacificus*	Jan 1943	Single bird in Manning Estuary	McGill 1955c
Kelp Gull *Larus dominicanus*	Nov 1958 – Jan 1959	Three birds at Moon Island including a breeding pair	Gwynne & Gray 1959
	1958-1959	Up to three birds often recorded in the Hunter Estuary and further south	Hobbs & Kavaney 1962
Sooty Tern *Onychoprion fuscatus*	Jan 1954	Single bird at Shoal Bay	Sefton 1958
White-winged Black Tern *Chlidonias leucopterus*	Jan 1960	Single bird at Swansea	Hobbs & Kavaney 1962
	Jan 1968	Single bird off Newcastle	Rogers 1969
Black Tern *Chlidonias niger*	Jan 1968	Single bird off Newcastle	Rogers 1969
Common Tern *Sterna hirundo*	Dec 1951	Single bird off Newcastle	Hitchcock 1965
	Jan 1952	Three records off Newcastle	
Red-tailed Tropicbird *Phaethon rubricauda*	Feb 1952	Two birds off Port Stephens	D'Ombrain 1952
	Jan 1955	Single bird off Broughton Island	Hindwood 1955b
White-tailed Tropicbird *Phaethon lepturus*	Feb 1956	Single bird inland near Bulahdelah	Hindwood 1957a
Wilson's Storm-Petrel *Oceanites oceanicus*	Sep 1941	Single bird off Port Stephens	Serventy 1952
White-chinned Petrel *Procellaria aequinoctialis*	Dec 1968	Beach-washed along Newcastle Bight	Holmes 1969
Buller's Shearwater *Ardenna bulleri*	Dec 1960	Single bird on Cabbage Tree Island	D'Ombrain & Gwynne 1961
Sooty Shearwater *Ardenna grisea*	Dec 1959	Two birds on Broughton Island	Hindwood & D'Ombrain 1960
Lesser Frigatebird *Fregata ariel*	Feb 1957	• Three birds off Cabbage Tree Island • One bird inside Port Stephens • Two birds off Broughton Island	Hindwood 1957c
Brown Booby *Sula leucogaster*	May 1954	Single bird off Port Stephens	Hindwood 1955a

Both the Common Tern *Sterna hirundo* and Australian Gull-billed Tern *Gelochelidon macrotarsa* were reported to occur only in small numbers in the Region (Hitchcock 1965; Hobbs & Kavaney 1962). However, this was in contradiction to near-contemporaneous reports of notably higher counts for both (Morris 1975). Morris commented that the Australian Gull-billed Tern was a common visitor to the Hunter Estuary in winter.

Records of uncommon and rare birds

The 1951-1980 literature included several reports of species which are now considered uncommon or rare in the Hunter Region or to be accidental visitors. Nowadays, all reports of these species are closely reviewed before being accepted. That was not always the case for older records. A list of 1951-1980 reports is presented in **Table 2** (excepting seabirds, which all appear in **Table 1**). This does not imply that they have been accepted as confirmed records although in most cases the birds almost certainly were correctly identified.

DISCUSSION

It has not seemed feasible to prepare a list of all the birds recorded in the Hunter Region after 1951. Whereas in 1901-1950 there were many reports in *The Emu* that included annotated bird lists for a specific part of a subregion (Stuart 2009, Stuart 2013), this hardly ever happened in the post-1950 period.

Within **Table 2** is a report of a pair of Red-backed Button-quail *Turnix maculosus* at Diamond Head in 1959. There have been more recent records from this general area (Stuart 2017) and it seems worthwhile to investigate if Crowdy Bay National Park has a resident population.

McGill reported some records of the Black-headed (Striated) Pardalote *Pardalotus striatus melanocephalus* (**Table 2**; McGill 1966). Those reports have recently been questioned (Stuart 2018).

Table 2. Rare and unusual birds (excepting seabirds) for the Hunter Region based on reports in *The Emu* during 1951-1980.

Species	Date	Details	Reference
Superb Fruit-dove *Ptilinopus superbus*	Dec 1918	An immature female at Belltrees homestead (via Scone)	Hindwood 1953
Black-eared Cuckoo *Chalcites osculans*	Feb 1959	A single bird near Dungog	Hobbs & Kavaney 1962
Oriental Cuckoo *Cuculus saturatus*	Feb 1961	A single bird at Fosterton	Dowling 1962
Black-tailed Native-hen *Tribonyx ventralis*	1951	Common around Hexham after heavy rain	D'Ombrain 1951
Banded Lapwing *Vanellus tricolor*	1951	Common around Hexham after heavy rain	D'Ombrain 1951
Red-backed Button-quail *Turnix maculosus*	Dec 1959	A pair at Diamond Head	Hobbs & Kavaney 1962
Glossy Ibis *Plegadis falcinellus*	pre-1951	Some records from the Hexham area	D'Ombrain 1951
	Feb 1960	Nine birds at Hinton	Hobbs & Kavaney 1962
Black Kite *Milvus migrans*	May 1960	A single bird at Dungog	Hobbs & Kavaney 1962
Spotted Harrier *Circus assimilis*	July 1960	A single bird at Pipers Bay Forster	Hobbs & Kavaney 1962
Barking Owl *Ninox connivens*	Dec 1959	Regularly at Dungog and Chichester State Forest	Hobbs & Kavaney 1962
Black-headed (Striated) Pardalote *Pardalotus striatus melanocephalus*	1959	A pair was nesting near Taree. Prior records from around the Manning River were also mentioned.	McGill 1966
Ground Cuckoo-shrike *Coracina maxima*	Apr 1959	Two birds near Dungog	Hobbs & Kavaney 1962
	pre-1959	A single bird at Barrington	
Common Blackbird *Turdus merula*	Sep 1959	A pair at Dungog (for ~5 months)	Hobbs & Kavaney 1962

The presence or absence of the Eastern Bristlebird *Dasyornis brachypterus* in the Hunter Region was uncertain. Chisholm's account of the diaries of the early 20th century collector S.W. Jackson stated that Jackson had found some at Wootton in 1922 (Chisholm 1958). However, a review of the status of Eastern Bristlebird (Chaffer 1954) made no mention of Jackson's records. Possibly they had not yet surfaced into the public domain. Jackson's diaries were unpublished until Chisholm's 1958 summary of them.

Apparent changes in distribution and abundance

Based solely on reports in *The Emu*, some species which we now find to be relatively common and/or widely distributed in the Region were not always so. Conversely, some species appear to have decreased in abundance in modern times or their ranges have contracted. The more noteworthy of the apparent changes are reviewed in this section.

Apparent range extensions

Table 3 summarises the first records for 14 species in new parts of the Region. Mostly these were cases of northern birds extending their range southwards or of inland birds extending their range eastwards. An exception was the Common Myna *Acridotheres tristis*. A small colony established in Newcastle in the 1950s (Hone 1978); subsequent records from some Newcastle suburbs possibly originated from that initial colony.

Hobbs & Kavaney (1962) reported Hawks Nest to be the southern limit for the White-breasted Woodswallow *Artamus leucorynchus* (**Table 3**). This is a good example of the limitations from only considering articles from *The Emu*: the species was well known to breed at Wyong to the south of the Hunter Region in the 1960s and 1970s (Morris 1975).

From the surveys of sites in the proposed Myall Lakes National Park, carried out in mid October 1972 (Recher 1975), there were three noteworthy omissions from the species list. Eastern Koel *Eudynamys orientalis*, Channel-billed Cuckoo *Scythrops novaehollandiae* and Rainbow Lorikeet *Trichoglossus moluccanus* were not recorded. All three now commonly occur in the area (Stuart 2017); they have expanded their ranges since 1972 (Cooper *et al.* 2016).

Table 3. Range limits and range extensions based on reports in *The Emu* during 1951-1980

Species	Date	Details	Reference
Australian Brush-turkey *Alectura lathami*	Nov 1957	Nesting record from Buttai which was now considered the southern range limit	Kavaney 1958
	Oct 1958	A pair near Ourimbah extended the southern range by ~30 km	Hobbs & Kavaney 1962
Crested Pigeon *Ocyphaps lophotes*	1943	First record in Taree	Breckenridge 1952
	1958	A single bird at Taree	Hobbs & Kavaney 1962
	July 1960	A few birds were near Forster	
	1962	Maitland was considered the eastern range limit in the Hunter Valley	
Bar-shouldered Dove *Geopelia humeralis*	1962	Hawks Nest was considered the southern range limit	Hobbs & Kavaney 1962
Channel-billed Cuckoo *Scythrops novaehollandiae*	Dec 1952	First record for Maitland	D'Ombrain 1952
Galah *Eolophus roseicapilla*	1962	Hinton was considered the eastern range limit	Hobbs & Kavaney 1962
Sulphur-crested Cockatoo *Cacatua galerita*	1962	The Williams River area was considered the eastern range limit	Hobbs & Kavaney 1962
Striped Honeyeater *Plectorhyncha lanceolata*	1962	Belford was considered the eastern range limit but a coastal population also existed	Hobbs & Kavaney 1962
Blue-faced Honeyeater *Entomyzon cyanotis*	1962	Woodville was considered the eastern range limit	Hobbs & Kavaney 1962
Mangrove Gerygone *Gerygone levigaster*	July 1960	Some birds at Darawank represented a southern range extension of ~50 km	Hobbs & Kavaney 1962
Western Gerygone *Gerygone fusca*	Sep 1955	Hollydene was now considered the eastern range limit	Hoskin 1957
	Oct 1956	A pair was nesting near Scone ~50 km north-east of Hollydene	
	July 1959	Birds in Belford area represented an eastern range extension of ~70 km	Hobbs & Kavaney 1962
Grey-crowned Babbler *Pomatostomus temporalis*	1962	Its spread through the lower Hunter Valley and elsewhere was discussed	Hobbs & Kavaney 1962
White-breasted Woodswallow *Artamus leucorynchus*	1962	Hawks Nest was considered the southern range limit	Hobbs & Kavaney 1962
Varied Triller *Lalage leucomela*	Dec 1952	Six birds at Harrington – first records from south of Clarence River	McGill 1954
Common Myna *Acridotheres tristis*	1950s	Some birds at Newcastle steelworks	Hone 1978
	1970	First records at Cardiff & Edgeworth	

Apparent range contractions

The anecdotal nature of many 1951-1980 reports about any particular species makes it difficult usually to differentiate whether lower numbers in modern times are due to range contractions or to changes in abundance. However, the Zebra Finch *Taeniopygia guttata* was described as being common in many parts of the Hunter Valley and Taree (Hobbs & Kavaney 1962) – it appears now to be absent from much of that range (Stuart 2017). Also, Southern Whiteface *Aphelocephala leucopsis* was recorded south as far as Cessnock and also occurred in small numbers around Dungog, Brookfield, Stroud and Gloucester (Hobbs & Kavaney 1962). The only modern records are from locations around Goulburn River National Park and Ulan (Stuart 2017).

Apparent changes in abundance

The Regent Honeyeater *Anthochaera phrygia* was present in large numbers (60+ birds) at Cooranbong in early February 1958 (Kavaney 1958b). Although there are occasional modern records from the Cooranbong-Morisset area, these have involved small numbers of birds in autumn-winter (Stuart 2017).

Both the Red-capped Robin *Petroica goodenovii* and Hooded Robin *Melanodryas cucullata* were considered common birds of the Hunter Region in the 1960s (Hobbs & Kavaney 1962). The former was found east as far as Branxton (with winter incursions to Maitland) while the latter was described as a common bird occurring in nearly all suitable open-forest habitat.

In 1955, the Little Egret *Egretta garzetta* was described as the rarest of Australia's egret species (the Cattle Egret *Bubulcus ibis* was not even in consideration at that time[3]) and the author provided tips for how to identify them (McGill 1955a). The Pink-eared Duck *Malacorhynchus membranaceus* was considered a rare visitor near the coast, such that a sighting of four birds near Hinton in 1960 was considered noteworthy (Hobbs & Kavaney 1962).

CONCLUSIONS

The Emu for about eight decades (1901 to around 1975) was a rich source of information about birds in the Hunter Region. It underwent a change in direction from the 1970s onwards, and rarely since then has contained much locally or regionally focussed information. However, around that same time other journals began to appear that were able to at least partially fill the gap. For example, the Newcastle Flora and Fauna Society's journal *Hunter Natural History* was published from 1969-1980 and contained many articles about Hunter Region birds. Similarly, *Australian Birds* (published by the NSW Field Ornithologists Club) first appeared in 1966 and was an important source *inter alia* of Hunter Region information (and NSW more generally) for around four decades. Since 2009, *The Whistler* has become a major repository for news about local bird studies. The Hunter Region's importance for Australian birds continues to be well documented.

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Below are details for every article relevant to the Hunter Region appearing in *The Emu* during 1951-1980, and for other cited references. Collectively the *Emu* articles gave insights into then-current local ornithological understandings; however not all of them were referenced in discussion.

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Some observations of Australian Pied Oystercatcher on Worimi Conservation Lands

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Survey records of Australian Pied Oystercatcher *Haematopus longirostris* (oystercatcher) on the Worimi Conservation Lands (WCL) from 2014-2017 were analysed. Nine breeding territories were identified on the northern section of WCL. Between 5 and 7 of these territories were occupied during the breeding season from August to December. The mean distance between occupied territories was 1.2-1.3 km and the annual breeding density was 0.7-0.9 pair/km. During the breeding season non-breeding birds were displaced to other sections of WCL or dispersed to other locations, returning in the non-breeding season.

The age structure of the population varied from month to month, although adult and immature (2+ year) birds predominated. Numbers declined temporarily in September at the start of the breeding season. The sporadic presence of banded birds from Victoria and northern New South Wales provides evidence of long-distance movement of birds to and from WCL.

The distribution of oystercatchers on WCL is strongly influenced by morphodynamic processes operating along the beach. The northern section of beach has dissipative characteristics that support higher biodiversity and prey availability. The southern section has reflective characteristics with potentially lower prey availability. Breeding pairs of oystercatchers preferentially established their territories along the northern section of the beach where high densities of Pipis *Donax deltoides* were present. The lower prey availability on southern sections of WCL may be a factor influencing temporary movements of non-breeding birds to other locations during the breeding season. The increase in oystercatcher numbers since 2009 corresponds with an increase in commercial Pipi stocks over the same period.

INTRODUCTION

Worimi Conservation Lands (32° 48'S, 151° 56'E), extend 25.5 km along Stockton Bight, between Fern Bay and Anna Bay. The southern boundary is 5.7 km north of the Hunter River mouth and the northern boundary is at Birubi Beach. WCL comprises 4,029 ha of ocean beach, sand hills and coastal woodland. The beach has a concave shape facing south-east. It is bordered in most part by a frontal dune with a wide swale to the rear. Mobile sand hills up to 40 m high and 1- 2 km wide are present behind the swale and merge to the north with stabilized sand hills covered by a variety of coastal woodland communities (Office of Environment and Heritage 2015). WCL forms part of the Worimi National Park. The location is shown in **Figure 1**.

WCL is owned by the Worimi community and leased to the NSW Government to be jointly managed on behalf of the traditional owners. NSW National Parks and Wildlife Service (NPWS) conducts day to day management (Office of Environment and Heritage 2015). Birubi Beach is located at the northern end of WCL and is used extensively by recreational beach goers while the remainder of the beach is used by 4WD vehicles and fishermen.

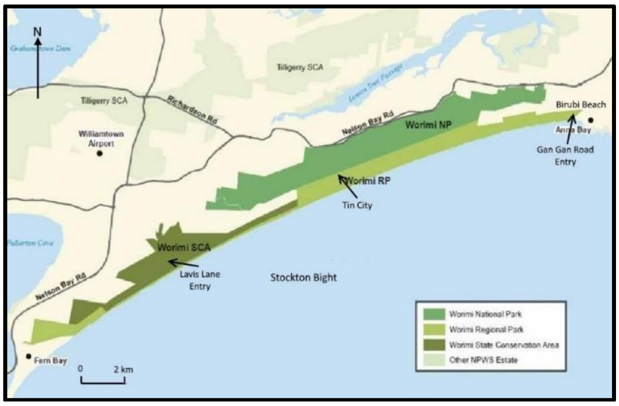

Figure 1. Location map, Worimi Conservation Lands

A population of 50-100 Australian Pied Oystercatcher (oystercatchers) is present along Stockton Beach on Worimi Conservation Lands (WCL). This population, together with those in adjacent Port Stephens, is the largest in New South Wales with 150-200 birds recorded in winter and summer surveys in 2016 (Stuart 2017). In the non-breeding season the birds are present as mixed flocks of adult and immature birds mainly along the northern two-thirds of the beach. During the breeding season, pairs of adult oystercatchers establish defended territories along the northern section of the beach. The displaced non-breeding birds form flocks generally to the south of these territories. The main source of food for oystercatchers on WCL was observed to be Pipis, *Donax deltoides.*

A previous review of avian surveys on WCL from 2009 to 2013 (Lindsey & Newman 2014) documented known breeding attempts by oystercatchers and noted that 'breeding is primarily restricted to an area approximately seven kilometres in length near and north of Tin City'. Russell & George (2012) identified five breeding territories within a 4 km section of beach in the north of WCL. The current study analysed records of oystercatchers from surveys of WCL to determine the following:

- the number, location and size of oystercatcher territories present during the breeding season;
- changes in the distribution of oystercatchers between the breeding and non-breeding seasons; and
- temporal changes in population numbers and age structure.

To provide context for these observations, some of the factors influencing the distribution of oystercatchers have been qualitatively assessed.

METHODOLOGY

Between 2009 and 2017, 70 surveys were conducted by members of the Hunter Bird Observers Club (HBOC) and NPWS Visitor Services personnel from the Worimi community. NPWS provided a 4WD vehicle for the surveys. The surveys commenced on an *ad hoc* basis in 2009, but since 2014 have been conducted monthly except when adverse weather makes the beach inaccessible. The surveys were conducted on the day prior to the monthly Hunter Estuary waterbird surveys and occur close to high tide, between 8.30 and 11.00 am. The survey route was divided into three sections of roughly equal length; from the southern boundary of WCL to the Lavis Lane entry (southern section), from the Lavis Lane entry to Tin City (central section) and from Tin City to the Gan Gan Road entry (northern section) (see **Figure 1**). Numbers of all species present were recorded and locations of oystercatchers were identified using GPS. Counts of oystercatchers with fully developed plumage (includes adult and 2-3 year-old immature birds), first-year birds and dependent young were recorded, together with any distinguishing features such as leg bands, injuries or deformities. First-year birds were distinguished by the partial dark bill tip, dull orange orbital ring and dull greyish-pink legs. Dependent young were birds that remained with parents, including when fledged (i.e. capable of flight).

The distribution of oystercatchers during the breeding and non-breeding seasons was analysed for years 2014-2017. The breeding season on WCL is from August to December. This corresponds with the southern Australian breeding season as reported by Marchant & Higgins (1994). There are no records of breeding outside of these months.

Records from breeding season observations for each year were plotted using a GIS system and the distance from the Gan Gan Road entry determined. Clusters of records of adult pairs and single adult birds occupying defended territories or adult birds accompanied by dependent young in the same location over several months were identified as breeding territories. (The latter point assumes adult pairs with recently fledged young do not relocate during the breeding season.) The location of territories was further confirmed by comparison with records of previous known breeding attempts (nests with eggs or adults with dependent young).

When a territory for a particular year was identified, the mean of monthly distances from the Gan Gan Road entry was determined to estimate the centre of each territory. Territory centres were determined separately for each year. Breeding density for each year was calculated as the number of breeding territories/km of occupied beach. The distance to flocks of non-breeding birds outside the territories was also determined. Reporting rates were determined for the presence of oystercatchers in the three survey sections for the breeding and non-breeding seasons.

To provide an insight into short-term change in total numbers, the mean monthly totals were determined for 2014-2017. The age structure of the population was analysed by determining the monthly numbers of first-year and older birds (2+ years) present from 2014-2017. Dependent young were included with first-year birds.

To provide some context for a discussion of the distribution patterns and temporal changes in oystercatcher numbers, the morphodynamics of the beach and information on Pipi abundance were qualitatively evaluated.

RESULTS

The monthly survey data showed that from January to July, oystercatchers were distributed along all sections of WCL in mixed flocks of adult and immature birds. However, from August to December, a number of pairs established defended territories along the northern section of WCL while the displaced non-breeding birds formed flocks away from these territories.

The distance to the centre of each territory from the Gan Gan Road entry and the distance apart for years 2014-2017 are summarized in **Table 1**. The data indicate that as many as nine territories were occupied at various times over the period with territories 2, 3, 4, 5 and 7 being the most consistently occupied.

In 2014 six territories were identified extending from 5.2 to 11.8 km south of the Gan Gan Road entry. The breeding density was 0.9 pair/km. The most distant territory was located 1 km south of Tin City. The mean distance between occupied territories was 1.2 ±0.4 km. All flocks of non-breeding birds were located 1.1 to 13.3 km south of the territories.

In 2015 seven territories were identified extending from 2.6 to 10.2 km south of the Gan Gan Road entry. The breeding density was 0.9 pair/km. The most distant territory was located 0.6 km north of Tin City. The mean distance between occupied territories was 1.3 ±0.4 km. All flocks of non-breeding birds were located 2.3 to 10.4 km south of the territories.

In 2016 six territories were identified extending from 3.0 to 11.2 km south of the Gan Gan Road entry. The breeding density was 0.7 pair/km. The most distant territory was located 0.4 km south of Tin City. The mean distance between occupied territories was 1.2 ±0.5 km. Flocks of non-breeding birds were located 0.7 to 2.9 km north and 1.0 to 14.3 km south of the territories.

In 2017 five territories were identified extending from 4.3 to 10.3 km south from the Gan Gan Road entry. The breeding density was 0.8 pair/km. The most distant territory was located 0.6 km north of Tin City. The mean distance between occupied territories was 1.2 ±0.5 km. Flocks of non-breeding birds were located 1.5 to 4.1 km north and 1.7 to 11.3 km south of the territories. In September a flock of non-breeding birds was recorded between the two southern territories.

The numbers of breeding territories identified for each year and their centres should be treated with some caution. As nesting sites are established in the swale some distance behind the beach (Russell and George 2012), all birds and territories may not have been located by the monthly surveys which were restricted to the beach. For some territories only one record was available over the annual breeding season and in other instances only one bird was observed on a territory. For some territories, the determined centre points were only 600 m apart creating some uncertainty regarding interpretation of which pair occupied a territory. Also, some pairs that established territories may not have bred.

All of the identified territories and confirmed breeding records are located on the northern section of WCL along a section of beach extending from 2.6 km south of the Gan Gan Road entry to 11.8 km south, near Tin City.

Table 1. Distance to centre of territories from the Gan Gan Road entry with standard deviation and distance apart for 2014-2017.

<table>
<tr><th></th><th>Territory</th><th colspan="2">1</th><th colspan="2">2</th><th colspan="2">3</th><th colspan="2">4</th><th colspan="2">5</th><th colspan="2">6</th><th colspan="2">7</th><th colspan="2">8</th><th colspan="2">9</th></tr>
<tr><td rowspan="2">2014</td><td>Distance from Gan Gan Rd. entry (km)</td><td colspan="2">-</td><td colspan="2">-</td><td colspan="2">5.2 ±0.1</td><td colspan="2">7.2 ±0.2</td><td colspan="2">8.1 ±0.0</td><td colspan="2">-</td><td colspan="2">9.9 ±0.0</td><td colspan="2">11.0 ±0.1</td><td colspan="2">11.8 ±0.2</td></tr>
<tr><td>Distance Apart (km)</td><td></td><td colspan="2"></td><td colspan="2"></td><td colspan="2">1.9</td><td colspan="2">1.0</td><td colspan="4">1.8</td><td colspan="2">1.1</td><td colspan="2">0.9</td><td></td></tr>
<tr><td rowspan="2">2015</td><td>Distance from Gan Gan Rd. entry (km)</td><td colspan="2">2.6 ±0.0</td><td colspan="2">4.5 ±0.1</td><td colspan="2">5.4 ±0.1</td><td colspan="2">6.6 ±0.4</td><td colspan="2">8.1 ±0.2</td><td colspan="2">9.5 ±0.2</td><td colspan="2">10.2 ±0.2</td><td colspan="2">-</td><td colspan="2">-</td></tr>
<tr><td>Distance Apart (km)</td><td></td><td colspan="2">1.9</td><td colspan="2">0.9</td><td colspan="2">1.2</td><td colspan="2">1.5</td><td colspan="2">1.4</td><td colspan="2">0.7</td><td colspan="2"></td><td colspan="2"></td><td></td></tr>
<tr><td rowspan="2">2016</td><td>Distance from Gan Gan Rd. entry (km)</td><td colspan="2">3.0 ±0.4</td><td colspan="2">3.8 ±0.1</td><td colspan="2">5.0 ±0.1</td><td colspan="2">6.9 ±0.0</td><td colspan="2">7.7 ±0.2</td><td colspan="2">-</td><td colspan="2">-</td><td colspan="2">11.2 ±0.3</td><td colspan="2">-</td></tr>
<tr><td>Distance Apart (km)</td><td></td><td colspan="2">0.8</td><td colspan="2">1.2</td><td colspan="2">2.0</td><td colspan="2">0.8</td><td colspan="6">3.5</td><td colspan="2"></td><td></td></tr>
<tr><td rowspan="2">2017</td><td>Distance from Gan Gan Rd. entry (km)</td><td colspan="2">-</td><td colspan="2">4.3 ±0.1</td><td colspan="2">5.4 ±0.1</td><td colspan="2">7.2 ±0.1</td><td colspan="2">7.8 ±0.1</td><td colspan="2">-</td><td colspan="2">10.3 ±0.0</td><td colspan="2">-</td><td colspan="2">-</td></tr>
<tr><td>Distance Apart (km)</td><td></td><td colspan="2"></td><td colspan="2">1.1</td><td colspan="2">1.8</td><td colspan="2">0.6</td><td colspan="4">2.5</td><td colspan="2"></td><td colspan="2"></td><td></td></tr>
</table>

To quantify the change in distribution of oystercatchers between the breeding and non-breeding seasons, reporting rates were determined for the three survey sections along WCL for the period 2014-2017. The results are presented in **Table 2**.

Table 2. Reporting rates, median and maximum counts of oystercatchers for breeding and non-breeding seasons for survey sections, 2014-2017.

Breeding Season August - December					
	No. of Surveys	Records	Reporting Rate	Median	Maximum
Southern section	19	9	47%	0	20
Central section	19	19	100%	6	36
Northern section	19	19	100%	13	48
Non-Breeding Season January - July					
	No. of Surveys	Records	Reporting Rate	Median	Maximum
Southern section	21	8	38%	0	11
Central section	18	16	89%	11	30
Northern section	18	18	100%	22	52

The data show that during the breeding season, fewer numbers of oystercatchers are present on the northern section compared to the non-breeding season. At the same time, numbers in southern and central sections increase. Conversely, during the non-breeding season, there are increased numbers present in central and northern sections while in the southern section they are recorded infrequently. The reporting rate in the northern section (100%) remains constant during both seasons. During the breeding season, the reporting rate in the central section increases from 89% to 100% while in the southern section it increases from 38% to 47%, consistent with the birds being displaced from the northern section. As breeding activity finishes, newly fledged birds, immature birds and non-breeding adults form large flocks along the northern and central sections of WCL.

The variation in the mean monthly count of oystercatchers for 2014-2017 is shown in **Figure 2.** It reveals a pronounced reduction in mean numbers in September.

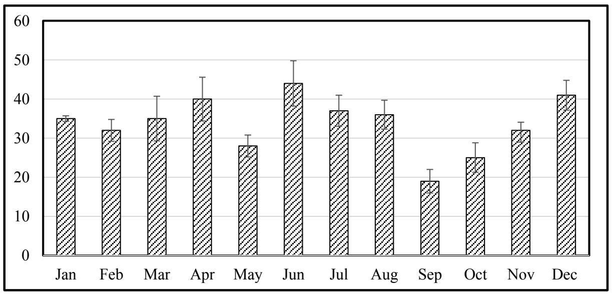

Figure 2. Mean monthly oystercatcher count and standard deviation, 2014-2017

The long-term trend of monthly oystercatcher numbers on WCL since the commencement of surveys in 2009 is shown in **Figure 3**. It shows that numbers vary considerably from month to month but overall there has been a steady increase since 2009. Post 2014 there may have been some under-reporting of numbers as surveys were restricted to the beach foreshore at high tide. Other factors that may impact the numbers recorded are prevailing weather, breeding activity, human disturbance and disturbance by predators.

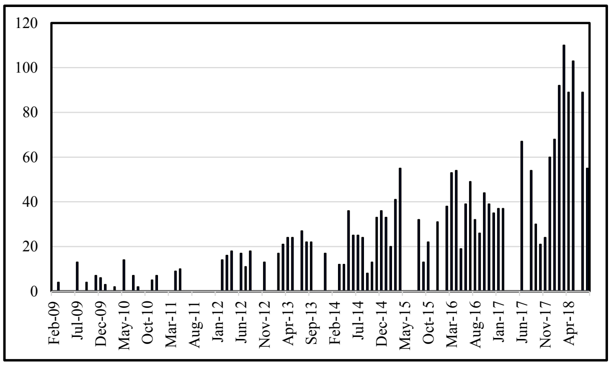

Figure 3. Monthly oystercatcher count, 2009-2018

To evaluate the age structure of the flocks on WCL and identify any temporal patterns of change, the numbers of first-year and older birds (2+ years) were determined for surveys from 2014 to 2017. The results are presented in **Figure 4**. There were a limited number of monthly surveys in which the age structure of all flocks was recorded. The chart shows that older birds comprise the majority (mean 63%) of the population over the survey period but with no consistent monthly trend. Difficulty in distinguishing the age of some 1- and 2-year-old birds may have resulted in first-year birds being over-reported as they approach their second year.

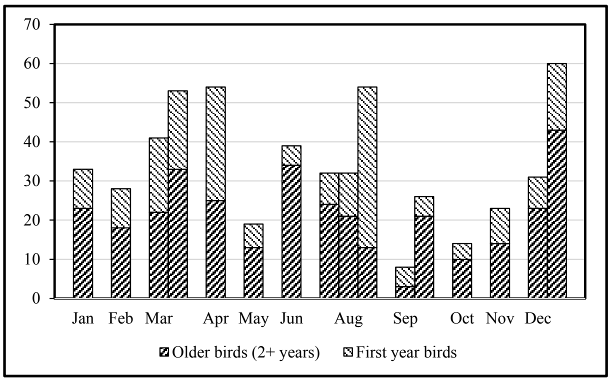

Figure 4. Age structure for monthly oystercatcher population, 2014-2018

DISCUSSION

Breeding season distribution

This study has further refined details of the number and distribution of breeding territories established by oystercatchers on the northern section of WCL, between August and December each year. Nine territories have been identified that were occupied at various times over the period 2014-2017 (**Table 1**). However, most pairs appear to be unsuccessful as there is a lack of evidence of recently fledged young on territories. The location of the territories agreed generally with the location of breeding territories reported by Lindsey & Newman (2014) and Russell & George (2012). However, the extent of territories was greater than recorded by Russell & George (2012) who identified five breeding pairs occupying approximately 4 km of beach, from a nest site 7.1 km south of the Gan Gan Road entry. This corresponded to territory 4 in **Table 1**. In all the years analysed in this paper, one or more territories was recorded north of those described by Russell & George (2012). This observed increase in the extent of defended territories corresponds to an increase in the total number of oystercatchers present and the number of breeding pairs in some years. The availability of food, Pipis, may have been a factor influencing the expanded distribution and will be discussed in a later section.

The distance between occupied territories varied from 0.6 to 1.9 km (**Table 1**) and the mean breeding density varied from 0.7 pair/km in 2016 to 0.9 pair/km in 2015. This is slightly less than the density recorded on Seven Mile Beach near Hobart in Tasmania, another ocean beach (M. Newman pers. comm.)

The southern limit of the territories remained around 1 km south of Tin City over the study period. In 2014, the closest territory to the Gan Gan Road entry was 5.2 km distant. In subsequent years territories were established as close as 2.6 km to the entry (**Table 1**). As territories extended to the north, pairs would have encountered increasing levels of disturbance due to recreational beach activity. This extension could represent newly established breeding pairs trading off greater levels of disturbance for access to greater food availability. However, it may also be a function of the change in nesting strategy from beach front to swales behind the fore dune, as described by Russell & George (2012). Potentially, this strategy was not impacted by recreational beach activity as much as nesting on the beach. However, it is highly inefficient with unknown, but probably minimal breeding success; forcing birds to fly in food for chicks and consequently delaying/limiting foraging lessons for young. (Oystercatchers are the only shorebird species which are fed by adults.) The New South Wales Scientific Committee (2008) nominated human disturbance as a key threatening process in oystercatcher breeding success.

During the breeding season the non-breeding birds are displaced to areas distant from the defended territories. Increased numbers were recorded along the southern section of WCL during the non-breeding season (**Table 2**). Non-breeding birds were recorded at least 1 km or more away from the nearest breeding territory. In 2014 and 2015 all of the displaced birds were recorded to the south of the defended territories. However, in 2016 and 2017 flocks of non-breeding birds were also recorded to the north of the defended territories. These northern flocks of non-breeding birds may be trading off increased human disturbance for access to greater food availability. The record of a flock of non-breeding oystercatchers between the two southern territories in September 2017 may have been a flock moving through the area of defended territories.

The displacement of non-breeding oystercatchers from territories at the commencement of the breeding season corresponds with a decrease in mean numbers in September (**Figure 2**) suggesting that some individuals move away from WCL to locations such as Port Stephens, 10 km to the north. There is a population of 100-150 oystercatchers in Port Stephens (Stuart 2010).

Non-breeding season distribution

During the non-breeding season, oystercatchers return from the southern and central sections of WCL to the northern section. Birds are rarely observed south of the Lavis Lane entry in the non-breeding season. This is illustrated in **Table 2** where the reporting rate for the southern section decreased from 47% to 38% and the median numbers of birds on the central and northern sections increased. Following the end of the breeding season, birds tend to become less dispersed and form large flocks along the northern section of WCL.

Marchant & Higgins (1994) report that some birds remain on their territories all year round, only defending them in the breeding season. This is the case for some pairs on WCL. However, the identifiable pair ‘Hoppy’ and ‘Peggy’ did not remain on their Tin City territory during the non-breeding season. They were recorded over the full length of the central section and part of the northern section, sometimes as part of a flock and on other occasions individually.

It should be noted that the interpretation of distribution of oystercatchers on WCL is based on monthly data which is biased by the survey methodology. All surveys were conducted in the morning, near high tide and only recorded birds on the beach. At that time the majority of birds were roosting in flocks on the upper sections of the beach and birds nesting in the swales and their unfledged offspring would not have been recorded. While the high-tide behaviour was in accordance with studies reported by Marchant & Higgins (1994), it is not known how foraging behaviour at low tide affects the distribution of oystercatchers on WCL.

Regional movements and variation in numbers

Interpretation of monthly change in numbers of oystercatchers on WCL was difficult as the extent of recruitment to WCL from other locations and dispersal from WCL were unknown. It is known that birds dispersing from southern Victoria and northern NSW are present on WCL on occasions (Lindsey & Newman 2014). A number of banded birds have been recorded on WCL: 'VW' banded Corner Inlet, Victoria; 'R4' banded Dart Island, Clarence Estuary, NSW; and 'S3' banded Red Rock, NSW. The latter has been recorded on several occasions in recent years. The bird 'Lucky' which was rescued, banded (silver metal band on right tibia) and released on WCL in March 2014 has been noted as absent from WCL for several months at a time during 2014-2017. Marchant & Higgins (1994) report seasonal movements by sections of oystercatcher populations along the coast and in some instances, long-distance movements. This was probably happening at WCL with regular movement to and from Port Stephens. The temporal change in the age structure of the population shown in **Figure 4** indicates that the composition was highly variable. Marchant & Higgins (1994) report oystercatchers show some fidelity to non-breeding flocking sites, although there was much movement between flocks by individuals, both between years and within one season. It is thought that a similar pattern of movement occurs on WCL.

Lindsey & Newman (2014) noted that the number of oystercatchers on WCL increased over the period 2009-2013. This trend has continued over the period of this study. **Figure 3** shows a steady increase since 2009. Lindsey & Newman (2014) concluded that until 2011, the majority of birds on WCL were 'mainly resident breeding adults' and since then it is thought that numbers have increased as more non-breeding adult and immature birds use the beach. Over the period 2014-2017, first-year oystercatchers was 37% (mean value) of the population but numbers were highly variable (**Figure 4**). It is noted that the number of breeding pairs on WCL could not produce the observed increase in oystercatcher numbers and that the majority of the increase has resulted from the recruitment of birds dispersing from other locations.

Factors influencing distribution

There are many factors that influence the distribution of oystercatchers along ocean beaches. Some of these are availability of food, beach morphodynamics, fore dune geometry, density of avian competitors, human disturbance, 4WD vehicles, weather and tidal cycles (Marchant & Higgins 1994, Owner & Rohweder 2003, Harrison 2009). Two of these, beach morphodynamics and availability of food, are considered here to explain the steady increase in numbers since 2009 and why defended territories are established exclusively on the northern section of WCL. Marchant & Higgins (1994) report oystercatcher food consists of molluscs, worms, crabs and small fish. On WCL, oystercatchers are observed foraging almost exclusively on Pipis plus the occasional beach worm. However, foraging observations have been made dominantly at high tide. Prey availability and foraging behaviour at low tide has not been observed.

Stockton Beach is continually nourished by sediment transported out to sea by the Hunter River and by material dredged from Newcastle Harbour and dumped offshore near the southern end of the beach. Hydrodynamic processes including waves, tides and inshore currents transport the sediment to the north along the beach (WorleyParsons Resources and Energy 2012) sorting the sand grains in the process. Sand on the southern end of the beach is poorly sorted, beach conditions are soft and the frontal dune is variable (Office of Environment and Heritage 2015). The beach tends to be steeper with a narrow face comprised of coarse-grained sediment and a narrow, shoaling surf zone. Waves break abruptly on the intertidal zone. Wright & Short (1984) and Short (1999) classify this as a reflective beach. On the northern section of the beach the sand is well sorted, fine grained and beach conditions are firm. The beach is wider and flatter, and the frontal dune system is more intact (Office of Environment and Heritage 2015). Waves break far from the intertidal zone, dissipating their energy progressively along a wide surf zone. Wright & Short (1984) and Short (1999) classify this as a dissipative beach. Observations from monthly surveys and examination of satellite imagery indicate the dissipative section of beach extends approximately 12.5 km south of the Gan Gan Road entry.

Jones & Short (1995) and Short (2000) have shown that as beaches become more dissipative, biodiversity increases. Murray-Jones (1999) recorded Pipis forming dense aggregations in the subtidal and intertidal regions of high-energy, dissipative beaches which support large blooms of surf diatoms from the mouth of the Murray River, South Australia to Fraser Island, Queensland. Owner & Rohweder (2003) and Harrison (2009) demonstrated that prey availability/Pipi abundance and hence oystercatcher abundance was directly related to beach morphology in northern NSW. It is therefore considered that maximum prey availability on WCL occurs on the northern dissipative section of beach, and that oystercatchers preferentially establish breeding territories in this area. The work of Murray-Jones (1999) who regularly sampled Pipi abundance on a 6 km section south of Birubi Beach from July 1995 to November 1997 supports this postulation. She reported high densities of large Pipis present in aggregations across the sample locations. Significant temporal variation in the density and location of Pipi aggregations was observed.

The Sydney Fish Market Annual Reports (2006-2017) show that for much of the period from 1993/94 to 2009/10, Stockton Beach provided over 50% of the total NSW commercial Pipi catch. In 2009/10 the fishery collapsed due to overfishing (McKenzie & Montgomery 2012). The New South Wales Scientific Committee (2008), when determining the endangered status of the Pied Oystercatcher, noted that a key prey species, the Pipi, had undergone a severe long-term decline as a result of commercial over-harvesting. Pipi resources, and consequently commercial catch in NSW have increased steadily since 2010/11 and in 2016/17, 298 tonnes were harvested (Sydney Fish Market Annual Reports 2006-2017). This has been supported by a regimen of enhanced regulation and improved management of Pipi stocks. Commercial harvesting of Pipis on Stockton Beach has continued over this period. Although there is only limited quantitative data for Pipi numbers on WCL, the increase in monthly oystercatcher numbers from 2009 to present (**Figure 3**) reflects the trend of increasing commercial catch over the same period.

A consequence of pairs establishing defended territories in areas of maximum prey availability is that non-breeding birds are forced onto the intermediate and reflective beach sections where prey availability was likely to be less. This may be the reason for the decrease in mean numbers in September (**Figure 2**) as non-breeding birds disperse temporarily to other areas such as Port Stephens. It may also explain why at times flocks of non-breeding birds remain on the northern-most part of WCL during the breeding season (**Table 1**) where they trade off increased levels of disturbance from recreational beach users for increased prey availability.

CONCLUSIONS

The establishment of breeding territories on the northern section of WCL by oystercatchers during the breeding season reflected the higher availability of prey in this area. The higher availability of prey in turn reflects the morphodynamics of a dissipative style of beach supporting high biodiversity including algal blooms that provide food for filter-feeding Pipis. The extent of breeding territories closely corresponds with the extent of dissipative beach.

During the breeding season, non-breeding birds are displaced to areas of WCL with lower prey availability or areas with higher levels of disturbance. This may be the reason some birds disperse temporarily to other locations. Following the breeding season, displaced birds return to areas of high prey availability on the northern section of WCL. Additional surveys at low tide are recommended to confirm distribution of oystercatchers along WCL in response to the varied foraging opportunities.

The age structure and population size of oystercatcher on WCL varies from month to month, indicating that interchange with other populations is continuous. Temporary recruitment from distant regions to the south and north is part of this process. Overall, the majority of the population are adult and immature (2+ years) birds. The increase in numbers of oystercatchers on WCL corresponds to an inferred increase in Pipi stocks over the period of the surveys. For this increase to be sustained long term it is essential that management practices by Fisheries NSW ensure high Pipi stock levels are maintained.

The mean breeding density of oystercatchers on WCL is slightly less than that of similar high-energy beaches studied in Tasmania. Furthermore, not all possible territories appear to be occupied from year to year, and the territories identified do not cover the full extent of potentially suitable beach. This suggests that WCL could support more breeding pairs, if conditions were suitable. It may also indicate the level of disturbance is a deterrent to breeding for less experienced birds.

The activities of recreational beach users and the secondary impact of their activities in attracting avian predators to WCL undoubtedly influence the manner in which oystercatchers use the beach at WCL, particularly in the northern section. However, the adoption of nesting behind the fore dune and the presence of flocks of non-breeding birds near the Gan Gan Road entry suggest that some birds are prepared to trade off a level of disturbance for access to areas with high prey availability.

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Birdlife at Belmont Wetlands State Park

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Belmont Wetlands is the 10th State Park of NSW, set on 514ha of coastal woodland and hind dunes at Belmont, Lake Macquarie. Despite historic degradation of this area, studies of its native birds continue to reflect and support ongoing rehabilitation of these coastal woodlands. This paper describes 105 species observed in a recent study (2015 to 2017). It also refers to an earlier study by Keith Laverick (LMCC 2001) in the same area and identifies significant differences in observed numbers of the same species. Differences are discussed in terms of known changes in habitat and Hunter Region trends.

INTRODUCTION

Belmont Wetlands State Park (BWSP) is located at 33°02'S, 151°40'E between the Pacific Ocean and Lake Macquarie. It is part of the largest remaining coastal wetlands system in the Lake Macquarie Local Government Area. This system has three separate vegetation communities, all connected by groundwater interactions (Brown 2003). BWSP includes Belmont Lagoon (see **Figure 1**), a spiritually and culturally important site for traditional owners and custodians, the Awabakal people. For over 100 years this landscape has been degraded due to prolonged mineral-sand mining, coal industry construction, erosion of natural sand dune barriers, changes in natural water flows, weed infestations, elevated nutrient levels and other urban factors.

Before the former owners, Broken Hill Propriety Holdings Company (BHP), returned this coastland to the NSW Government in 2002, several site condition assessments of the wetlands were made which included fauna and flora surveys. In 2006 the NSW Government proclaimed this land Crown Reserve and a State Park Trust was formed to rehabilitate and manage its natural resources as a recreational reserve (BWSP Trust 2010). Section 6.6 of that report cites several earlier avian studies of interest. This study aimed to document the current avian population, both resident and migratory, and to document changes since the last published studies.

Previous Studies

BWSP Trust noted 81 species were identified by Peddle Thorp in a study for BHP in 1994. Ongoing records in Lake Macquarie City Council's (LMCC) fauna database for this area include three vulnerable avian species (Black Bittern *Ixobrychus flavicollis*, Great Knot *Calidris tenuirostris*, Powerful Owl *Ninox strenua*) and two endangered species (Little Tern *Sterna albifrons* and Swift Parrot *Lathamus discolor*). M.K. Laverick presented a three-year Bird Life Study report to LMCC (2001) which included detailed information on 113 species observed over 104 surveys in a three-year period. It also provided site photographs and a scale drawing of the study area showing four adjacent open freshwater areas (North and South Railway Swamps, Swan Lake, Big Swamp). In **Figure 1** these lie within the red and yellow sail-shaped area bordered by George Fire Trail and Merleview Fire Trail. Laverick's report helped shape the current study's survey route and enabled comparison of observations of the same species 20 years apart.

Study Area Vegetation

Section 6.5 of BWSP Trust (2010) identifies three dominant vegetation communities in these wetlands. Coastal Sand Scrub is relatively dense and low, with Coast Tea Tree *Leptospermum laevigatum*, Coast Banksia *Banksia integrifolia* and Coast Wattle *Acacia sophorae.* Swamp Mahogany-Paperbark Forest features Broad-leaved Paperbark *Melaleuca quinquenervia*, Swamp Mahogany *Eucalyptus robusta* and Swamp Oak *Casuarina glauca.* The third is fragmented areas of Coastal Sand Apple-Blackbutt Forest, with Smooth-barked Apple *Angophora costata*, Blackbutt *E. pilularis* and Old Man Banksia *Banksia serrata.* Red Bloodwood *Corymbia gummifera* also appears along the Fernleigh Track – Kalaroo Fire Trail border. Besides natural regrowth, considerable new planting has occurred. Golden Wreath Wattle *Acacia saligna* and Bitou Bush *Chrysanthemoides* *monilifera* were planted extensively from c. 1960, rapidly colonizing and dominating the hind dunes. Introduced grasses (Coolatai grass *Hyparrhenia hirta* and Natal grass *Melinis repens*) are also prevalent along fire trails. Together these sources yield abundant seed and blossom.

Figure 1 Aerial Map of Belmont Wetlands and Belmont Lagoon (State Park boundaries in red; original Belmont Swamp area is sail-shaped between George Fire Trail and Merleview Fire Trail). Adapted with permission from Belmont Wetlands State Park Trust, Final Plan of Management 2010, Fig. 12, p.43.

METHODS

Between 9/4/2015 and 29/12/2017 this author completed 73 regular bird surveys at Belmont Wetlands about every 10 days – except from 1/6/2016 to 28/9/2016 due to closure of George Fire Trail (GFT). The 6.7km route included the triangular route used by Laverick (4.4km). It started where Fernleigh Track (FT) and GFT meet at Belmont rail siding, but continued past Merleview Fire Trail (MFT) to Jewells siding. Then it crossed to the adjacent, dirt Kalaroo Fire Trail (KFT), went south along the eastern side of Laverick's "Big Swamp" until it met the south end of GFT. The final leg went north on GFT back to FT. A survey took approx. 2 hours and was completed between 6.00 and 10.30am. All observations were audiotaped; these data were later transcribed and analysed using MS Excel.

Survey results are presented in two sections. The first identifies species observed in 2015-17, each with maximum number and average count (median) per survey, number of surveys a species was recorded (Obs) and its Belmont Wetlands Reporting Rate (BW RR%). The latter measures a species' detection, given as the percentage of all surveys it was recorded.

The second section compares species observations in the current study with Laverick's report. The Chi-squared Test (1df, with Yates correction factor) was used to identify very significant differences between pairs of observations for each species (Fowler & Cohen 1999). A very high level of significance ($p<.01$) was set for detecting differences in observed numbers of each species in 2001 from 2018 in order to minimize spurious effects from additional flora on the current survey route. This approach seems valid; the survey effort was similar in number of surveys and each study was conducted in a 3-year period.

RESULTS

Section 1: The Present Study

A total of 105 species were recorded at Belmont Wetlands (BW) over 73 surveys between April 2015 and December 2017 (see Appendix on website: http://hboc.org.au/publications/the-whistler). This list includes generalist and specialist (coastal woodlands) species, plus several waterbird species. Criteria specified in Stuart (2016) identify most of these as resident or visitor, breeding in the Hunter Region. Species in the Appendix are listed in decreasing order of Reporting Rate. Fifteen species were most commonly observed (BW RR%>80). No obvious feature (e.g. dietary preference) or taxonomic grouping is evident.

Principal dietary influences

To further examine birds' use of BW habitats, **Table 1** shows the 40 most common species recorded (BW RR%>28). Nectar, seed or insects are the main diet for many of these, as indicated by seasonal spikes in median species counts at times of abundance. Nectar feeders commonly seen along FT are: Little Wattlebird *Anthochaera chrysoptera*, Lewin's Honeyeater *Meliphaga lewinii*, White-cheeked Honeyeater *Phylidonyris niger*, Striped Honeyeater *Plectorhyncha lanceolata,* Rainbow Lorikeet *Trichoglossus moluccanus,* Eastern Spinebill *Acanthorhynchus tenuirostris* and Yellow-faced Honeyeater *Caligavis chrysops*. Fairley & Moore (2010) support current BW survey records by offering the following times of year for floral abundance: Smooth-barked Apple (Nov.-Dec.), Coast Banksia and Old Man Banksia (Jan.-Jun.), Red Bloodwood (Feb.-Mar.), Broad-leaved Paperbark (Feb.-May), Swamp Mahogany (Apr.-Sep.), Swamp She-oak (Jun.-Sep.), Coast Tea Tree and Coast Wattle (Jul.-Oct.), and Golden Wreath Wattle (Aug.-Nov.). Small passerines (Silvereye *Zosterops lateralis*, Superb Fairy-wren *Malurus cyaneus*, White-browed Scrub-wren *Sericornis frontalis*) feed on flowering weeds (Bitou Bush *Chrysanthemoides monilifera* and Lantana *Lantana camara*) in spring and autumn.

Table 1. Comparison of survey data for the 40 most common species at Belmont Wetlands (BW) 2015-2017 and Belmont Swamp (BS) 1997-1999 (Laverick in LMCC 2001).

Common Name	**Scientific Name**	**Max**	**Median**	**Obs**	**BW RR%**	**BS RR%**	χ^2
Little Wattlebird	*Anthochaera chrysoptera*	57	20.0	73	100	97	0.0
Eastern Whipbird	*Psophodes olivaceus*	43	14.0	73	100	96	0.1
Lewin's Honeyeater	*Meliphaga lewinii*	21	9.0	72	99	54	**11.7***
Grey Butcherbird	*Cracticus torquatus*	18	5.0	72	99	73	3.2
Australian Raven	*Corvus coronoides*	22	6.0	70	96	100	0.0
Grey Fantail	*Rhipidura fuliginosa*	32	5.0	68	93	91	5.3
Laughing Kookaburra	*Dacelo novaeguineae*	13	4.0	68	93	70	2.7
Bar-shouldered Dove	*Geopelia humeralis*	30	6.0	65	89	70	1.9
Superb Fairy-wren	*Malurus cyaneus*	21	5.0	63	86	47	**10.0***
Spotted Dove	*Streptopelia chinensis*	9	2.0	63	86	89	0.0
Australian Magpie	*Gymnorhina tibicen*	12	3.0	62	85	92	0.1
White-cheeked Honeyeater	*Phylidonyris niger*	33	10.5	60	82	85	0.0
Pied Currawong	*Strepera graculina*	8	2.0	61	84	28	**25.3***
Striped Honeyeater	*Plectorhyncha lanceolata*	12	3.0	59	81	1	**79.2***
Red-browed Finch	*Neochmia temporalis*	53	8.0	59	81	66	1.2
Rainbow Lorikeet	*Trichoglossus moluccanus*	188	5.0	54	74	11	**44.0***
Grey Shrike-thrush	*Colluricincla harmonica*	7	2.0	51	70	35	**10.4***
Welcome Swallow	*Hirundo neoxena*	37	3.0	49	67	60	0.3
Olive-backed Oriole	*Oriolus sagittatus*	13	4.0	49	67	32	**10.5***
Eastern Yellow Robin	*Eopsaltria australis*	6	1.0	49	67	31	**12.3***
Silvereye	*Zosterops lateralis*	90	5.0	46	63	95	4.9
Brush Bronzewing	*Phaps elegans*	12	5.0	40	55	21	**11.4***
White-browed Scrubwren	*Sericornis frontalis*	46	3.0	41	56	71	0.2
Eastern Spinebill	*Acanthorhynchus tenuirostris*	15	2.0	41	56	19	**16.3**
Golden Whistler	*Pachycephala pectoralis*	8	3.0	41	58	9	**34.6**
Rufous Whistler	*Pachycephala rufiventris*	15	2.0	39	53	52	0.1
Brown Thornbill	*Acanthiza pusilla*	17	3.0	36	49	44	0.2
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	7	2.0	36	49	68	2.1
Pacific Black Duck	*Anas superciliosa*	5	2.0	33	45	81	**7.4***
Yellow-faced Honeyeater	*Lichenostomus chrysops*	28	3.0	33	45	29	3.0
Grey Goshawk	*Accipter novaehollandiae*	2	1.0	26	36	2	**28.9***
White-breasted Woodswallow	*Artamus leucorhynchus*	33	6.0	29	40	75	**8.1**
Willie Wagtail	*Rhipidura leucophrys*	3	1.0	28	38	76	**9.3**
Dollarbird	*Eurystomus orientalis*	13	2.0	25	34	20	2.8
Brown Quail	*Synoicus ypsilophora*	21	3.0	23	32	7	**14.3***
Eastern Rosella	*Platycercus eximius*	11	2.0	22	30	63	**8.3**
Noisy Miner	*Manorina melanocephala*	8	2.0	22	30	7	**13.1***
Galah	*Eolophus roseicapilla*	10	2.0	21	29	47	3.1
Eastern Koel	*Eudynamys orientalis*	3	1.0	23	32	1	**27.8***
Crested Pigeon	*Ocyphaps lophotes*	6	1.0	21	29	56	6.2

Note: Obs is the number of surveys a species was recorded. Chi-square values (χ^2>6.63) in bold indicate a very significant difference (p<.01) in observed frequencies of the same species between the current study and Laverick's study (LMCC 2001). An asterisk (*) indicates 15 species that are reported far more often in 2018 than in 2001.

Table 1 also suggests that BW habitats provide abundant insects, attracting obligate aerial summer visitors (Dusky Woodswallow *Artamus cyanopterus*, White-breasted Woodswallow *Artamus superciliosus*, Welcome Swallow *Hirundo neoxena* and Dollarbird *Eurystomus orientalis*) and generalist woodland species (Grey Butcherbird *Cracticus torquatus*, Australian Magpie *Gymnorhina tibicen*, Pied Currawong *Strepera graculina*, Black-faced Cuckoo-shrike *Coracina novaehollandiae* and Grey Fantail *Rhipidura fuliginosa*). Pied Butcherbird *Cracticus nigrogularis* was observed rarely at BW (RR=8%).

BW's flora attract a wide range of granivorous species in season. Bar-shouldered Dove *Geopelia humeralis*, Spotted Dove *Streptopelia chinensis*, Red-browed Finch *Neochmia temporalis*, Brush Bronzewing *Phaps elegans*, Brown Quail *Coturnix ypsilophora,* Galah *Eolophus roseicapilla* and Crested Pigeon *Ocyphaps lophotes* are seen feeding on acacia, melaleuca, leptospermum and grass seeds on tracks. Seed-fruits attract: Eastern Rosella *Platycercus eximius*, Pied Currawong, and Yellow-tailed Black-Cockatoo *Zanda funereus* in autumn, while Little Corella *Cacatua sanguinea,* Australian King-Parrot *Alisterus scapularis* and Scaly-breasted Lorikeet *Trichoglossus chlorolepidotus* are uncommon spring-summer foragers. Sulphur-crested Cockatoo *Cacatua galerita* visit in any month, but typically in spring-summer. Small flocks of Plum-headed Finch *Neochmia modesta* and Double-barred Finch *Taeniopygia bichenovii* have also been reported foraging on BW fire trails (Stuart 2016).

Small rodents, reptiles, birds, frogs, crustaceans and native fish in BW's marshes contribute to the diet of various carnivorous and omnivorous species listed as common residents in **Table 1**. Open woodland generalists include: Australian Raven *Corvus coronoides*, Laughing Kookaburra *Dacelo novaeguineae*, Pied Currawong and Grey Shrike-thrush *Colluricincla harmonica*. Eight raptor species were recorded hawking over BW in 2015-17, including Osprey *Pandion haliaetus* and Swamp Harrier *Circus approximans*. In autumn 2016 and 2017 a pair of White-bellied Sea-Eagle *Haliaeetus leucogaster* was observed, and separately, a single juvenile each season. All but Grey Goshawk *Accipiter novaehollandiae* are uncommon residents of this coastal area.

Use by vagrants and migrants

Monthly records of juvenile birds along the survey route suggest that BW provides a range of habitats that support breeding by most resident and visitor species listed in **Table 1**, although nest sites are not usually obvious nor is confirmation easy within the current survey schedule. However, Dusky Woodswallow was observed at BW in 2015, and one pair bred successfully in summer 2016/17 - at the eastern-most part of its range in the Hunter (Stuart 2016). White-breasted Woodswallow are summer visitors which nest in small knot-holes of tall, dead Broad-leaved Paperbark stumps in the middle of BW's open marshes. Few of these stumps remain now, but alternative nest sites have not been detected.

A species of interest which nests in dense acacia, melaleuca, or leptospermum scrub on coastal hind dunes is Brush Bronzewing (Higgins & Davies 1996). This secretive ground pigeon was recorded in good numbers per survey during its breeding period from September to March in 2015/16 and 2016/17 (Feletti 2017). However this pattern did not repeat in spring-summer 2017/18. The species was occasionally seen or heard in winter (16 observed; 8 surveys) but rarely in spring or summer. No breeding behavior was observed in summer 2017/18.

Other summer-breeding or autumn visitors include: Olive-backed Oriole *Oriolus sagittatus*, Sacred Kingfisher *Todiramphus sanctus,* Dollarbird, Black-faced Cuckoo-shrike and two cuckoo species - Eastern Koel *Eudynamys orientalis* and Channel-billed Cuckoo *Scythrops novaehollandiae*.

Apart from the 40 common species, Belmont Wetlands has had over 60 occasional visitors, listed in the Appendix (RR<15%). Most of these visitors are nomadic, but resident in the Hunter Region (Stuart 2016), including: Brown Honeyeater *Lichmera indistincta*, Scarlet Honeyeater *Myzomela sanguinolenta*, Mistletoebird *Dicaeum hirundinaceum*, Spangled Drongo *Dicrurus bracteatus*, Crested Shrike-tit *Falcunculus frontatus*, Red-whiskered Bulbul *Picnonotus jocosus*, Black-faced Monarch *Monarcha melanopsis*, Leaden Flycatcher *Myiagra rubecula*, Rufous Fantail *Rhipidura rufifrons,* Yellow-tailed Black-Cockatoo*,* Sulphur-crested Cockatoo *Cacatua galleria,* and Little Corella.

Section 2: Comparison of Current Study Results with Laverick's 2001 Report

Laverick's detailed Bird Life Study (LMCC 2001) allows us to compare ecological and avian data at the same site. Climate-wise, the two 3-year periods were quite similar. In 1997, his first year of surveys at Belmont Swamp (BS), an intense El Nino and drought conditions occurred. Above-average rainfall (La Nina) was recorded in 1998, stopping surveys for several months until a more normal weather pattern followed in 1999. In 2015-16 Belmont Wetlands experienced a similar weather pattern; then in 2017 intense, prolonged and record heatwaves began in early spring and continued into mid-autumn 2018. Marshes (occasional wetlands) remained dry and overgrown with cumbungi and weeds.

This reflects a major ecological change in 20 years because Laverick's photographs and sketch map of BS (the 'sail-area' in **Figure 1**) in his report show open-water swamps. He also recorded medium to large waterbird species and their breeding attempts (e.g. Dusky Moorhen *Gallinula tenebrosa*, Black Swan *Cygnus atratus*, Great Cormorant *Phalacrocorax carbo,* Australian White Ibis *Threskiornis moluccus*, Grey Teal *Anas gracilis,* Royal Spoonbill *Platalea regia,* Australasian Grebe *Tachybaptus novaehollandiae*, Nankeen Night-Heron *Nycticorax caledonicus*, Straw-necked Ibis *Threskiornis spinicollis*, Latham's Snipe *Gallinago hardwickii*, Intermediate Egret *Ardea intermedia*, Little Egret *Egretta garzetta*, Caspian Tern *Hydroprogne caspia*.) None of these was observed at BW in the current study. Occasionally a pair of Pacific Black Duck *Anas superciliosa*, Chestnut Teal *Anas castanea* and Purple Swamphen *Porphyrio porphyrio* have been recorded for short periods on a brackish marsh beside FT.

Table 1 enables us to compare detection of each species in the two studies (BS in 2001, BW in 2018). For example, Striped Honeyeater (BW RR% 81) and Rainbow Lorikeet (BW RR% 74) were commonly observed species in 2015-17 but recorded rarely by Laverick (1%, 11% respectively in 1997-99). Of the 15 most common species in this study (BW RR%>80) 7 were also most commonly recorded by Laverick (BS RR%>80). These are: Little Wattlebird, Eastern Whipbird *Psophodes olivaceus*, Australian Raven, Grey Fantail, Spotted Dove, Australian Magpie, White-cheeked Honeyeater. Further comparisons (BW RR%, BS RR%) show another 8 species are most common in 2018 (BW RR%>80) but were not in 2001 (BS RR%<80). These are: Lewin's Honeyeater, Grey Butcherbird, Laughing Kookaburra, Bar-shouldered Dove, Superb Fairy-wren, Pied Currawong, Striped Honeyeater, Red-browed Finch. This Appendix (2018) and Laverick's study (LMCC 2001) each lists over 60 uncommon species (RR%<28). Many were detected in both studies, sometimes only once (Crested Shrike-tit *Falcunculus frontatus*, Cicadabird *Edolisoma tenuirostris,* Leaden Flycatcher *Myiagra rubecula,* Black-faced Monarch *Monarcha melanopsis*).

It is useful to determine which species have responded to woodland habitat changes (e.g. flora rehabilitation efforts, natural maturation) since 2001. This is done statistically, by comparing observation tallies (BW Obs, BS Obs) for each of 40 common species in this study. Using the chi-squared test on these data (frequencies, not RR%) yields χ^2 values in **Table 1**. Nineteen of these species have very significant differences ($p<.01$ $\chi^2>6.63$) in observed frequencies. An asterisk (*) indicates 15 species are reported far more often in 2018 than in 2001. These include: Lewin's Honeyeater, Superb Fairy-wren, Pied Currawong, Striped Honeyeater, Rainbow Lorikeet, Grey Shrike-thrush, Olive-backed Oriole, Eastern Yellow Robin, Brush Bronzewing, Eastern Spinebill, Golden Whistler, Grey Goshawk, Brown Quail, Noisy Miner and Eastern Koel. The other four species in bold were reported far more often in 2001 (Pacific Black Duck, White-breasted Wood-swallow, Eastern Rosella and Willie Wagtail). χ^2 values for the other 21 species (40%) in **Table 1** indicate no measurable difference in detection (at this confidence level) between 2018 and 2001.

DISCUSSION

Results from the current study (2018) indicate that avian fauna continue to thrive in the surveyed area (BW) of Belmont Wetlands State Park. The observed total of 105 species at Belmont Wetlands in 2018 compares favourably with Laverick's 2001 list of 113 species. Similar weather patterns in 1997-99 and 2015-17 enable comparison of this coastal habitat and its birdlife in those two study periods (BOM 2017). Two main ecological changes have occurred there in the past 20 years. First, open wetlands reported in 2001 are now marshes overgrown with reeds and weeds. This probably explains nomadic movement of at least 13 common waterbird species to adjacent sites; they are occasionally seen flying over the area. The marshes now support significantly more observations of woodlands insectivores (Olive-backed Oriole, Superb Fairy-wren, Eastern Yellow Robin, Golden Whistler, Eastern Koel), and insects supplement the diet of many other species.

Second, along with hydrological changes at BW has been ongoing maturation of woodlands flora on these coastal hind dunes, and an increased floral abundance from substantial replanting of native flora since the 1960s. Together these influences have resulted in greater diversity in nectar-eaters, granivorous and other carnivorous/omnivorous species which regularly use this coastal woodlands habitat. Similar, high reporting rates for these broad dietary groups in 2001 and 2018 studies indicate most of their respective species were common originally but survey maxima data indicate they are thriving in 2018. Two notable species predictably very common within the next decade are Striped Honeyeater and Rainbow Lorikeet, due to their breeding and dietary behaviour (Moffat *et al*. 1983).

The change from open water swamps to occasional marshes has likely affected reporting rates of some coastal raptors (Osprey 2001 19%; 2018 4%), White-bellied Sea-Eagle (2001 13%; 2018 22%), Grey Goshawk (2001 2%; 2018 36%) but most of the woodlands raptors reported by Laverick are still seen hawking infrequently, here and over Belmont Lagoon (Feletti 2016).

Apart from the (13) waterbird species not observed at BW in 2015-17, four species showed significant decline in observations from 1997-99. Of no concern is Pacific Black Duck, now observed at Cold Tea Creek. Eastern Rosella and White-breasted Woodswallow each nest in mature tree-hollows – generally in short supply at BW due to urbanisation and industry. These nest sites are under increased competition from bird and possum species. Open nests of Willie Wagtail are also now at greater risk of predation from woodlands carnivores or cuckoo species. However these data may be cyclical, since 2018 records show increased sightings of each species and 20-year records in the Hunter Region indicate their status is 'of no concern' (Stuart 2016).

Both Belmont Lagoon and Belmont Wetlands contain a considerable amount of reed bed habitat potentially suitable for the Australasian Bittern *Botaurus poiciloptilus* which has recently been reported from several adjacent suburbs. It was not recorded in this study as it would have low probability of detection with the survey method used. For the same reason Powerful Owl was not detected in the current study, despite recent confirmed reports (B. Ciezak pers. comm. 2/5/2018) at BWSP.

CONCLUSION

Both 2001 and 2018 studies reported a mix of generalist and specialist (coastal) woodlands birds. Many common species in 2001 have increased in observed numbers per survey, and in detectability (RR%). This reaffirms BWSP as a sustainable coastal woodlands. There is also strong overlap between species listed as uncommon (e.g. RR%<15), which supports BWSP's ongoing legacy as a short-term refuge and food source for numerous woodlands visitors and birds of passage. Laverick provided LMCC with key information on the adverse consequences of further residential development or neglect of this natural asset. In summary, the 2001 and 2018 reports are encouraging; despite ecological changes to this habitat over the last 20 years, birdlife is flourishing at Belmont Wetlands State Park.

Several other species, listed in the *Threatened Species Conservation Act 1995* (NSW) and more recently in the *Biodiversity Conservation Act 2016* (NSW), have a regular presence in this area: White-bellied Sea-Eagle, Osprey, and Dusky Woodswallow (Roderick & Stuart 2016).

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The birds of Mambo Wetlands Reserve, Port Stephens

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The avian population of the Mambo Wetlands Reserve was surveyed in 2017 and 2018. The majority of the species recorded were common woodland birds. Six uncommon species were recorded and there was an unconfirmed record of the Australian Little Bittern *Ixobrychus dubius,* which is a rare species in the Hunter Region. Records were compared with those from surveys conducted between 1999 and 2016. The number of species recorded declined from 116 to 90 and a comparison of reporting rates suggested that many species had decreased in abundance.

INTRODUCTION

In 2017 a proposed development of land adjoining the Mambo Wetlands Reserve provided impetus for a proposal to have the reserve nominated as a site of international importance under the Ramsar Convention on Wetlands (1999). Previous studies by General Flora and Fauna (2004), Gary Worth Project Consulting (2009) and Wildthing Environmental Consultants (2016), and a report by Port Stephens Council (2006) identified a number of threatened avian, mammal, amphibian and flora species within the reserve and adjacent areas. Four endangered ecological communities were also identified in the reserve.

To support the above nomination, regular surveys of the reserve were conducted over 13 months in 2017 and 2018 to confirm the avian threatened species present (Fraser 2018). The objective of this paper is to document the full suite of avian species present and compare with previous studies.

METHOD

Mambo Wetlands Reserve is located at Salamander Bay, Port Stephens NSW (32° 44' 00"S, 152° 05' 45"E). Covering 175 hectares of saltwater and freshwater wetlands and coastal forest, the reserve is connected to Salamander Bay by the tidal Mambo Creek. It is bounded by Foreshore Drive to the north, Port Stephens Drive to the west, Salamander Way to the south and Sandy Point Road to the east. Residential properties adjoin the reserve to the southwest, south and east, while the Salamander Bay shopping and council precinct adjoins to the southeast. To the north it meets the shores of Salamander Bay. Seven vegetation communities are present in the reserve; Coastal Sand Woodland (CSW), Estuarine Mangrove Complex (EMC), Estuarine Saltmarsh Complex (ESC), Freshwater Gahnia Swamp Forest (FGSF), Moist Coastal Apple Forest (MCAF), Mahogany/Paperbark Swamp Forest (MPSF) and Paperbark/Swamp Oak Complex (PSOC) (Port Stephens Council 2006). The location of the reserve is shown in **Figure 1** and the vegetation communities in **Figure 2**.

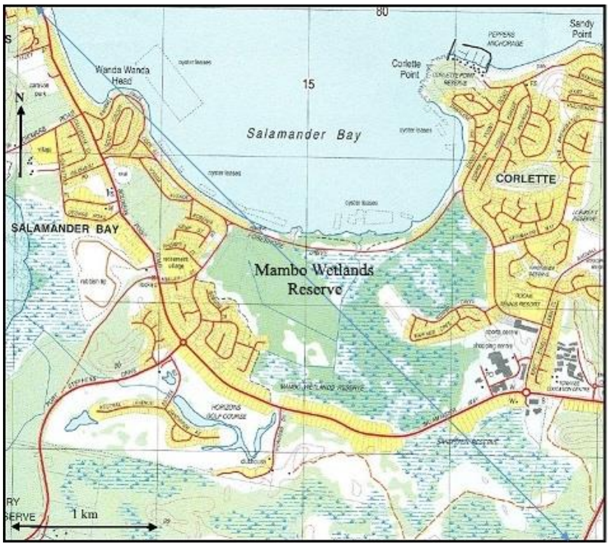

Figure 1. Mambo Wetlands Reserve location map

The reserve is the hub of several wildlife corridors. To the east, it is linked via Kingfisher Reserve to Gan Gan Hill. To the south it adjoins Sandpiper Reserve and thence links to Tomaree National Park to the east and Tilligerry Nature Reserve to the west. To the west the reserve is connected through Boronia Gardens Reserve and Wanda Wetlands Reserve to reserves along Cromarty Bay and the Tilligerry Nature Reserve. The tidal areas of the reserve are part of the Port Stephens -

Great Lakes Marine Park. The reserve is zoned Environmental Protection - Urban Conservation by the Port Stephens Council.

The study initially involved identifying records from technical reports and previous bird surveys. A total of 29 surveys were conducted between 1999 and 2016; Tom Clarke (1), Graeme Stevens (16), BirdLife Australia Atlas (2), General Flora and Fauna (3), Mambo-Wanda Reserve Committee (1), Tomaree Bird Watchers (5) and Wildthing Environmental Consultants (1).

The routes and methods used for these surveys are unknown. Shorebird records of Salamander Bay included in some surveys were not used. This study was limited to birds recorded within the boundaries of the reserve.

A programme of standardised surveys was conducted to confirm the presence of previously recorded species, particularly threatened species. As previous surveys had neglected the areas of EMC, ESC and CSW, a more uniform survey approach was adopted across all vegetation communities. Two survey routes were adopted. One route accessed the reserve from the east and covered areas of the CSW, FGSF, MPSF and PSOC communities. The other route accessed the reserve from the west sampling the ESC, MCAF and MPSF communities. The EMC community was only surveyed at low tide. Due to difficult access, the FGSF and PSOC communities were surveyed using a 2-ha 20-minute method from fixed points. Other communities were surveyed using a 500 m-area search method (BirdLife Australia 2018). Species present in each community were recorded separately. The eastern and western routes were surveyed on alternate occasions. Surveys took 3.0-3.5 hours. The routes were a combination of existing tracks and regular circuits through the less accessible areas. A total of 41 surveys were conducted approximate weekly between February 2017 and February 2018. The survey routes and survey points are shown on **Figure 2**.

Most of the 1999-2016 surveys recorded species' presence only and this was continued for the 2017-2018 surveys. Reporting rates (RR) were calculated for both data sets by dividing the number of surveys and expressing the result as a percentage. To identify if recorded differences for species between the two datasets were significant, the chi square test was applied using the Yates correction.

RESULTS

The 29 surveys conducted from 1999 to 2016 recorded 116 species. In 2017-2018, 41 surveys were conducted and 90 species were recorded. The majority of the species recorded are common throughout the Hunter Region (Stuart 2017) and are dominantly woodland birds (see **Table 1**). The species recorded and their reporting rates (RR) are presented in the **Appendix.**

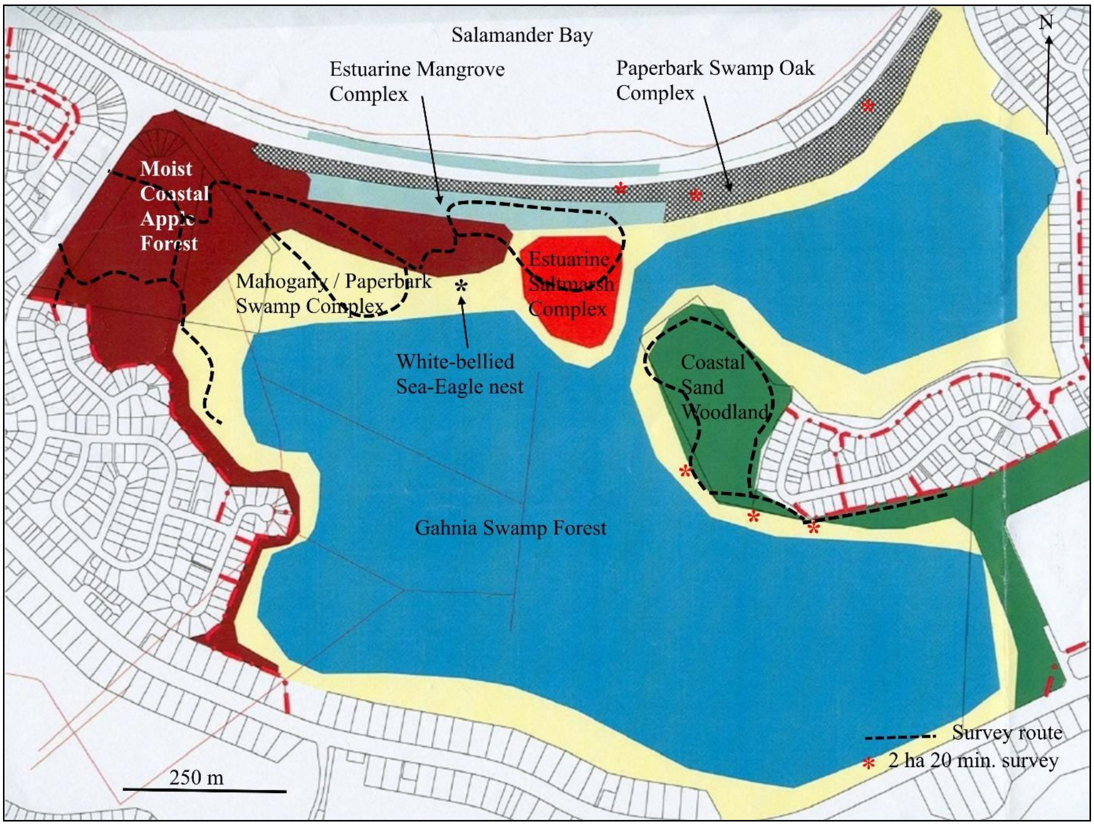

Figure 2. Vegetation communities and survey routes, Mambo Wetlands Reserve

Table 1. Summary of species categories, Mambo Wetlands Reserve.

	1999-2016 Surveys	**2017-2018 Surveys**
Woodland Birds	92	73
Waterbirds	15	13
Raptors	9	4
Total Species	116	90

There was an unconfirmed record of the Australian Little Bittern *Ixobrychus dubius*, a rare species in the Hunter Region (Stuart 2017). Six uncommon species (Stuart 2017) were also recorded; Fork-tailed Swift *Apus pacificus*, Lewin's Rail *Lewinia pectoralis*, Powerful Owl *Ninox strenua*, Glossy Black-Cockatoo *Calyptorhynchus lathami,* White-browed Woodswallow *Artamus superciliosus* and Restless Flycatcher *Myiagra inquieta*. The presence of Fork-tailed Swift has no significance with respect to the importance of the reserve. The Lewin's Rail and the possible Australian Little Bittern were identified by call.

The species most commonly recorded varied between the two survey periods. Those with RR>60% (in most instances) comprise two groups occupying different strata in the habitat. The canopy was dominated by Rainbow Lorikeet *Trichoglossus moluccanus*, Yellow-faced Honeyeater *Caligavis chrysops*, Laughing Kookaburra *Dacelo novaeguineae*, Australian Magpie *Gymnorhina tibicen,* Grey Butcherbird *Cracticus torquatus*, Little Wattlebird *Anthochaera chrysoptera*, Noisy Miner *Manorina melanocephala* and Eastern Rosella *Platycercus eximius*. Smaller, less obvious species were mainly recorded in the shrub layer and ground layer; Brown Thornbill *Acanthiza pusilla,* Grey Fantail *Rhipidura albiscapa*, White-cheeked Honeyeater *Phylidonyris niger*, Variegated Fairy-wren *Malurus lamberti*, Eastern Yellow Robin *Eopsaltria australis*, Golden Whistler *Pachycephala pectoralis,* White-browed Scrubwren *Sericornis frontalis*.

Almost half (48%) of the species recorded in both survey periods were recorded infrequently and had RR<15%. The majority of these species are common in the Hunter Region (Stuart 2017). Fewer species (90) were recorded in the recent surveys than in previous data sets (116). The majority of this decrease was in woodland species (**Table 1**).

Of the species recorded in 1999-2016 surveys, 31 were not recorded in 2017-2018, and 26 of these have not been recorded for 10 years or more. Among those not recorded in 2017-2018 were four of the uncommon species, Fork-tailed Swift, Powerful Owl, Glossy Black-Cockatoo and Restless Flycatcher. Seven new species were recorded in the reserve in 2017-2018 (see **Appendix**). Comparison of the two datasets, shows that the RRs of some larger, more mobile species have increased, many of the smaller woodland species have decreased while some have remained relatively unchanged. Four species that illustrate this are White-bellied Sea-Eagle *Haliaeetus leucogaster,* Lewin's Honeyeater *Meliphaga lewinii,* White-throated Treecreeper *Cormobates leucophaea* and Varied Sittella *Daphoenositta chrysoptera*. All four species are residents in the Hunter Region (Stuart 2016). The White-bellied Sea-Eagle had a RR of 17.2% for 1999-2016 which increased to 39% in the 2017-2018 surveys while Lewin's Honeyeater increased from 10.3% in 1999-2016 to 46.3% in 2017-2018. In contrast, the White-throated Treecreeper had a RR of 44.8% in 1999-2016, but was not recorded during the 2017-2018 surveys. The RR for the Varied Sittella was almost unchanged with RRs of 17.2% in 1999-2016 and 14.6% in 2017-2018.

To test significance of the difference in numbers recorded between the two datasets, the chi square test was applied. Species that exhibited a statistically significant difference are shown in **Table 2**.

The only confirmed breeding record from both survey periods was for the White-bellied Sea-Eagle. An occupied nest was located in January 2018 on the edge of the MPSF (32° 43′ 52.14"S, 152° 05′ 34.62"E) where it overlooked Salamander Bay. The location is shown in **Figure 2**. It is not known how long this nest has been at this site.

Seven of the species recorded in the reserve are classified as vulnerable in NSW (*Biodiversity Conservation Act 2016*); Eastern Osprey *Pandion haliaetus*, White-bellied Sea-Eagle, Powerful Owl, Glossy Black-Cockatoo, Little Lorikeet *Glossopsitta pusilla*, Varied Sittella and Dusky Woodswallow.

Table 2. Species exhibiting significant differences in reporting rate (RR) between the 1999-2016 and the 2017-2018 surveys. Chi square values are 3.841 for p<0.05 and 6.635 for p<0.01.

Species	1999-2016 Surveys		2017-2018 Surveys		Chi square values	P	Significance
	RR	Records	RR	Records			
Peaceful Dove	17.2%	5	0.0%	0	4.86	p<0.05	Significant
Bar-shouldered Dove	17.2%	5	53.7%	22	4.93	p<0.05	Significant
Whistling Kite	24.1%	7	0.0%	0	7.63	p<0.01	Very significant
Galah	51.7%	15	12.2%	5	7.96	p<0.01	Very significant
Little Corella	24.1%	7	61.0%	25	4.27	p<0.05	Significant
Musk Lorikeet	17.2%	5	0.0%	0	4.86	p<0.05	Significant
White-throated Treecreeper	44.8%	13	0.0%	0	16.04	p<0.01	Very significant
Scarlet Honeyeater	31.0%	9	2.4%	1	7.82	p<0.01	Very significant
Lewin's Honeyeater	10.3%	3	46.3%	19	5.90	p<0.05	Significant
Mangrove Gerygone	0.0%	0	43.9%	18	11.08	p<0.01	Very significant
Rufous Whistler	37.9%	11	7.3%	3	6.50	p<0.05	Significant
Grey Shrike-thrush	58.6%	17	19.5%	8	6.22	p<0.05	Significant
Pied Currawong	27.6%	8	2.4%	1	6.51	p<0.05	Significant

DISCUSSION

Interpretation of the difference in RRs between the two data sets is complicated by the difference in survey methods, the observers conducting the surveys and the extent to which different ecological communities were sampled. It is not possible to separate the impact of these differences from the underlying changes in the status of each species. Consequently, the statistical evaluation merely reflects differences in the frequency species were recorded and does not necessarily imply a statistically significant change in the status of a species.

Factors contributing to statistically significant changes of Reporting Rate (Table 2)

The increased numbers of Mangrove Gerygone *Gerygone mouki* and Bar-shouldered Dove *Geopelia humeralis* recorded reflect increased survey effort in ecological communities that were not previously surveyed. The Mangrove Gerygone was only recorded in the EMC and ESC communities and the Bar-Shouldered Dove was mostly frequently recorded in the CSW community.

The decreased occurrence of Peaceful Dove recorded is attributed to habitat changes following the fire in the reserve in 2003. Open areas produced by burning of ground cover and shrub layer would have provided habitat suitable for ground foraging, granivorous species such as the Peaceful Dove. Subsequent regrowth would have resulted in loss of habitat and permanent relocation of the species. Other species with similar habitat requirements that exhibit the same pattern of change are Brown Quail *Coturnix ypsilophora*, Double-barred Finch *Taeniopygia bichenovii* and Australasian Pipit *Anthus novaeseelandiae.* The Brown Quail was last reported in 2004 and the other two species were last reported in 2006 (see **Appendix**). The Black-shouldered Kite *Elanus axillaris*, an open grassland foraging species, was similarly last reported in 2006.

The absence of Musk Lorikeet records in 2017-2018 is attributed to their nomadic behaviour and the absence of suitable species flowering during the survey period. They have since been recorded in the reserve in response to flowering of Blackbutt *Eucalyptus pilularis*.

Reasons for the increase in records of Little Corella and Lewin's Honeyeater are not clear. The Little Corella is reported as 'stable or possibly increasing' in the Hunter Region from BirdLife Australia Atlas records, while the Lewin's Honeyeater is reported as 'probably stable' (Stuart 2017). Newman (2009) reported Lewin's Honeyeater at Green Wattle Creek had benefitted from increased understorey growth, however there has been no apparent change in reserve habitat since recovery from the 2003 fire that would account for the increase. In contrast to the regional trends, the Atlas of NSW and ACT Birds (Cooper *et al.* 2014, Cooper *et al.* 2016) records the Little Corella RR trend as declining and the trend for

Lewin's Honeyeater on the North Coast as increasing since 2003.

Other species with decreased occurrence are Whistling Kite *Haliastur sphenurus*, Galah *Eolophus roseicapillus*, White-throated Treecreeper, Scarlet Honeyeater *Myzomela sanguinolenta,* Rufous Whistler *Pachycephala rufiventris*, Grey Shrike-thrush *Colluricincla harmonica* and Pied Currawong *Strepera graculina*. The absence of White-throated Treecreeper in 2017-2018 is considered to be a short-term anomaly as the species was recorded by Wildthing Environmental Consultants in 2016. All these species are reported as 'probably stable' in the Hunter Region from BirdLife Australia Atlas records (Stuart 2017). However, the Atlas of NSW and ACT Birds (Cooper *et al.* 2014, Cooper *et al.* 2016) records declining RR trends for Whistling Kite, Galah, Scarlet Honeyeater and Rufous Whistler. The Atlas reports the trend for the White-throated Treecreeper as being unclear and the trends for Grey Shrike-thrush and Pied Currawong as showing an increase. The Birds Australia report on Woodlands and Birds (2005), revealed a national decline for many woodland species. This report highlighted loss and fragmentation of habitat as a major contributing factor to the decline. In the Port Stephens region, continuing housing development in the Salamander Bay/Corlette area has resulted in the destruction of local habitat. The declining occurrence of these woodland species within the reserve most likely reflects the broader-scale trend for these species.

Species recorded infrequently or recorded in one survey only

A large number of species (48%) were recorded infrequently (RR<15%) in both survey periods. This is a normal outcome for this type of survey. However, of particular note are the 33 species recorded in the 1999-2016 surveys that were not recorded in the 2017-2018 surveys. Of these, 21 have not been recorded in 10 years or more. These were mainly woodland species plus some waterbirds that are common in the Hunter Region (Stuart 2017). The cryptic Buff-banded Rail *Gallirallus philippensisare* which has not been recorded since 2005, has been recorded in the reserve more recently (L. Wooding pers. comm.), but was not recorded during the 2017-2018 surveys. Confirmation of the occurrence of the Australian Little Bittern, which was an unconfirmed record during the recent surveys, should be a focus of future surveys.

The location of the reserve and the changing nature of its surrounds is probably a factor influencing the above observations. The reserve is closely surrounded by residential and commercial development, much of which has been built in the past 10-20 years. The reserve is impacted by noise from motor vehicles, residential and commercial sources and recreational activities within the reserve. This activity may have progressively made the reserve a less desirable habitat for some species. Many of the infrequently recorded woodland birds possibly use the reserve temporarily as they move between adjacent habitats in response to changing foraging opportunities. The absence of records for 10 years or more for some species may be a reflection of the broader-scale decreases in the diversity of woodland birds highlighted in the State of Australia's Birds study (Birds Australia 2005).

Resident threatened species

The reserve is an important habitat for two resident threatened species, White-bellied Sea-Eagle and Varied Sittella. The proximity of the reserve to the shoreline of Port Stephens and the presence of tall, mature trees in a secluded location makes the reserve an ideal nest site for White-bellied Sea-Eagle. The increased numbers of records in 2017-2018 is due to the presence of an active nest.

Varied Sittella is in decline in the Hunter Region (Newman 2010). The similar RRs (17.2% and 14.6%) for the two survey periods suggest that the local population may be stable. The RR is considerably higher than the overall Hunter Region RR of 3.7% from the BirdLife Australia Atlas (Stuart 2017). Newman (2015) has shown the importance of connectivity between remnant woodlands in providing habitat to support local strongholds for sittellas in the Paterson area of the Hunter Region. Newman suggested that the species was being locally nomadic. Varied Sittella clans require large territories, 13-20 ha with mature, rough-barked trees with hollows and dead branches to provide sufficient foraging and nesting opportunities (Noske 1998). The reserve provides these resources and is considered to be an important habitat and probable stronghold for the species, particularly in the near coastal areas of the Hunter Region.

CONCLUSION

Despite the urban location, surrounding development and associated habitat loss, the reserve provides suitable habitat for a number of resident, larger, more mobile species that dominate the canopy and smaller species that occupy the shrub and ground layers. These are dominantly woodland species. Around half the species recorded appear to use the reserve temporarily. The reduction in number of species present between the two survey periods is an indication of declining bird diversity in the reserve. The RRs of species that have declined and the previously reported species that are no longer recorded in the reserve may be a reflection of a broader-scale decline of woodland birds. The results of the surveys support efforts to obtain further recognition and protection for the reserve and other undeveloped areas of local habitat.

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APPENDIX

Species reporting rates all surveys, Mambo Wetlands Reserve, 1999-2018

Species	1999-2016 Surveys		2017-2018 Surveys		Last Record
	RR	Records	RR	Records	
Rainbow Lorikeet	93.1%	27	92.7%	38	Feb-18
Yellow-faced Honeyeater	86.2%	25	92.7%	38	Feb-18
Brown Thornbill	79.3%	23	95.1%	39	Feb-18
Laughing Kookaburra	72.4%	21	85.4%	35	Feb-18
Australian Magpie	72.4%	21	73.2%	30	Feb-18
Grey Fantail	69.0%	20	61.0%	25	Feb-18
White-cheeked Honeyeater	65.5%	19	34.1%	14	Feb-18
Grey Shrike-thrush	58.6%	17	19.5%	8	Feb-18
Grey Butcherbird	58.6%	17	92.7%	38	Feb-18
Variegated Fairy-wren	55.2%	16	65.9%	27	Feb-18
Little Wattlebird	55.2%	16	90.2%	37	Feb-18
Noisy Miner	55.2%	16	80.5%	33	Feb-18
Silvereye	55.2%	16	51.2%	21	Feb-18
Australian White Ibis	51.7%	15	46.3%	19	Feb-18
Galah	51.7%	15	12.2%	5	Feb-18
Eastern Yellow Robin	51.7%	15	63.4%	26	Feb-18
Eastern Spinebill	48.3%	14	26.8%	11	Feb-18
Golden Whistler	48.3%	14	82.9%	34	Feb-18
Eastern Rosella	44.8%	13	63.4%	26	Feb-18
White-throated Treecreeper	44.8%	13			Dec-16
Black-faced Cuckoo-shrike	44.8%	13	41.5%	17	Feb-18
Superb Fairy-wren	41.4%	12	17.1%	7	Jan-18
Australian Raven	41.4%	12	53.7%	22	Feb-18
Sacred Kingfisher	37.9%	11	36.6%	15	Feb-18
Rufous Whistler	37.9%	11	7.3%	3	Feb-18
Little Pied Cormorant	34.5%	10	7.3%	3	Feb-18
Olive-backed Oriole	34.5%	10	58.5%	24	Feb-18
Spotted Dove	31.0%	9	36.6%	15	Feb-18
Scarlet Honeyeater	31.0%	9	2.4%	1	May-17
Dusky Woodswallow	31.0%	9	14.6%	6	Dec-17
White-faced Heron	27.6%	8	19.5%	8	Feb-18
Pied Cormorant	27.6%	8	2.4%	1	Mar-17
Noisy Friarbird	27.6%	8	22.0%	9	Jun-17
Red Wattlebird	27.6%	8	19.5%	8	Nov-17
White-browed Scrubwren	27.6%	8	61.0%	25	Feb-18
Pied Currawong	27.6%	8	2.4%	1	Nov-17
Welcome Swallow	27.6%	8	51.2%	21	Feb-18
Crested Pigeon	24.1%	7	53.7%	22	Feb-18
Whistling Kite	24.1%	7			Oct-16
Dollarbird	24.1%	7	24.4%	10	Feb-18
Little Corella	24.1%	7	61.0%	25	Feb-18
Scaly-breasted Lorikeet	24.1%	7	7.3%	3	Jun-17
Spotted Pardalote	24.1%	7	26.8%	11	Feb-18
Channel-billed Cuckoo	20.7%	6	14.6%	6	Jan-18
Masked Lapwing	20.7%	6	9.8%	4	Jan-18
Brown Honeyeater	20.7%	6	26.8%	11	Feb-18

APPENDIX continued

Species reporting rates all surveys, Mambo Wetlands Reserve, 1999-2018

Species	1999-2016 Surveys		2017-2018 Surveys		Last Record
	RR	Records	RR	Records	
White-throated Gerygone	20.7%	6	31.7%	13	Feb-18
Eastern Whipbird	20.7%	6	14.6%	6	Feb-18
Pacific Black Duck	17.2%	5	4.9%	2	Apr-17
Peaceful Dove	17.2%	5			Dec-06
Bar-shouldered Dove	17.2%	5	53.7%	22	Feb-18
Pheasant Coucal	17.2%	5	29.3%	12	Feb-18
Eastern Koel	17.2%	5	41.5%	17	Feb-18
Fan-tailed Cuckoo	17.2%	5	26.8%	11	Nov-17
White-bellied Sea-Eagle	17.2%	5	39.0%	16	Feb-18
Musk Lorikeet	17.2%	5			Apr-09
Yellow Thornbill	17.2%	5	4.9%	2	Jan-18
Varied Sittella	17.2%	5	14.6%	6	Feb-18
Magpie-lark	17.2%	5	29.3%	12	Feb-18
Red-browed Finch	17.2%	5	7.3%	3	Oct-17
Purple Swamphen	13.8%	4	19.5%	8	Nov-17
Sulphur-crested Cockatoo	13.8%	4	2.4%	1	Feb-18
Pied Butcherbird	13.8%	4	2.4%	1	Apr-17
Willie Wagtail	13.8%	4	19.5%	8	Feb-18
Chestnut Teal	10.3%	3			Nov-17
Great Egret	10.3%	3	7.3%	3	Feb-18
Blue-faced Honeyeater	10.3%	3	9.8%	4	Feb-18
Brown-headed Honeyeater	10.3%	3	2.4%	1	Sep-17
White-naped Honeyeater	10.3%	3			Sep-07
Lewin's Honeyeater	10.3%	3	46.3%	19	Feb-18
Striated Thornbill	10.3%	3			Apr-09
White-breasted Woodswallow	10.3%	3	7.3%	3	Dec-17
Leaden Flycatcher	10.3%	3			Oct-08
Tree Martin	10.3%	3	4.9%	2	Apr-17
Shining Bronze-Cuckoo	6.9%	2	2.4%	1	Apr-17
Tawny Frogmouth	6.9%	2	2.4%	1	Sep-17
Dusky Moorhen	6.9%	2			Jan-06
Swamp Harrier	6.9%	2	2.4%	1	Feb-18
Grey Goshawk	6.9%	2			Jul-08
Powerful Owl	6.9%	2			Jul-16
Yellow-tailed Black-Cockatoo	6.9%	2	19.5%	8	Feb-18
Little Lorikeet	6.9%	2			Oct-16
Southern Emu-wren	6.9%	2	29.3%	12	Feb-18
Brown Gerygone	6.9%	2	2.4%	1	May-17
Australasian Figbird	6.9%	2	12.2%	5	Oct-17
Grey Teal	3.4%	1			Jan-06
Australian Wood Duck	3.4%	1	7.3%	3	Sep-17
Brown Quail	3.4%	1			Jun-05
Brown Cuckoo-Dove	3.4%	1			Dec-14
Horsfield's Bronze-Cuckoo	3.4%	1	4.9%	2	Aug-17
White-throated Needletail	3.4%	1	9.8%	4	Feb-18
Fork-tailed Swift	3.4%	1			Jan-06
Buff-banded Rail	3.4%	1			Jun-05

APPENDIX continued

Species reporting rates all surveys, Mambo Wetlands Reserve, 1999-2018

Species	1999-2016 Surveys		2017-2018 Surveys		Last Record
	RR	Records	RR	Records	
Nankeen Night-Heron	3.4%	1			Jan-06
Royal Spoonbill	3.4%	1	9.8%	4	Feb-18
Black-shouldered Kite	3.4%	1			Jan-06
Pacific Baza	3.4%	1			Jan-06
Brown Goshawk	3.4%	1	2.4%	1	Mar-17
Southern Boobook	3.4%	1			Dec-16
Rainbow Bee-eater	3.4%	1			Jan-06
Glossy Black-Cockatoo	3.4%	1			Jan-06
Long-billed Corella	3.4%	1			Dec-16
Striated Pardalote	3.4%	1			Sep-06
Yellow-rumped Thornbill	3.4%	1			Jan-06
White-winged Triller	3.4%	1			Jan-06
Spangled Drongo	3.4%	1	9.8%	4	Feb-18
Rufous Fantail	3.4%	1	4.9%	2	Dec-17
Restless Flycatcher	3.4%	1			Jan-06
Black-faced Monarch	3.4%	1			Jan-06
Rose Robin	3.4%	1	2.4%	1	Jan-18
Mistletoebird	3.4%	1	9.8%	4	Jan-18
Double-barred Finch	3.4%	1			Jan-06
Australasian Pipit	3.4%	1			Jan-06
Australian Reed-warbler	3.4%	1			Jan-06
Common Starling	3.4%	1			Dec-16
Common Myna	3.4%	1	2.4%	1	Mar-17
Lewin's Rail			7.3%	3	Nov-17
Australian Little Bittern			4.9%	2	Oct-17
Little Egret			2.4%	1	Dec-17
Eastern Osprey			9.8%	4	Feb-18
Striped Honeyeater			4.9%	2	Aug-17
Mangrove Gerygone			43.9%	18	Feb-18
White-browed Woodswallow			2.4%	1	Oct-17

Does the Black-headed Pardalote *Pardalotus striatus melanocephalus* occur in the Hunter Region?

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Recent observations of Striated Pardalote *Pardalotus striatus* in the north of the Hunter Region have revealed the birds to be intergrades between Black-headed Pardalote *P. s. melanocephalus* and another subspecies. This supports previous taxonomic conclusions and calls into question earlier reports of Black-headed Pardalotes from Taree and elsewhere in the Manning Valley.

INTRODUCTION

The Striated Pardalote *Pardalotus striatus* is widely distributed in Australia. Six subspecies are recognised, which intergrade where they abut (Higgins *et al.* 2002):

- *striatus*, the nominate subspecies, breeding in Tasmania and the Bass Strait islands and migrating to the mainland in the non-breeding period;
- *ornatus*, found in coastal and sub-coastal eastern and south-eastern Australia;
- *substriatus*, widely distributed in central and Western Australia;
- *melanocephalus* (often called the Black-headed Pardalote), occurring mainly in Queensland and northern NSW;
- *uropygialis*, found in much of northern Australia; and
- *melvillensis*, present on the Tiwi Islands.

In the field it can be a challenge to identify a Striated Pardalote to subspecific level. Birds are small and often high in the foliage, making it difficult to discern all the features needed for identification. The recent growth in use of digital cameras by many birdwatchers has made the task easier.

Hunter Region records of the Striated Pardalote

The Striated Pardalote is considered a common bird of the Hunter Region (Stuart 2017). Two subspecies are considered to be resident – *ornatus* and *substriatus*, favouring the eastern and western parts of the Region respectively. There are many sites where they meet (for example, in the Cessnock woodlands and areas around Singleton). There have been several records of them interbreeding (e.g. M. Roderick pers. comm.; P. Alexander pers. comm.) and possibly that is a relatively common occurrence.

The migratory subspecies *striatus* also occurs in the Region. All confirmed records to date have been for the period May to July (e.g. M. Roderick pers. comm.; A. Richardson pers. comm.; Hobbs & Kavaney 1962). Its distribution seems widespread: birds have been recorded in areas around Morisset, Cessnock and Singleton. There are several instances of them associating with one or other of the two resident subspecies.

Black-headed Pardalote

Recently, the distribution of *melanocephalus* was described as 'south to Laurieton' (Cooper *et al.* 2016). Laurieton is ~20km to the north of the Hunter Region boundary. However, in the past, this subspecies has been reported as occurring in the Hunter Region. A well-known early 20th century reference book described the distribution as being 'to the Hunter River' (Cayley 1931, and subsequent editions). In a review of the birds of New South Wales, it was stated that the distribution of *melanocephalus* in NSW was 'south to the Wollomba River' (Morris *et al.* 1981). This is assumed to mean the Wallamba River near Forster as there is no 'Wollomba River' in NSW.

The basis for such descriptions of the distribution is unclear. Just one Hunter Region record containing any detail has been located. A pair was reported nesting near Taree in August 1959 (Hobbs & Kavaney 1962; McGill 1966). The nest was found by A.R. McGill, who showed it to the other two. All three observers apparently were convinced of the identity of the occupants. McGill later noted that the record involved pairs each of *ornatus* and *melanocephalus* birds competing for a nest hollow, with the latter pair emerging victorious (McGill 1966).

McGill stated he had recorded *melanocephalus* sometimes in the Manning Valley but gave no details other than that they were uncommon in that area (McGill 1966).

Recent Observations

In 2017, birds initially identified as *melanocephalus* were found at two locations in the north of the Hunter Region. One bird was near Old Bar, seen by AS on 1 November 2017. The other bird was at Tea Gardens, seen intermittently for two months from late September (A. Rogers pers. comm.; L. Wooding pers. comm.).

In both cases, four important features of *melanocephalus* (unstreaked black cap, orange eyebrow, red wing spot, broad white wing bar) were clearly discerned in the field, leading to an initial tentative identification as that subspecies. However, close inspection of photographs revealed that both were intergrade birds. In both cases, the dark line below the bird's eye was faintly flecked with white feathers. This feature (flecking) was not obvious in the field. For a true *melanocephalus* subspecies bird the line should be all dark (Higgins *et al.* 2002; R. Cooper pers. comm.; R. Noske pers. comm.; M. Roderick pers. comm.).

Both the 2017 records were accepted by the Hunter Bird Observers Club (HBOC) Records Appraisal Committee as *melanocephalus* intergrade birds (RAC Case No. 502).

DISCUSSION

Given the coastal locations of the two sightings, both birds are suspected to have been intergrades between *ornatus* and *melanocephalus* subspecies. The regional distribution of *substriatus* is unclear; however, there are no confirmed records in the HBOC database from east of Cessnock. Also, a distribution map of Striated Pardalote subspecies, based mainly on museum specimens, shows *ornatus* as the only one occurring on the south-eastern seaboard (Schodde & Mason 1999).

That same map also indicates that *melanocephalus* does not occur at all in NSW or indeed south approximately of Brisbane (Schodde & Mason 1999). The coastal area from Brisbane to about Port Macquarie is indicated on the map as comprising intergrades of *ornatus* X *melanocephalus* and/or *substriatus* X *melanocephalus*.

The 1959 report of a pair of *melanocephalus* near Taree (Hobbs & Kavaney 1962; McGill 1966) therefore seems questionable. The pair was nesting, which should have offered opportunities for close-up views. However, because the observers apparently were limited to using binoculars (no mention was made of photographs being taken), they probably would not have been able to notice features such as the faint flecking below the eye which from photographs revealed the two 2017 birds to be intergrades. Similar comments apply to McGill's report of sightings of *melanocephalus* in the Manning Valley (McGill 1966).

CONCLUSIONS

Previous studies have indicated that the Black-headed Pardalote *P. s. melanocephalus* does not occur in the Hunter Region, but that intergrades with other Striated Pardalote subspecies may be present. Recent observations support this conclusion. Reports from the 1950s and 1960s of *melanocephalus* birds at Taree and elsewhere in the Manning Valley should be considered unconfirmed.

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Some recent breeding observations of threatened shorebird species in Port Stephens

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The first successful breeding record of Australian Pied Oystercatcher *Haematopus longirostris* in Port Stephens is documented and a second Beach Stone-curlew *Esacus magnirostris* breeding pair is reported, indicating a local expansion of the population.

INTRODUCTION

Port Stephens has been shown to be an important site for a number of migratory and resident shorebirds (Stuart 2011), including Australian Pied Oystercatcher *Haematopus longirostris* and Beach Stone-curlew *Esacus magnirostris*. It often hosts more than 1% of the world population of Australian Pied Oystercatcher making it internationally significant for this species (Hansen *et al.* 2016). There have also been regular Beach Stone-curlew *Esacus magnirostris* records since 2012. The Australian Pied Oystercatcher and Beach Stone-curlew are respectively classified as Endangered and Critically Endangered under the *Biodiversity Conservation Act 2016* (NSW).

The Beach Stone-curlew was first recorded in 2012 and now breeds in Port Stephens (Hunter Bird Observers Club 2015; Mo 2016). Australian Pied Oystercatcher nests with eggs have been previously reported (Orobillah Island, early 2000s, G. Little pers. comm.; Corrie Island, late 2000s, L. Penman pers. comm.) but it is not known if those attempts were successful. Stuart (2011) reported average counts of 110 Australian Pied Oystercatchers in summer and 142 birds in winter, from boat-based surveys of known roosting sites. This account describes recent successful breeding-related observations of Australian Pied Oystercatcher and the expansion of the Beach Stone-curlew breeding population within Port Stephens.

RESULTS

Australian Pied Oystercatcher

The first record of successful breeding-related activity was obtained in September 2017. A nest with two eggs was located on a sandspit at Winda Woppa (32º40'46"S, 152º08'44"E). One egg was subsequently predated; the remaining egg hatched 15 October 2017. The fledgling was observed on site with two adult birds in early December. A second pair established a nest with two eggs 160 m from the first site in late November. The nest was predated 19 days later by a Lace Monitor *Varanus varius.*

A nest with three eggs was located on Corrie Island (32º41'8"S, 152º08'15"E) in early October 2017. Another nest with one egg was reported in that month on Dowardee Island (32º42'8"S, 152º03'33"E) (T. Murray pers. comm.). The fate of these two nests is unknown. The nests on Winda Woppa and Corrie Island were located 5 and 5.5 km respectively west of the entrance to Port Stephens.

Beach Stone-curlew

In early October 2017, a Beach Stone-curlew nest with a single egg was located on a sandspit on the southwest end of Corrie Island (32º41'4"S, 152º 07'43"E). An adult bird was observed nearby. Inspection of photos revealed the egg was in the process of hatching. A subsequent visit five days later found the nest to be deserted. The fate of this chick is uncertain.

DISCUSSION

The recent record of successful breeding-related activity by an Australian Pied Oystercatcher pair follows much previous conjecture regarding the status of the species in Port Stephens. Although there are records as early as 1882 (W. Boles pers. comm.), the population was not considered in the review of the species by the NSW Scientific Committee (2008). The significance of the population has only recently been recognised

(Stuart 2011). Stuart quotes M. Newman (pers. comm.) stating there are very few suitable locations for pairs to establish breeding territories in Port Stephens. Stuart further suggested that most of the birds are from breeding sites elsewhere and that they only spend part of their life cycle in the port. The nearest known breeding site is on Worimi Conservation Lands, Newcastle Bight, around 10 km south. Here, Fraser & Lindsey (2018) recorded nine breeding territories, five to seven of which were occupied annually between 2014 and 2017. This number of breeding pairs cannot provide sufficient recruitment to sustain the population in Port Stephens (M. Newman pers. comm.).

Records of nesting-related activity by Australian Pied Oystercatchers now extend from Winda Woppa in the east of the port to Orobillah Island, 20 km west of the port entrance. The sites are on isolated shoreline or secluded islands within the port and habitat varies from shallow, marine tidal shoals to estuarine sand and mud flats. Preferred food is reported to be molluscs, worms and crabs (Marchant & Higgins 1993; Harrison 2009). These have been shown to be common along part of the shoreline (Stuart & Wooding 2018). Areas with similar characteristics are present west of Soldiers Point at Cromarty Bay, Taylors Beach, Fenninghams Island, Bull Island and Swan Bay. Future surveys of these locations during the peak breeding season (September/October) may identify further nest sites.

Beach Stone-curlew favour nest sites in secluded locations such as Corrie Island and Dowardee Island. These two sites are 7 km apart. The foraging ecology of Beach Stone-curlew is not well known but preferred food is reported to be crabs and other invertebrates (Marchant & Higgins 1993). Both groups of fauna are common along parts of the shorelines (Stuart & Wooding 2018).

CONCLUSIONS

Recent confirmation of successful breeding-related activity by Australian Pied Oystercatcher further enhances the importance of Port Stephens for this species. However, the contribution of breeding by resident birds is likely to be modest and the conclusion that the natal origin of most of the population lies outside the area is unchanged. The discovery of a second breeding pair of Beach Stone-curlew in the port further demonstrates the southern expansion of this species in NSW.

Despite the intensive residential development along the southern shores of Port Stephens, areas of suitable nesting habitat are present over a wide area of varying ecology. Undoubtedly the presence of several National Parks, Conservation Areas and Nature Reserves covering parts of the shoreline and islands plays a significant role in conserving this habitat.

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Latham's Snipe counts at Irrawang Swamp, NSW

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During the Austral summer Latham's Snipe *Gallinago hardwickii,* a cryptic species, is typically encountered in small numbers widely distributed across the landscape. However, these birds also congregate in large numbers on shallow lagoons with muddy edges. This note places on record historical counts of large congregations in the Hunter Region of NSW. The reasons for recent decreases, often exceeding 50%, in the size of these congregations are discussed. Variations in the seasonal occurrence of peak numbers of snipe at Irrawang Swamp near Raymond Terrace demonstrate the importance of water levels, with snipe attracted by muddy margins.

INTRODUCTION

Maddock (2008) in his overview of the degradation of the wetlands of the Hunter Region used Latham's Snipe as a case study. His publication is the most comprehensive compilation of historical information on the species for the Hunter Region of NSW. However, although referenced in Cooper *et al.* (2016) this publication is not widely known. In view of the current national interest in Latham's Snipe (Hansen 2017; https://lathamssnipeproject.wordpress.com/) and concerns about the species' trajectory, the purpose of this note is to make the data more readily available and comment on its current implications.

Latham's Snipe breeds in Japan and migrates to Australia during the Austral summer, where it is widely distributed in freshwater wetlands. While often encountered in small numbers dispersed across wetlands and marshy areas Latham's Snipe sometimes congregates in large numbers at favoured wetlands (Maddock 2008). Irrawang Swamp 32.724°S 151.748°E, a wetland of approximately 10 ha area, which is located between Raymond Terrace and Seaham in NSW is an example. Arguably the Irrawang data set, which is the focus of this note, is the most comprehensive source of information on the manner in which snipe congregate at suitable wetlands in the Hunter Estuary.

METHODS

It is difficult to conduct comprehensive counts of Latham's Snipe, a cryptic species prone to hiding in dense vegetation and remaining invisible unless flushed. However, under favourable conditions, when water levels fall and extensive areas of mud are exposed, Irrawang Swamp provides good opportunities to obtain accurate counts of the number of snipe present.

Surveys were conducted by walking the circumference of the swamp and noting all birds on the mudflat and flushed from peripheral vegetation. Surveys were first conducted in 1985/86 and have been repeated intermittently since that time. In years when monitoring occurred, surveys were conducted between August and March, often at approximately monthly intervals. A number of partial counts, made from Newline Road adjacent to the swamp, were excluded from the analysis.

A number of observers were involved including Wilma Barden and Geoff Winning in 1986/86, Anne Heinrich in 1986/87, Max Maddock from 2002/03 to 2006/07 and Bruce Watts in 2017/18.

RESULTS

Peak annual counts of Latham's Snipe at Irrawang Swamp for the 31-year period 1986/87 to 2017/18 are shown in **Table 1**. In six summers multiple surveys were conducted (*n* ranged from 3 to 7). The month in which peak numbers occurred varied from October to March, although most of the peak counts were made in November. Although comprehensive records of water levels are not available, conditions ranged from full to completely dry with no snipe present. Counts involving high numbers of snipe were characterised by conditions involving extensive areas of drying mud.

Table 1. Summary of Latham's Snipe survey results for Irrawang Swamp

Period	Peak numbers	Peak Month	Number of surveys
1986/87	32	Jan	5
1987/88	61	Nov	6
2001/02	35		
2002/03	40	Mar	3
2004/05	73	Nov	7
2005/06	24	Nov	6
2007/08	4		
2017/18	21	Oct	7

In **Table 2** (copied from Maddock 2008) the peak counts at Irrawang Swamp are compared with those at other wetlands where large numbers of Latham's Snipe occurred historically. These peak counts demonstrate the manner in which key wetlands in the Hunter Valley can hold very large numbers of Latham's Snipe.

Table 2. Records of Latham's Snipe maximum counts Lower Hunter key wetlands 1978-2007/08

Year	Lorna St (NWR/MSw)	Wetlands Centre	Cedar Hill (PNR)	Seaham Swamp	Irrawang Swamp
1978				105	
1984	100+	12+			
1985/86	104	9	44	30	32
1987/88	55				61
1988/89	47	5	115		
1996/97	20 (MSw)				
1997/98			475		
1998/99			230		
1999/00			115	1	
2000/01			66	11	
2001/02			35	0	
2002/03			7	0	40
2003/04	30	9	35	0	
2004/05			45	0	73
2005/06	34		66	1	24
2006/07			97	0	23
2007/08			5	0	4

Derived from Anon (1984), Crawford (2008), Gilligan (1980), Barden (1988, 1989), Barden and Winning (1986), Maddock (unpublished data), Stuart (1994-2006).
Abbreviations: NWR Newcastle Wetlands Reserve, MSw Market Swamp, PNR Pambalong Nature Reserve.

DISCUSSION

During the Austral summer lagoons in the lower Hunter Valley progressively dry out, but can be rapidly filled by storms resulting in highly variable annual conditions. At Irrawang Swamp the occurrence of snipe was favoured by drying conditions which resulted in extensive muddy margins. It is a shallow wetland with many trees and small islands of vegetation providing opportunities for snipe to loaf when not actively foraging, a feature which facilitates counting the numbers present. Indeed the functions of these lagoons may be to provide diurnal shelter, because the Latham's Snipe are crepuscular and nocturnal feeders and may disperse to forage in other areas at dusk (Newman 2008; B. Hansen pers. comm.). The variable conditions at the lagoons result in fluctuations in the timing of peak numbers, a result which, as discussed later, has implications for snipe monitoring programs.

In the area surrounding Irrawang Swamp and the other locations mentioned in **Table 2** there are many small dams, ephemeral marshy areas and extensive flood plains which support snipe dispersed in small numbers. Under dry conditions these wetlands may become unsuitable for Latham's Snipe resulting in the progressive movement and concentration of birds at the larger wetlands as water levels drop and muddy margins are exposed. Conversely, following storms involving torrential rain, the lagoons rapidly fill to capacity, the muddy margins disappear and snipe must seek other foraging options. At this stage they disperse and exploit ephemeral water meadows and marshy areas on the surrounding flood plains of the Lower Hunter Valley (Newman 2008). In extreme drought conditions, when even the larger lagoons such as Irrawang Swamp are dry, the snipe must seek other opportunities which include foraging in dry paddocks, where they probe for spiders and other insects (Newman 2008).

The peak counts listed in **Table 2** demonstrate the manner in which key wetlands in the Hunter Valley can hold very large numbers of Latham's Snipe. Indeed the 475 recorded at Pambalong Nature Reserve in 1997/98 in December is one of only two sites in the East Asian-Australasian Flyway where more than 1% of the population of the species has been recorded in a count (Bamford *et al*. 2008). The other was a count of 430 at the Powling Street Wetlands, at Port Fairy in Victoria in 2010 (B. Hansen pers. comm.).

The results in **Table 2**, particularly those conducted at Cedar Hill Drive, now Pambalong Nature Reserve, provide clear evidence of a decrease at that location. During the past decade the highest count at any wetland in the Hunter Region was 53 on 15 Jan 2015 at Wallsend Wetlands (Stuart 2017). While this contemporary evidence suggests that the species has decreased, at least within the Hunter Region (Cooper *et al*. 2016), it may be prudent to consider the possibility that these decreases are the consequence of the degradation of habitat with invasive aquatic vegetation encroaching on areas of open water (Maddock 2008) at locations where snipe used to congregate in large numbers as water levels fell.

The variation in the timing of peak counts at Irrawang Swamp (**Table 1**), which is driven by fluctuations in water levels, exacerbates the difficulty in using such counts to estimate regional population levels and their trends. For instance, the Pambalong Nature Reserve counts were conducted during December, and may not have represented the peak population for the summer if water levels were unsuitable.

CONCLUSIONS

Contemporary records of Latham's Snipe at wetlands in the Lower Hunter Valley where they congregate are considerably lower (typically < 50%) than in previous decades when numbers in the range 100 to 500 were recorded at several locations (**Table 2**). Surveys at Irrawang Swamp demonstrate that snipe are most numerous when shallow lagoons are drying out and the timing of optimal conditions varies between years.

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