A myna problem: alien species no obstacle to recovery for the Mangaia kingfisher

Tom Thacker, Philip J. Seddon, Yolanda van Heezik and Gerald McCormack

Abstract Endemic island species are of conservation interest as unique taxa, often with restricted populations, but many are data poor. The Mangaia kingfisher *Todiramphus ruficollaris*, known locally as the tangaʻeo, is endemic to the island of Mangaia in the Cook Islands, and categorized as Vulnerable on the IUCN Red List. The population size has not been estimated since 1996, despite concerns over habitat loss and competition with an invasive species, the common myna *Acridotheres tristis*. We provide new population estimates for both the tangaʻeo and the common myna, using the same methodology as previous estimates. During December 2018–February 2019 we surveyed with distance sampling along 73 line transects walked across the six habitat types on Mangaia. We estimate there are 4,106 tangaʻeo on Mangaia (95% CI 3,191–5,283), a dramatic 7–8 fold increase compared to the previous estimate of 393–764. We estimate there are 13,350 common myna (95% CI 10,998–16,206), a slight increase, although densities in the two most favoured habitats for myna have declined. There is no evidence that the common myna poses a threat to the viability of the tangaʻeo population, as the latter has increased despite a much larger population of common myna. Presumed declines in the tangaʻeo population in the past were probably a result of habitat loss as a result of the cultivation of pineapples *Ananas comosus* for export, an industry that collapsed in the 1980s. We recommend a review of the IUCN Red List status of the tangaʻeo.

Keywords *Acridotheres tristis*, common myna, Cook Islands, distance sampling, Mangaia kingfisher, population size, tangaʻeo, *Todiramphus ruficollaris*

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Introduction

In the face of unprecedented losses in global biodiversity, conservation efforts have increasingly targeted endemic species in biodiversity hotspots (Brooks et al., 2002). The islands of Polynesia and Micronesia comprise one such hotspot (Myers et al., 2000). However, for many species, a lack of available data has impeded attempts to devise conservation management plans (Bland et al., 2015). The Mangaia kingfisher *Todiramphus ruficollaris*, known locally as the tangaʻeo, is endemic to the island of Mangaia in the Cook Islands in the tropical South Pacific, and is categorized as Vulnerable on the IUCN Red List (BirdLife International, 2016).

Although the tangaʻeo has previously been described as a distinct species (Pratt et al., 1987; Sibley & Monroe, 1990), it has also been listed as a subspecies of *Todiramphus tuta* (Fry et al., 1992; Rowe & Empson, 1996a), the only other member of this genus found within the Cook Islands, on the islands of Atiu and Mauke. However, phylogenetic analyses of the genus *Todiramphus* support the distinction of the tangaʻeo as a separate species (Andersen et al., 2015, 2018).

As a forest-dwelling species, the loss of available habitat is a significant threat (Rowe & Empson, 1996a): the past intensive cultivation of pineapples *Ananas comosus*, in particular, led to the removal of tracts of forest on Mangaia. Furthermore, the introduction of the common myna *Acridotheres tristis* may have contributed to the decline of the tangaʻeo (Rowe & Empson, 1996a). The common myna was first introduced to Mangaia in the early 20th century but did not become abundant until an introduction from Rarotonga in 1964 as a biocontrol agent for a coconut stick insect (*Graeffea crouanii*; Parkes, 2006). Common myna have been observed harassing adult tangaʻeo and have been responsible for the failure of some tangaʻeo nests (McCormack et al., 2008). Pineapple plantations are no longer present to the same extent following the collapse of the export industry in the 1980s (Fleming & Blowes, 2003), but Mangaia does continue to support a large population of common myna (Baker et al., 1996). Rats *Rattus rattus* and *Rattus exulans* are present on Mangaia, but are not thought to affect the tangaʻeo population seriously (Baker et al., 1996).

There are few published studies of the tangaʻeo, although anecdotal evidence suggests the species might have declined in abundance following the introduction of the common myna (T. George, pers. comm., 2018). The most recent estimates of population size were made > 20 years ago. Rowe & Empson (1996a) estimated a total tangaʻeo population of 250–450 individuals, and Baker et al. (1996) estimated 393–764 individuals using distance sampling. Distance sampling has been used effectively to estimate the abundance of other bird species (Garcia-Del-Rey, 2009), and is considered superior

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to techniques using count data to derive indices of abundance (Rosenstock et al., 2012).

Current baseline data on the population size of the tanga'eo are needed before any conservation action can be considered. Here we estimate the tanga'eo population using distance sampling and, given the concerns around potential negative interactions between the common myna and the tanga'eo, we also estimate the size of the common myna population on Mangaia.

Study area

Mangaia is the southernmost island in the Cook Islands (Fig. 1) and is the second largest by land area (5,140 ha). The wet season is November–May, and both mean rainfall and temperature are typically greater during these months (monthly means of 194.4 mm and 24.9 °C) than during the rest of the year (114.2 mm and 21.7 °C; IUCN, 2014).

The central region of the island is dominated by the weathered remnant of an extinct shield volcano (Merlin, 1991). Swamplands lie in the lowland area at the foot of these volcanic hills, just above sea level. Both the volcanic interior of the island and the surrounding lowlands are completely surrounded by a rampart of ancient and eroded coral limestone (known as the makatea) that extends to nearly 70 m altitude. The height of the makatea decreases towards the shoreline such that the terrace along the coast is at c. 15 m altitude.

The land area of Mangaia has six habitat types, based on geophysical areas (Merlin, 1991). Coastal *Barringtonia asiatica* forests occur along the lower makatea terrace, except in the south-east where this forest gives way to coastal scrubland, characterized by large stands of pandanus *Pandanus tectorius* trees. Primary forest can be found along the upper terrace of the makatea, comprising a number of indigenous species such as *Elaeocarpus tonganus* and the lantern tree *Hernandia moerenhoutiana*, and introduced species such as candlenut *Aleurites moluccana*. Secondary forest, characterized by albizia trees *Falcataria moluccana*, occurs in the low-lying regions around the base of the volcanic hills and around the swamps, where taro *Colocasia esculenta* is grown. Plantation forests of Caribbean pine *Pinus caribaea* and *Acacia* species cover the central volcanic hills. There are three villages on Mangaia, and these constitute the sixth habitat type. Coconut palms *Cocos nucifera* grow across the island, but are uncommon on the volcanic hills.

Methods

Distance sampling uses data, describing the distances of individuals detected during sampling, to model a detection function that can estimate the number of individuals not detected within the sampling area and thus produce a total population estimate. We walked line transects during
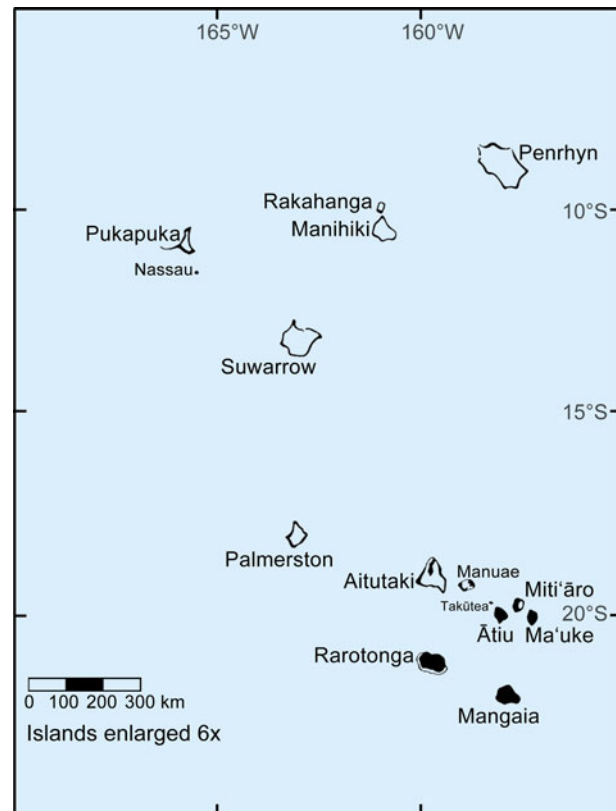

Fig. 1 Mangaia is the southernmost island in the Cook Islands, c. 200 km south-east of Rarotonga. The land area of each island is shaded black, including the reef flat.

daylight hours during December 2018–February 2019, attempting to survey each of the six habitat types equally. This was achieved across four of the six habitats (primary, secondary and plantation forest, and village habitats); *Barringtonia* forests were sampled more extensively, and coastal scrubland was sampled the least. We walked fewer transects in the latter habitat because of the limited number of detections of either species there.

Transects were established along a randomly set compass bearing, from random start coordinates. We recorded start and end coordinates using a GPS; transect length and start and end times were also recorded. The distance from the observer to each detection was recorded using a rangefinder, and the angle from the transect line determined by a compass, facilitating calculation of the perpendicular distance from the transect line to each detection (Supplementary Material 1). Detections were by either sight or sound. In instances where detections were by sound, we recorded the distances of objects determined to be closest to the source of the call. Although transect start points and bearings were chosen randomly, in some cases the topography, particularly on the makatea, dictated what was practical, and transects had to be adjusted or shortened.

Distance 6.0 was used to analyse the line transect data (Buckland et al., 2015), and Akaike's information criterion

(AIC) was then used to determine the fit of the detection functions to the data. For both species, the AIC of a detection function modelled using data pooled across habitat types was greater than the sum of the AIC values produced when modelling the detection function separately using data from each habitat type. Consequently, data were stratified by habitat type with the detection function fitted separately for each habitat. No truncation or grouping of the data was used. Analyses were run using the Conventional Distance Sampling engine, and the models with the lowest AIC score and the smallest pooled coefficient of variation were selected. A half-normal key function and a cosine series expansion were chosen as the simplest best fit for the tangaʻeo data; for the common myna, a hazard-rate key function with a simple polynomial series expansion was selected.

Spatial analysis of the data was carried out in *ArcMap 10.6.1* (Esri, Redlands, USA). Examination of satellite imagery of Mangaia combined with ground truthing allowed the total areas of each habitat type to be calculated. These areal estimates were subsequently used in the *Distance* analysis to derive abundance estimates from estimated habitat-specific densities.

Results

Across 73 transects totalling 38.17 km (Fig. 2), 189 tangaʻeo and 986 common myna were detected. Tangaʻeo were not detected in the coastal scrubland, but were otherwise recorded in each of the other five habitats; common myna were detected in all six habitat types.

The total tangaʻeo population was estimated to be 4,106 individuals (95% CI 3,191–5,283; Table 1). The highest densities of tangaʻeo were recorded in the secondary and primary forest habitats (Fig. 3); few were found in and around the villages. The coastal *Barringtonia* forests had high densities of tangaʻeo but contributed relatively few tangaʻeo to the overall estimate because of the small extent of this habitat. The area covered by the plantation forest meant this habitat harboured a larger number of tangaʻeo despite the lower density than in the *Barringtonia* forests. The pooled coefficient of variation (CV) for all of the habitats was 12.7%; for each of the individual habitats the CV was higher. Secondary forest had the smallest CV (16.1%), whereas *Barringtonia* forest had the largest (39.9%).

The common myna was estimated to have a population size of 13,350 individuals (95% CI 10,998–16,206; Table 1). The highest densities were in and around the villages and in the secondary forest, where the majority of individuals were found (Fig. 3). The coastal habitats supported fewer individuals than the inland habitat types, and the habitat with the lowest density of common myna was the plantation forest. The pooled CV for all habitat types was 9.8%; secondary forest had the smallest CV (11.7%) and coastal

FIG. 2 Locations of the 73 transects surveyed across Mangaia during December 2018–February 2019. The area in the northeast of the island without any transects is where the airstrip is located; this was not considered a separate habitat type and was ignored for sampling purposes.

scrubland and plantation forest had the highest (80.1 and 50.3%, respectively).

Discussion

This is only the second population estimate for the tangaʻeo on Mangaia using distance sampling, and it indicates that the population has increased since 1996. Our findings suggest that the common myna population has also increased during this time. Both species occur at high densities within the secondary forest habitat on Mangaia.

We surveyed a greater number of habitats than Baker et al. (1996), although our results suggest that the density of tangaʻeo has increased in each of the three habitats (primary, secondary and *Barringtonia* forests) surveyed in 1996. The areal extents of primary and *Barringtonia* forest (1,309 and 301 ha, respectively, in 2019) do not appear to have changed substantially since 1996 (1,118 and 194 ha). Secondary forest appears to have increased (1,270 ha compared to 397 ha; Baker et al., 1996). This disparity appears to be a result of Baker et al.'s (1996) decision to distinguish agricultural land occurring in the lowlands surrounding the central volcanic hills from secondary forest. In contrast, our study considered secondary forest and agricultural land as a single, continuous habitat type, based on our observations that the tangaʻeo and common myna occurred at similar frequencies in these two areas. Consequently, although our results suggest that the tangaʻeo population has increased from 218 (Baker et al., 1996) to 2,102 in the secondary forest, the initial estimate from 1996 might have underestimated the number of tangaʻeo in this habitat. Another factor potentially contributing to an underestimate in 1996 was that neither the central plantation forest nor the village habitats were surveyed (Baker et al., 1996). Rowe & Empson (1996a)

TABLE 1 Density and abundance of the tangaʻeo *Todiramphus ruficollaris* and the common myna *Acridotheres tristis* across the six habitat types on Mangaia.

|	Area (ha)	Total survey effort (km)	No. of transects	Density per ha (95% CI)		Abundance (95% CI)	
				Tangaʻeo	Common myna	Tangaʻeo	Common myna
Barringtonia forest	301	8.67	13	0.70 (0.31–1.58)	1.03 (0.50–2.09)	209 (92–474)	339 (166–691)
Primary forest	1,309	6.56	17	0.72 (0.35–1.48)	1.49 (0.77–2.87)	941 (458–1,934)	1,946 (1,008–3,759)
Secondary forest	1,270	6.32	20	1.65 (1.19–2.30)	5.96 (4.71–7.55)	2,102 (1,513–2,918)	7,570 (5,976–9,589)
Plantation forest	1,231	5.91	13	0.67 (0.42–1.06)	0.73 (0.27–1.95)	821 (514–1,309)	901 (338–2,397)
Village	282	6.57	7	0.12 (0.06–0.23)	8.14 (4.91–13.48)	33 (17–66)	2,290 (1,382–3,795)
Coastal scrubland	372	4.14	3		0.82 (0.12–5.40)		304 (46–2,006)
Total	4,765	38.17	73			4,106 (3,191–5,283)	13,350 (10,998–16,206)
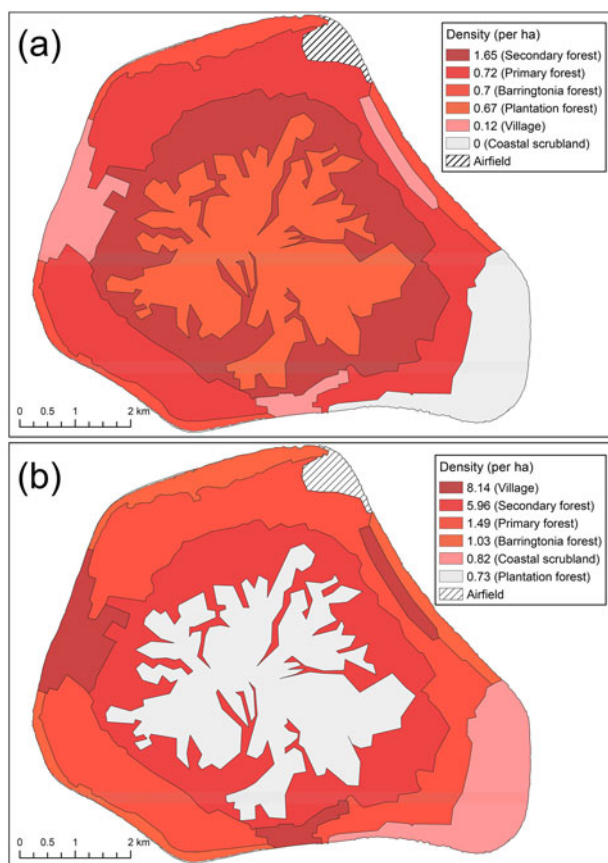
FIG. 3 Density heat maps for (a) the tangaʻeo and (b) the common myna across Mangaia.

did not report that these two habitats contained large numbers of tangaʻeo at that time, suggesting that the population size estimated by Baker et al. (1996) was unlikely to have been significantly affected by this omission.

To give context to our findings, the density of tangaʻeo on Mangaia is similar to those of other Pacific *Todiramphus* species, although caution is needed when comparing estimates derived from different methodologies and at different scales. As the majority of densities reported previously for Pacific *Todiramphus* species are for entire islands and not individual habitats, it is helpful to consider that the average density of tangaʻeo weighted for the different sizes of the habitats on Mangaia is 0.86 individuals per ha. This is greater than the density reported for the Pohnpei kingfisher *Todiramphus reichenbachii* (0.5 per ha) and for the Tuamotu kingfisher *Todiramphus gambieri* (0.2 per ha), but less than that of the Society kingfisher *Todiramphus veneratus* on the island of Moʻorea in French Polynesia (2.4 per ha; Kesler et al., 2010).

For the common myna, estimates of density and abundance in 1996 included the villages, but the coastal scrubland and plantation forest were not surveyed (Baker et al., 1996). Common myna occurred at much greater densities within the three villages and in secondary forest in 1996 (20.4 and 12.9 per ha, respectively; Baker et al., 1996) than in our study (8.14 and 5.96 per ha). The estimated population of common myna on Mangaia has increased since 1996 (13,350 compared to 8,877; Baker et al., 1996), and this seems mainly related to the increased areas of secondary forest and village habitat reported here. Despite occurring at a lower density, we recorded more myna within the secondary forests in 2019 than in 1996 (7,570 compared to 5,121), and similarly within the villages (2,290 compared to 1,306). The decline in the density of the common myna in these two apparently preferred habitat types suggests that the population has declined since 1996.

Pineapple plantations were historically extensive throughout the island's interior, on the volcanic hills and in the surrounding fertile lowlands. Following the collapse of the industry in the 1980s, the pineapple plantations on the volcanic hills were replaced by plantation forest. In contrast, the land used for the cultivation of pineapples in the lowlands was left to regenerate into secondary forest. Furthermore, following the industry's collapse and the Cook Islands' financial crisis in 1996, the human population of Mangaia decreased from c. 1,500 people to < 500 (G. Vaiimene, pers. comm., 2018). With a reduced population, the rate of deforestation probably slowed, although there are no empirical data to confirm this. It seems likely that the increase in tangaʻeo reported here is linked to the increased availability of favourable habitat, as well as a reduction in habitat fragmentation and loss. Similarly, any past decline in tangaʻeo would likely

have been the result of habitat loss brought about by the widespread cultivation of pineapples.

We observed aggressive interactions between common myna and tangaʻeo throughout our fieldwork, with tangaʻeo more frequently the aggressor in such interactions. There is no evidence, however, to indicate that a decline in common myna is linked to an increase in the tangaʻeo. Grarock et al. (2013) reported that the density of common myna declined with increasing tree density in Australia. Moreover, the rate of egg success (the proportion of eggs laid that hatched) for common myna in Australia dropped from > 90% in sites with a low density of trees to 56% in sites with high tree density (Grarock et al., 2013). The regeneration of secondary forest following the decline in pineapple cultivation could have affected the island's population of common myna negatively, although no data are available to confirm this.

Our findings suggest that the common myna population on Mangaia is not adversely affecting the tangaʻeo population, despite the population size being approximately three times that of the tangaʻeo. The 7–8 fold increase in the tangaʻeo population since 1996 further suggests that intensive management of this species is not required. The tangaʻeo is currently categorized as Vulnerable on the IUCN Red List because there are < 1,000 mature individuals (criterion D1), and it is confined to fewer than five locations (D2; BirdLife International, 2016). With an estimated total population of several thousand birds, the D1 criterion should be dropped in future assessments.

The species remains restricted to a single location, and this has been the basis for the D2 criterion in the past. The spectre of introduced alien species and diseases remains. Avian malaria, for example, has had a devastating effect on endemic species in Hawaii since its introduction in the early 20th Century, with numerous species having been driven to extinction or having suffered reductions in their ranges (van Riper et al., 1986). However, although avian malaria has already spread to both French Polynesia (Beadell et al., 2006) and the Cook Islands (Ishtiaq et al., 2006) there is no evidence that the tangaʻeo has been adversely affected. *Rattus rattus* and *R. exulans* have been introduced to Mangaia but do not appear to be exerting significant pressure upon the tangaʻeo. Cyclones do not appear to pose a serious threat to this species, as the raised makatea encircling the island protects the primary and secondary forests where tangaʻeo are most commonly found.

The D2 criterion is warranted only in circumstances where there exists a plausible future threat that could drive the species to become either Critically Endangered or extinct within a short period of time, in either one or two generations or 3–5 years, depending on which is longer (IUCN Standards & Petitions Committee, 2019). Although habitat fragmentation remains a concern, it is unlikely that the degree of habitat loss necessary to propel the species to become Critically Endangered would occur within this time

frame. The IUCN distributional criteria, such as the D2 criterion, have been criticized as inadequate for species endemic to small islands, and for unnecessarily increasing the number of species categorized as threatened (Martín, 2009). None of these threats are likely to drive the species to become Critically Endangered or extinct within the specified window required for the D2 criterion. We therefore recommend that the D2 criterion for the tangaʻeo is re-evaluated.

We recommend that distance sampling surveys of the tangaʻeo are undertaken at least every 5 years, to track population size and allow managers to anticipate any need for intervention in the event of declines. Future analyses should include the method of detection (seen or heard) as a covariate, as this has been found to improve the fit of models for a number of other bird species (van Heezik & Seddon, 2012).

Author contributions Study design: TT, PJS, YvH; fieldwork: TT; data analysis, writing: TT, PJS, YvH, GM.

Conflicts of interest None.

Ethical standards This research abided by the *Oryx* guidelines on ethical standards.

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