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Population trends of native and non-native forest birds on the island of Molokaʻi, Hawaiʻi

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ABSTRACT

Hawaiʻi's avifauna has undergone profound changes over the past 1,000 years, with many endemic bird species driven to extinction and non-native introductions reshaping forest ecosystems. On the island of Molokaʻi, habitat degradation and the introduction of mosquito-borne disease, mainly avian malaria (*Plasmodium relictum*), have caused extinctions and severe declines in native forest bird populations. To assess these changes, point-transect distance sampling surveys were initiated in 1979 and 1980 and repeated six times, most recently in 2021, covering a 3,527 ha area. This study analyzes species composition, population densities, abundances, and long-term trends for species with sufficient detections using log-linear regression. Since 1979–1980, a total of 17 non-native and 3 native species were detected across all six surveys. The non-native Warbling White-eye (*Zosterops japonicus*) had the highest density and abundance in every survey, while abundance of Red-billed Leiothrix (*Leiothrix lutea*) significantly trended upward. The Japanese Bush Warbler (*Horornis diphone*) irrupted in the survey area, increasing in abundance by more than 99% since 1979–1980. Among native species, ʻApapane (*Himatione sanguinea*) populations increased by 33%, while Hawaiʻi ʻAmakihi (*Chlorodrepanis virens*) declined by 78% from their peak in 1995. Federally threatened ʻIiwi (*Drepanis coccinea*), highly susceptible to avian malaria, have not been recorded on Molokaʻi since 2010, and the endemic Olomaʻo (*Myadestes lanaiensis*) has not been detected since 1980 and is possibly extinct. Our results underscore the ongoing conservation crisis for Molokaʻi's forest birds and the continued vulnerability of native birds to disease and habitat loss.

KEYWORDS

Avian malaria; distance sampling; extinction; Hawaiian forest birds; non-native species

PALABRAS CLAVE

Malaria aviar; muestreo de distancias; extinción; aves de bosque hawaianas; especies introducidas

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Tendencias poblacionales de aves forestales autóctonas y no autóctonas en la isla de Moloka'i, Hawai'i

RESUMEN

La avifauna de Hawai'i ha sufrido profundos cambios en los últimos 1.000 años, con muchas especies de aves endémicas llevadas a la extinción e introducciones de especies no autóctonas que han remodelado los ecosistemas forestales. En la isla de Moloka'i, la degradación del hábitat y la introducción de enfermedades transmitidas por mosquitos, principalmente la malaria aviar (*Plasmodium relictum*), han causado extinciones y graves descensos en las poblaciones de aves forestales autóctonas. Para evaluar estos cambios, en 1979 y 1980 se iniciaron estudios de muestreo de distancias punto-transecto, que se repitieron 6 veces, la última vez en 2021, cubriendo un área de 3.527 ha. Este estudio analiza la composición de las especies, las densidades de población, las abundancias y las tendencias a largo plazo (mediante regresión log-lineal) para las especies con suficientes detecciones. Desde 1979–1980, se han detectado un total de 17 especies introducidas y 3 autóctonas en los 6 muestreos. La especie introducida *Zosterops japonicus* presentó la mayor densidad y abundancia en todos los muestreos, mientras que la abundancia de *Leiothrix lutea* tendió significativamente al alza. *Horornis diphone* irrumpió en la zona de estudio, aumentando su abundancia en más de un 99% desde 1979–1980. Entre las especies autóctonas, las poblaciones de *Himatione sanguinea* aumentaron un 33%, mientras que las de *Chlorodrepanis virens* disminuyeron un 78% desde su máximo en 1995. La especie *Drepanis coccinea*, muy susceptible a la malaria aviar y amenazada a nivel federal, no se ha registrado en Moloka'i desde 2010, y la endémica *Myadestes lanaiensis* no se ha detectado desde 1980 y posiblemente esté extinta. Nuestros resultados subrayan la actual crisis de conservación de las aves forestales de Moloka'i y la continua vulnerabilidad de las aves autóctonas a las enfermedades y la pérdida de hábitat.

Estimating abundances and monitoring trends through time are important components of conservation and management, particularly in Hawai'i, where nearly 500 threatened or endangered species of plants and animals account for >30% of all listed species in the United States (USFWS 2024). The first humans arrived in Hawai'i between 780 and 1130 AD (Kirch 2011; Athens et al. 2014), and since that time approximately 75% of the 142 endemic Hawaiian species of birds have gone extinct (Olson and James 1994; Banko and Banko 2009; Elphick et al. 2010). Only 21 species of endemic forest birds persist in the main Hawaiian Islands, and of those, 12 are federally threatened or endangered (USFWS 2024). The causes of decline are multifaceted: invasive plants and ungulates have degraded and modified habitat, while non-native rats (*Rattus* sp.), mongooses (*Herpestes javanicus*), and feral cats (*Felis catus*) raid nests and prey on adults (Atkinson 1977; Pratt 1994). Introduced vector-borne diseases such as avian malaria (*Plasmodium relictum*) and avian pox (*Avipoxvirus* species) have decimated Hawaiian honeycreepers, with susceptibility varying by species (Atkinson et al. 2001; van Riper and Scott 2001; van Riper et al. 2002; USFWS 2006; Atkinson and LaPointe 2009; Paxton et al. 2016). Avian malaria and its vector, the southern

house mosquito (*Culex quinquefasciatus*) are cold-intolerant, creating a system that limits the distribution of most native forest birds to high-elevation refugia (>1,500 m) where disease and vector are restricted (Samuel et al. 2011, 2015). As native birds disappeared, over 170 non-native bird species were intentionally introduced, and 54 species have become naturalized (Foster 2009; VanderWerf et al. 2018). The Warbling White-eye (*Zosterops japonicus*), Japanese Bush Warbler (*Horornis diphone*), and Red-billed Leiothrix (*Leiothrix lutea*) are common on many islands, even in predominantly native forests where the remaining native forest birds persist (Richardson 1949; Pyle and Pyle 2017).

Part of the Maui Nui complex that includes Maui, Lānaʻi, and Kahoʻolawe, the island of Molokaʻi has seen some of the most dramatic changes to its avifauna and native habitat. The island is approximately 670 km², with elevations ranging from sea level to 1,510 m (Fig. 1), almost completely overlapping with the range of *C. quinquefasciatus* and avian malaria. Molokaʻi was first settled by the Polynesians, and the majority of its residents remain of native Hawaiian and Pacific Islander descent. It is one of the least populated and least visited of the main Hawaiian islands, home to approximately 7,300 people and receiving less than 50,000 visitors per year (Hawaiʻi Tourism Authority 2023; World Population Review 2024). The west side of the island is below 600 m, is very dry, and has been transformed into grasslands and

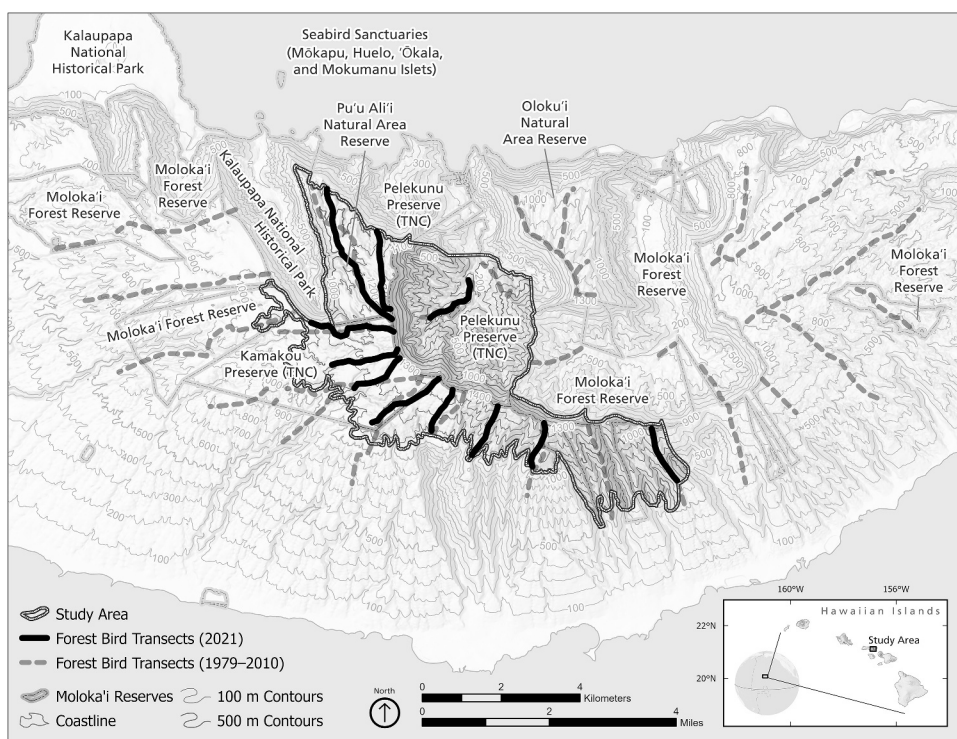


Figure 1. Hawaiʻi forest bird survey transects and reserves on Molokaʻi. Solid black lines represent transects surveyed in 2021, while dashed gray lines show transects originally surveyed in 1979–1980 but inconsistently sampled in later years. The 3,527 ha survey area, used for abundance estimates, is outlined with a double black line. Base map: World Geodetic System 1984 (WGS84) zone 5; coastline: US Geological Survey's National Elevation Dataset (USGS 2014).

non-native kiawe (*Prosopis pallida*) woodlands from overgrazing by cattle (*Bos taurus*), goats (*Capra hircus*), and axis deer (*Axis axis*; Hess 2016; Leopold and Hess 2017). The northeast plateau is dominated by steep mountain ridges and valleys, and the moisture gradient ranges from seasonal mesic to very wet (Price et al. 2012). While some native wet forest remains, the plateau is becoming invaded by strawberry guava (*Psidium cattleianum*), Himalayan ginger (*Hedychium gardnerianum*), and Koster's curse (*Miconia crenata*) that are spread by feral pigs (*Sus scrofa*; Scott et al. 1986; Gon et al. 2018). To the north lies a low-elevation peninsula (below 100 m elevation), which is protected as the historical Hansen's disease settlement, Kalaupapa National Historical Park.

Expeditions to inventory birds on Moloka'i were conducted by western naturalists in the late 1800s (Perkins 1903; Bryan 1908; Olson and James 1994). The first specimens of the Moloka'i-endemic Bishop's 'Ō'ō (*Moho bishopi*), Maui Nui-endemic Black Mamo (*Drepanis funerea*), 'Ākohekohe (*Palmeria dolei*), Kākāwahie (Moloka'i Creeper, *Paroreomyza flammea*), and Oloma'o (*Myadestes lanaiensis*) were collected in the 1880s and 1890s (Olson and James 1994). At that time, 'Ō'ū (*Psittirostra psittacea*), 'Īiwi (*Drepanis coccinea*), Hawai'i 'Amakihi (*Chlorodrepanis virens*), and 'Apapane (*Himatione sanguinea*) were considered abundant (Evenhuis 2007). By 1907, however, the Bishop's 'Ō'ō was presumed extinct (Bryan 1908), and the Black Mamo, 'Ākohekohe, and 'Ō'ū were extirpated by the 1930s (Richardson 1949). The Kākāwahie, Oloma'o, and 'Īiwi were sufficiently rare that they were not detected on Moloka'i from 1908 until the 1960s (Munro 1944; Richardson 1949; Pekelo 1963; Scott et al. 1977). The last known detections of the Kākāwahie were in 1963, and it was recently removed from the endangered species list due to extinction (USFWS 2023).

To monitor these changing populations, shifting ranges, and management impacts of forest bird and plant communities, the US Fish and Wildlife Service (USFWS) initiated the Hawai'i Forest Bird Survey (HFBS) as a long-term, systematic, landscape-scale survey within suitable native forest bird habitat (Scott et al. 1986). These surveys have been repeated on the islands of Kaua'i, Moloka'i, Maui, and Hawai'i since the 1970s to monitor the status and trends of native forest birds in the state. On Moloka'i, 22 transects were initially established in 1979 and 1980 on 131 km² of the eastern portion of the island at 250–1,515 m elevation (Scott et al. 1986). Transects were surveyed according to point-transect distance sampling methods with stations 150 m apart (Scott et al. 1986; Camp et al. 2011). Results of the initial HFBS were published by Scott et al. (1986), which provided a baseline of bird populations in the early 1980s. Repeated surveys across multiple years are necessary to establish trends and obtain a better understanding of threats and management priorities for native species in such disturbed and invaded habitat as that on Moloka'i. Therefore, the Hawai'i Department of Land and Natural Resources (DLNR)-Division of Forestry and Wildlife, the Maui Forest Bird Recovery Project, The Nature Conservancy (TNC), and the National Park Service prioritized and resurveyed many of the original transects using the same sampling method over a 42 year period. For the first time, we present the results of all six surveys analyzed together, including reanalysis of the original results using updated techniques, and provide species composition, indices of occurrence, and number of birds per station surveyed for every species detected. For birds with sufficient detections to characterize detection probability (Buckland et al. 2015), we provide species-specific density estimates and assess trends from 1979–1980 to 2021.

Methods

Survey area

The survey area was located on reserves across the north-central and eastern portions of Molokaʻi (Fig. 1). In 2021, we surveyed 156 sampling points on 11 transects that spanned state and private management boundaries, including the TNC Kamakou and Pelekunu Preserves, DLNR Puʻu Aliʻi Natural Area Reserve and Molokaʻi Forest Reserve, and various small privately owned lands. Portions of the original HFBS transects were resampled during subsequent surveys and transects were 1.2–4.0 km long with sampling points approximately 150 m apart. The elevation of transects ranged from 200 to 1,500 m, encompassing low-elevation valleys and the highest ridgelines and peaks on the island. Climate zones and habitat types of survey areas ranged from very wet forest and bogs, to moderately wet-mesic on leeward-facing slopes, to moist mesic in the valleys (Price et al. 2012). Above 700 m elevation, forest canopy was composed of native ʻōhiʻa lehua (*Metrosideros polymorpha*), ʻōlapa (*Cheirodendron trigynum*), and kopiko (*Psychotria marianiana*). Below 700 m, weedy non-native species such as strawberry guava, Christmasberry (*Schinus terebinthifolia*), and java plum (*Syzygium cumini*) dominated the tree canopy. We defined our study area using locations of transects that were surveyed in 1979–1980, 1988, 1995, 2004, 2010, and 2021. Although it was sampled on only four of the previous surveys, we included the transect in the Pelekunu Preserve to broaden inference to the valley bottom and maximize spatial coverage. We used elevation contours, management boundaries, and topographic features (i.e., ridges and streams; Fig. 1) and mapped a 3,527 ha area using ArcGIS Pro (Environmental Systems Research Institute, CA, USA).

Point-transect distance sampling methods

Bird surveys were conducted from 13 to 17 April 2021, using point-transect distance sampling to estimate abundance as a function of the distance between the observer and birds (Buckland et al. 2015). To improve probability of detection, surveys were conducted during the breeding season of most native forest birds, when adults and juveniles are active (Scott et al. 1986; Camp et al. 2009). Robust estimates rely on the critical assumptions that all birds are detected with certainty at the station center point, birds farther from the observer are harder to detect, birds are detected prior to any responsive movement, and distances are measured with low error. Prior to the start of the survey, six primary surveyors were trained to calibrate for distance estimation and learn local dialects of bird vocalizations, thereby minimizing variability among observers and standardizing for local conditions.

Sampling points were surveyed with single visits in 1979–1980, 1988, 1995, 2004, 2010, and 2021. Unlimited-distance point counts occurred starting half an hour before sunrise and ending no later than 1200 h HST, according to methods established in Camp et al. (2011). Species, horizontal distance from the station center point to individual birds detected, and detection type (heard, seen, or both) were recorded during each 8 min count. Distances were estimated to the nearest meter, and surveyors used laser rangefinders to determine distances to reference objects (e.g., shrubs, trees, rocks) to assist in accurately estimating distances. Birds only flying over or through the survey area were excluded. Observers also recorded cloud cover (0–100% by 10s), rain intensity, wind and gust strength

on the Beaufort scale, and time of day of each survey. Sampling was halted when rain intensity, wind strength, or gust strength exceeded prescribed levels (i.e., heavy rain, high winds; Camp et al. 2011).

Data analysis

We estimated species-specific densities (birds/ha) for five species—Hawai‘i ‘Amakihi, ‘Apapane, Japanese Bush Warbler, Warbling White-eye, and Red-billed Leiothrix—using the R (R Core Team 2021) package *DISTANCE* (Miller et al. 2019) following methods described in Thomas et al. (2010) and Buckland et al. (2015). We assumed that birds were uniformly distributed with respect to distance from sampling points. We limited the candidate detection function models to half-normal and hazard-rate detection functions with expansion series of order 2. We paired the half-normal detection function with cosine and Hermite polynomial adjustments, and we paired the hazard-rate detection function with cosine and simple polynomial adjustments. Model precision can be improved by incorporating sampling covariates in the multiple covariate distance sampling (MCDS) engine of *DISTANCE* (Buckland et al. 2015). We included covariates for cloud cover, rain, wind, gust, observer, type of detection, and survey year; all covariates were treated as categorical variables. Each model was fit to data pooled across years for each species, and we selected the model with the lowest Akaike’s Information Criterion (AIC; Buckland et al. 2015; Supplemental Table S1). Following recommendations in Buckland et al. (2015), we truncated detections at a distance where the detection probability using a preliminary half-normal detection function model was about 0.1. This procedure facilitates modeling by deleting outliers and reducing the number of parameters needed to modify the detection function. We estimated species-specific densities for each survey year from the global detection function using the post-stratification procedure, and we derived variances and confidence intervals by bootstrap methods in *DISTANCE* from 1,000 iterations (Thomas et al. 2010). We calculated absolute abundances as mean density times the size of our study area.

We assessed long-term population trends (1979–2021) from the bootstrap sample estimates with a log-linear regression model for the 5 species for which we estimated abundances. Evidence for a trend was derived from the bootstrap distribution of slopes following methods described in Camp et al. (2016). Diagnostics (visual inspection of residual plots) demonstrated that the log-linear regressions of trends met all model assumptions. All statistical analyses were conducted using R in the RStudio version 1.4.1717 interface (RStudio Team 2021), data were manipulated using the tidyverse package (Wickham et al. 2021), and figures were generated using the ggplot2 package (Wickham et al. 2021).

Results

Species composition and occurrence

A total of 17 non-native and 3 native species were detected across all six surveys from 1979 to 2021 (Supplemental Table S2). After initial surveys in 1979–1980, 1988, and 1995, three additional non-native species were detected: the White-rumped Shama (*Copsychus*

malabaricus) and Eurasian Skylark (*Alauda arvensis*) in 2004, and the Hwamei (*Garrulax canorus*) in 2021. Game birds such as the Ring-necked Pheasant (*Phasianus colchicus*) and Chukar (*Alectoris chukar*) were detected in low numbers initially, but apparently did not establish in the survey area and have not been detected since 1995. The Zebra Dove (*Geopelia striata*) and Northern Mockingbird (*Mimus polyglottos*) were only detected in 1979 and 1995, respectively. ‘Apapane and Hawai‘i ‘Amakihi were detected during each of the six surveys and were the only native species detected in 2021. The most ‘Iiwi detected were three individuals in 1979, the last on-count detection was a single individual in 2004, and the last confirmed ‘Iiwi sighting was in 2010, when a surveyor saw an individual outside the 8 min count period. The Oloma‘o was not detected within the survey area during any of the six surveys since HFBS, with the last confirmed detections occurring off-count during the 1979–1980 surveys within the Oloku‘i Natural Area Reserve (hereafter Oloku‘i Plateau) and Kamakou Preserve (three detections each site).

In 2021, ‘Apapane and Hawai‘i ‘Amakihi together accounted for one-third (33%) of all 1,791 detections. There were 574 detections of ‘Apapane, occurring at 94.2% of stations surveyed. There were 11 detections of Hawai‘i ‘Amakihi, at 5.8% of stations surveyed. Eight of these detections occurred on one low-elevation ridge in the Pelekunu Preserve, with 3 detections within the higher Kamakou Preserve to the southwest. Three non-native species—Japanese Bush Warbler, Warbling White-eye, and Red-billed Leiothrix—accounted for two-thirds (65%) of all detections in 2021. Across the 42 years of surveys, Warbling White-eye occurred at >90% of stations for each of the six surveys. After occurring at 6.1% of stations surveyed in 1979, the Japanese Bush Warbler rapidly increased to 94.2% of stations surveyed in 1988. Occurrence of the Red-billed Leiothrix ranged between 43.6% and 81.7% of stations in each survey. Additional non-native species such as the Black Francolin (*Francolinus francolinus*), Erckel’s Francolin (*Pternistis erckelii*), Spotted Dove (*Spilopelia chinensis*), Common Myna (*Acridotheres tristis*), Scaly-breasted Munia (*Lonchura punctulata*), House Finch (*Haemorhous mexicanus*), and Northern Cardinal (*Cardinalis cardinalis*) occurred inconsistently across the six surveys at <16% of stations surveyed.

Population status and trends

The Warbling White-eye had the highest density and abundance for each of the six surveys, with a 2021 abundance of $90,388 \pm 9,324$ (SE) birds (Fig. 2; Supplemental Table S3). ‘Apapane was the second most abundant species at $81,896 \pm 10,836$ birds in 2021, followed by Japanese Bush Warbler at $21,403 \pm 1,787$ birds, and Red-billed Leiothrix at $9,813 \pm 1,051$ birds. Hawai‘i ‘Amakihi was the least abundant species in 2021 at $1,085 \pm 510$ birds, and densities were <1.0 bird/ha for every survey since 1995.

The Red-billed Leiothrix showed an upward trend from 1979 to 2021, despite leveling off between 2004 and 2021 (slope = 0.047, SE = 0.015, $F_{1,4} = 9.678$, $P = 0.040$), while trends for Hawai‘i ‘Amakihi, ‘Apapane, Japanese Bush Warbler, and Warbling White-eye were non-significant (alpha-level = 0.05; Fig. 2). Hawai‘i ‘Amakihi abundances tended to decrease over the time series, although higher abundances in 1988 and 2004 resulted in a non-significant decline (slope = -0.033, SE = 0.019, $F_{1,4} = 3.010$, $P = 0.16$). ‘Apapane abundances tended to increase overall, with a substantial increase between 1979 and 1995, but declines between 2004 and 2021 brought abundances closer to the 1979 estimate (slope = 0.004, SE = 0.009, $F_{1,4} = 0.419$, $P = 0.69$). The abundance of the

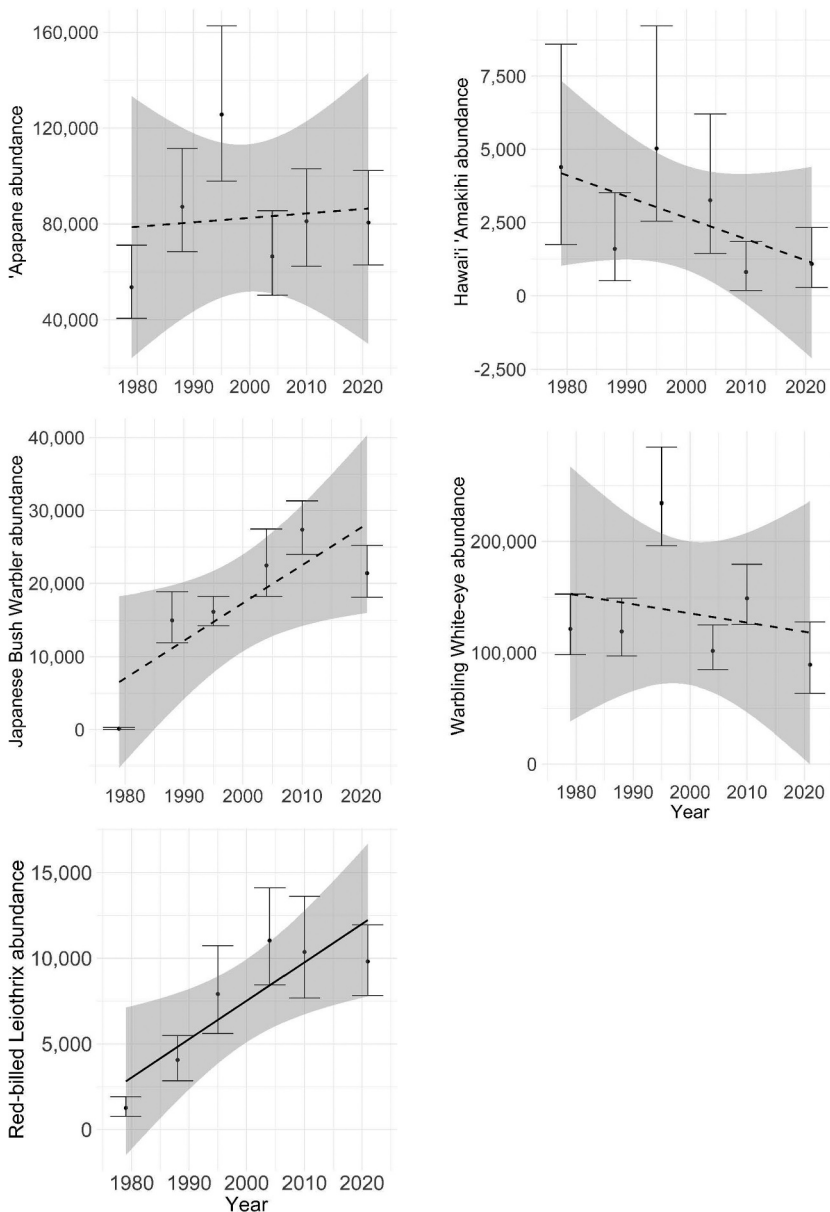


Figure 2. Abundance estimates and trends for Hawai'i 'Amakihi, 'Apapane, Japanese Bush Warbler, Red-billed Leiothrix, and Warbling White-eye on Moloka'i from 1979–1980 to 2021. Only the trend for Red-billed Leiothrix (solid line) is significant; All other trends are non-significant (dashed lines). Whisker bars show 95% confidence intervals (CIs) for abundance estimates, and gray ribbons indicate 95% CIs for trends.

Japanese Bush Warbler was very low in 1979 and increased by more than 100-fold by 1988 and continued to increase thereafter; however, the overall trend was non-significant, likely influenced by the decline between 2010 and 2021 (slope = 0.096, SE = 0.047, $F_{1,4} = 4.179$, $P = 0.11$). The abundance of the Warbling White-eye tended to

decrease between 1979 and 2021, and the high estimate in 1995 appears to be an outlier (slope = -0.007 , SE = 0.011 , $F_{1,4} = 0.410$, $P = 0.56$).

Discussion

In 2021, we observed an avian community on Molokaʻi that was a remnant of its composition reported in the 19th century. Of the 26 native forest bird species documented from subfossil and historical records (Olson and James 1994), only ʻApapane and Hawaiʻi ʻAmakihi have been detected during surveys over the past 12 years. Since the onset of surveys in the late 1970s, non-native Passeriformes, Columbiformes, and Galliformes have come to dominate the avifauna, now totaling 17 species, with the introduced Warbling White-eye, Japanese Bush Warbler, and Red-billed Leiothrix becoming nearly ubiquitous across the survey area. While many of our trends were inconclusive, they highlight the complicated nature of community composition in highly disturbed habitat and the need for regularly repeated, long-term surveys to understand complicated community dynamics.

Of the two native species detected during the 2021 survey, ʻApapane were found throughout the survey area at relatively high densities. The 2021 density estimate of 23.22 ± 3.07 birds/ha (Supplemental Table S3) is comparable to or higher than those in some high-elevation native forests on the islands of Hawaiʻi and Maui (Camp et al. 2009; Kendall et al. 2023). Although trends for ʻApapane were non-significant due to high variance, abundance estimates in both 2010 and 2021 exceeded 80,000 birds, indicating that the population may have stabilized in the survey area, especially in ʻōhiʻa lehua-dominated forest.

In contrast, Hawaiʻi ʻAmakihi have been observed at lower densities since their highest estimate of 1.43 ± 0.48 birds/ha in 1995. By 2021, their density was estimated at only 0.31 ± 0.14 birds/ha—lower than even some critically endangered species, such as the ʻĀkohekohe on Maui, estimated at 0.75 ± 0.1 birds/ha (Judge et al. 2021). Because the Pelekunu Preserve transect with the most Hawaiʻi ʻAmakihi detections was not surveyed in 1988 and 2010, the density for those years was lower than expected. While some populations of Hawaiʻi ʻAmakihi on the Island of Hawaiʻi have demonstrated resistance to avian malaria (Woodworth et al. 2005; Foster et al. 2007; Atkinson et al. 2013), the rarity of the species on Molokaʻi indicates that disease remains a significant stressor. On Lānaʻi, the combination of disease and habitat degradation led to the extinction of the Hawaiʻi ʻAmakihi population (Lindsey et al. 2020), and the Molokaʻi population now faces a similar threat.

The ʻĪiwi is especially vulnerable to avian malaria (Atkinson et al. 1995; Atkinson and LaPointe 2009) and was listed as federally threatened in 2017 due to population declines (USFWS 2017). ʻĪiwi have been considered extinct on Lānaʻi for nearly a century (Hobdy 1993), and while the population on Oʻahu was estimated at fewer than 10 individuals (USFWS 2017), local experts consider the population extirpated (E.A. VanderWerf, Pacific Rim Conservation, 2022, pers. comm.). On Kauaʻi, Maui, and the Island of Hawaiʻi, downward trends and range contractions of ʻĪiwi populations are correlated with warming climate trends and the expansion of avian malaria and its mosquito vector (Paxton et al. 2013, 2016; Fortini et al. 2015). During their extensive survey efforts on Molokaʻi, Scott et al. (1986) estimated a range-wide population of only 80 birds. Numbers of detections continued to dwindle, and the last confirmed sighting was in 2010.

One of five endemic Hawaiian thrushes, the Olomaʻo has been rare since it was first described in the early 1800s, with the last confirmed sightings in 1979–1980 on the Olokuʻi Plateau and Kamakou Preserve (Scott et al. 1986). Historical data suggest the Olomaʻo may have gone extinct on Lānaʻi by 1934, and on Molokaʻi between 1988 and 2009 (Elphick et al. 2010), but surveys on the Olokuʻi Plateau have not been repeated since 1979–1980. Three other Hawaiian thrush species have faced similar declines. On Oahu, the ʻAmaui (*M. woahensis*) was extinct by the early 1800s (Henshaw 1902; Evenhuis 2007). On Kauaʻi, the Kamaʻo (*M. myadestinus*) was recently declared extinct (USFWS 2023), and the critically endangered Puaiohi (*M. palmeri*) has managed to survive in the hundreds, despite severe population declines caused by rat predation (Crampton et al. 2017). In contrast, the ʻŌmaʻo (*M. obscurus*) thrives in many windward forests on the Island of Hawaiʻi because of its tolerance to avian malaria (Atkinson et al. 2001; Judge et al. 2012). To confirm the extinction of Olomaʻo and potential extirpation of ʻŌmaʻo, it would be useful to conduct targeted rare bird searches (Camp et al. 2011) and deploy autonomous recording units (Sebastián-González et al. 2018) in remote areas like the Olokuʻi Plateau.

After intentional introductions in the early 1900s (Foster 2009), non-native species have increased to be some of the most numerous birds in Hawaiʻi. The influence of non-native species on trends of native species has been challenging to quantify, but there are important factors through which non-natives can contribute to native population declines. Non-native birds, resistant to avian malaria, act as reservoirs that increase rates of disease transmission to native species (McClure et al. 2020). On Molokaʻi, non-native birds showed an average disease prevalence of 64% (Paxton 2022), making them important sources of pathogen transmission. Furthermore, non-native generalists contribute to the degradation of native forest bird habitat by dispersing some of Hawaiʻi's most invasive plants (Foster and Robinson 2007; Chimera and Drake 2010; Vizentin-Bugoni et al. 2019). Finally, the overlap in diet of certain native and non-native passerines suggests that competition could impact native species if foraging resources are limited (Banko et al. 2015).

To secure the future of native forest birds on Molokaʻi, increased understanding of current habitat and disease trends is needed. Habitat management including weed control, native plant reforestation, and control of ungulates and introduced predators has resulted in stable or increasing populations of native forest birds elsewhere in Hawaiʻi (Camp et al. 2010; Paxton, Laut, et al. 2018; Paxton, Yelenik, et al. 2018), and could help restore and preserve essential habitat on Molokaʻi. Surveying more frequently throughout the study areas could help to better monitor and detect trends in Molokaʻi forest bird populations (Brinck et al. 2012). While climate-change-induced expansion of mosquito-borne disease remains the most significant challenge for Hawaiian forest birds throughout their range (Fortini et al. 2015; Paxton, Laut, et al. 2018), innovative mosquito suppression tools such as the *Wolbachia*-based Incompatible Insect Technique (IIT) are being developed and implemented elsewhere in Hawaiʻi, and could be utilized on Molokaʻi as well (Zheng et al. 2019; Beebe et al. 2021). IIT involves overflooding the wild mosquito population with male *C. quinquefasciatus* with a different strain of the *Wolbachia* bacterium. When these males mate with wild females, the eggs produced will be inviable due to cytoplasmic incompatibility, and with continuous releases, wild mosquito populations would collapse and disease transmission would be suppressed. This could help reverse declines in native species and create disease-free habitats for the translocation and reintroduction of imperiled Hawaiian birds. By managing habitat, monitoring bird populations long-term, and developing and

implementing tools for disease suppression, other islands and similarly imperiled areas may help ensure the remaining native forest birds do not follow the path of extirpation and extinction that has occurred on Molokaʻi.

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