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2017-2018 PALILA ABUNDANCE ESTIMATES AND TREND

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ABSTRACT

The palila (*Loxioides bailleui*) population was surveyed annually from 1998–2018 on Mauna Kea Volcano to determine abundance, population trend, and spatial distribution. In the latest surveys, the 2017 population was estimated at 1,177–1,813 birds (point estimate: 1,461) and the 2018 population was estimated at 778–1,420 (point estimate: 1,051). Only two palila were detected outside the core survey area during a mountain-wide survey in 2017, suggesting that most, if not all, palila inhabit the western slope during the annual survey period. Since 1998, the size of the area containing palila detections on the western slope did not show a significant change, suggesting that the range of the species has remained stable; although this area represents only about 5% of its historical extent. During 1998–2003, palila numbers fluctuated moderately (coefficient of variation [CV] = 0.20). After peaking in 2003, population estimates declined steadily through 2011; since 2010, estimates have continued to decline at a slower rate. The average rate of decline during 1998–2018 was 168 birds per year with very strong statistical support for an overall declining trend in abundance. Over the 21-year monitoring period, the estimated rate of change equated to a 76% decline in the population.

INTRODUCTION

The palila (*Loxioides bailleui*) is an endangered, seed-eating, finch-billed honeycreeper (a distinct group within family Fringillidae: subfamily Drepanidinae) found only on Hawai`i Island. Once occurring on the islands of Kaua`i and O`ahu, as well as Mauna Loa and Hualālai volcanoes of Hawai`i, palila are now found only in subalpine, dry-forest habitats on Mauna Kea (Banko *et al.* 2002a). Previous analyses showed that palila numbers fluctuated throughout the 1980s and 1990s, but since 1998 palila have declined and they appear to have declined steadily since 2003 (Jacobi *et al.* 1996, Leonard *et al.* 2008, Banko *et al.* 2009, Gorresen *et al.* 2009, Banko *et al.* 2013, Camp *et al.* 2016).

Palila tend to move up and down the western slope of Mauna Kea seasonally as they track the availability of their main food, seeds of the endemic māmane (*Sophora chrysophylla*) tree (Hess *et al.* 2001). During population surveys, usually in January, māmane seedpods are most abundant at higher elevations, but seedpod abundance increases at lower elevations by May (Banko *et al.* 2002b). Although the distribution of palila shifts in response to food availability, the areas that are occupied seasonally overlap extensively and the area that is surveyed each winter provides a stable and representative basis for evaluating population abundance and trends.

The aim of this report is to update abundance estimates and population trends for the palila since 1998, based on the 2017 and 2018 surveys. Additional transects were added to the original Hawaiian Forest Birds Survey (HFBS) transects in 1998 to produce a more precise population estimate and provide more complete coverage of the palila distribution during the survey period (Johnson *et al.* 2006).

METHODS

Bird Sampling

Since 1980, 95% of the palila population has occurred in a 64.4 km² area on the southwestern slope of Mauna Kea (Scott *et al.* 1984, Banko *et al.* 2013; Figure 1). We refer to this area hereafter as the “core survey area.” During 31 January–16 February 2017 and 6–15 February 2018, point-transect sampling was conducted on Mauna Kea to estimate palila abundance and range. In 2017, 13 bird survey transects inside the 64.4 km² palila core survey area (transects 101–108, 122–126, Figure 1), and an additional 18 transects outside (transects 109–116, 119–121, and 127–133, Figure 1, Table 1), were surveyed one or more times. Prior to 2008 surveys were conducted mountain-wide, but lack of detections outside the southwest slope led to focusing survey effort to the core survey area, with the intent to survey the entire mountain every five years, starting in 2012 (D. Leonard, pers. comm.). Therefore, the 2017 survey encompassed the whole mountain. In 2018, the same 13 transects within the core survey area were surveyed one time. In 2017, a majority of the stations ($n = 389$) within the core survey area were counted twice, while 23 stations were counted once and eight stations were counted three times. In 2018, all stations ($n = 419$) in the core area were counted once. At stations outside of the core area in 2017, the western or lower extensions of transects 101, 124, and 125 were sampled twice (54 stations; 108 counts; not shown on Figure 1), with all other stations around Mauna Kea were counted once (313 stations; 313 counts). In 2018, one of the 97 stations outside of the core area was counted twice for a total of 98 counts. In general, multiple counts provide more data and improve precision of results, and repeated counts in the same year allow for explicit estimate of within-year variation (Camp *et al.* 2016), but they are not required to produce useful abundance estimates.

Most Hawaiian forest bird surveys last eight minutes (Camp *et al.* 2009), however, six minutes is used for palila counts because their woodland habitat is more open than mesic and wet forest habitats, allowing for easier and more rapid detections. Counts commenced at sunrise and continued up to four hours (approximately 11:00 HST). During each count, trained and calibrated observers recorded the species, detection type (heard, seen, or both), and distance of each bird from observer. Time of sampling and weather conditions (cloud cover, rain intensity, wind strength, and wind gust strength [hereafter gust strength]) were also recorded, and surveying was postponed when conditions hindered the ability to detect birds (wind and gust strengths > 20 kph or heavy rain).

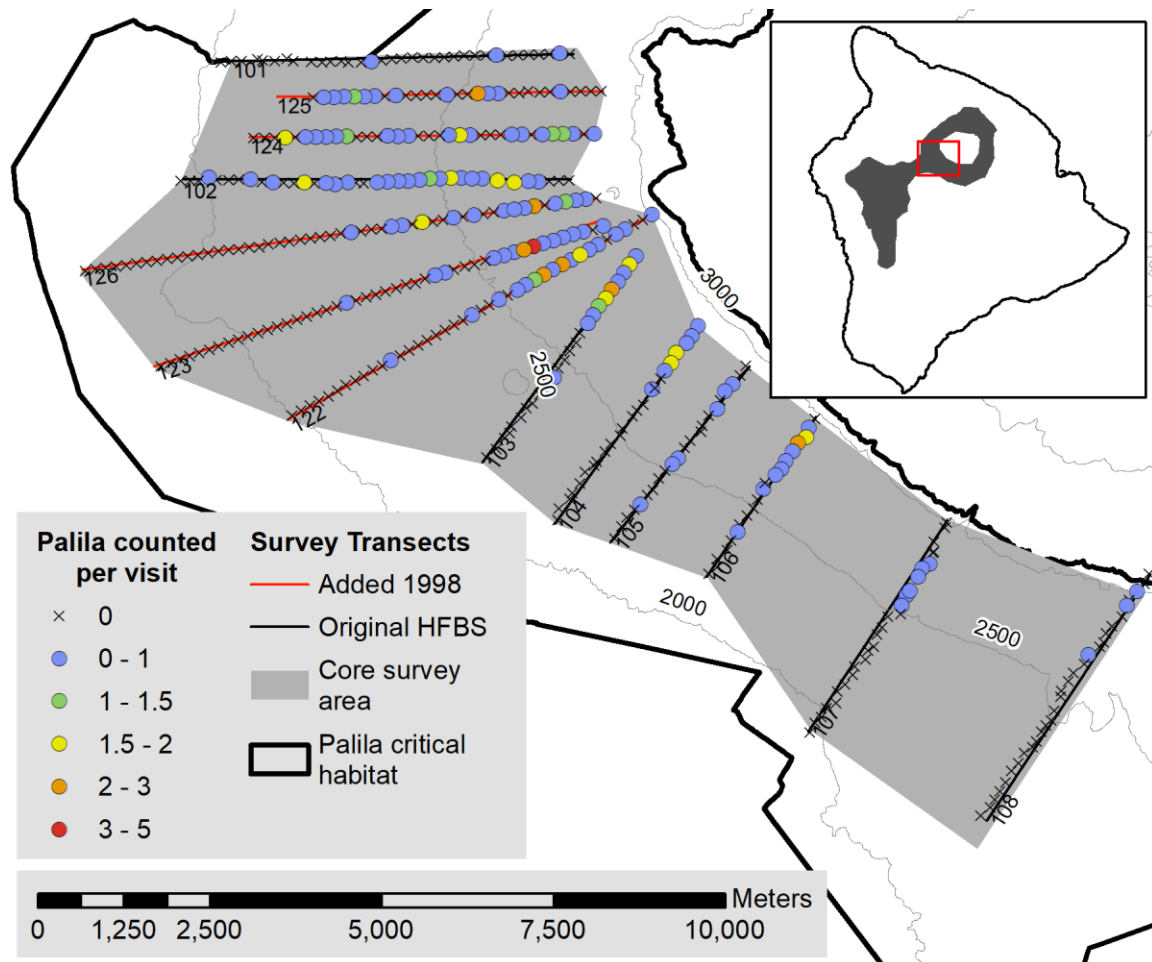


Figure 1. Palila detected per visit across 2017–2018 surveys (mean 2.9 visits / station, minimum 2, maximum 4 for both years combined). × symbols mark stations where no palila were detected regardless of survey effort. The shaded region makes the survey area, and is the area of inference used to produce abundance estimates. Lines depict the original HFBS transects (black) plus those added in 1998 (red). The inset map shows the historical palila range on Hawai`i Island.

Table 1. Number of transects and stations sampled by year, inside and outside the core survey area.

Year	Inside core Survey Effort			Outside core Survey Effort	
	Transects	Stations	Counts	Stations	Counts
1998	12	355	358	186	186
1999	13	417	418	192	206
2000	13	418	428	224	224
2001	13	416	417	223	223
2002	13	417	417	270	270
2003	13	404	410	258	258
2004	13	399	399	244	244
2005	13	403	428	352	352
2006	13	386	386	353	353
2007	13	408	412	253	253
2008	12	387	387	53	53
2009	13	416	422	0	0
2010	13	415	420	0	0
2011	13	411	432	0	0
2012	13	420	843	426	426
2013	13	418	889	0	169
2014	13	407	817	115	0
2015	13	420	839	63	125
2016	13	420	837	79	97
2017	13	420	825	367	421
2018	13	419	419	97	98

Abundance Estimation

Distance analysis fits a detection function to estimate the probability of detecting a bird at a given distance from the observer. This detection function is fitted with covariates, accounting for the effect of the observer, detection type, weather conditions, year, and block years (pooled years in four blocks). With each additional year of data, estimates of these effects become more precise and the improved detection function tends to cause population estimates of previous years to change slightly.

Density estimates (birds/km²) were calculated from point-transect sampling data using program DISTANCE, version 7.1, release 1 (Thomas *et al.* 2010). The 2017 and 2018 data were pooled with detections from previous surveys since 1998. Candidate models were limited to half-normal and hazard-rate detection functions with expansion series of order two (Buckland *et al.* 2001:361, 365). Survey effort in a given year was adjusted by the number of times the station was counted in that year. To improve model precision, potential sampling covariates were incorporated in the multiple covariate distance sampling engine of DISTANCE (Thomas *et al.* 2010). Covariates included the weather conditions, time of sampling, type of detection, observer, and year of survey. Right-tail truncation was set at 87.5 m, the distance where the detection probability was approximately 10%. This procedure facilitates

modeling by deleting outliers and reducing the number of adjustment parameters needed to modify the detection function. The detection probability model selected was the one having the lowest Akaike's Information Criterion (AICc) corrected for small sample size (AICc; Buckland *et al.* 2001, Burnham and Anderson 2002; Figure 2, Table 2). Annual population densities for each survey were calculated using the global detection function, and the pooled data was post-stratified by year and location (inside/outside core survey area). The 95% confidence intervals for the annual density estimates were derived from the 2.5th and 97.5th percentiles using bootstrap methods in DISTANCE for 999 iterations (Buckland *et al.* 2001, Thomas *et al.* 2010). Population abundance estimates were the product of the density estimate times the area of the core survey area (64.4 km²).

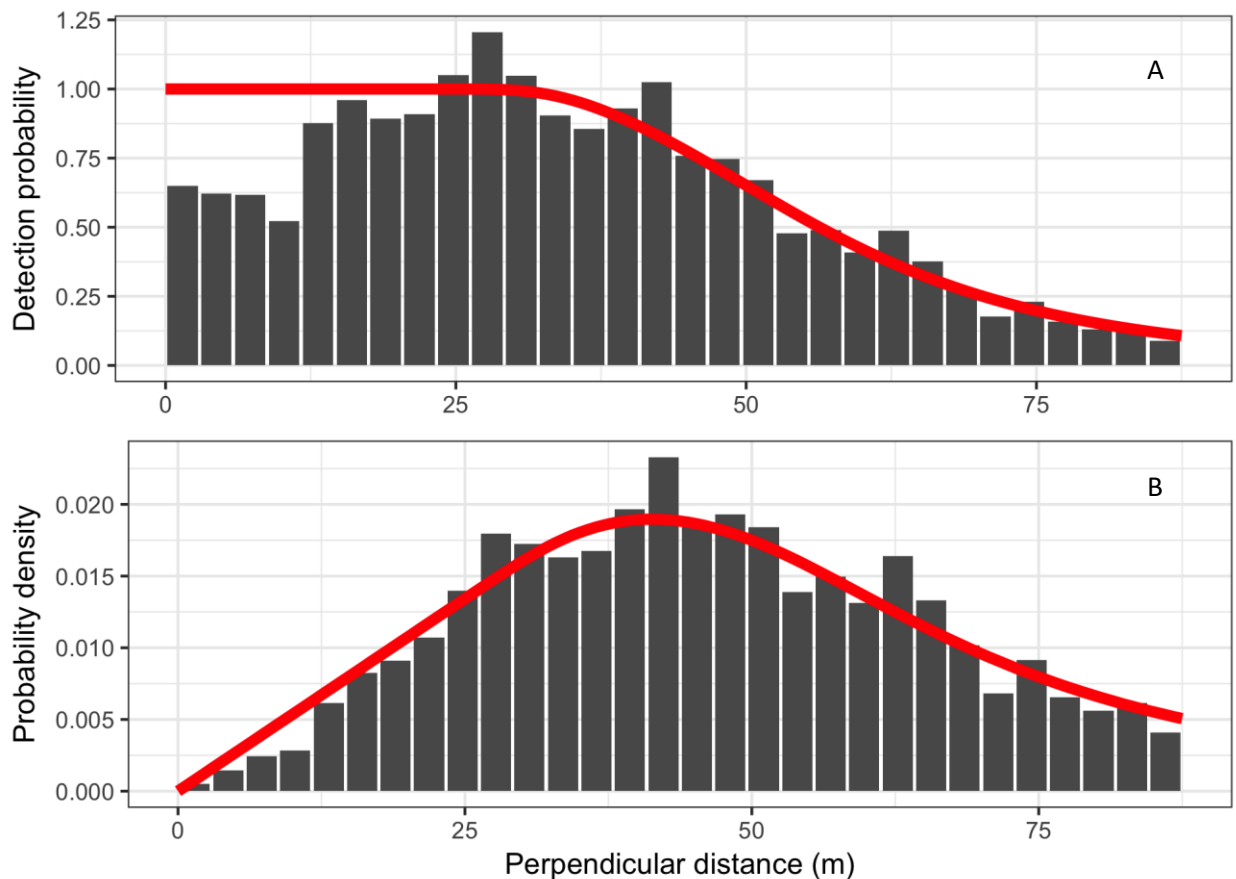


Figure 2. Hazard-rate detection function (A), and probability density (B) of the best-fit detection model. There are no expansion series and blocked years are included as a detection covariate. The model was fit using palila distance data pooled across all surveys from 1998 to 2018. Data were truncated at 87.5 m. Figure 2B shows how the observed probability distribution of distances (grey bars) matches the modeled detection function, especially at short distances.

Table 2. Results of fitting 19 detection function models to the 1998–2018 palila distance histogram. ΔAICc is the difference in AICc scores between each model and the overall best-fit model, and w is the discrete model probability. The hazard-rate detection with blocks of years as a factor was chosen (bold) as the best model.

Model ^{1,2}	# parameters	Ln(Likelihood)	AICc	ΔAICc	w
H-rate Key YearBlock(f)	5	23138.8	46287.5	0	0.99
H-rate Key Year(f)	22	23126.7	46297.7	10.2	0.01
H-rate Key VBDetType	3	23154.1	46314.2	26.7	0
H-rate Key DetType	4	23153.6	46315.2	27.7	0
H-rate Key HBDetType	3	23273.4	46552.7	265.2	0
H-rate Key Gust	6	23279.5	46571.1	283.6	0
H-rate Key Wind	5	23286.9	46583.8	296.3	0
H-rate Key Obs	5	23293.5	46597.1	309.6	0
H-rate Key	2	23302.3	46608.6	321.1	0
H-rate Key Rain	3	23301.6	46609.1	321.6	0
H-rate Key Time	3	23303.7	46613.5	326.0	0
H-rate Key Cloud	4	23304.0	46616.1	328.5	0
H-norm Key	1	23374.5	46751.1	463.6	0

¹Models are hazard-rate (H-rate) and half normal (H-norm); adjustment terms are cosine (Cos), simple polynomial (S-poly) and hermite polynomial (H-poly); and covariates are cloud cover (Cloud), detection type (DetType = heard, visual or both, HBDetType = pooled heard and both, VBDetType = pooled visual and both), gust strength (Gust), observer (Obs), time of detection (Time), wind strength (Wind), year (Year; continuous = c and factor = f), and block year (pooled years in four blocks; continuous = c and factor = f).

²Models H-norm Cos, H-norm H-poly, H-rate Cos, H-rate S-poly, H-rate Key Year(c), and H-rate Key YearBlock(c) failed to converge.

Trend Detection

The trend in palila abundance was assessed in two different ways. The bootstrap samples generated to estimate uncertainty in population abundance were also used to analyze the long-term population trend (1998–2018) with a log-linear regression model. The evidence of a trend was derived from the bootstrap distribution of slopes, following Camp *et al.* (2015). Diagnostics demonstrated that the log-linear regressions of trends met all model assumptions (visual inspection of residual plots; Shapiro-Wilk normality test, $W = 0.095$, $P = 0.36$), except that temporal autocorrelation was evident. We looked at four additional models with autoregressive structures to see how well they were supported by the observations. The best model was still the simple log-linear regression with 68% of the AIC weight. A lag-one (AR1) model had an AICc 2 units higher and 25% of the weight. An AR2 model had 6% of the weight, and the autoregressive moving average (ARMA)(1, 1) and ARMA(2, 2) models made up the remainder (< 1% of the weights combined). Because of the temporal autocorrelation, to be conservative in estimates

of a trend, and for comparison with the previous estimate we opted to use an AR1 log-linear model to assess for trend.

We also assessed trend using a Bayesian state-space model on the log-scale. This approach partitions model uncertainty into portions attributable to observation error (due to random noise in the environment affecting detectability and measurement) and process error (due to stochastic fluctuations in population outside of the overall trend). Such a state-space model can be interpreted as a biologically informed smoother, and provides annual estimates consistent with the observed inter-annual noise. We used diffuse priors for the model parameters: a normal distribution with mean 0 and standard-deviation of 10 for the mean annual population change and exponential priors with mean 0.1 as priors for the standard deviation of slope and the observation error.

The state-space model was fit using Stan (Carpenter *et al.* 2016) run from an R environment (R Core Team, 2018) using the rstan package (Stan Development Team 2018). The model parameters were estimated from 1,000 iterations for each of four chains (i.e., model runs) after first discarding 500 iterations as a “warm-up” period. The four chains were pooled (4,000 total samples) to calculate the posterior distribution. Gelman-Rubin convergence statistics for all estimated parameters were below 1.001; less than the 1.1 threshold that indicates convergence (Gelman and Rubin 1992).

Both methods were assessed in an equivalence-testing approach (Camp *et al.* 2008), using the observed distribution of slopes of the bootstrap distribution, and the posterior distribution of the slope from the Bayesian state-space model. We chose biologically meaningful thresholds for the overall population trend as a 25% change in the population over a 25-year period (annual rate of change equal to -0.0119 and 0.0093 on the log-scale). A biologically meaningful trend occurs when the posterior probability distribution of the slope lies outside the equivalence region, whereas a negligible trend occurred when the slope is within the equivalence region. An inconclusive result occurs when small sample size and high variation in estimates results in the posterior distribution of the slope providing weak evidence in all three outcomes (increasing, stable, and decreasing).

RESULTS

Abundance

Within the 64.4 km² core survey area of Palila Critical Habitat on the southwestern flank of Mauna Kea, the number of palila detected decreased by 22% between 2016 and 2017 (319 in 2016 and 248 in 2017). There was another 60% decrease in palila detections between 2017 and 2018 (248 in 2017 and 99 in 2018). An additional 10 palila were detected on the lower extensions of transects 124 and 125 (seven in 2017 and three in 2018; Figure 1, Table 3). Two palila were also detected on the north slope in 2017 (transects 117 and 130; Table 3). The model that best fit the distance histogram was a hazard-rate detection function with no adjustment terms, and blocked years (pooled years in four blocks; Figure 3) as a covariate (Figure 2, Table 2). In 2017, the palila population in the core survey area was estimated at 1,177–1,813 birds (point estimate: 1,461; Figure 4, Table 3). In 2018, the palila population in the core area was estimated at 778–1,420 (point estimate of 1,051).

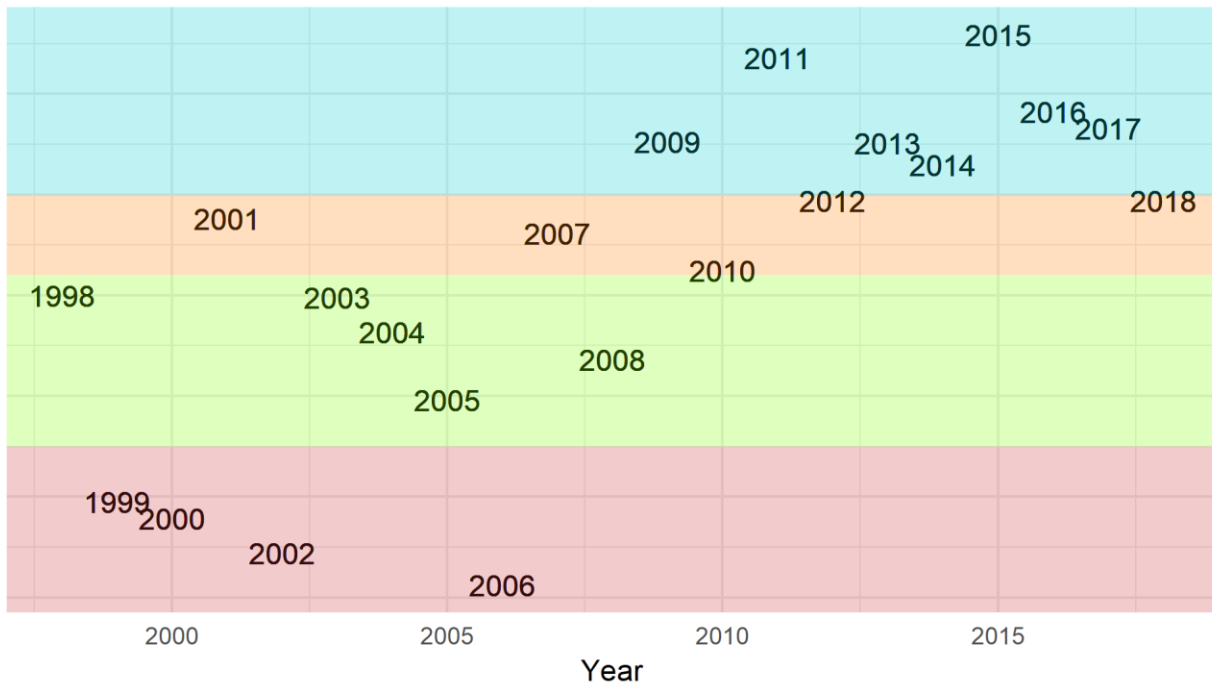


Figure 3. Four blocks of years grouped by their point estimate from the hazard-rate detection function with year as a factor model. Colored bands show years with a similar effect. Vertical axis is the relative coefficient value for that year's influence on the detection function.

Trend

Between 1998 and 2003, palila numbers fluctuated moderately (coefficient of variation [CV] = 0.20 change from local minimum to maximum value; Figure 4). After peaking in 2003, palila population estimates declined steadily through 2011. During 2010–2018, estimates also fluctuated moderately about the 2011 level (CV = 0.24) with a local peak in 2012. The observed average rate of decline during 1998–2018 has been 168 birds per year. The bootstrap log-linear AR1 regression model showed very strong evidence (posterior probability; $P = 1.0$) of overall declining trend in palila abundance since 1998. The log-linear state-space model (Figure 5) also shows very strong evidence of a decline with 92% posterior probability showing a declining trend, with only 4% each probability of a negligible or increasing trend. The regression model fit the observed data with a Bayesian R^2 of 0.86, and simulated model runs with log-normal random error were greater than observed errors 56% of the time, where a value of 50% would indicate the observed error fit a log-normal distribution exactly (Gelman *et al.* 2004). Over the 20-year monitoring period, the estimated rate of change equated to a 76% decline in the palila population.

Table 3. Annual palila detections and population estimate parameters. Detections are given for palila recorded inside the core survey area and for stations outside the core area during six-minute counts. Population parameters include the population estimate, % coefficient of variation (CV), standard error (SE), and lower and upper limits of the 95% confidence interval inside the core survey area. From 2012–2017 most stations in the core were surveyed at least twice, so the decreased number of detections between 2017 and 2018 (from 248 to 99) is partly due to reduced sampling effort.

Year	# Detections inside	# Outside core area detections	Estimate	%CV	SE	Lower Limit	Upper Limit
1998	315	2	4412	9.38	414	3671	5303
1999	389	0	5069	8.53	432	4288	5992
2000	242	12	2970	9.74	289	2454	3596
2001	331	4	4369	9.28	405	3642	5241
2002	339	9	4395	8.41	370	3726	5184
2003	442	4	5625	8.23	463	4786	6610
2004	371	8	4786	7.77	372	4110	5575
2005	315	0	4038	9.17	370	3373	4834
2006	267	15	3721	9.49	353	3089	4483
2007	210	3	2805	10.01	281	2305	3414
2008	192	0	2560	10.37	265	2089	3137
2009	187	na	2308	11.34	262	1849	2883
2010	151	na	1546	11.75	182	1228	1946
2011	119	na	1299	13.79	179	992	1702
2012 ¹	362	0	2063	10.30	212	1686	2525
2013 ²	337	na	1696	8.29	141	1442	1996
2014 ³	351	4	1960	9.13	179	1639	2345
2015 ⁴	192	1	1076	12.7	137	839	1380
2016 ⁵	319	4	1861	11.01	205	1500	2309
2017 ⁶	248	9	1461	11.03	161	1177	1813
2018	99	3	1051	15.40	162	778	1420

¹Of 362 total detections, 194 recorded on first count, 168 recorded on subsequent counts.

²Of 337 total detections, 178 recorded on first count, 159 recorded on subsequent counts.

³Of 351 total detections, 163 recorded on first count, 188 recorded on subsequent counts.

⁴Of 192 total detections, 99 recorded on first count, 93 recorded on subsequent counts.

⁵Of 319 total detections, 178 recorded on first count, 141 recorded on subsequent counts.

⁶Of 248 total detections, 138 recorded on first count, 110 recorded on subsequent counts.

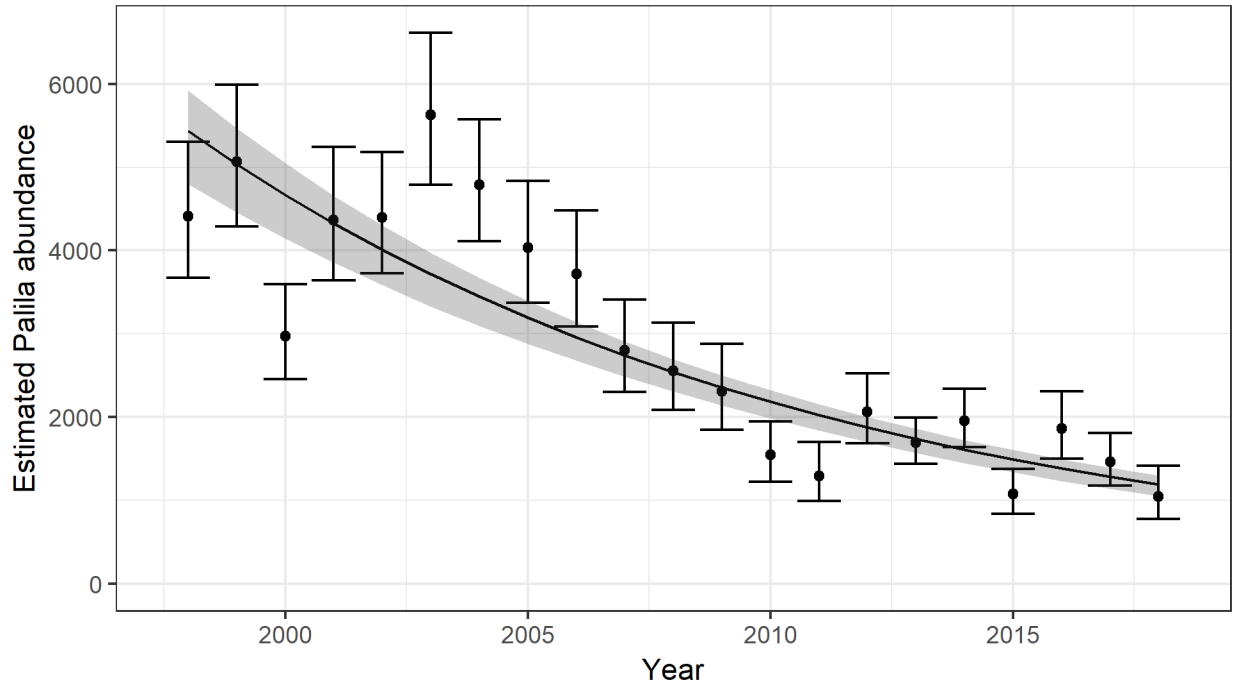


Figure 4. Annual palila population estimates from 1998 through 2018 inside the core survey area on the western slope of Mauna Kea. The line represents the best fit log-linear regression (with lag-one autocorrelation), error bars show 95% bootstrap intervals around the point estimates, and the shaded area shows the 95% band around bootstrapped regression.

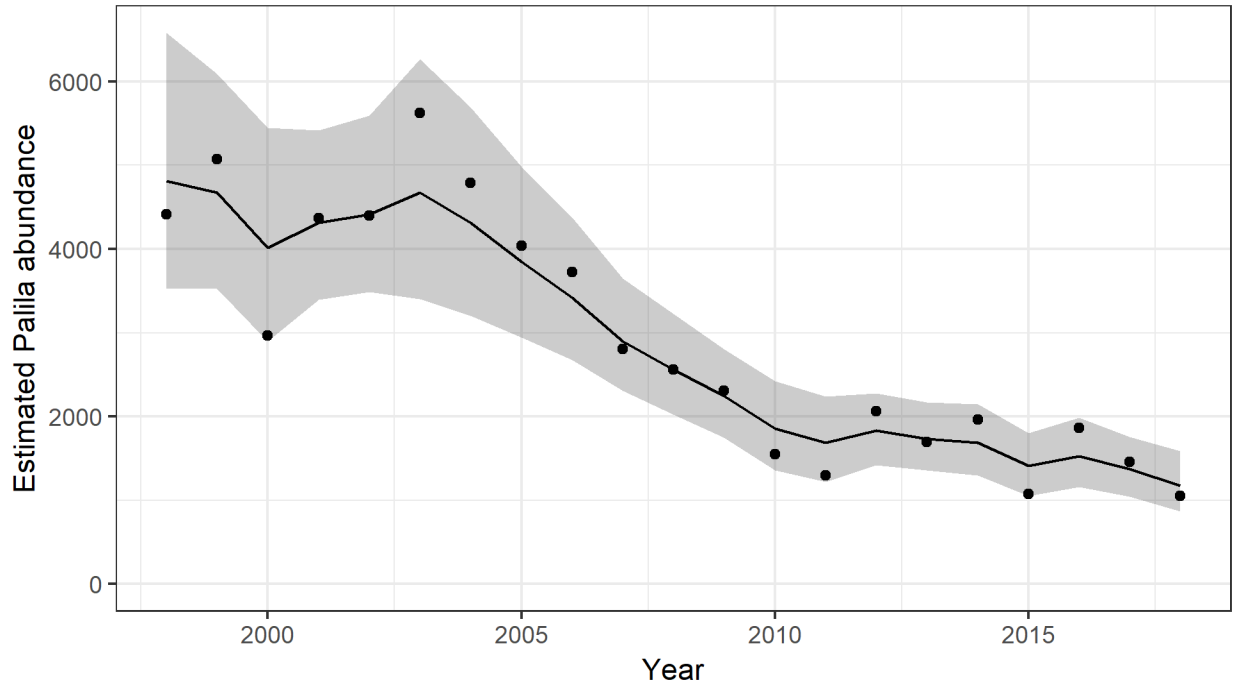


Figure 5. State-space model estimates of palila abundance from 1998 through 2018. Points are estimates from DISTANCE models, the line shows the median estimate from the Bayesian posterior distribution of abundance, and the shaded area shows the 95% credible interval of abundance posteriors.

CONCLUSIONS

The 2018 palila population was estimated at 778–1,420 birds (point estimate of 1,051). There was very strong evidence that the palila population declined after 1998, with the greatest decline occurring after 2003. The average rate of decline during 1998–2018 was 168 birds per year, resulting in a 76% decline in the population over the 20-year period.

Trends assessment shows very strong evidence that the palila population is in decline. The most optimistic interpretation shows only an 8% chance of it being stable or increasing. Despite the 2017 survey having almost twice the survey effort in the core area (825 vs. 419 survey visits) the width of the bootstrap uncertainty interval for the two years was almost the same (636 vs 642).

MANAGEMENT IMPLICATIONS

The palila population has been in decline since 1998. The 2018 abundance estimate is the lowest published since regular surveys began in 2018, although Johnson *et al.* (2006) attempted to correlate historical survey methods with modern techniques, and suggests it may have been lower during the mid-1980's. A naïve projection of the decline since 2009 would predict the population will half the current abundance in 10 years and become extinct in another 20.

In the 2017 mountain-wide survey, no palila were detected along the southeast and eastern slopes of Mauna Kea, where they have been detected historically. There were two detections on the north slope of Mauna Kea where wild birds were translocated (1997–1998, 2004–2006) and captive-reared birds were released (2003–2005, 2009; Banko and Farmer 2014). This area is adjacent to and upslope of the Pu`u Mali mitigation area where former pasture land has been taken out of grazing and reforestation of native trees has begun. There were also palila detected below the core area on the southwest slope (lower stations on transects 124 and 125), another area where former pasture lands are subject to a reforestation project. Despite these hopeful signs that reforestation has begun to rehabilitate former pasture land into palila habitat, palila range has been relatively constant across the annual surveys and is only about 5% of its historical extent (Figure 1 inset; Banko *et al.* 2013).

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