How many Laysan Teal *Anas laysanensis* are on Midway Atoll? Methods for monitoring abundance after reintroduction

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Abstract

Wildlife managers often request a simple approach to monitor the status of species of concern. In response to that need, we used eight years of monitoring data to estimate population size and test the validity of an index for monitoring accurately the abundance of reintroduced, endangered Laysan Teal *Anas laysanensis*. The population was established at Midway Atoll in the Hawaiian archipelago after 42 wild birds were translocated from Laysan Island during 2004–2005. We fitted 587 birds with unique markers during 2004–2015, recorded 21,309 sightings until March 2016, and conducted standardised survey counts during 2007–2015. A modified Lincoln-Petersen mark-resight estimator and ANCOVA models were used to test the relationship between survey counts, seasonal detectability, and population abundance. Differences were found between the breeding and non-breeding seasons in detection and how maximum counts recorded related to population estimates. The results showed strong, positive correlations between the seasonal maximum counts and population estimates. The ANCOVA models supported the use of standardised bi-monthly counts of unmarked birds as a valid index to monitor trends among years within a season at Midway Atoll. The translocated population increased to 661 adult and juvenile birds (95% CI = 608–714) by 2010, then declined by 38% between 2010 and 2012 after the Tōhoku Japan earthquake-generated tsunami inundated 41% of the atoll and triggered an Avian Botulism type C *Clostridium botulinum* outbreak. Following another severe botulism outbreak during 2015, the population experienced a 37% decline. Data indicated that the Midway Atoll population, like the founding Laysan Island population, is susceptible to catastrophic population declines. Consistent standardised monitoring using simple counts, in place of mark-recapture and resightings surveys, can be used to evaluate population status over the long-term. We estimate there were 314–435 Laysan Teal (95% CI for population estimate;

point estimate = 375 individuals) at Midway Atoll in 2015; *c.* 50% of the global population. In comparison, the most recent estimate for numbers on Laysan Island was of 339 individuals in 2012 (95% CI = 265–413). We suggest that this approach can be used to validate a survey index for any marked, reintroduced resident wildlife population.

Key words: Chapman estimate, count index, Laysan Teal, Lincoln-Petersen estimate, mark-resight models.

Species reintroduction programmes are being used increasingly to restore biodiversity and reduce extinction risk (Seddon *et al.* 2007; Miskelly & Powlesland 2013; Batson *et al.* 2015). Intensive post-release monitoring, such as radio-tracking founder birds, is important during the early post-release stage or initial breeding seasons to yield precise estimates of survival and reproduction. As a next step to understanding the outcome of a reintroduction attempt and to inform future management, marking a proportion of the population (*e.g.* with leg rings) facilitates monitoring individuals for estimating survival and abundance using capture-recapture or mark-resight analyses (Fischer & Lindenmayer 2000; Armstrong & Seddon 2008). After the population has increased and become established, a reduction in monitoring intensity might be warranted if systematic and accurate population data can be collected with reduced effort (Parker *et al.* 2013). Indices are often used to express comparisons of changes over a period of time and are often applied to infer population abundance from surveys of unmarked birds (*i.e.* direct counts). The valid application of a population index using survey counts requires testing of the assumption that the index is proportional to population size (Nichols 1992; White 2005).

In a previous study we tested the assumptions of survey monitoring protocols for providing a valid population index for Laysan Teal *Anas laysanensis* (classed as Critically Endangered globally; IUCN 2016) on Laysan Island. The Laysan Island study used a Lincoln-Petersen estimator to relate the survey counts to abundance derived from 15 years of mark-recapture and resightings data (Reynolds *et al.* 2015b). The reintroduced population at Midway Atoll National Wildlife Refuge was established with 42 birds translocated from Laysan Island during 2004–2005 (Reynolds *et al.* 2008), and the population increased rapidly to > 500 adult and juvenile birds by 2008 (Reynolds *et al.* 2011). In the study presented here, we estimate abundance over the period 2004–2015 for Laysan Teal reintroduced to Midway Atoll (hereafter, Midway), then provide linear regression equations for the 2007–2015 data to relate the maximum counts to estimates of abundance based on the Lincoln-Petersen estimator by season using analysis of covariance (ANCOVA). Our approach of transitioning from labour-intensive radio tracking, to less intensive mark-resight data, and then to the least intensive index of population abundance from unmarked birds, may be useful for other reintroductions, monitoring or restoration efforts. Our study using 8 years of data from marked Laysan Teal can serve as a model of how to validate a count index for other marked, resident wildlife populations undergoing systematic monitoring. Thus, this approach has utility for wildlife managers of reintroduced populations seeking to transition to a simple index to monitor population abundance for long-term trend analysis.

Methods

Study area

Midway is a remote Pacific atoll 2,300 km northwest of Honolulu (25°46'N, 171°44'W; Fig. 1) and is a part of the Papahānaumokākuea Marine National Monument (Executive Order 13022; Presidential Proclamation 8031 15 June 2006). The atoll consists of three islands

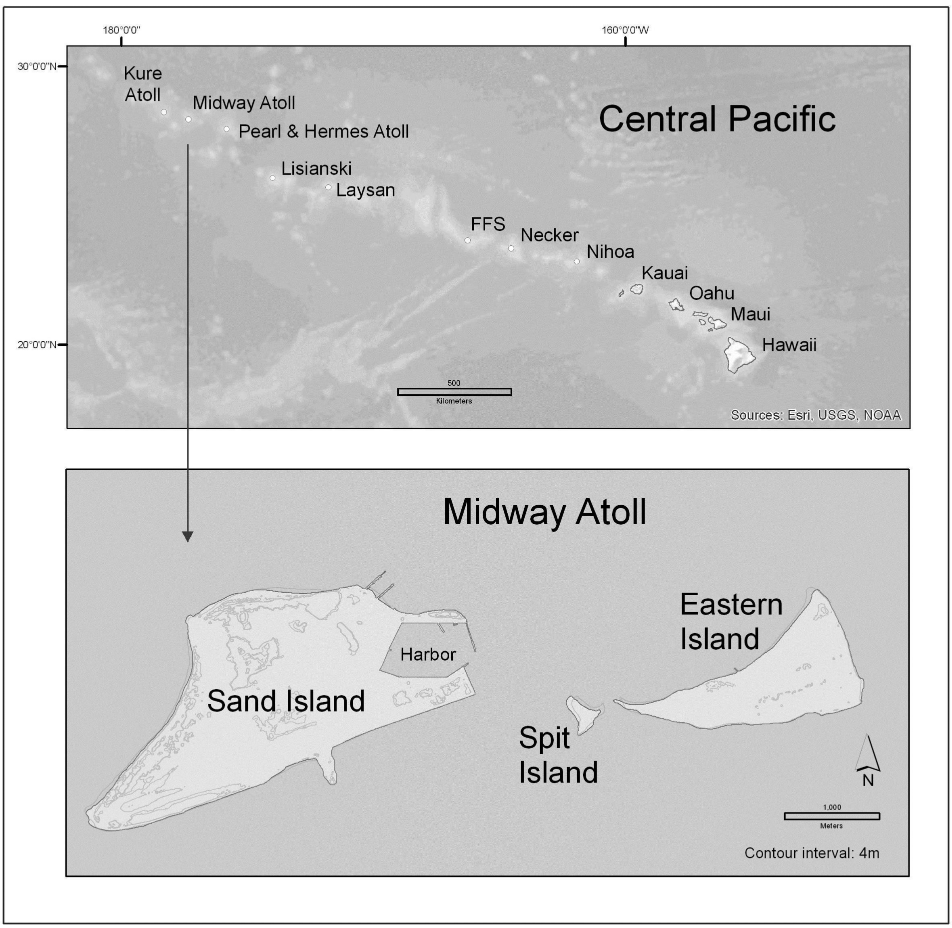

Figure 1. Map of the Hawaiian Islands, with detail of Midway Atoll.

totalling 604 ha, with a mean elevation of 3.0 m (Reynolds *et al.* 2015a).

Study species

The Laysan Teal is a formerly widespread Hawaiian dabbling duck that inhabited diverse habitats, but it had been confined to a small, remote low-lying atoll for about 150 years (Olson & Ziegler 1995). Bones of Laysan Teal are widespread and indicate that the species once occurred throughout the Hawaiian archipelago (Cooper *et al.* 1996) but, like many other Hawaiian birds (Olson & James 1991), it was extirpated from the larger islands following the arrival of humans and introduced rats 800–900 years ago (Burney *et al.* 2001). Unlike most of these species, the Laysan Teal survived on small islands in the remote northwestern chain, including Lisianski Island, from which it was extirpated in the mid-1800s, and Laysan Island, where the last relict population persisted. The Laysan Teal is non-migratory, primarily insectivorous and nests on the ground in dense terrestrial vegetation (see also Warner 1963; Reynolds *et al.* 2006, 2007, 2009). Wild-to-wild translocations have been made to Midway Atoll and to Kure Atoll, to reduce extinction risk and restore the species' range (USFWS 2009).

Population monitoring

Initially radio telemetry was used to monitor translocated individuals and their offspring during 2004–2007 (see Reynolds *et al.* 2008). As the population grew, however, resightings of individually marked birds, then standardised surveys were initiated. Laysan Teal have been caught on Midway during nine capture periods (May–December 2005, June 2006–February 2007, June–November 2007, August–November 2008, March–October 2009, October 2011, September–October 2013, August 2014 and July–December 2015; see also Reynolds *et al.* 2011), with most ducks captured using noose-carpet traps during crepuscular periods, or at night using a flexible hand-held net. Birds were marked with a numbered aluminium ring on one leg and a unique field-readable plastic engraved colour ring (Haggie Engraving, Crumpton, Maryland) or a field-readable engraved aluminium ring (Gey Band and Tag Company, Norristown, Pennsylvania) on the other leg.

The first standardised mark-resightings survey (hereafter survey) of Laysan Teal at Midway was conducted 23 October 2007, when collection of systematic survey data was discontinued until September 2008. Additional survey gaps occurred during February–May 2010 and December 2012–October 2014. During September 2008 to March 2016 surveys were typically conducted weekly or bi-monthly (see Reynolds *et al.* 2011 for detailed survey methods). Because the surveys and resightings were made weekly or bimonthly (*i.e.* over a relatively short time period), and most individually marked birds were sighted frequently on the small atoll (approximately once per week), the assumption of population closure is appropriate for mark-recapture models (see Reynolds *et al.* 2011 for detailed treatment of model assumptions). The survey started at sunrise and included all wetlands, persistent standing water, and freshwater guzzlers (*i.e.* water troughs) in the atoll. The start location and direction of survey routes were assigned randomly to reduce spatial-temporal bias. Observers recorded the ring status of each bird observed (as ringed, unringed or undetermined), and identified as many individual ringed birds as possible by colour-coded ring combinations or by reading the aluminium rings. Birds were classified as downy ducklings or post-fledglings (adult and feathered independent young of the year), with the population estimates including both juveniles and pre-breeders.

Data from uniquely marked individuals, identified from trapping, systematic surveys, incidental resightings and collection of carcasses, were used to determine the last date on which the birds were observed alive and to calculate each individual's lifetime median resighting frequency (*i.e.* 50th percentile of intervals between resightings). Each individual's median resightings interval was used to determine whether a missing bird (not seen again during the time series) was likely to be alive on a given survey date (details below).

Statistical analysis

Population estimates. Lincoln-Petersen (hereafter LP) based estimators have been used previously to estimate Laysan Teal abundance on Laysan Island and Midway (Moulton & Weller 1984; Marshall 1992; Reynolds & Citta 2007; Reynolds *et al.* 2011, 2015b). To estimate post-fledgling population abundance we used a mark-recapture sampling framework and a Chapman (1951) bias-corrected modification to the LP estimator:

$$\hat{N}_t = \frac{(M_t + 1)(n_t + 1)}{m_t + 1} - 1$$

where $\hat{N}_t$ is the population estimate, M_t is the total marked population, n_t is the number of animals counted, and m_t is the number of marked animals counted (*i.e.* resighted), all at a given time t.

Data were divided into two periods to correspond with a breeding year: typical breeding (March–August, covering laying, incubation, brood rearing and moult) and typical non-breeding (September–February, with January and February grouped with the previous year, covering late moult, flocking, courtship, pairing and pre-breeding). Transition months may however need adjustment in future estimates to reflect actual breeding phenology for a given survey year, although such adjustment was not required in the current study. The post-fledgling population is geographically closed because there is no immigration or emigration between atolls, and demographically closed because timing of recapture-resighting periods was relatively short compared to the time interval between such periods. We removed individuals from total marked (M_t) live birds for the next survey if their marked carcass was recovered. Additionally, we estimated M_t using an individual's resightings history following Reynolds *et al.* (2011) and Reynolds *et al.* (2015b) to account for mortality of marked birds (M_t) where the carcass was not recovered. If a bird was not resighted after ringing it was excluded and assumed dead (n = 12). If a bird was not sighted after its median resightings interval, and never seen again, we assumed it was dead and excluded it from M_t at the next survey date. Thus, we inferred an individual's survival or mortality for each individually marked bird based on their individual resight frequency. This inferred mortality was calculated for every survey and every marked individual to better meet assumptions of the LP estimator.

Since LP estimators tend to overestimate population sizes, we used criteria, based on Robson & Regier (1964), to reduce overestimation bias and identify the highest quality survey for estimating abundance. These criteria were: highest counts within a period where the coefficient of variation (CV) of the LP estimator was < 10%, and where the percentage of teal known to be ringed or unringed (*i.e.* their ring status was known) identified during the survey was ≥ 60%. If multiple seasonal surveys met these criteria, we selected the count with the maximum percentage of known ducks.

Index validation. We used SAS v9.4 (SAS Institute Inc. 2012) to conduct analysis of covariance (ANCOVA) to investigate the relationship between the maximum of the observed counts (dependent variable) each season and year (hereafter maximum seasonal count) and the population abundance estimates for that survey date (independent variable). The ANCOVA allowed us to determine if the maximum count per season was a suitable index for population abundance and to test for differences between seasons. The full model had different slopes and intercepts for each season on comparing the linear relationship between the LP estimate and the maximum count. The reduced model (main effects: count and season, no interaction term) allowed for identical slopes for each season, but different intercepts (*i.e.* parallel relationships). A further reduced model was examined that allowed for the same slope and intercepts across seasons (*i.e.* a simple linear regression, or correlation, of LP estimate *vs.* maximum count, pooling over the two periods). In addition, one survey was selected at random for each month in each year, and of those randomly selected counts, the maximum count per season was selected (hereafter maximum random count) and these data were analysed using ANCOVAs as described above. This model may be more applicable to future survey efforts by managers under funding limitations because it requires only one quality survey per month to generate maximum counts over seasons.

Results

Population counts and estimates

During 2004–2015, 587 Laysan Teal were fitted with unique leg rings and 21,309 recaptures and resightings were recorded through 07 March 2016. The median resightings interval across all individuals was eight days, and varied from one day for frequently seen birds to 399 days for a rarely seen bird. The median number of sightings per individual bird was 27 (range = 1–208). The maximum number of marked individuals alive in the population (339); 58% of the estimated 581 total birds (95% CI = 540–623), occurred in December 2009. When a ringed carcass was recovered (n = 139 reported), the median difference between the estimated date of death and actual carcass recovery was 47 days.

The highest rate of detection of post-fledglings occurred during the non-breeding period (see below); therefore we used counts from this period to estimate maximum annual post-fledgling abundance. We identified the best quality surveys, which were expected to yield the most accurate estimates to within 10–25% of the population abundance (Robson & Regier 1964); LP estimates in the non-breeding period ranged from 209 (95% CI = 185–232) in 2007, to 661 (95% CI = 608–714) in 2010 (Table 1, Fig. 2). A population decline of 38% occurred between the non-breeding seasons of 2010 and 2012 (Table 1). This was observed after winter storms, followed by the Tōhoku tsunami in March 2011, and a Botulism type C *Clostridium botulinum* outbreak as a result of massive seabird die-offs from sudden flooding (Reynolds *et al.* 2017). By February 2015 adult and juvenile abundance grew to 599 (95% CI = 518–680); however, the population declined by 37% following another severe Botulism type C outbreak later that same year (Table 1, Fig. 2; USGS National Wildlife Health Center, Honolulu, Hawai'i, unpubl. data, 22 Mar–26 Sep 2011 and 21 Apr–27 Nov 2015).

Table 1. Maximum counts and modified Chapman bias-corrected Lincoln-Petersen mark-resight population abundance (95% confidence intervals; CI) for Laysan Teal at Midway Atoll, Hawai'i, for the years 2007–2015. The estimates shown are derived for the best survey during the non-breeding period that had the greatest proportion of teal with known ring status (*i.e.* know to be ringed or unringed) and met Chapman's (1951) standards for an unbiased estimator or criteria for marked samples sizes (Robson & Regier 1964). In all cases we chose best quality surveys, defined as being the highest count within a period where the coefficient of variation in relation to the LP estimator was < 10% and where the percentage of teal with known ring status identified during the survey was ≥ 60%.

Year	Count	Proportion of the population ringed	Proportion of ringed birds with known ring status	95% CI of population estimate
2007	135	0.43	0.90	185–232
2008	361	0.41	0.92	458–520
2009	349	0.52	0.95	508–571
2010	375	0.41	0.67	608–714
2011	263	0.59	0.92	369–414
2012	284	0.38	0.97	374–441
2013				No estimate
2014	352	0.19	0.95	518–680
2015	211	0.22	0.86	314–435

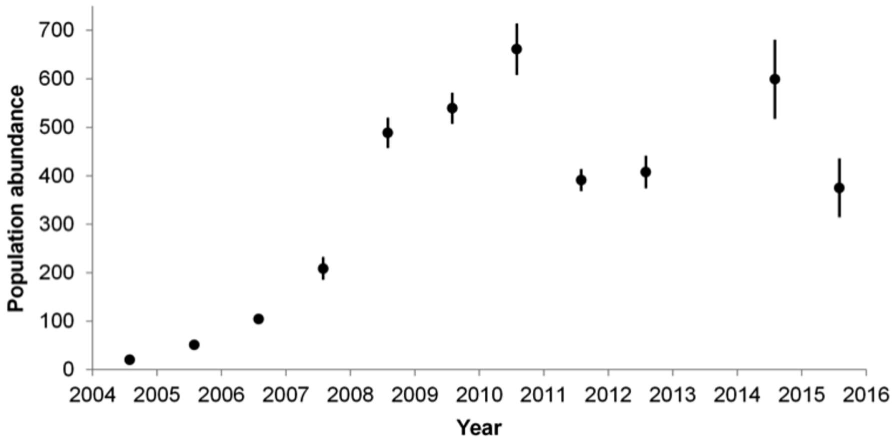

Figure 2. Modified Chapman bias-corrected Lincoln-Petersen mark-resightings population abundance estimates with 95% confidence intervals (CI) for Laysan Teal at Midway Atoll, Hawai'i, for the years 2007–2015. Before 2007 all adults were given radio transmitters, so the abundance was known exactly for those 3 years. No abundance estimate was generated for 2013 because no surveys were undertaken in this calendar year.

Index validation

The percentage of birds observed (*i.e.* the ratio of the maximum seasonal count to the LP estimate) averaged 62% (s.d. = 10%, range = 47–74%) during the 6-month non-breeding season and 33% (s.d. = 11%, range = 15–43%) during the 6-month breeding season. In the ANCOVAs the interaction of count and season was not significant ($P > 0.05$, n.s.), so the interaction term was dropped from the model. Models fitted with the main effects of year and season when comparing the direct counts to LP estimated counts explained most of the variability in the data (maximum seasonal count $r^2 = 0.77$, $n = 12$; maximum random count $r^2 = 0.81$, $n = 12$). The maximum seasonal count and the maximum random count models each had significant terms for count ($P = 0.0005$ and $P = 0.0004$, respectively) and season ($P = 0.0088$ and $P = 0.0006$, respectively). Since statistical power and results of the ANCOVAs were qualitatively similar between analyses, we present results only from the maximum random count analysis. If we ignore season, the correlation between maximum counts and the LP estimates (or the slope in the simple linear regression) is not significant ($r = 0.05$, $n = 12$, $P = 0.09$, n.s.). The relatively large correlations shown in Fig. 3 imply that the counts within a season are a good index of population abundance.

A seasonal Laysan Teal population estimate (y), and the confidence bounds around the estimate ($t_{\alpha/2,n-2}\sqrt{MSE\ 1/n+\frac{(x-\bar{x})^2}{\Sigma_i(x_i-\bar{x})^2}}$; Sokal & Rohlf 1995), can be calculated based on a season's highest count (x) and season-specific equations from the most appropriate model (Fig. 3). Equations to estimate season-specific abundance apply only to seasonal maximum counts derived

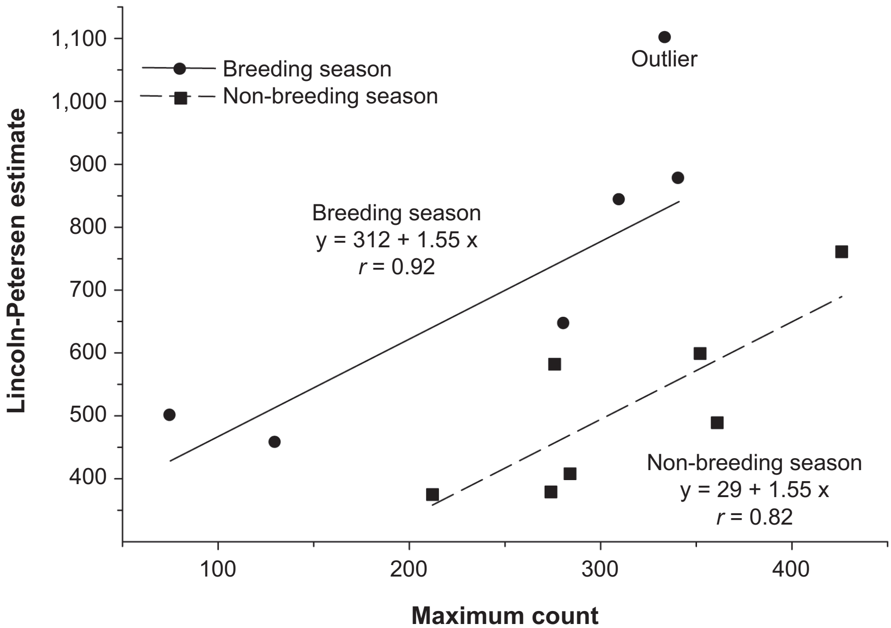

Figure 3. Modified Chapman bias-corrected Lincoln-Petersen estimates *vs.* maximum observed counts for Laysan Teal on Midway Atoll, Hawai'i, for the years 2007–2015 assuming equal slopes among seasons (*i.e.* reduced model). Seasons (breeding, non-breeding) are shown separately, with the regression line for each season also plotted separately. One outlier (LP = 1,100) was excluded from this analysis. Equations to estimate season-specific abundance apply only to seasonal maximum counts within the 8-year observed range of maximum counts (breeding = 75–341 birds; non-breeding = 212–426 birds).

from surveys conducted once or twice monthly within the observed count range (breeding season = 75–341 birds, non-breeding season = 212–426 birds); survey results outside of these ranges are not validated. Season-specific regression equations for deriving abundance from the maximum count and 95% confidence bounds are as follows:

Non-breeding:

$$y = 29.41 + 1.55x \pm$$
$$2.571\sqrt{6824.58\ 1/7 + \tfrac{(x-312.14)^2}{30520.86}}$$

Breeding:

$$y = 312.09 + 1.55x \pm$$
$$3.182\sqrt{5633.08\ 1/5 + \tfrac{(x-227.40)^2}{55313.20}}$$

Discussion

The progression for monitoring the translocated population first included intensive radio-tracking during the early post-release period (2004–2007), followed by population monitoring using mark-resightings and recapture data combined with systematic counts (in 2007–2015).

Now, a direct standardised survey count, without requiring the capture and marking of birds, will be a substantially less labour- and data- intensive approach for monitoring Laysan Teal abundance at Midway. A similar study undertaken on Laysan Island (Reynolds *et al.* 2015b) showed that the highest detection of birds also occurred in the non-breeding season (autumn and winter), and that both seasons showed a high correlation (r = 0.82–0.92) between estimated abundance and the maximum counts. The equations for the linear regressions, along with equations to estimate 95% confidence intervals, allow for a simpler survey approach that can utilise previous time series data from surveys and provide simpler analyses for managers than previously applied models (Marshall 1992; Reynolds & Citta 2007; Seavy *et al.* 2009).

The population at Midway Atoll grew to a total of 661 birds (95% CI = 608–714) in 2010, then a population decline of 38% was observed between 2010 and 2012 after the 2011 Tōhoku earthquake-generated tsunami. By 2014, the population had begun to recover from the tsunami (LP estimate = 599 birds, 95% CI = 518–680), but following a severe botulism outbreak during 2015 the population again experienced a 37% decline. Data indicate that the Midway population, like the founding Laysan Island population, is susceptible to catastrophic population declines (Seavy *et al.* 2009), and consistent standardised monitoring using simple counts can be used to evaluate population status over the long-term. For 2015, we estimated that there were 314–435 (95% CI for population estimate, point estimate = 375) teal on Midway Atoll, or approximately 50% of the global population. In comparison, the 2012 estimate for Laysan Island was 339 individuals (95% CI = 265–413; Reynolds *et al.* 2015b).

Future surveys and estimates:

Our model relies on at least one or two high-quality atoll-wide surveys per month at Midway for this count index to have utility for estimating population abundance or detecting population declines. Care should be taken if using count data during transition months between breeding and non-breeding seasons because our models are based on detection probabilities that may vary in relation to bird behaviour, which changes once breeding commences (Reynolds *et al.* 2015b). Linear regression could be used to validate long-term monitoring indices and evaluate population status of other reintroduced populations that are also marked and systematically monitored. Non-overlap between 95% CIs in any two years indicates a significant difference ($P < 0.05$) in population abundance recorded in those two years, which would serve to alert managers to major changes in the population trajectory, whether it be increasing or in decline.

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Photograph: Laysan Teal with Hawaiian Monk Seal *Neomonachus schauinslandi* on Laysan Island, by Matthew Chauvin.

