Recent recovery and expansion of Guam’s locally endangered Sāli (Micronesian Starling) *Aplonis opaca* population in the presence of the invasive brown treesnake

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**Summary**

Assessing the impacts of invasive predators on the demography and distribution of native species is critical for understanding mechanisms of species persistence and informing the design of recovery programmes. On the oceanic island of Guam, the introduction of the predatory brown treesnake *Boiga irregularis* after World War II caused the near-total loss of the native forest avifauna. Localised snake control measures have been implemented since the early 1990s, yet it remains poorly understood how they have impacted Guam’s remaining native bird populations. To address this question, we combined intensive area searches of Andersen Air Force Base (AAFB) with island-wide transect surveys and opportunistic sightings to provide a comprehensive update on the distribution and abundance of Sāli (*Micronesian Starling, Aplonis opaca*) – one of Guam’s last extant native bird species. Area searches of AAFB, where the largest remnant of the Sāli population persists, revealed a 15-fold population increase since the last survey in the early 1990s, and transect surveys and opportunistic sightings indicate incipient recolonisation of other urbanised areas of northern and central Guam. We estimate the current island-wide population size at ~1,400 individuals. The population increase can likely be attributed to a combination of snake control measures and the Sāli’s ability to exploit urban refugia for nesting and roosting. Although these trends demonstrate some population recovery, a skewed age ratio (>90% adults and subadults) at AAFB and a highly urbanised distribution and low abundance outside AAFB indicate that snake predation continues to strongly impact the population. More intensive snake suppression efforts, particularly in forested areas, may allow for the Sāli population to attain its former distribution and abundance on Guam. More broadly, our findings reinforce the importance of urban areas as refugia for some threatened species.
Keywords: abundance, *Aplonis opaca*, brown tree snake, distribution, Guam, islands, Micronesian Starling, predator control, urbanization

Introduction

Invasive predators are a primary threat to biodiversity at a global scale (Clavero and Garcia-Berthou 2005, Doherty et al. 2016). Native species often lack a shared evolutionary history with novel predators, and thus lack the requisite adaptations for coexisting with invaders (Sih et al. 2010). Under these favourable conditions, introduced predators can achieve ecological release and exponential population growth in the presence of naïve prey (Sih et al. 2010), with devastating ecological and economic effects (Savidge 1987, Clavero and Garcia-Berthou 2005, Reaser et al. 2007, Clavero et al. 2009). Some of the most extreme impacts of invasive predators have been documented on oceanic islands (Medina et al. 2011, Spatz et al. 2017), which are disproportionately vulnerable due to their isolation from continental terrestrial systems and high levels of endemism (Kier et al. 2009). For example, invasive predators have caused population declines, local extirpations, and extinctions of native bird species across island systems such as Hawai‘i, New Zealand, and the Mascarenes (Atkinson 1977, Clout 2001, VanderWerf 2009, Cheke and Hume 2010, Doherty et al. 2016).

Despite the wide-ranging and severe impacts of invasive predators on native island biota, population recovery has been documented in response to predator control. Eradications of invasive mammals on islands have already resulted in substantial conservation benefits to native species, such as positive demographic or distributional responses (Jones et al. 2016), and further gains are expected from future eradication projects (Holmes et al. 2019). For logistical reasons, eradication projects to date have occurred largely within fenced predator-proof exclosures (Tanentzap and Lloyd 2017) and on relatively small, uninhabited islands, although larger, inhabited islands are increasingly being targeted (Glen et al. 2013). Where successful eradication is not currently feasible, predator control can also substantially increase reproductive success and survival of island populations (Moorhouse et al. 2003, Whitehead et al. 2008, VanderWerf 2009). Adaptation of native species to introduced predators has also resulted in some examples of improved fitness and range recovery (Strauss et al. 2006).

One location where recovery of native species has been particularly challenging is the Pacific island of Guam in the Mariana Archipelago, Micronesia. Following the introduction of the predatory brown treesnake *Boiga irregularis* to Guam after World War II, nine of the island’s 11 native forest bird species were extirpated in a matter of decades (Savidge 1987, Rodda et al. 1992, Wiles et al. 2003). During the peak of the irruption in the early 1990s, brown treesnake densities are estimated to have reached 50–100 individuals/hectare or higher, eventually declining to 25–50/ hectare by the late 1990s (Rodda et al. 1999), with an estimated island-wide population size of 1–2 million snakes (Rodda and Savidge 2007). Nevertheless, some bird species have managed to persist in the presence of the brown treesnake (Wiles et al. 2003), including the endangered Yáyaguak (Mariana Swiftlet) *Aerodramus bartschi* (Apodidae) which roosts in caves that may be relatively inaccessible to snakes, and the locally endangered Sāli (Micronesian Starling) *Aplonis opaca* (Sturnidae). The Sāli is a cavity-nesting omnivore and important seed disperser in the Marianas (Rehm et al. 2017, 2019, Pollock et al. 2020), with a broad geographic distribution across much of Micronesia (Craig and Feare 2018). Although historically common-to-abundant throughout Guam across all habitat types (Jenkins 1983, Craig and Feare 2018), the Sāli’s distribution and abundance on Guam declined precipitously along with the rest of the avifauna after the introduction of the brown treesnake (Savidge 1987, Wiles et al. 2003). The last census in the early 1990s estimated the population at only 60–120 individuals, primarily restricted to Andersen Air Force Base (AAFB), a military installation in northern Guam (Wiles et al. 1995).

The Sāli population continues to persist on Guam, but there has been no formal assessment of its status since the early 1990s (Wiles et al. 1995). Recent observations indicate that the population
may be expanding, particularly at AAFB, where snake population control and containment aimed at protecting infrastructure and preventing spread to other islands has been ongoing since 1993 (reviewed in Clark et al. 2018). Although recent studies of radio-tagged Sāli fledglings at AAFB have documented high post-fledging mortality due primarily to brown tree snake predation (Wagner et al. 2018, Pollock et al. 2019), regular sightings of Sāli in urban areas in northern and central Guam not occupied since the 1980s suggest that its distribution may be expanding southward even without widespread snake control.

To assess the current status of Guam’s Sāli population, we conducted an island-wide survey of their distribution and abundance. Our primary objectives were to obtain a current estimate of Sāli population size and explore how distribution and abundance have changed over time. To do so, we leveraged multiple recent data sources (i.e. opportunistic sightings, transect surveys and standardized area searches) combined with a review of historical literature on the population on Guam. We discuss the potential reasons for a population increase and range expansion on Guam and describe possible management actions to facilitate Sāli recolonization across the island.

**Methods**

**Study site**

Guam is the largest (541 km²) and most economically developed island in Micronesia with the region’s largest human population (~160,000 inhabitants as of 2010; Spies et al. 2019). More than 20% (>11,000 ha) of the island’s area is developed (Spies et al. 2019). The northern half of Guam is a limestone plateau that supports most of the island’s remaining intact karst forest, whereas the southern half is volcanic in origin, more mountainous, and composed largely of ravine forest and savanna habitat (Donnegan et al. 2004). Most of the island’s human population and developed habitats are concentrated in northern and central Guam, whereas southern Guam is less developed and more sparsely populated. Although Sāli prefer forested areas (Rehm et al. 2018), they are generalists and historically were present in all available habitats on Guam, from roadside and urban areas to savanna and forest (Baker 1947, Jenkins 1983, Engbring and Ramsey 1984).

**Literature review**

To assess changes in Sāli population size and distribution over time and contextualize our current survey results, we gathered all available published and grey literature that referred to Sāli abundance and distribution on Guam. To do so, we searched Web of Science and Scopus in November 2020 using the search terms “Sāli” AND “Aplonis opaca” AND “Micronesian Starling” AND “Guam” AND “population” AND “abundance” AND “distribution”. We also supplemented this literature with unindexed reports familiar to the authors.

**Population size and age structure at AAFB**

To estimate the size and age structure of the Sāli population at AAFB, we conducted three consecutive week-long area searches of the base’s main developed area (its administrative and housing areas) in September-October 2018 (Figure 1). We also sampled areas to the south and west of the base perimeter once each week (Figure S4 in the online supplementary material) to ensure that we were not omitting appreciable numbers of birds off-base during our surveys. The extent of our sampling area was smaller than the Sāli surveys in the 1980s and 1990s, which encompassed the flight line and large swaths of forest throughout northern Guam (Engbring and Ramsey 1984, Wiles et al. 1995). For example, Engbring and Ramsey (1984) conducted point-counts at 178 stations on or in proximity to AAFB, all within forest habitat. The primary reason we limited our survey to the main developed area of the base was to encompass the core Sāli roosting habitat, where virtually all individuals appear to currently roost. Extensive radiotelemetry has
demonstrated that Sāli range widely throughout the forested areas along the eastern and southern perimeter of AAFB during the day (H. S. Pollock and H. S. Rogers pers. obs.). However, because of high snake predation in forested areas (Pollock et al. 2019), birds of all age classes (>99% of \( n = 44 \) individuals, \( n = 444 \) roosting observations) return to the developed area in the afternoon (around 1500) prior to roosting, where they are relatively sedentary and easier to count (H. S. Pollock and M. Kastner pers. obs.). More than 350 individuals in the AAFB population were colour-banded in 2017–2018 as part of a larger project on Sāli demography (see Wagner et al. 2018, Pollock et al. 2019) and our method for estimating population size relies on resights of these colour-banded individuals (see Statistical analysis below). We assumed a closed population with no births, deaths, emigration, or immigration occurring between the successive counts. We are confident that emigration and immigration were minimal based on the aforementioned tracking of radio-tagged individuals, all of which used forest extensively and travelled off-base but returned to the core roosting area at night (H. S. Pollock and H. S. Rogers pers. obs.). By repeating intensive area searches each week for three consecutive weeks, we obtained three replicates while minimizing the confounding effect of mortality on population size estimates (Kendall 1999).

To count Sāli, we divided the main developed area of AAFB into 28 search areas of roughly similar size, comprising three habitat types: urban (UR), residential housing (HW, HE, HN), and golf course (GC; Figure 1). Each day, we randomly selected four search areas (thus allowing all 28 search areas to be surveyed per week) and assigned groups of two observers to survey two search areas each. Observers traversed a given search area together, which increased overall detection probability and the accuracy of colour-band identifications. To minimise the risk of double-counting individuals, observers never searched adjacent search areas in a given day, remained in constant contact during surveys, and communicated movements of any birds throughout a given search area. Surveys lasted until the entire extent of the search area had been covered (50.4 ± 14.7 minutes; range: 27–91 minutes) and were allocated to one of two time blocks: ‘early’
Distribution and abundance outside of AAFB

To estimate the distribution and abundance of Sāli outside the developed area at AAFB, we combined two data sources – opportunistic sightings and transect surveys. We excluded a small population (~200 individuals) of Sāli that has remained stable on nearby Cocos Island, a small islet 2.4 km south of Guam (Engbring and Ramsey 1984, Engbring and Fritts 1988, Wiles et al. 2003; L. Barnhart Duenas pers. obs.). First, we collated opportunistic sightings of Sāli from three complementary sources of information: (1) eBird records (eBird 2019) from 2009 to 2018 (n = 9 observations; Table S1), (2) a database of Sāli sightings from 2005 to 2018 (n = 39 observations; Table S1) maintained by the Guam Division of Aquatic and Wildlife Resources (DAWR), and (3) a database of Sāli sightings from 2009 to 2019 maintained by MK (n = 16 observations; Table S1). For eBird data, we took a conservative approach and excluded sightings that did not include a detailed description of the bird or the specific location and date of the sighting. All sightings included in the DAWR database were independently verified by DAWR biologists through detailed discussions with observers who reported sightings as well as site visits after each report. All sightings collected in MK’s database were recorded by biologists familiar with the species. For each sighting, we recorded the village and specific location, the observer, the number of Sāli, GPS coordinates, and year of the sighting.

We also surveyed 46 transects distributed across the island once each in April–May 2018 to aid in determining the island-wide distribution. First, we surveyed 19 spring bird count (‘SBC’) transects situated along trails or roads that were previously established by DAWR in 1985. Ten SBC transects (two northern, three central, and five southern) were located in rural areas with little development and nine (five northern and four central) were located in suburban areas near residential homes within 1 km of forest habitat. We sampled birds at 10 points along each transect. Average transect length was 5,189 ± 1,496 m (average distance between points: mean ± SD = 605 ± 210 m) and varied in length due to differences in landscape configuration and accessibility. At each point, experienced observers conducted 10-minute unlimited distance point-counts (sensu Matsuoka et al. 2014) and recorded all individuals that were seen or heard. Second, we used the opportunistic sightings compiled by DAWR to inform placement of 27 additional ‘Sāli’ transects located where Sāli had recently been observed. Sāli transects were ~500 m in length, did not overlap with SBC transects, and all except one were located within 500 m of forested habitat (either scrub forest or ravine forest; sensu Taborosi 2013). The number of Sāli transects per village depended on the number of opportunistic sightings in that area – we placed more transects in areas with more prior sightings to increase detectability given that Sāli were often present in low numbers. Transect surveys began at dawn, lasted approximately 75 minutes, and were conducted by the same experienced observers as the SBC transects. We conducted 10-minute unlimited-distance counts...
at six points along each transect, all equally spaced 100 m apart, and recorded all individual Sāli seen or heard at each point.

**Statistical analysis**

All analyses were conducted in R version 3.3.3 (R Core Team 2017). To estimate Sāli population size on AAFB, we followed six steps. (1) We used unique resights of colour-banded individuals across the duration of the study period (21 days) to generate accumulation curves using function `specaccum` in the ‘vegan’ package (Oksanen et al. 2013). (2) We used the function `specpool`, which uses three non-parametric estimators (Chao’s estimator and two separate jackknife estimators) and a bootstrap estimator, to extrapolate these accumulation curves and estimate the total number of banded individuals present at the study site during the study period. Standard errors were lowest for the bootstrap estimator (Figure S2), so we opted for this approach. (3) We calculated the weekly detection probability of colour-banded birds for each week of the survey, estimated as the number of banded individuals detected in a given week divided by the total number of individuals counted that week. (4) We divided the extrapolated estimate of the total number of banded individuals present at the time of the counts in step (2) by the weekly detection probability in step (3) to generate a weekly population size estimate. (5) To quantify uncertainty in each weekly population size estimate, we used the standard errors from the bootstrap estimator in step (2) to create a range for each weekly estimate, bounded on the lower end by mean–SE and on the upper end by mean + SE. (6) We averaged the three weekly mean estimates generated in step (4) to create a total mean estimate of AAFB population size.

Because opportunistic sightings were not collected in a standardised way and transects were surveyed only once each, we were unable to provide a quantitative estimate of Sāli abundance outside of AAFB. Therefore, we used our cumulative expertise and anecdotal repeat sightings to provide a semi-quantitative estimate for each region of Guam (northern, central, southern). We then summed this semi-quantitative estimate with the AAFB estimate to provide a rough island-wide estimate of abundance.

**Results**

**Literature review**

We found 16 papers in the literature from 1901 to 1995 that mentioned either the abundance or distribution of Sāli on Guam (Table 1). Eleven of the 12 studies published prior to 1970 provided qualitative estimates only, describing Sāli as ‘common’, ‘very common’, or the ‘most common’ bird species on the island. Baker (1947) detected Sāli on 100% of his surveys (n = 125) and found that the species comprised nearly 60% of all birds counted. By 1978–1979, the species had become rare across southern Guam and was uncommon over most of northern and central Guam (Jenkins 1983).

The first quantitative estimate of Sāli population size on Guam was made in 1981 by Engbring and Ramsey (1984). Excluding the small Cocos Island population, they counted 1,667 individuals during island-wide point-counts and used distance-sampling accounting for imperfect detection to estimate an overall population size of 15,132–18,602 (mean estimate: 16,776 individuals). At this time, the range had contracted substantially relative to the study by Jenkins (1983) conducted only a few years prior in 1978–1979, and Sāli were completely absent from southern and central Guam except for a small group of birds in the village of Hagåtña (Figure 2). Population size at and around AAFB’s airfield, administrative and housing areas, and adjoining plateau forest was estimated at only 231 individuals (Engbring and Ramsey 1984).

The overall population continued to decline as snakes reached higher densities across northern Guam, with almost no birds detected on any island-wide long-term survey routes after 1985 (Wiles et al. 2003). By the early 1990s, Wiles et al. (1995) estimated an island-wide population (excluding Cocos Island) of only about 60–120 birds, including 50–100 Sāli in the developed portion of AAFB.
and nearby areas of Mt. Santa Rosa and Gayinero, Yigo; two much smaller groups of birds numbering no more than five individuals each at the Conventional Weapons Storage Area (CWSA, now called ‘Munitions Storage Area’) on AAFB and at Naval Computer and Telecommunications Area Master Station (NCTAMS, now called ‘Naval Base Guam Telecommunications Site’) in Dededo; and a scattering of solitary birds along the southern coast.

### Distribution and abundance at AAFB

In three successive week-long surveys of the AAFB Sâli population, we counted 683, 609, and 844 birds, respectively (Table 2). We counted only 3–6 birds each week in the forests along the southern and eastern peripheries of the base (Figure S4), confirming that we were not omitting large numbers of Sâli from our weekly counts. Birds were concentrated towards the centre of the base’s main developed area, with more Sâli detected in interior search areas (n = 16 search areas) than in peripheral search areas adjacent to forest edge (n = 12 search areas; Table S2). The majority

| Source          | Observation type | Abundance              | Distribution                  |
|-----------------|------------------|------------------------|-------------------------------|
| Seale 1901      | Qualitative      | Common                 | –                             |
| Safford 1902    | Qualitative      | Very common            | –                             |
| Bryan 1936      | Qualitative      | Common                 | –                             |
| Marshall 1945   | Qualitative      | Most common            | Island-wide distribution     |
| Stophlet 1946   | Qualitative      | Most common            | –                             |
| Watson 1946     | Qualitative      | Most common            | –                             |
| Baker 1947      | Semi-quantitative| Most common; seen in 100% of 125 counts; 57.3% of total 2,428 birds | –                             |
| Kibler 1950     | Qualitative      | Very common            | Island-wide distribution     |
| Hartin 1961     | Qualitative      | Very common            | Island-wide distribution     |
| King 1962       | Qualitative      | Very common            | –                             |
| Tubb 1966       | Qualitative      | Most common            | Island-wide distribution     |
| Wood 1968       | Qualitative      | Common                 | –                             |
| Pratt et al. 1979 | Qualitative     | Uncommon relative to other islands and declining | –                             |
| Jenkins 1983    | Semi-quantitative| Most common native species in northwest Guam, rare across northern plateau and central Guam, very rare in southern Guam; 51.4% juveniles | Island-wide distribution |
| Engbring and Ramsey 1984 | Quantitative (island-wide point-counts) | Most common; ~17,000 individuals | Widespread throughout northern Guam, with a small group in Hagåtña |
| Wiles et al. 1995 | Semi-quantitative (qualitative observations, island-wide point counts, driving transects) | ~60-120 individuals | Restricted to AAFB and Yigo (~50-100 individuals), two small groups (<5 individuals) at CWSA and NCTAMS, and a few scattered birds along the southern coast |
of birds were unbanded, with <5% banded birds counted each week (mean = 3.6%, range = 3.0–4.3%; Table 2). We registered 25 colour-banded individuals in week 1, 26 in week 2, and 24 in week 3, for a cumulative total of 42 unique individuals (Figure S5). Thirteen (46.4%) of the colour-banded birds detected in week 2 were unique resights, compared to only four (16.7%) in week 3. Extrapolation of the resight accumulation curve using the bootstrap metric estimated 50 unique colour-banded individuals on AAFB (mean ± SE: 49.9 ± 3.4 individuals; Figure S2), suggesting that our sampling approach was reasonably thorough (i.e. we detected 42/50 = 84% of projected banded birds present on AAFB).

Our weekly estimates of the AAFB population size were as follows: 1,351 (range: 1,257–1,441) individuals for week 1; 1,160 (range: 1,081–1,240) individuals for week 2; and 1,663 (range: 1,550–1,777) individuals for week 3. By averaging the proportions of banded birds across weeks, our mean estimate of population size was 1,391 (range: 1,245–1,538) individuals. The age structure of the population was heavily skewed, with adults and subadults comprising a mean of 91.1% of the population across the three-week survey period (Table 2).

**Distribution and abundance outside of AAFB**

We compiled opportunistic sightings of Sāli at 64 unique locations based on eBird data, Guam DAWR records, and MK’s records between 2005 and 2019 (Table S1). Sightings extended across
nearly half of the island, but were largely concentrated in villages of northern and central Guam (excluding AAFB), with a few sightings in southernmost Guam (Figures 2, 3). Overall, we tallied 156 birds in 64 sightings in 12 of the island’s 19 villages, as follows: Tamuning-Tumon-Harmon (21), Hagåtña (11), Yigo (10), Dededo (6), Merizo (4), Santa Rita (3), Mongmong-Toto-Maite (2), Inarajan (2), Barrigada (2), Piti (1), Umatac (1), and Asan-Mainai (1). Numbers of individuals per sighting averaged 2.4 ± 2.2 birds (range: 1–12 birds), with 69% (44/64) of sightings involving ≥2 birds. Observations occurred primarily in urbanised areas, including the island’s main business districts (particularly at large malls and shopping centres), residential areas, and city parks. Sightings were frequently made along main roads, streets, or trails, and none were more than 2 km from a built-up area or major arterial road. Birds were most often observed perched on power lines, or power poles in Yigo, Hagåtña, Tamuning-Tumon-Harmon, and Dededo. No opportunistic sightings were recorded from the villages of Agana Heights, Chalan Pago-Ordot, Sinajana, Mangilao, Yona, Agat, and Talofofo. Regions without sightings included the developed east-central side of the island and nearly all of southern Guam (Figures 2, 3).

During the 19 SBC and 27 Sålì transect surveys combined, we registered 91 unique Sålì sightings on 20 of 46 (43%) transects surveyed (Figure 3, Table 3). On SBC transects, Sålì were only present in Yigo and were not detected in any of the other eight villages. In contrast, on Sålì transects, we detected Sålì in five of six villages, except Merizo. All sightings occurred in northern and central Guam, with the highest concentration occurring in Yigo and along the periphery of AAFB (Figure 3). No Sålì were detected in the southern villages of Merizo or Umatac (Table 3), despite the presence of the nearby Cocos Island population (~200 birds) and the opportunistic sightings noted above. We observed nesting behaviour (i.e. the presence of an active nest, birds transporting nesting material) and/or juvenile birds on four transects – two in Yigo (Figure S3) and one each in Hagåtña and Tamuning-Tumon-Harmon.

Combining the opportunistic sightings and transect surveys, we estimated the population in urbanised areas of northern Guam (Yigo and Dededo) at 30–40 individuals, central Guam at 30–50 individuals (20–30 in Hagåtña, 10–20 in Tamuning-Tumon-Harmon), and up to 10 individuals scattered outside those areas. Thus, we estimated that 60–100 Sålì were present outside of AAFB. Adding these 60–100 birds to the estimate for AAFB produces an overall island-wide population size estimate of 1,450–1,490 individuals (1,650–1,690 individuals if the population of ~200 Sålì on Cocos Island is included).

Discussion

Using a combination of area searches, opportunistic sightings and transect surveys, we provide the first update on the distribution and abundance of Sålì on Guam in more than 25 years. We found a 15-fold increase in population size (~100 vs. ~1,500 birds) since the last survey in the early 1990s (Wiles et al. 1995), with an estimated 93–96% of the population concentrated in the main developed area of AAFB. Sålì are also in the process of recolonising urbanised areas elsewhere in northern and central Guam, where the species has been absent since the expansion of brown tree snakes in the 1970s and early 1980s (Figure 2). As noted by Wiles et al. (1995), a few birds also continue to occur along the coast of southern Guam (Figure 2), but their status is unclear and most likely represent temporary residents originating from the small separate population on nearby Cocos Island or individuals regularly commuting from that island.

Outside of AAFB, Sålì observations were largely limited to Guam’s main urban areas (Figure 2, Table 3), with most sightings taking place in four of the island’s most heavily developed and populated villages (Tamuning-Tumon-Harmon, Hagåtña, Yigo, and Dededo; Table 3, Table S1). In particular, Yigo’s proximity to AAFB likely accounts for its large number of sightings. Whether or not the birds in these areas form an established self-sustaining population is unknown. Most sightings in these areas involved pairs or small groups of birds, suggesting ample potential for breeding, yet we documented relatively few nests or juveniles. Additionally, birds were only
Table 3. Summary of Division of Aquatic and Wildlife Resources (DAWR) transect survey sampling effort in April–May 2018 (i.e. 27 newly established Sāli transects and 19 spring bird count [SBC] historical transects) across the island of Guam. Included are the region and village where transects were located, number of transects per village, and the total number of Sāli detected across all transects in each village.

| Region          | Village          | Number of Transects | Number of individuals detected |
|-----------------|------------------|---------------------|--------------------------------|
| Sāli transects  |                  |                     |                                |
| Northern Guam   | Yigo             | 5                   | 35                             |
|                 | Dededo           | 3                   | 1                              |
| Central Guam    | Tamuning-Tumon   | 9                   | 13                             |
|                 | Hagåtña          | 6                   | 6                              |
|                 | Mongmong         | 1                   | 2                              |
| Southern Guam   | Merizo           | 3                   | 0                              |
| SBC transects   |                  |                     |                                |
| Northern Guam   | Yigo             | 6                   | 34                             |
|                 | Dededo           | 2                   | 0                              |
| Central Guam    | Asan-Maina       | 1                   | 0                              |
|                 | Barrigada        | 2                   | 0                              |
|                 | Piti             | 1                   | 0                              |
|                 | Toto             | 1                   | 0                              |
| Southern Guam   | Inarajan         | 2                   | 0                              |
|                 | Merizo           | 1                   | 0                              |
|                 | Umatac           | 1                   | 0                              |
|                 | Yona             | 2                   | 0                              |
| **Totals**      |                  | **46**              | **91**                         |
reliably present at large shopping malls except in Hagåtña, and sightings from other locations may have been transient individuals, especially given the high mobility of the species. For these reasons, we cannot rule out the possibility that Saåli presence in this region of the island remains strongly dependent on birds dispersing from the AAFB population, which likely functions as a source population for areas farther south.

Avoidance of brown treesnakes is critical to the survival of all birds on Guam (Savidge et al. 1987, Wiles et al. 2003). To that end, our findings suggest that two factors, ongoing snake control measures (reviewed in Vice 2011, Clark et al. 2018, Engeman et al. 2018) and the Saåli’s adaptation to urban habitats (Wiles et al. 2003), are likely responsible for the species’ partial population recovery on the island. AAFB’s main developed area, which covers about 17 km², has been a focal point of snake interdiction efforts on Guam since 1993 (Vice 2011), with thousands of individuals captured and removed annually (USDA APHIS Wildlife Services, pers. comm.). Control measures include mouse-baited traps installed on fencing and other structures along the base’s perimeter, airfield, and electrical infrastructure, and in the base’s cargo storage, administration, and housing areas; plastic tube bait stations containing dead mice implanted with acetaminophen placed on vegetation and structures along forest roads and the eastern forest edge; and nocturnal spotlight searches along fencing (Vice 2011, Engeman et al. 2018, USDA APHIS Wildlife Services pers. comm.). These efforts are likely reducing snake abundance in the main developed area of AAFB, especially in the centre, where we recorded the highest numbers of roosting and nesting Saåli.

Similar snake control efforts are also conducted at other military installations and port facilities on the island (Vice 2011, Engeman et al. 2018), but all of these operations except the one at the Guam International Airport cover considerably smaller geographic units and none have thus far enabled the establishment of resident Saåli as on AAFB.

Developed areas on Guam appear to be serving as refugia from snake predation by providing safe roosting and nesting locations for birds, as hypothesised by Wiles et al. (2003). Indeed, brown treesnakes tend to avoid roads (Siers et al. 2014), highly lit areas (Campbell et al. 2008), and open expanses such as grass lawns and parking lots, all of which typify developed areas. AAFB’s main developed area – the core area for Saåli nesting and nighttime roosting (H. S. Pollock and H. S. Rogers pers. obs.) – is characterised by such habitat features including asphalt roads, runways, taxiways, and parking areas; expansive mowed lawns with isolated ornamental trees; and numerous buildings. Nesting sites on the base typically include solitary trees, building cavities, lamp posts and artificial nest boxes (Savidge et al. 2018).

A number of key differences exist between the main developed area of AAFB and Guam’s off-base urban areas, which likely explain the lower abundance of Saåli outside of AAFB. In contrast to AAFB, off-base developed areas receive almost no intensive large-scale snake interdiction (the exception being at the Guam International Airport), contain scattered pockets of secondary vegetation and remnant forest that provide habitat for snakes, and possess far fewer areas of large, mowed lawns. Indeed, the limited available data indicate that snakes still occur in fairly high densities and that the largest-sized individuals tend to be found in developed areas, likely due to the increased availability of avian and mammalian prey (Siers et al. 2017, Wagner et al. 2018). Nevertheless, Guam’s off-base urban areas possess other features absent from AAFB that inhibit snake presence and movement, and thus are probably beneficial to Saåli. These include the presence of major roads with heavier traffic volumes, large parking lots with isolated trees (e.g., at large malls and commercial shopping centres), higher densities of larger buildings, and the presence of artificial nesting structures such as power poles.

Despite the growth and expansion of Guam’s Saåli population since the early 1990s, several lines of evidence clearly indicate that current brown treesnake control measures (primarily intended to prevent off-island spread and damage to electrical infrastructure) and the island’s existing urban environment are insufficient to neutralise the continuing impacts of snakes on the population. First, snake capture rates along the perimeter of AAFB (USDA APHIS Wildlife Services pers. comm.) have remained relatively constant since the mid-1990s, rather than declining in number. While this is certainly causing a localised reduction in snake abundance (Siers et al. 2019), it has
not translated to an overall population suppression. Second, recent studies of Sâli at AAFB have found very low fledgling survival (~26%), primarily due to predation by brown treesnakes (56% of mortality) but also by feral or domestic cats (19% of mortality; Pollock et al. 2019). Third, the age ratio of the Sâli population has shifted drastically, from immatures forming an apparent majority of the population in the mid-1940s (Baker 1951) and 51.4% of birds counted in late 1970s (n = 138 observations; Jenkins 1983) to a ratio of only 8.9% juveniles in 2018 on AAFB. This shift is consistent with the low fledgling survival rate (Pollock et al. 2019) and suggests exceedingly limited recruitment into the population. Fourth, Sâli have failed to expand into a large, suburbanised area in east-central Guam, where few or no sightings have yet occurred. This area, composed of the villages of Barrigada, Mangilao, Chalan Pago-Ordot, and Yona, features high human populations, but less of the heavily urbanised setting found in Tamuning-Tumon-Harmon, Hagåtña, Yigo, and Dededo. Sâli have also not yet recolonised the interior of southern Guam, which is largely covered in forest and grassland and has only minor development. Taken together, these factors suggest that brown treesnakes still pose a considerable threat to Guam’s remaining bird populations, including Sâli. These findings support the generally shared presumption that there can be no island-wide recovery of native forest species without effective snake suppression on Guam. Application of novel control methods such as the automated aerial bait delivery system (Siers et al. 2019a, b, 2020), which deploys dead mice implanted with acetaminophen across the landscape, will likely be required to suppress the snake population to levels sufficient for making Guam habitable again for extirpated native birds. Our results provide a reference point for future studies of the Sâli population and its expansion and inform conservation projects focused on reintroducing birds to Guam.

One possible additional factor aiding the population on AAFB has been the deployment of nest boxes for use by the cavity-nesting Sâli. Nest boxes can boost reproductive success of cavity-nesting birds by increasing the availability of nest sites, providing adequate shelter from the elements and protection from predators. Small numbers were placed in the base’s main developed area in the late 1990s and early 2000s (D. Lujan, U.S. Navy Joint Region Marianas pers. comm.), and an expanded installation program of more than 50 predator-resistant boxes has been ongoing since 2015 (Savidge et al. 2018). These have improved the nesting success of Sâli relative to that of unprotected nests and allowed over 800 nestlings to fledge successfully (J. Savidge and H. S. Rogers pers. obs.). The overall benefit to the population, however, has probably been marginal to date due to the high levels of snake and cat predation on fledglings (Wagner et al. 2018, Pollock et al. 2019). To date, given a population of ~1,250 breeding birds on AAFB and only 50–70 nest boxes, the majority of juveniles entering the population likely still originate from natural nests in cavities of buildings, power poles and trees (M. Kastner, personal observations).

The population size and distribution of Sâli are crucial factors for ecosystem functioning on Guam. Sâli have a very broad diet and are the only native frugivorous bird species remaining on the island (Pollock et al. 2020). An important consequence of their constrained distribution is the limitation of ecosystem services related to seed dispersal and consequent forest regeneration (Rehm et al. 2018, 2019, Kastner et al. 2021), which are currently geographically restricted on Guam. Although the persistence of native wildlife in urban refugia may be beneficial from a species conservation perspective (Shaffer 2018), significant range expansion into historical forest habitats is necessary to fulfil a broader vision of rewilding on Guam (Thierry and Rogers 2020). Cohesive integration of technical advances in predator control (e.g. Siers et al. 2019a, b) with appropriate economic and social policy (Peltzer et al. 2019) are necessary to achieve the successful implementation of species reintroduction on Guam.

Supplementary Materials

To view supplementary material for this article, please visit http://dx.doi.org/10.1017/S0959270920000726.
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References

Atkinson, I. A. E. (1977) A reassessment of factors, particularly Rattus rattus L., that influenced the decline of endemic forest birds in the Hawaiian Islands. *Pac. Sci.* 31:109–133.
Baker, R. H. (1947) Size of bird populations at Guam, Mariana Islands. *Condor* 49: 124–125.
Baker, R. H. (1951) The avifauna of Micronesia, its origin, evolution, and distribution. *Univ. Kansas Publ. Mus. Nat. Hist.* 3: 1–359.
Bryan Jr., E. H. (1936) The birds of Guam. *Guam Rec.* 13: 18–20.
Campbell, S. R., Mackessy, S. P. and Clarke, J. A. (2008) Microhabitat use by brown treesnakes (*Boiga irregularis*): effects of moonlight and prey. *J. Herpetol.* 42: 246–251.
Chække, A. and Hume, J. P. (2010) *Lost land of the Dodo: the ecological history of Mauritius, Réunion and Rodrigues*. London, UK: Bloomsbury Publishing.
Clark, L., Clark, C. and Siers, S. (2018) Brown tree snakes: methods and approaches for control. Pp. 107–134 in W. Pitt, J. Beasley and G. Wittmer, eds., *Ecology and management of terrestrial vertebrate invasive species in the United States*. Boca Raton, FL: CRC Press.
Clavero, M. and García-Berthou, E. (2005) Invasive species are a leading cause of animal extinctions. *Trends Ecol. Evol.* 20:110.
Clavero, M., Brotons, L., Pons, P. and Sol, D. (2009) Prominent role of invasive species in avian biodiversity loss. *Biol. Conserv.* 142: 2043–2049.
Clout, M. (2001) Where protection is not enough: active conservation in New Zealand. *Trends Ecol. Evol.* 16: 415–416.
Craig, A. and Feare, C. (2018) Micronesian Starling (*Aplonis opaca*). In J. A. del Hoyo, A. Elliott, J. Sargatal, D. A. Christie and E. de Juana, eds., *Handbook of the birds of the world alive*. Barcelona, Spain: Lynx Edicions.
Doherty, T. S., Glen, A. S., Nimmo, D. G., Ritchie, E. G. and Dickman, C. R. (2016) Invasive predators and global biodiversity loss. *Proc. Natl. Acad. Sci. USA* 113: 11261–11265.
Donnegan, J. A., Butler, S. L., Grabowiecki, W., Hiserote, B. A. and Limtiaco, D. (2004) Guam’s forest resources, 2002. *Resource Bulletin PNW-RB-243*. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station.
eBird (2019) eBird: an online database of bird distribution and abundance. Accessed 8 April 2019. eBird, Ithaca, New York. Available at: [http://www.ebird.org](http://www.ebird.org).
Engbring, J. and Fritts, T. H. (1988) Demise of an insular avifauna: the brown tree snake on Guam. *Trans. West. Sect. Wildl. Soc.* 24: 31–37.
Engbring, J. and Ramsey, F. L. (1984) Distribution and abundance of the forest birds of Guam: results of a 1981 survey. Honolulu, HI: U.S. Fish and Wildlife Service.

Engeman, R. M., Shiel, A. B. and Clark, C. S. (2018) Objectives and integrated approaches for the control of brown tree snakes: an updated review. J. Environ. Manage. 219: 115–124.

Glen, A. S., Atkinson, R., Campbell, K. J., Hagen, E., Holmes, N. D., Keitt, B. S., Parkes, J. P., Saunders, A., Sawyer, J. and Torres, H. (2013) Eradicating multiple invasive species on inhabited islands: the next big step in island restoration? Biol. Invasions 15: 2589–2603.

Hartin, M. H. (1961) Birds of Guam: observations – July to November 1960. ‘Elepaio 22: 34–48.

Holmes, N. D., Spatz, D. R., Oppel, S., Tershy, B., Croll, D. A., Keitt, B. D., Genovesi, P., Burfield, I. J., Will, D. J., Bond, A. L. and Wegmann, A. (2019) Globally important islands where eradicating invasive mammals will benefit highly threatened vertebrates. PLoS One 14: e0212128.

Jenkins, J. M. (1983) The native forest birds of Guam. Ornithol. Monogr. 31: 1–61.

Jones, H. P., Holmes, N. D., Butchart, S. H., Tershy, B. R., Kappes, P. J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D. P., Bonnaud, E., Burridge, A. A. and Campbell, K. (2016) Invasive mammal eradication on islands results in substantial conservation gains. Proc. Natl. Acad. Sci. USA 113: 4033–4038.

Kastner, M., Pollock, H. S., Savidge, J. A., Fricke, E. C. and Rogers, H. S. 2021.- Functional rescue of seed dispersal by a remnant frugivore population on a defaunated tropical island. Biotropica. doi: 10.1111/btp.12926.

Kendall, W. L. (1999) Robustness of closed capture–recapture methods to violations of the closure assumption. Ecology 80: 2547–2552.

Kibler, L. F. (1950) Notes on the birds of Guam. Auk 67: 400–403.

Kier, G., Kreft, H., Lee, T. M., Jetz, W., Ibisch, P. L., Nowicki, C., Mutke, J. and Barthlott, W. (2009) A global assessment of endemism and species richness across island and mainland regions. Proc. Natl. Acad. Sci. USA 106: 9322–9327.

King, B. (1962) Guam field notes. ‘Elepaio 23: 29–31.

Marshall, J. T., Jr. (1945) Field notebook, 1944–1945. Volume 1472, Section 3: Marshall Islands, Marianas Islands (Saipan, Tinian, and Guam), and Palau Islands. Berkeley, California: Museum of Vertebrate Zoology Archives, University of California.

Matsuoka, S. M., Mahon, C. L., Handel, C. M., Sólymos, P., Bayne, E. M., Fontaine, P. C. and Ralph, C. J. (2014) Reviving common standards in point-count surveys for broad inference across studies. Condor 116: 599–608.

Medina, F. M., Bonnaud, E., Vidal, E., Tershy, B. R., Zavaleta, E. S., Donlan, C. J., Keitt, B. S., Corre, M., Horwath, S. V. and Nogales, M. (2011) A global review of the impacts of invasive cats on island endangered vertebrates. Glob. Change Biol. 17: 3503–3510.

Moorehouse, R., Greene, T., Dilks, P., Powlesland, R., Moran, L., Taylor, G., Jones, A., Knegtmans, J., Will, D., Pryde, M. and Fraser, I. (2003) Control of introduced mammalian predators improves Kaka Nestor meridionalis breeding success: reversing the decline of a threatened New Zealand parrot. Biol. Conserv. 110: 33–44.

Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P. R., O’Hara, R. B., Simpson, G. L., Solymos, P., Stevens, M. H. H., Wagner, H. and Oksanen, M. J. (2013) Package ‘vegan’. Community Ecology Package, version 2. http://CRAN.Rproject.org/package=vegan.

Peltzer, D. A., Bellingham, P. J., Dickie, I. A., Houlston, G., Hulme, P. E., Lyver, P. O. B., McGlone, M., Richardson, S. and Wood, J. (2019) Scale and complexity implications of making New Zealand predator-free by 2050. J. R. Soc. N. Z. 49: 412–439.

Pollock, H. S., Fricke, E. C., Rehm, E. M., Kastner, M., Suckow, N., Savidge, J. A. and Rogers, H. S. (2020) Sáli (Micronesian Starling – Aplonis opaca) as a key seed dispersal agent across a tropical archipelago. J. Trop. Ecol. 36: 56–64.

Pollock, H. S., Savidge, J. A., Kastner, M., Seibert, T. F. and Jones, T. M. (2019)
Pervasive impacts of brown treesnakes drive low fledgling survival in endangered Micronesian Starlings (Aplonis opaca) on Guam. Condor 121: 1–11.

Pratt, H. D., Bruner, P. L. and Berrett, D. G. (1979) America’s unknown avifauna: the birds of the Mariana Islands. Amer. Birds 33: 227–235.

R Core Team (2017) R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. URL: https://www.R-project.org.

Reaser, J. K., Meyerson, L. A., Cronk, Q., De Poorter, M., Eldrege, L. G., Green, E., Kairo, M., Latasi, P., Mack, R. N., Mauremootoo, J., O’Dowd, D., Orapa, W., Sastrowtomo, S., Saunders, A., Shine, C., Thrainsson, S. and Vaiutu, L. (2007) Ecological and socioeconomic impacts of invasive alien species in island ecosystems. Environ. Conserv. 34: 98–111.

Rehm, E. M., Chojnacki, J., Rogers, H. S. and Savidge, J. A. (2017) Differences among avian frugivores in seed dispersal to degraded habitats. Rest. Ecol. 26: 760–766.

Rehm, E. M., Balsat, M. B., Lemoine, N. P. and Savidge, J. A. (2018) Spatial dynamics of habitat use informs reintroduction efforts in the presence of an invasive predator. J. Appl. Ecol. 55: 1790–1798.

Rehm, E., Fricke, E., Bender, J., Savidge, J. and Rogers, H. (2019) Animal movement drives variation in seed dispersal distance in a plant–animal network. Proc. Roy. Soc. B 286: 20182007.

Rodda, G. H. and Savidge, J. A. (2007) Biology and impacts of Pacific island invasive species. 2. Boiga irregularis, the brown tree snake (Reptilia: Colubridae). Pac. Sci. 61: 307–324.

Rodda, G. H., Fritts, T. H. and Conry, P. J. (1992) Origin and population growth of the brown tree snake, Boiga irregularis, on Guam. Pac. Sci. 46: 46–57.

Rodda, G. H., Fritts, T. H., McCoid, M. J. and Campbell, E. W. (1999) An overview of the biology of the brown tree snake (Boiga irregularis), a costly introduced pest on Pacific Islands. Pp. 44–80 in G. H. Rodda, Y. Sawai, D. Chiszar and H. Tanaka, eds., Problem snake management: the habu and the brown tree snake, Ithaca, NY: Cornell University Press.

Safford, W. E. (1902) The birds of the Marianne Islands and their vernacular names – I. Osprey 6: 39–42.

Savidge, J. A. (1987) Extinction of an island forest avifauna by an introduced snake. Ecology 68: 660–668.

Savidge, J. A., Kastner, M. and Seibert, T. F. (2018) Developing a predator-resistant nest box for Micronesian Starlings with application to endangered Guam Micronesian Kingfishers final report. Prepared for U.S. Department of Navy, NAVFAC Marianas, Santa Rita, Guam.

Seale, A. (1901) Report of a mission to Guam. Honolulu, HI: Bishop Museum Press.

Shaffer, H. B. (2018) Urban biodiversity arks. Nat. Sustain. 1: 725–727.

Siers, S. R., Savidge, J. A. and Reed, R. N. (2014) Invasive brown treesnake movements at road edges indicate road-crossing avoidance. J. Herpetol. 48: 500–505.

Siers, S. R., Savidge, J. A., and Reed, R. N. (2017) Ontogenetic and ecological variation in invasion risk of brown treesnakes (Boiga irregularis) on Guam. Manag. Biol. Invas. 8: 469–483.

Siers, S. R., Shiels, A. B., Payne, C. G., Chlarson, F. M., Clark, C. S., and Mosher, S. M. (2019a) Photographic validation of target versus nontarget take of brown treesnake baits. Wildl. Soc. Bull. 43: 752–759.

Siers, S. R., Pitt, W. C., Eisemann, J. D., Clark, L., Shiels, A. B., Clark, S. C., Gosnell, R. J. and Messaros, M. C. (2019b) In situ evaluation of an automated aerial bait delivery system for landscape-scale control of invasive brown treesnakes on Guam. Pp. 348–355 In C. R. Veitch, M. N. Clout, A. R. Martin, J. C. Russell and C. J. West, eds., Island invasives: scaling up to meet the challenge. Dundee, Scotland, UK: Winter and Simpson.

Siers, S. R., Shiels, A. B. and Barnhart, P. D. (2020) Invasive snake activity before and after automated aerial baiting. J. Wildl. Manage. 84: 256–267.

Sih, A., Bolnick, D. I., Luttbeg, B., Orrock, J. L., Peacock, S. D., Pintor, L. M., Preisser, E., Rehage, J. S. and Vonesh, J. R. (2010)
Predator–prey naïveté, antipredator behavior, and the ecology of predator invasions. *Oikos* 119: 610–621.

Spatz, D. R., Zilliacci, K. M., Holmes, N. D., Butchart, S. H., Genovesi, P., Ceballos, G., Tershy, B. R., and Croll, D. A. (2017) Globally threatened vertebrates on islands with invasive species. *Science Advances* 3: e1603080.

Spies, N. P., Mizerek, T., Reeves, M. K., Amidon, F. and Miller, S. E. (2019) Developed systems in the Mariana Islands Archipelago. In M. Goldstein and D. DellaSala, eds,* Encyclopedia of the world’s biomes*. Amsterdam, Netherlands: Elsevier Publishing.

Stopfleth, J. J. (1946) Birds of Guam. *Auk* 63: 534–540.

Strauss, S. Y., Lau, J. A. and Carroll, S. P. (2006) Evolutionary responses of natives to introduced species: what do introductions tell us about natural communities? *Ecol. Lett.* 9: 357–374.

Taboroši, D. (2013) *Environments of Guam*. Honolulu, HI: Bess Press.

Tanentzap A. J. and Lloyd, K. M. (2017) Fencing in nature? Predator exclusion restores habitat for native fauna and leads biodiversity to spill over into the wider landscape. *Biol. Conserv.* 214: 119–126.

Thierry, H. and Rogers, H. (2020) Where to rewild? A conceptual framework to spatially optimize ecological function. *Proc. Roy Soc. B.* 287: 20193017.

Tubb, J. A. (1966) Notes on birds of Guam. *Nat. Hist. Bull. Siam Soc.* 21: 135–138.

VanderWerf, E. A. (2009) Importance of nest predation by alien rodents and avian poxvirus in conservation of Oahu Elepaio. *J. Wildl. Manage.* 73: 737–746.

Vice, D. S. (2011) *Brown tree snake introduction and prevention of spread*. Barrigada, Guam: U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services.

Wagner, C., Tappe, C., Jaramillo, O., Kastner, M., Van Ee, N., Savidge, J. A. and Pollock, H. S. (2018) First reported predation of fledgling Micronesian Starlings (*Aplonis opaca*) by brown tree snakes (*Boiga irregularis*) on Guam. *Micronesica* 6: 1–7.

Watson, R. J. (1946) Bird notes from Guam. *Raven* 17: 40–42.

Whitehead, A. L., Edge, K. A., Smart, A. F., Hill, G. S., and Willans, M. J. (2008) Large scale predator control improves the productivity of a rare New Zealand riverine duck. *Biol. Conserv.* 141: 2784–2794.

Wiles, G. J., Aguon, C. F., Davis, G. W. and Grout, D. J. (1995) The status and distribution of endangered animals and plants in northern Guam. *Micronesica* 28: 31–49.

Wiles, G. J., Bart, J., Beck Jr., R. E. and Aguon, C. F. (2003) Impacts of the brown tree snake: patterns of decline and species persistence in Guam’s avifauna. *Conserv. Biol.* 17: 1350–1360.

Wood, H. (1968) Birds of Guam. Pp. 47–53 in R. E. Key, ed. *A naturalist’s guide to Guam*. Agana, Guam: Guam Science Teachers Association.