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Factors affecting shorebird hatching outcomes at the Ashley River/Rakahuri-Saltwater Creek estuary, New Zealand

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Abstract: Shorebird nest outcomes can be affected by factors such as predation, human disturbance, and habitat characteristics. Over two breeding seasons between 2022–2024, we monitored the hatching success of banded dotterels (*Anarhynchus bicinctus*), southern black-backed gulls (SBBGs) (*Larus dominicanus*), black-fronted terns (*Chlidonias albostriatus*), pied stilts (*Himantopus leucocephalus*), and variable oystercatchers (*Haematopus unicolor*) at the Ashley River estuary, New Zealand, and compared these values to those in the literature. We also recorded habitat variables at the nest sites of the two species with the largest sample sizes: banded dotterels and SBBGs. Hatching success was lowest for black-fronted terns and highest for SBBGs. Overall, failure was predominantly due to predation and flooding. SBBG hatching success was unrelated to the measured nest site variables but may have been influenced by seasonal changes, with earlier nests appearing more successful. Banded dotterel nests that were closer to water appeared to be more successful, as did nests in the first year of the study. Cats (*Felis catus*) were recorded depredating banded dotterel nests, highlighting the importance of monitoring and controlling invasive species to protect native birds in New Zealand's estuaries.

Keywords: hatching success, cat predation, invasive predators, black-fronted tern, southern black-backed gull, banded dotterel, pied stilt, variable oystercatcher

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INTRODUCTION

Shorebirds worldwide are threatened by many factors, including habitat loss, predation, climate change, and human disturbance (Dowding & Murphy 2001; Sutherland *et al.* 2012; Iwamura *et al.* 2013). These factors can affect shorebird hatching success (Dowling & Weston 1999; O'Connell & Beck 2003) and the availability of suitable nesting habitat for different species (Sutherland *et al.* 2012; von Holle *et al.* 2019).

In New Zealand, the greatest threat to native shorebirds comes from introduced mammalian predators such as rats (*Rattus* spp.), cats (*Felis catus*), mustelids (*Mustela* spp.), and European hedgehogs (*Erinaceus europaeus*) (Dowding & Murphy 2001). Because New Zealand's native birds evolved without mammalian predators, their nest defence adaptations, such as camouflage and parental displays, provide limited defence against these predators (Dowding & Murphy 2001). Other threats to New Zealand shorebirds include native avian predators, habitat loss, and human disturbance (Dowding & Murphy 2001; Steffens *et al.* 2012). For shorebirds breeding on braided rivers, the main cause of hatching failure is often predation from introduced

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mammals (Sanders & Maloney 2002; Cruz *et al.* 2013) but also in some cases from southern black-backed gulls | karoro (hereafter SBBGs; *Larus dominicanus*) (McClellan 2009; Schlesselmann *et al.* 2018).

Compared to braided rivers, there has been less focus on measuring hatching success in New Zealand coastal habitats (Kearvell 2011). On the Kaikoura Peninsula, 72% of variable oystercatcher | tōrea pango (*Haematopus unicolor*) eggs hatched between 1999–2006, but the causes of failure were not identified (Rowe 2008). At Matakana Island, ~35% of northern New Zealand dotterel | tūturiwhatu (*Anarhynchus obscurus aquilonius*) nests hatched between 1992–2000 (Wills *et al.* 2003). Failure in this species was due to predation and human disturbance (Wills *et al.* 2003). In a 1993 study at the Ashley River estuary, 96.3% of banded dotterel | pohowera (*Anarhynchus bicinctus*) nests failed due to flooding, burial, and crushing by vehicles (Kearvell 2011). SBBG egg loss in Wellington 1963–1965 varied condsiderably between colonies, ranging from 6 to 100% (Fordham 1966). Causes of egg loss were due to ferret (*Mustela furo*) predation, addling, disappearance, and flooding (Fordham 1966).

At a local scale, nest microhabitat can affect hatching success (Hong & Higashi 2008; Que *et al.* 2015). For example, substrate type and vegetation cover can protect nests from flooding (Hong & Higashi 2008; Que *et al.* 2015), vegetation cover can reduce heat stress (García-Borboroglu & Yorio 2004), and both substrate and vegetation play a role in camouflaging nests against predators (García-Borboroglu & Yorio 2004; Colwell *et al.* 2011). However, a relationship between habitat variables and hatching success is not always observed (Mabee & Estelle 2000; Miller *et al.* 2014), and no studies to our knowledge have assessed the role of microhabitat on hatching success for New Zealand shorebirds.

As coastal habitats could present different threats to those in braided rivers, understanding the threats facing New Zealand's shorebirds in these environments is vital (Kearvell 2011). Studying the role of microhabitat variables on hatching success could also identify the habitat features that promote the nest success of threatened species. Therefore, our aims were threefold: 1) to measure hatching success and identify the causes of failure for banded dotterels, SBBGs, black-fronted terns | tarapirohe (*Chlidonias albostriatus*), pied stilts | poaka (*Himantopus leucocephalus*), and variable oystercatchers at the Ashley River/Rakahuri-Saltwater Creek Estuary, New Zealand; 2) to compare hatching success rates to those recorded in other studies; and, 3) to test whether microhabitat features affect the hatching success of banded dotterels and SBBGs.

METHODS

Study site

The Ashley River/Rakahuri-Saltwater Creek estuary or Te Akaaka (-43.2780, 172.7211) (Fig. 1), hereafter the Ashley estuary, is located ~30 km north of Ōtautahi/Christchurch in Aotearoa/New Zealand (Kearvell 2011). The Ashley estuary is separated from the Pacific Ocean by a spit, with openings to the sea varying in number and location over time. Two openings were present during the 2022/23 breeding season (Fig. 1), but the southern opening closed in early 2023, extending the length of the eastern spit. Freshwater input to the Ashley estuary comes from the Ashley River/Rakahuri, Saltwater Creek, and Taranaki Stream (Bolton-Ritchie 2016; Fig. 1). The land surrounding the estuary is dominated by agricultural fields to the north and west, and by the settlement of Waikuku Beach to the south.

Figure 1. Aerial view of the Ashley River/Rakahuri-Saltwater Creek estuary, with the two openings to the estuary present in the 2022/23 breeding season, the sources of freshwater input, and the spit labelled. Image from drone photos from the Ashley-Rakahuri Rivercare Group in 2022 (G. Davey, *unpubl. data*), overlayed on a Google Satellite (2016) image.

Nest monitoring

We located nests of five shorebird species (banded dotterels, SBBGs, black-fronted terns, pied stilts, and variable oystercatchers) at the Ashley estuary over two successive breeding seasons between 2022 and 2024. Because these species mainly breed between August and February (Cruz *et al.* 2013; Schlesselmann *et al.* 2017), monitoring occurred between September 2022 and February 2023 and August to December 2023. We monitored SBBGs in five colonies throughout the estuary in 2022/23. However, we excluded them from monitoring during 2023/24 because of a culling effort carried out for conservation purposes at the Ashley estuary that year (Greg Stanley *pers. comm.* to EG). From 1 to 22 Sep 2023, we had to modify monitoring following a wastewater spill at the estuary, to avoid direct contact with water (ECan 2023). Some areas of the estuary were unable to be monitored during this period, so it is possible that some early nests, particularly of banded dotterel, may have been missed.

The Ashley estuary was visited two to four times per week to locate and monitor nests, depending on the weather, tidal cycle, and height of the Ashley River. We located nests by walking through the study site, aided by observations of parental behaviour and, for colonial nesting species, birds gathering in a local area. We recorded the location, species, and number of eggs in each nest in QField (QField. https://qfield.org/), a phone app linked to QGIS (QGIS Geographic Information System. https://www.qgis.org/en/site/), an open-source geographic information system programme.

Once a nest was located, we monitored it both from a distance (with binoculars) and by approaching nests. We limited time spent close to each nest as much as possible to reduce disturbance. We monitored nests until they had successfully hatched, with at least one egg having hatched (Rebergen *et al.* 1998), or until they were either empty or any remaining eggs were deemed unable to hatch (i.e., deserted). We then recorded the hatching outcome using the criteria adapted from Cowell *et al.* (2011) and Schlesselmann *et al.* (2018) (Table 1).

Table 1. Nest hatching outcomes and their criteria, adapted from Colwell *et al.* (2011) and Schlesselmann *et al.* (2018) and observations from this study.

Nest outcome	Criteria
Failed – burial	Eggs not visible and sand built up at nest site. Can be supported by knowledge of strong winds.
Failed – desertion	Eggs cold and/or known to have been left unattended.
Failed – flooding	Water in nest, or signs of water having previously covered the nest. Eggs may be discoloured or absent.
Failed – predation	Eggs present but damaged, or eggshell fragments present at nest site, or eggs gone but too early for them to have hatched (if hatching date known).
Succeeded	At least one chick present in or near the nest.
Unknown	Unclear whether nest hatched or failed, e.g., no sign of chicks, but bird faeces present at nest site.

During the 2023/24 breeding season, we placed trail cameras (Nextech 1080P, Eastern Creek, NSW, Australia) at five banded dotterel, two variable oystercatcher, and two pied stilt nests to determine causes of nest failure. We selected nest sites at random from those with a suitable location to place a camera. We anchored cameras to a rock or piece of driftwood 1-2 m away from a nest, facing north or south to reduce glare. All work was approved by the University of Canterbury's Animal Ethics Committee (AEC Application 2023/10R).

Nest microhabitat measurements

We recorded habitat measurements at SBBG and banded dotterel nests, as they had the largest sample sizes. To measure the nest microhabitat, we centred a 1 m^2 quadrat over the nest site as recommended by Nguyen *et al.* (2003) and Miller *et al.* (2014). Within the quadrat, we visually estimated the percentages of silt/sand (< 2 mm), small pebbles (2-10 mm), gravel (11-64 mm), cobbles (65-256 mm), vegetation, wood, and "other" substrate (Stucker *et al.* 2013). Wood was included when the substrate beneath it was not visible and when it appeared to be sufficiently anchored into the ground to prevent it being moved by the wind. "Other" substrate did not fall into any other categories; for example, one SBBG nest was partly constructed on a corrugated metal sheet. In some cases, nest material was no longer present (e.g., had been blown away by the wind), and so all the substrate within the quadrat was visible. Where nest material was still present, we visually estimated the substrate type beneath it based on the surrounding substrate. Observations of nests where the nest material was no longer present indicated that this provided accurate estimates.

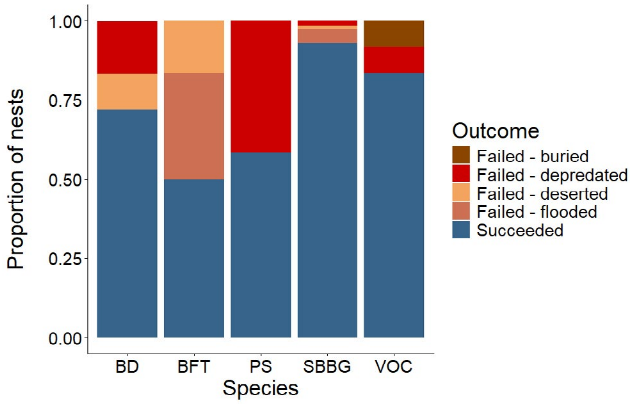

Figure 2. Causes of hatching failure at the Ashley estuary across 2022–2024 (except for southern black-backed gulls (SBBG), where data comes from 2022/23). Shown is the proportion of nests of each species that successfully hatched and the proportion of nests that failed (and cause of failure). Succeeded = at least one egg hatched, failed = no eggs hatched. See Table 2 for sample sizes. Other species codes: BD = banded dotterel, BFT = black-fronted tern, PS = pied stilt, VOC = variable oystercatcher.

For each nest, we also measured the distance to: 1) the nearest vegetation ≥ 1 m high, 2) the nearest neighbouring nest of any species, and 3) the nearest neighbouring SBBG nest. As vegetation on the study site < 1 m tall was predominantly thin and patchy, we excluded vegetation shorter than 1 m to ensure only plants that provided thicker cover were included. These habitat features were measured in the field using a tape measure if distances were ≤ 20 m. Where the distance was > 20 m, measurements were taken in QGIS using the "measure line" function because this was more accurate. We measured the distance of nests to the nearest open water with the QGIS "measure line" function, using the high tide line visible in the drone images from September 2022 (G. Davey, *unpubl. data*) in QGIS to determine the location of open water.

To reduce disturbance, we recorded habitat features once a nest, and any close neighbouring nests, were no longer active. Therefore, we sometimes could not measure habitat features for several months. While this could have affected some measures, this has not previously been identified as a problem in studies of shorebird nest sites (Colwell *et al.* 2011). Based on knowledge of the study site, we would only expect major changes to habitat features if there was a large flood before the variables were measured, which did not occur during the timeframe of our study.

Hatching success analysis

Apparent hatching success was calculated by dividing the number of nests that hatched at least one chick by the total number of nests of that species. We plotted this in R version 4.4.1 (R Core Team 2024) using the ggplot2 package (Wickham 2016). We then calculated actual hatching success in program MARK (v 10.1; Program MARK. http://www.phidot.org/software/mark) assuming a constant daily survival rate (DSR) over time, an approach equivalent to the Mayfield method (Mayfield 1961, 1975; Rotella 2021). This requires knowing the day a nest was found, the last day it was observed active, the last day it was checked, and its fate (succeeded or failed) (Rotella 2021). We then raised the DSR and standard error (SE) values from MARK to the power of the known incubation period for each species obtained from NZ Birds Online (NZ Birds Online. https://www.nzbirdsonline.org.nz/) to provide the likelihood that a given nest would hatch (Cruz *et al.* 2013; Rotella 2021).

For species where the hatching date spans multiple days, we calculated the likelihood of hatching using both the shortest and longest average incubation periods. For example, the incubation period of a banded dotterel ranges from 25–28 days (Pierce 2013), and so we raised the banded dotterel DSR to the power of 25 and also to the power of 28, with the likelihood of hatching being somewhere within the range of the two values.

In total, we located 18 banded dotterel nests, 254 SBBG nests, six black-fronted tern nests, 23 pied stilt nests, and 15 variable oystercatcher nests. However, nests that could not be followed to determine a final outcome were excluded from hatching success analyses (numbers included in analyses in Table 2). Nests that were first found once chicks were already present were also excluded from the actual hatching success analysis.

Hatching success compared to the literature

We conducted a literature review to determine how our hatching success values compared to those for the same species elsewhere in New Zealand. Scopus was used to search for all relevant papers, from which hatching success estimates were extracted and classified according to whether these were apparent or actual values (e.g., the Mayfield method or other modelling approaches). This allowed us to select the equivalent hatching success value for comparison. We then calculated the increase or decrease in hatching success from a previous study to this study using the formula: $[(V_1 - V_2) / (V_2)] * 100$, where V_1 is the hatching success recorded at the Ashley estuary in this study and V_2 is the hatching success value from a previous study.

Microhabitat analysis

Eighteen banded dotterel nests and 165 SBBG nests with known outcomes were included in the microhabitat analysis. To avoid multicollinearity, we calculated correlations between variables using Spearman-rank correlations (because the data did not approximate a normal distribution even after transformation) and selected non-correlated variables for the analysis. For SBBGs, we selected distance to water and to the nearest neighbouring SBBG nest, and the month a nest was initiated. Month was split into October, November, and December/January (which were combined because few nests were initiated in either month). Distance to the nearest SBBG nest was chosen instead of distance to the nearest neighbour because conspecifics were the nearest neighbour at all but two nests, and because SBBGs are known nest predators of New Zealand shorebirds (Wills *et al.* 2003; Schlesselmann *et al.* 2018). For banded dotterels, we selected the percentage of silt/sand, distance to water, distance to the nearest patch of vegetation, and breeding season (2022/23 or 2023/24). It was not possible to analyse the microhabitat data using binomial generalised linear models (GLMs), due to a lack of natural variability in some aspects of the data. Instead we used a descriptive approach by comparing the mean +/- SE of the selected variables at hatched and failed nests, with box-and-whisker plots made in R version 4.4.1 (R Core Team 2024) using the ggplot2 package (Wickham 2016). The package ggbreak (v 0.1.2; Xu *et al.* 2021) was used to add axis breaks to plots with large outliers.

RESULTS

Hatching success at Ashley estuary

Across the 2022–2024 breeding seasons, apparent hatching success varied between species, being lowest for black-fronted terns and highest for variable oystercatchers (Table 2; Fig. 2). Banded dotterel hatching failure was caused by three cases of predation (two documented as cat predation; Fig. 3) and two of nest desertion (one of which had a cat visit twice before desertion; Fig. 2). Black-fronted tern nest failure occurred from two cases of flooding and one of desertion (Fig. 2). All five cases of pied stilt nest failure were caused by predation, though predator identities were not determined (Fig. 2). Variable oystercatcher nest failure was caused by one instance of nest burial by sand and one of predation (Fig. 2). SBBG hatching success in 2022/23 was the highest of any species (Table 2). Failure occurred due to nine cases of flooding, three of predation, and two of desertion (Fig. 2).

Actual success calculated using program MARK provided the likelihood of a nest of a given species surviving to hatch. This varied between species but was highest for SBBG and lowest for black-fronted terns (Table 2). Apparent hatching success values were greater than the actual hatching success for all species (Table 2). However, the difference between these values varied. It was greatest for black-fronted terns, where the apparent hatching success was 45.8% greater than the actual hatching success, and lowest for SBBGs, where the difference was only 9.5%.

Hatching success compared to previous studies

The differences between hatching success in this study and previous studies varied markedly (Table 3). For pied stilts and variable oystercatchers, our values were greater than

Table 2. The apparent and actual hatching success of shorebird nests at the Ashley estuary during 2022–2024 (except for black-backed gulls, where data comes from 2022/23 only). For actual nest success, the likelihood of hatching is given as a range between the shortest and longest average incubation periods for species with average hatching dates that span multiple days. N = the number of nests used for each type of analysis, N with cameras = number of nests with trail cameras, DSR = daily survival rate, SE = standard error.

		Apparent hatching success		Actual hatching success		
Species	**N with cameras**	**N**	**Apparent hatching success (%)**	**N**	**DSR +/- SE**	**% chance of hatching (SE range in brackets)**
Banded dotterel	5	18	72.2	17	0.981 +/- 0.009	57.7 - 61.2 (45.1 - 76.2)
Black-backed gull	0	196	92.9	156	0.993 +/- 0.002	84.0 – 85.7 (80.5 – 88.9)
Black-fronted tern	0	6	50.0	5	0.958 +/- 0.024	34.3 (18.3 – 63.4)
Pied stilt	2	12	58.3	12	0.969 +/- 0.014	45.9 (32.3 – 65.0)
Variable oystercatcher	2	12	83.3	11	0.989 +/- 0.008	72.8 (58.2 – 91.1)

Figure 3. A cat preying on a banded dotterel nest at the Ashley estuary (Photograph: Eleanor Gunby).

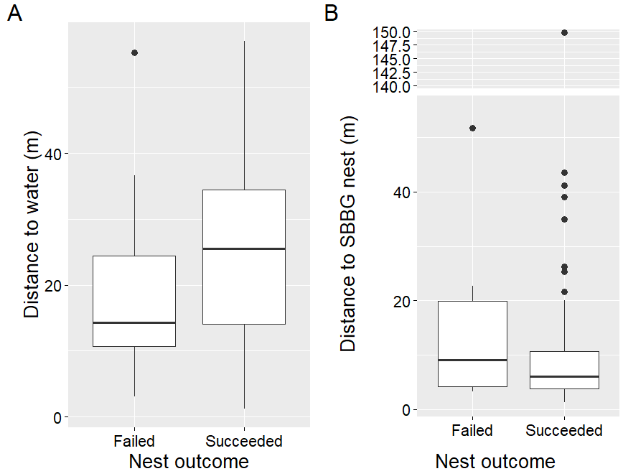

Figure 4. The mean distance to water (a) and distance to the nearest neighbouring SBBG nest (b) at SBBG nests that failed (n = 11) and successfully hatched (n = 154) at the Ashley estuary during 2022/2023. Thick line = median, upper and lower thin lines of box = quartiles, black dots = outliers.

reported previously. For banded dotterels and black-fronted terns, rates of hatching success reported in previous studies were both higher and lower than we observed, confirming a high degree of variability in hatching outcomes for these species.

Microhabitat features and hatching success

SBBGs typically nested in sites dominated by silt/sand and vegetation, with the average substrate surrounding a nest composed of 68% silt/sand, 29% vegetation, and 3% wood. No SBBG nest sites contained pebbles, small gravel, or cobbles. Neither distance to water (Fig. 4a) nor distance to the nearest neighbouring SBBG nest (Fig. 4b) appeared to differ between nests that hatched and failed. However, there appeared to be a possible decline in hatching success depending on the month in which a nest was initiated. Of the nests initiated in October, 100% hatched (53/53), compared to 93% initiated in November (77/83) and 83% in December/January (24/29).

Banded dotterel nest sites contained a variety of substrates: on average, 54% silt/sand, 20% gravel, 9% cobbles, 9% vegetation, 7% small pebbles, and 1% wood. The percentage of silt/sand (Fig. 5a) and the distance to the nearest patch of vegetation (Fig. 5b) were similar between nests that hatched and failed. However, nests that hatched appeared to be closer to water on average than nests that failed (Fig. 5c). There also appeared to be a possible difference in hatching success between years, with 90% of nests in 2022/23 (n = 10) hatching compared to 50% in 2023/24 (n = 8).

DISCUSSION

Hatching success

Shorebird hatching success and the causes contributing to hatching failure are highly variable. Banded dotterel hatching success in our study was intermediate between previous findings (Rebergen *et al.* 1998; Kearvell 2011; Cruz *et al.* 2013; Table 3). It was higher than that recorded at the

Table 3. Hatching success estimates of shorebirds from the literature, and the differences from hatching success observed in this study. A minus sign indicates that hatching success in this study was lower than for that species in a previous study.

Species	Authors	Location	N	Calculation method	Hatching success (%)	Hatching success in this study (%)	Increase or decrease (%)
Banded dotterel	Cruz *et al.* (2013)	Tasman River	161	Actual	74	57.7 – 61.2	-17.3 to -22.0
	Kearvell (2011)	Ashley estuary	33	Actual	3.3	57.7 – 61.2	1649.4 to 1755.5
	Rebergen *et al.* (1998)	Ahuriri River	50	Actual	74	57.7 – 61.2	-17.3 to -22.0
		Ohau River	50	Actual	32	57.7 – 61.2	80.4 to 91.3
		Tekapo River	53	Actual	40	57.7 – 61.2	44.3 to 53.1
	Sanders & Maloney (2002)*	Upper Waitaki Basin	110	Apparent	57.3	72.2	26.1
	Smith (2006)	Ahuriri River	14	Actual	51.8	57.7 – 61.2	11.6 to 18.3
Black-fronted tern	Bell (2017)*	Upper Clarence River	710	Apparent	39.4	50.0	26.9
		Acheron River	800	Apparent	45.8	50.0	9.2
	Cruz *et al.* (2013)	Tasman River	243	Actual	40	34.3	-14.2
	Keedwell (2005)	Ohau River	1022	Apparent	50.2	50.0	-0.4
	Sanders & Maloney (2002)*	Upper Waitaki Basin	35	Apparent	51.4	50.0	-2.8
	Schlesselmann *et al.* (2018)	Waitaki River	266	Actual	50.5-56.4	34.3	-32.0 to -39.1
Pied stilt	Pierce (1986)	Cass River	125	Actual	34.9	45.9	31.6
Variable oystercatcher	Michaux (2013)	Long Bay Regional Park and Okura Estuary	7	Apparent	57.1	83.3	45.9

*Hatching success values were extracted from data provided 652 in these papers.

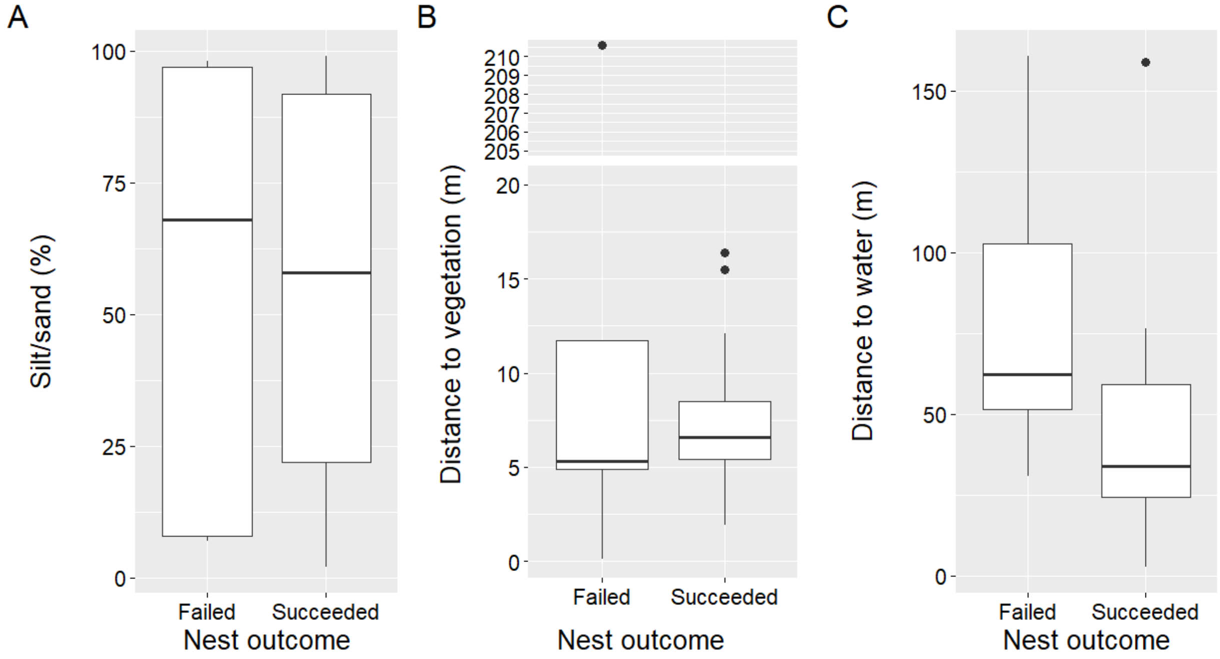

Figure 5. The mean percentage of silt/sand (a), distance to the nearest patch of vegetation (b), and distance to water (c) at banded dotterel nests that failed (n = 5) and successfully hatched (n = 13) at the Ashley estuary between 2022 and 2024. Thick line = median, upper and lower thin lines of box = quartiles, black dots = outliers.

Ashley estuary in 1993 (Table 3); however, the three most common causes of failure in 1993 (flooded, crushed, and buried by sand) (Kearvell 2011) were not recorded by us. Instead, banded dotterel nests failed due to predation and desertion: we found that 16% of nests were depredated, compared to 12% in 1993, while 11% were deserted, which was not recorded as a cause of failure in 1993 (Kearvell 2011). All predation occurred in the 2023/24 breeding season, with two of the three instances caused by cats. While the causes of desertion were not determined, one nest was visited by cats twice before being deserted shortly after the second visit. This suggests that cats currently may be an important cause of banded dotterel hatching failure at the Ashley estuary.

The differences in banded dotterel hatching success and causes of failure among studies may be influenced by several factors, one of which is habitat. Banded dotterel hatching success can be higher on islands than the mainland (Rebergen *et al.* 1998). Islands provide some protection from mammalian predators; for example, banded dotterel hatching success on the Tasman River decreased at lower flows, likely due to increased predator access to islands in the river channel (Cruz *et al.* 2013). At the Ashley estuary, few islands were present and instead the nests in our study were located along the edges of mudflats or on raised gravel areas alongside waterways, which may have facilitated predator movement.

Unlike Kearvell (2011), who only studied banded dotterel nests on the eastern spit, we monitored nests across the entirety of the Ashley estuary. We found fewer nests than were present in 1993 (Kearvell 2011), and only one was located on the spit. Banded dotterels may now be nesting in different areas of the estuary due to factors such as changes in physical habitat or human disturbance. It is also possible that nests in 1993 were located sooner after initiation and so captured more instances of hatching failure from causes such as flooding. While this seems unlikely, given that both studies began at a similar time in the breeding season, we could not monitor some areas of the estuary for approximately three weeks in September 2023 following a wastewater spill (ECan 2023). This may have caused us to miss some banded dotterel nests early in the breeding season. It is also possible that the difference between our and Kearvell's (2011) results is due to a local or regional decline in banded dotterel numbers and subsequent contraction in their range within the local area.

Black-fronted tern hatching success was similar to that found in several previous studies (Keedwell 2005; Cruz *et al.* 2013; Table 3). The main causes of black-fronted tern failure have been attributed to predation and desertion (Keedwell 2005; Bell 2017), while we found failure was caused by flooding and desertion. Flooding, which has led to hatching failure elsewhere (Bell 2017; Schlesselmann *et al.* 2018), caused the failure of two black-fronted tern nests in 2022/23. The effect of flooding on hatching success is expected to vary between locations and years; for example, for snowy plovers (*Charadrius nivosus*) in the U.S.A. it depended on the amount of rainfall and the elevation of nesting substrate (Sexson & Farley 2012).

Like black-fronted terns, SBBGs nest in colonies, meaning that hatching failure can affect many nests simultaneously. Colonial breeding can have benefits (e.g., mobbing of predators or decreasing per capita likelihood of a nest being depredated); however, it can also have costs (e.g., increased conspicuousness or conspecific aggression) (Hernández-Matías *et al.* 2003). For example, multiple nests are likely to be affected by a single predator (O'Donnell *et al.* 2010). We found that predation occurred at two of five SBBG colonies, while flooding occurred at three, demonstrating variation in the causes of nest failure among the different colonies.

Variable oystercatcher hatching failure was caused by one instance each of burial and predation. Sand was observed blowing on the eastern spit, particularly in the most exposed areas, during fresh to strong winds (>30 km/hr). This could have buried the nest, similar to previously recorded burials of banded dotterel nests on the spit (Kearvell 2011). In the single documented nest predation, the identity of the predator could not to be determined.

The hatching success of pied stilts was higher than that recorded on the Cass River, with predation being the main cause of failure on the Cass River (Pierce 1986) and the sole cause of failure we recorded. However, the identity of the predators involved were not determined. Using trail cameras at a greater number of pied stilt nests would be required to identify their main nest predators.

Alongside showing specific causes of hatching failure, our trail cameras showed evidence of 13 additional disturbance events for incubating birds. Ten of these were caused by photographers and pedestrians, including one with a dog, and one each by a cat, a SBBG, and a vehicle. Seven disturbances occurred at a banded dotterel nest in the eastern area of the estuary, while five were at a variable oystercatcher nest on the spit (Fig. 1). This highlights the high levels of anthropogenic disturbance, which can lead to decreased hatching success (Que *et al.* 2015) and nest abandonment (Toland 1999), in specific areas of the Ashley estuary.

Habitat variables and SBBG hatching success

SBBG hatching success appeared to be unrelated to the distance to the nearest neighbouring SBBG nest or the nearest patch of open water. While this could suggest their hatching success is not affected by habitat features, there may have been sufficient good quality habitat for the nesting colonies. SBBG appeared to favour nesting on vegetation and silt/sand and avoided stony substrates. In Argentina, SBBG hatching success was higher for nests on softer substrates (clay and silt) and in highly vegetated areas (García-Borboroglu & Yorio 2004), suggesting a potential reason for this habitat preference. SBBGs in Wellington also seemed to prefer nesting near vegetation when it was present, likely to provide shelter for chicks (Fordham 1966).

SBBG nests that were initiated earlier in the breeding season appeared to have higher hatching success. Similarly, SBBG egg losses in Wellington were lower earlier in the breeding season (Fordham 1966). Such temporal variation has been recorded in other species (Maxson *et al.* 2007; Grant & Shaffer 2012), with potential explanations including increased predator numbers (Grant *et al.* 2005) or decreased food availability (Siikamäki 1998) later in the season.

Habitat variables and banded dotterel hatching success

Similarly to previous studies (Bomford 1988; Norbury & Heyward 2008), banded dotterels appeared to prefer nest sites that contained a variety of substrates but were predominantly comprised of silt/sand and gravel. However, there was no apparent relationship between banded dotterel nest success and the percentage of silt/sand surrounding the nest site. This may have occurred if there was enough silty/sandy habitat available, or if dotterels that could not find such habitat moved elsewhere to nest.

Banded dotterel nests that successfully hatched appeared to be located closer to water on average than unsuccessful nests. This differs from research on braided rivers, where the distance of banded dotterel nests to water did not affect their hatching success (Rebergen *et al.* 1998). One benefit of nesting close to water is that it may be useful in deterring predators. Cats may avoid mudflats (Molsher *et al.* 2005), and so nests located near mud close to the water may have reduced likelihoods of cat predation, which was the main cause of nest failure in our study.

Banded dotterel hatching success appeared lower in 2023/24 than the previous breeding season, largely due to cat predation, which was not observed during 2022/23. It is possible that new cats had arrived in the area, or that changes in the physical habitat of the estuary made it more accessible to predators. While we did not measure changes in habitat between the two years of our study, vegetation in the southwestern area of the estuary (where most banded dotterels nested) appeared to have increased in height and extent during the 2023/24 season.

Management implications

Our research provides an indication of the causes and rates of shorebird hatching failure in a New Zealand estuary. However, estuaries are dynamic environments and as our results demonstrate, shorebird hatching success can vary between breeding seasons and between different studies. Research over longer time frames (e.g., five to 10 years) should be encouraged to promote a better understanding of changes in the hatching success of shorebird species and for estimating fledging success, which is not always feasible in shorter-term studies. Given that we found apparent hatching success values were always higher than actual nest success, it is also important that future studies include calculations of actual hatching succes. Such a skew in apparent hatching success is expected because of differences in the likelihood of researchers locating nests that succeed or fail (Rotella 2021).

Predation was the main cause of hatching failure for pied stilts and banded dotterels. While predators were not always identified, cat predation appeared to be a particular threat to banded dotterels in this area. Residents could be encouraged to keep their cats indoors, particularly at night, which is when we recorded cats visiting nests. Predator exclusion cages for nests could also be used, although the potential for unintended consequences such as increased adult predation or nest abandonment must be considered (Isaksson *et al.* 2007).

The other main cause of hatching failure was flooding. One approach to preventing flooding is to move nests to higher ground (Moore 2008). However, we observed that much of the higher elevation habitat at the Ashley estuary was vegetated and could be accessed by 4WD vehicles. Therefore, any nest relocations would likely need to occur in conjunction with habitat quality improvements. Clearing vegetation may also help to reduce predation rates, given cats use vegetation for cover when hunting (Moseby & McGregor 2022).

Our findings also provide an initial indication of the influence that microhabitat features may have on the hatching success of banded dotterels and SBBGs. Hatching success can be affected by a variety of habitat features at different scales, such as food resources, that we did not measure. To date, there has been a lack of published research on this topic for New Zealand shorebird species. Future studies on microhabitat use, including how individuals select nest sites based on habitat availability, may provide additional information on factors affecting hatching success and habitat selection in New Zealand birds. It would also help guide restoration efforts to create the microhabitats that will maximise hatching success for species of conservation concern.

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