



Technical Report HCSU-033

## PALILA ABUNDANCE ESTIMATES AND TREND

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October 2012



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This product was prepared under Cooperative Agreement CA03WRAG0036 for the Pacific Island Ecosystems Research Center of the U.S. Geological Survey.



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## BACKGROUND

The Palila (*Loxioides bailleui*) is an endangered, seed-eating, finch-billed honeycreeper found only on Hawai`i Island. Once occurring on the islands of Kaua`i and O`ahu and Mauna Loa and Hualālai volcanoes of Hawai`i, Palila are now found only in subalpine, dry-forest habitats on Mauna Kea (Banko *et al.* 2002). Previous analyses showed that Palila numbers fluctuated throughout the 1980s and 1990s but declined rapidly and steadily since 2003 (Jacobi *et al.* 1996, Leonard *et al.* 2008, Banko *et al.* 2009, Gorresen *et al.* 2009, Banko *et al.* in press).

The aim of this report is to update abundance estimates for the Palila based on the 2012 surveys. We assess Palila trends over two periods: 1) the long-term trend during 1998–2012 and 2) the short-term trajectory between 2003 and 2012. The first period evaluates the population trend for the entire time series since additional transects were established (Johnson *et al.* 2006). These additional transects were established to produce a more precise population estimate and provide more complete coverage of the Palila range. The initial year for short-term trajectory was chosen subjectively to coincide with the recent decline in the Palila population. Additionally, stations in the core Palila habitat were surveyed on two occasions in 2012, thus allowing us to address the question of how repeat samples improve estimate precision.

## METHODS

### Bird Sampling

Between 24 January and 8 February 2012, point-transect sampling was conducted to estimate Palila abundance and range. All 33 bird survey transects on Mauna Kea, including transects inside and outside the 64.4 km<sup>2</sup> area of the core Palila population, were surveyed one or more times. The survey yielded 1,269 counts, each lasting six minutes, at 846 stations (Figure 1, Table 1). Most Hawaiian forest bird surveys last eight minutes (Camp *et al.* 2009), however, this shorter count period was used on Mauna Kea because the woodland habitat is more open than mesic and wet forest habitats, allowing for easier and more rapid detection. About half of the stations (440) were counted once, while 396 stations were counted twice, three stations were counted three times, and seven stations were counted four times. Transects in the core area included 101–108 and 122–126, and a majority of the stations on these transects were counted two or more times (486 stations; 909 counts). Transects outside the core area, 109–121 and 127–133, were sampled once (360 stations). Counts commenced at sunrise and continued up to four hours. During each count, trained and calibrated observers recorded the species, detection type (heard, seen, or both), and distance of each bird from the survey station center. Time of sampling and weather conditions (cloud cover, rain, wind, and wind gust [hereafter gust]) were also recorded, and surveying was postponed when conditions hindered the ability to detect birds (wind and gust > 20 km per hour or heavy rain).

### 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, and a global effect for the year. With each additional year of data, estimates of these effects become more precise and the improved detection function may cause population estimates of previous years to change slightly.

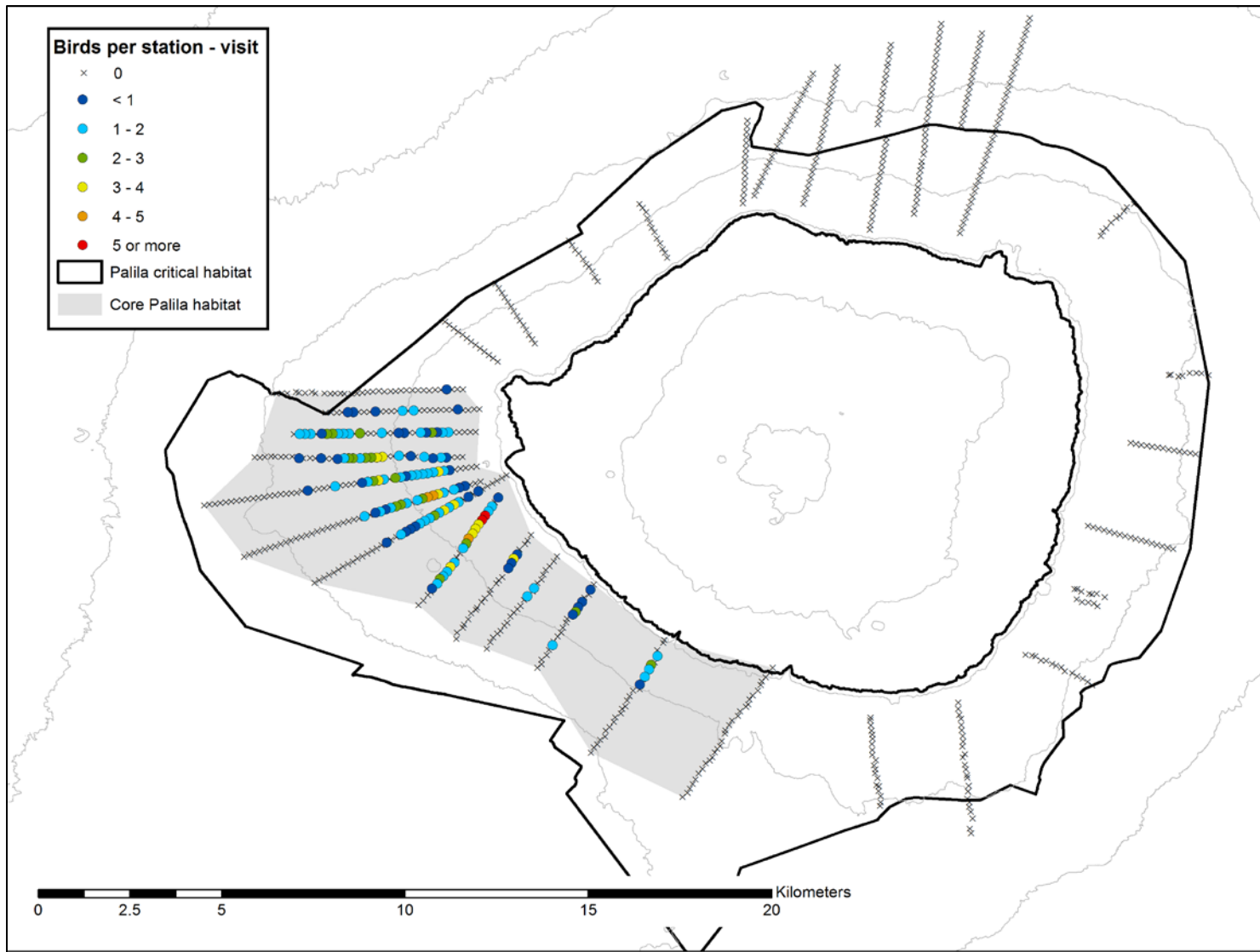


Figure 1. Palila detected on transect stations during the 2012 Mauna Kea bird survey. The core portion of the Palila population is demarcated in the shaded area.

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

Year	Inside core			Outside core		
	Transects	Stations	Counts	Transects	Stations	Counts
1998	12	355	357	14	186	186
1999	13	414	418	14	192	212
2000	13	418	424	17	224	224
2001	13	414	416	17	221	223
2002	13	416	417	20	270	271
2003	13	403	403	20	258	258
2004	13	397	397	18	240	251
2005	13	402	428	20	340	351
2006	13	386	398	20	323	356
2007	12	387	412	20	256	256
2008	12	386	432	0	0	0
2009	13	416	416	0	0	0
2010	13	415	420	0	0	0
2011	13	411	432	0	0	0
2012	13	486	909	20	360	360

Density estimates (birds/km<sup>2</sup>) were calculated from point-transect sampling data using program DISTANCE, version 6.0, release 2 (Thomas *et al.* 2010). The 2012 data were pooled with detections from previous surveys going back to 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). The uniform detection function was not considered because covariates cannot be modeled. Survey effort was adjusted by the number of times the station was counted. To improve model precision, 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 87.0 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 detectability model selected was the one having the lowest Akaike's Information Criterion, when corrected for small sample size (AICc; Buckland *et al.* 2001, Burnham and Anderson 2002). 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 population). The 95% confidence intervals for the annual density estimates were derived from the 2.5<sup>th</sup> and 97.5<sup>th</sup> 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 population (64.4 square kilometers).

### Trend Detection

The trend in Palila abundance was assessed for both short-term and long-term trajectories in the time series using linear regression models. The short-term trajectory coinciding with population declines between 2003 and 2012 was assessed using log-linear regression. The initial point, 2003, was subjectively chosen because the population estimate was the highest of all years. Bayesian regression was used to assess the long-term population trend (1998–2012), where the evidence of a trend was derived from the posterior probability of the slope using a

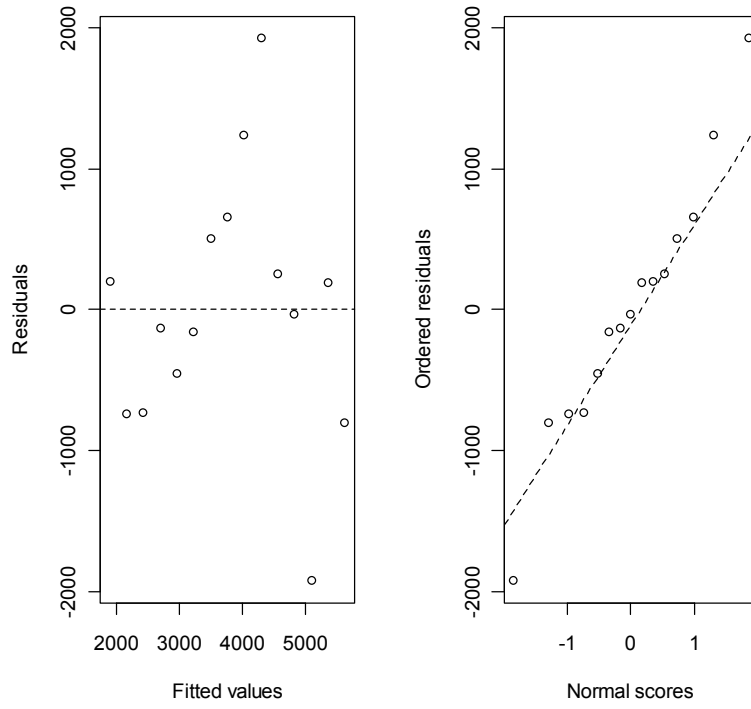


log-link regression model, following Camp *et al.* (2008). Diagnostics demonstrated that the log-linear regressions of trends met all model assumptions, except that temporal autocorrelation was evident (Figure 2). The lag-one autoregressive model had the lowest AIC value and was a substantially better fit than a model that assumed temporal independence (lag-zero; 8.1 AIC units greater). The abundance at time  $t$  was dependent on the previous population abundance at time  $t-1$ , and the simple linear regression model was modified to account for temporal dependence using an autoregressive moving average (ARMA) model. Standard autoregressive models require that the time series is stationary, i.e., no trend exists. ARMA models allow for non-stationary time series where the trend is estimated with the moving average model, while density dependence is accounted for with the autoregressive model. An ARMA model was used to calculate the distribution of the posterior probabilities of the slope in WinBUGS (Lunn *et al.* 2000) in program R (R version 2.7.0; 2008-04-22; The R Foundation for Statistical Computing). The parameter  $\alpha$  describes the density at time  $t = 0$  (i.e., intercept),  $\beta$  is the rate of change (i.e., slope) with each unit increase in time  $t$ , and  $\tau$  equals variance<sup>-1</sup> (i.e., precision). A parameter  $\phi$  describes the covariance coefficient. The parameters  $\alpha$ ,  $\beta$ , and  $\phi$  were given uninformative normal priors, and an uninformative gamma prior was given for  $\tau$ . An uninformative prior distribution was chosen to restrict the posterior distribution to the likelihood. The trends were centered on the year 2005 to reduce multicollinearity (condition where two or more variables are highly correlated). The model parameters were estimated from 500,000 iterations for each of three chains (i.e., model runs) after discarding the first 2,000 iterations (a "burn-in" period). The three chains were pooled (1,500,000 total samples) to calculate the posterior distribution. An equivalence-testing approach was applied to the regression model to incorporate natural variability in the estimates, and abundances were considered equivalent within pre-specified bounds equal to a 25% change in the population over a 25-year period (annual rate of change equal to -0.0119 and 0.0093). Changes in the population were defined as increasing, decreasing, negligible trend (i.e., stable population), or an inconclusive result. An ecologically meaningful trend occurred when the slope lay outside the equivalence region, whereas a negligible trend occurred when the slope lay within the equivalence region. An inconclusive result occurred when the sample size was insufficient to produce precise variation estimates (Camp *et al.* 2008).

### **Repeat Surveys**

Most stations within the core population were sampled twice in 2012. Multiple counts increase the numbers of detections, thereby reducing the total uncertainty in the abundance estimates and improving the overall power to detect population changes. In addition, repeat surveys allow for state-space modeling. State-space modeling can be used to identify observation and process error in the abundance estimates. Process error is the inherent uncertainty in estimating the population due to temporal and spatial variability and natural population fluctuations. Observation error is associated with sampling, detecting birds, and population modeling. Once identified, observation error can be eliminated, allowing the remaining unaccounted for variability to be ascribed to the process error (noise) in the population estimates. Using AIC, the model with covariates of count and year was compared to the model with year alone. This approach compared the fit of each model to the data to determine whether the distributions of the first and subsequent counts differed.

A)



B)

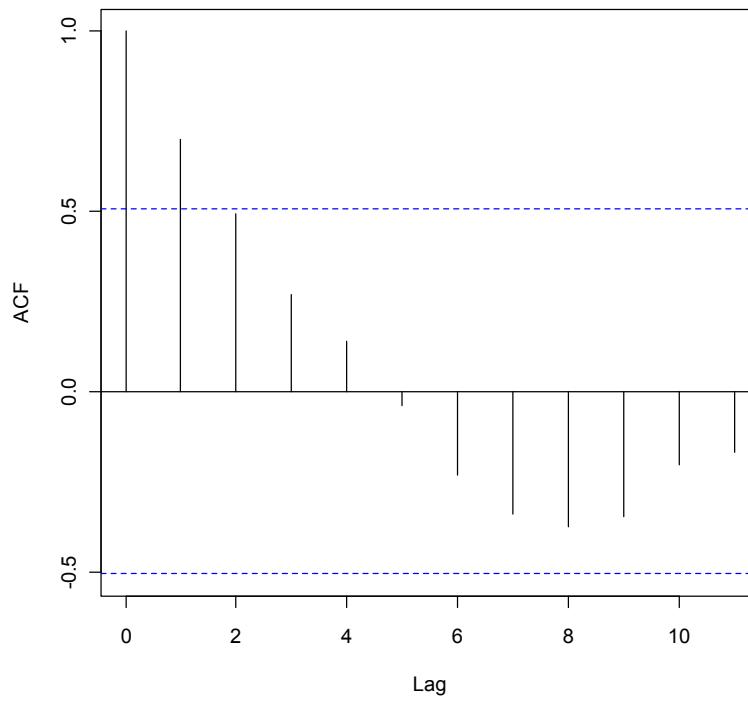


Figure 2. Diagnostics to assess regression model assumptions: panel A) residuals and panel B) autocorrelation.

## RESULTS

### Bird Abundance

The 2012 Palila population was estimated to be between 1,749 and 2,640 birds (point estimate of 2,176; Figure 3, Table 2). This estimate was for the core population on the western side of Mauna Kea and included 411 Palila observations (362 detections during counts). An additional 49 Palila were detected outside of the six-minute count period and were used to fit the detection function but were not used to estimate population abundance. The model that best-fit the distance histogram was a hazard-rate detection function with no adjustment terms and included the covariate detection type (detection types 2 and 4 pooled; Figure 4, Table 3).

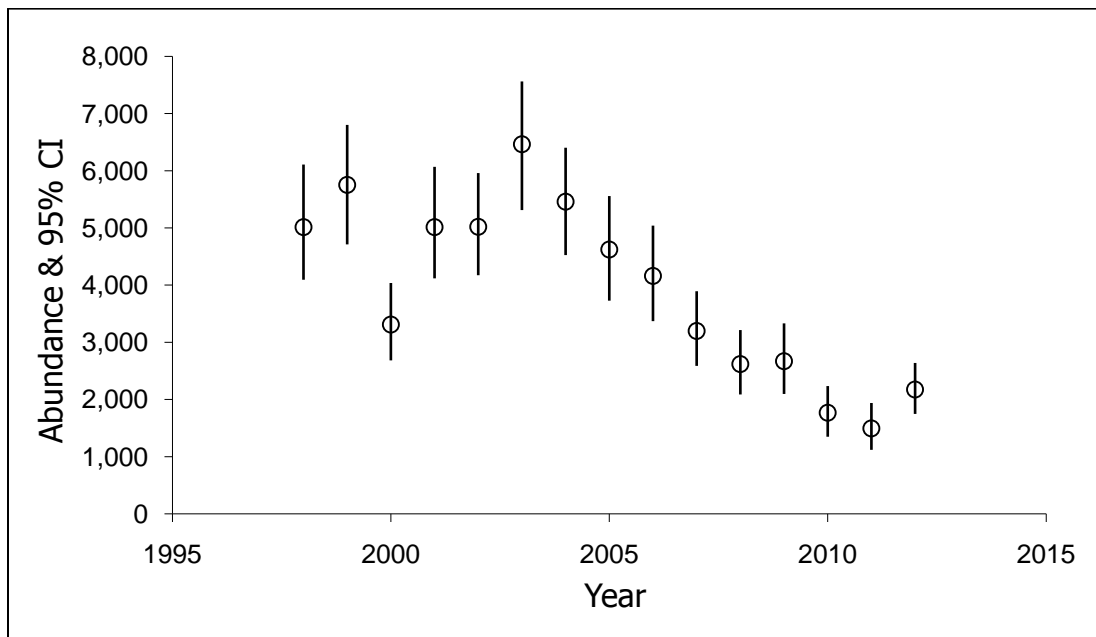


Figure 3. Palila population abundance estimates and 95% confidence intervals (CI) by year inside the core area.

No Palila were detected outside the core population, therefore the population estimate was derived entirely from detections obtained inside the core area during six-minute count periods. In 2007, point-transect surveys were conducted on all 20 transects outside the core population, and three Palila were detected on the northern slope of Mauna Kea (Table 2). Between March 2008 and February 2012, quarterly, non-systematic surveys were conducted on the northern slope, but no Palila were detected after March 2011 (Chris Farmer, American Bird Conservancy, pers. comm.). These birds likely represented the remnants of a small population that was reintroduced to the northern slope from 2003–2009 (Banko *et al.* 2009).

### Trends

Between 1998 and 2003 Palila numbers fluctuated widely (Figure 3). Since peaking in 2003, estimated Palila numbers had declined steadily through 2011, increasing only slightly in 2012. The average rate of decline since 2003 was 519 birds per year for a 66% decline over 10 years.

Table 2. Number of Palila detected by year inside and outside the core area during six-minute counts and the population estimate, % coefficient of variation (CV), standard error (SE), and lower and upper limits of the 95% confidence interval inside the core area.

Year	# detections inside	# detections outside	Estimate	% CV	SE	Lower limit	Upper limit
1998	313	2	5,014	10.22	512	4,095	6,109
1999	388	1	5,753	9.61	553	4,713	6,802
2000	234	14	3,310	10.37	343	2,685	4,038
2001	345	4	5,014	10.06	504	4,119	6,069
2002	339	9	5,021	9.14	459	4,174	5,960
2003	439	7	6,463	9.04	584	5,313	7,562
2004	371	9	5,458	8.87	484	4,526	6,403
2005	315	1	4,623	10.10	467	3,729	5,558
2006	271	16	4,162	10.24	426	3,371	5,041
2007	210	3	3,198	10.52	336	2,589	3,894
2008	186	na	2,619	10.81	283	2,089	3,215
2009	189	na	2,669	11.87	317	2,098	3,332
2010	151	na	1,768	12.84	227	1,350	2,236
2011	119	na	1,495	14.09	211	1,121	1,940
2012	362 <sup>1</sup>	0	2,176	10.69	233	1,749	2,640

<sup>1</sup> Of the 362 detections made during counts, 194 were observed on the first count and 168 during subsequent counts.

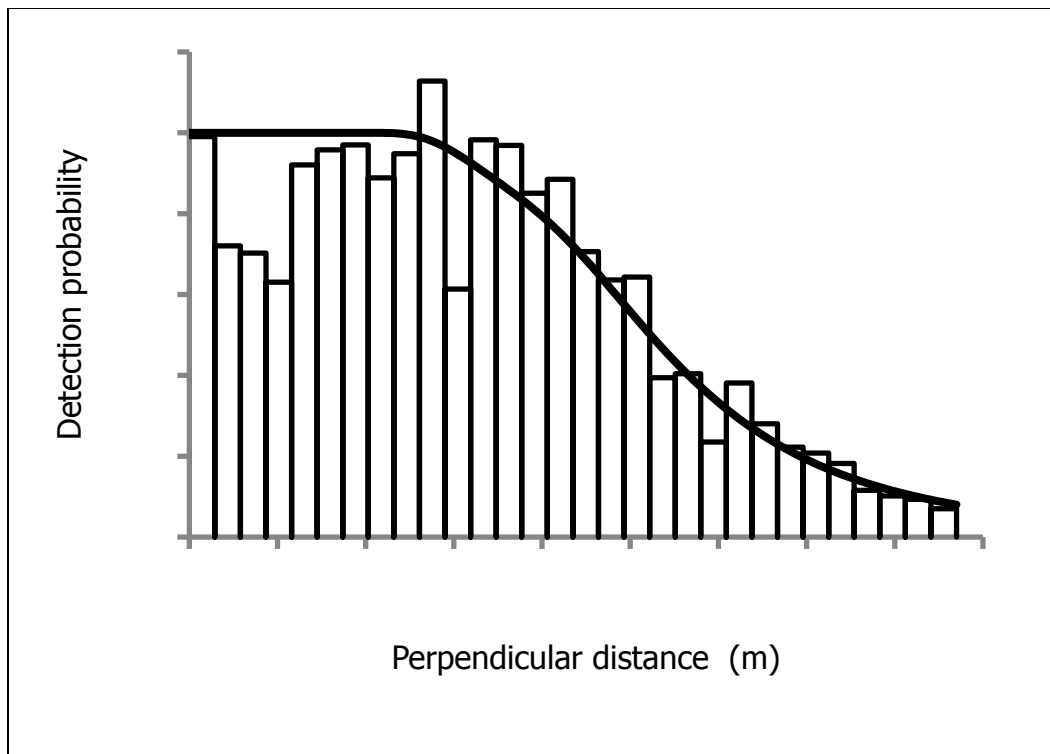


Figure 4. Hazard-rate detection function, with no expansion series and including the covariate detection type (line), and Palila distance data (histogram) pooled across all surveys from 1998 to 2012. Data were truncated at 87.0 m.

Table 3. Results of fitting 15 detection function models to the 1998–2012 Palila distance histogram.  $\Delta\text{AICc}$  is the difference in AICc scores between each model and the overall best-fit model, and  $w_i$  is the discrete model probability.

Model <sup>1,2</sup>	# parameters	Ln(likelihood)	AICc	$\Delta\text{AICc}$	$w_i$
H-rate Key DetType	4	-18495.28	36998.57	0	1.00000
H-rate Key Year (f)	17	-18511.34	37056.82	58.25	0.00000
H-rate Key Wind	5	-18614.04	37238.09	239.52	0.00000
H-rate Key Obs	4	-18615.38	37238.76	240.19	0.00000
H-rate Key Year (c)	3	-18616.69	37239.39	240.82	0.00000
H-rate Key Gust	5	-18615.35	37240.72	242.15	0.00000
H-rate Key Time (c)	3	18621.82	37249.64	251.07	0.00000
H-rate Key	2	-18623.11	37250.22	251.65	0.00000
H-rate Key Cloud	4	-18622.36	37252.72	254.15	0.00000
H-rate Key Time (f)	10	-18616.67	37253.39	254.82	0.00000
H-norm Key	1	-18673.64	37349.28	350.71	0.00000

<sup>1</sup> Models 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), gust strength (Gust), observer (Obs), time of detection (Time; continuous [c] and factor [f]), wind strength (Wind), and year (Year; continuous [c] and factor [f]).

<sup>2</sup> Models H-rate Cos, H-rate S-poly, H-norm Cos, and H-norm H-poly failed to converge.

Log-linear regression indicated that the short-term trajectory (2003–2012) had entered the leveling off phase of the exponential curve (intercept =  $8.73 \pm 0.09$  SE; slope =  $-0.153 \pm 0.02$ ), suggesting that the population may persist at low levels for a short span (55 years) before the curve intersects the x-axis (i.e., equivalent to extinction). This model assumes that no stochastic or demographic events alter this trajectory, which is unrealistic considering that Palila have a low reproductive rate, are subject to further habitat degradation due to browsing by non-native ungulates, and simultaneously face further uncertainty from climate change manifested by severe drought (Banko *et al.* in press).

Over the 1998–2012 time series there was very strong evidence (posterior probability;  $P = 0.92$ ) of an overall declining trend in Palila abundance. Using a lag-one ARMA model to account for temporal autocorrelation in the annual estimates yielded a steeply declining slope ( $-0.045$ , 95% credible interval =  $-0.088$  to  $0.005$ ). This slope was well below the lower threshold level of a 25% change over a 25 year period ( $-0.0119$ ), and although the credible interval about the slope bracketed zero, the upper interval was well within the upper threshold level, indicating little support for a negligible ( $P = 0.06$ ) or increasing trend ( $P = 0.02$ ). From 1998 to 2012, the overall rate of change equated to a 57% decline in Palila over 15 years.

### Repeat Surveys

In 2012, slightly less than half of the Palila detections were made during repeat counts (46%; 168 of 362 detections). The model incorporating year as a covariate had an AICc value of 37056.82 (Table 3). Nevertheless, including count with the year covariate inflated the AICc value by 1.15 units over the year-only model and thus did not improve the model fit. There was a small difference in the coefficients for the first and subsequent counts ( $0.2693 \pm 0.0659$  and

0.2210 ± 0.0688, respectively). These values were not statistically different (two-sample z-test: z-score = 0.507,  $P = 0.61$ ). Thus, there was no evidence that the numbers and distributions of birds detected in the first and subsequent counts were different.

## CONCLUSIONS

Although the 2012 Palila estimate was the highest since 2010, both the short-term trajectory and long-term trend indicated a declining population. No Palila were detected outside the core area, suggesting that the range of the species has continued to contract to less than 5% of its historical extent. The detection model was not improved by including time of detection or count covariates, indicating that there was no difference in the numbers of Palila detected or in their detectability by time or count. Similar numbers of Palila were detected during the first and subsequent counts of 2012. Thus, greater precision in population estimates can be gained if future surveys include repeat visits.

## ACKNOWLEDGEMENTS

Funding for annual Palila surveys since 1998 was provided by Federal Highway Administration, U.S. Army Garrison Hawai`i, Hawai`i Division of Forestry and Wildlife, U.S. Fish and Wildlife Service, American Bird Conservancy, and the U.S. Geological Survey Wildlife Program. Funding for analyses of the data was provided by the Hawai`i Division of Forestry and Wildlife. We are grateful to the many agency staff and volunteers who helped collect survey data and to Kevin Brinck and Marcos Gorresen for helpful discussions about statistical and analytical methods and to Steve Hess and Jay Nelson for reviews of an early draft.

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