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Distribution and Trends of Endemic Hawaiian Waterbirds

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Abstract.—Four endemic species of wetland-dependent waterbirds occur on the main Hawaiian Islands, all of which have experienced sharp population declines and are listed as endangered species. Twice per year, state-wide surveys are conducted to count waterbirds, but these surveys are evaluated only infrequently. We used a state-space approach to evaluate long-term (1986–2016) and short-term (2006–2016) trends and current distribution and abundance of endemic Hawaiian waterbirds. The most numerous species was the Ae‘o, or Hawaiian Stilt (*Himantopus mexicanus knudseni*), with a 5-year estimated average abundance of 1,932 individuals, followed by ‘Alae Ke‘oke‘o, or Hawaiian Coot (*Fulica alai*), with 1,815 individuals, Alae ‘Ula, or Hawaiian Common Gallinule (*Gallinula galeata sandvicensis*) with 927 individuals, and the Koloa Maoli, or Hawaiian Duck (*Anas wyvilliana*) with 931 individuals. All four species had positive trends over the long-term, but short-term and island specific trends were more variable, and in some cases negative. These results provide valuable information to help guide management of Hawaii's threatened and endangered endemic waterbirds. *Received 4 Nov 2020, accepted 13 March 2021.*

Key words.—*Anas wyvilliana***,** endemic Hawaiian waterbird, *Fulica alai*, *Gallinula galeata sandvicensis*, *Himantopus mexicanus knudseni*, population trends, wetlands.

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The Hawaiian Islands historically supported a diverse range of endemic and migratory (non-resident, wintering) waterbirds in both wetland and riparian forest habitats. At least 30 endemic waterbird species are known from historical and subfossil records, but during the last 1,000–1,200 years of human occupation all of Hawaii's endemic rails (*Porzana* spp.), flightless waterfowl, and a flightless ibis species have gone extinct (Olson and James 1984). Both Polynesian and European settlers played significant roles in these extinctions through direct hunting, alteration of Hawaiian ecosystems (both from agricultural practices and urban development), and the introduction of non-native plants and animals (Kirch 1982; 1983; Olson and James 1991).

Today, only four endemic species of wetland-dependent waterbirds persist on the main Hawaiian Islands. The Koloa Maoli, or Hawaiian Duck (*Anas wyvilliana*), is a small duck that was once considered to be a subspecies of the Mallard (*Anas platyrhynchos*), but genetic studies have shown is a distinct species closely related to the Mallard (Browne *et al.* 1993; Lavretsky *et al.* 2015). Nonetheless, Hawaiian Ducks readily hybridize with Mallard ducks, with only the Kaua‘i Island population considered genetically pure (Fowler *et al.* 2008; Wells *et al.* 2019), and hybridization is considered the largest threat to the species (USFWS 2011). The Hawaiian Duck historically used a wide variety of natural wetland habitats for nesting and feeding, including freshwater marshes, flooded grasslands, coastal ponds, streams, montane pools, and forest swamplands at elevations ranging from sea level to 3,000 m (9,900 feet), and in the 1800s they were fair-

ly common in natural and farmed wetland habitats (Engilis *et al.* 2002). Hawaiian Ducks mostly breed in high montane habitat, although some ducks will breed in lowland wetlands, and population distributions shift seasonally, likely as birds disperse to remote breeding areas (Gee 2007).

The ʻAlae Keʻokeʻo, or Hawaiian Coot (*Fulica alai*) was once considered a subspecies of the American Coot (*Fulica americana*), but it is now regarded as a distinct species endemic to the Hawaiian Islands (Brisbin *et al.* 2002). The Hawaiian Coot has historically occurred on all of the main Hawaiian Islands, and is typically found in coastal plain wetlands ranging from sea level to 260 m, rarely to 1,067 m, preferring wetland habitats with suitable emergent plant growth interspersed with open water (Brisbin *et al.* 2002) and fresh water (non-saline) areas for nesting (Byrd *et al.* 1985).

The ʻAlae ʻUla, or Hawaiian Common Gallinule (*Gallinula galeata sandvicensis*) is an endemic subspecies of the Common Gallinule (*Gallinula galeata*), and likely originated from North America (Bannor and Kiviat 2002). The gallinule was historically found on all of the main Hawaiian Islands except Lānaʻi (Munro 1944), but in recent years have only been detected on the islands of Kauaʻi and Oʻahu. Hawaiian Common Gallinule (hereafter referred to as Hawaiian Gallinule) are predominantly a species of the lowland wetlands, using natural ponds, reservoirs, marshes, streams, lagoons, grazed wet meadows, flooded agricultural fields, shrimp aquaculture ponds, and other areas associated with water (Shallenberger 1977; Bannor and Kiviat 2002). Connectivity of wetlands is key for gene flow in this species (van Rees *et al.* 2018).

The fourth species, the Aeʻo, or Hawaiian Stilt (*Himantopus mexicanus knudseni*) is considered a subspecies of the Black-necked Stilt (*Himantopus mexicanus*) of North and South America (Robinson *et al.* 1999). They were historically found on all of the major islands except Lānaʻi and Kahoʻolawe (Paton and Scott 1985), and utilize a wide range of wetland and upland habitats (Kawasaki *et al.* 2020). All four species are listed as State and Federally Endangered Species, with surveys in the mid-1900s indicating small (less than 1,000 individuals) populations of each species (USFWS 2011). However, the stilt is being proposed for federal downlisting to Threatened Species status due to higher population numbers.

The quantity and quality of wetland habitat has changed considerably since human settlement and is a key factor affecting declines in waterbird numbers and distributions. Wetland habitat loss across the islands since pre-historic time is estimated at 31% (Dahl 1990) but is as high as 65% on some islands (USFWS 2011; Van Rees and Reed 2014). The arrival of the Polynesian people to Hawaiʻi 1,000–1,200 years ago (Kirch 2011) and the cultivation of kalo (taro, *Colocasia esculenta*), which requires flood-irrigation, caused substantial changes to wetland habitat across the islands, including plant composition, water levels, and wetland distribution (Culliney 2006). Waterbirds regularly use agriculture lands, and some agricultural practices such as flood irrigation may have increased the amount of wetland habitat in the islands (Swedberg 1967). Other flood-irrigation crops such as rice (*Oryza sativa*) and massive water development projects for other agricultural crops continued to affect wetland habitat availability for waterbirds beginning in the late 1800s through mid-1900s. Around 1780 there was an estimated 8,990 ha of coastal plain wetlands (primary habitat for waterbirds), but this amount is estimated to have declined to 6,190 ha by 1990 (Dahl 1990). Wetlands today continue to be threatened with urban development, invasive plant species, changes in precipitation from climate change, and threats of flooding and saltwater intrusion from sea-level rise (Timm *et al.* 2015).

Hawaiian waterbirds are conservation reliant species (Reed *et al.* 2012; Underwood *et al.* 2013), and active management is needed to provide suitable habitat and reduce threats. Conservation efforts since the 1970s have focused on securing and protecting wetland habitats for endangered waterbirds. Federal and state reserves protect an estimated 27% of coastal wetlands (USFWS

2011), with private and non-governmental organizations providing protection for additional wetlands.

Wetlands managed for waterbirds require extensive, active management to provide the habitat qualities needed by the different species. In particular, the two largest threats are invasive plants and predators. Invasive plants such as California grass (*Urochloa mutica*), pickleweed (*Batis maritima*), water hyacinth (*Eichhornia crassipes*), Indian fleabane (*Pluchea indica*), and mangrove (*Rhizophora mangle*) have overrun many Hawaiian wetlands, reducing or eliminating the suitability of wetlands for native waterbirds (Shallenberger 1977). Non-native plants likely affect nesting success of stilts, which prefer sites with little or no cover surrounding nests, and even coots, although they use emergent vegetation for nesting (Coleman 1981; Morin 1998). Overall, control of invasive plants can lead to increased wetland use by all species (Rauzon and Drigot 2002). Water properties (e.g., salinity, depth) affect food resources, foraging availability, and the types of plants that are able to grow (Stone 1989), thereby influencing habitat selection.

Current efforts to minimize predation pressure include multiple removal and exclusion techniques (Underwood *et al.* 2013), although data regarding the effectiveness of current predator control efforts and the contribution of such efforts to metapopulation productivity are largely unknown. Non-native predators, including small Indian mongoose (*Herpestes javanicus*), cats (*Felis catus*), dogs (*Canis lupus familiaris*), rats (*Rattus* spp.), and to a lesser extent feral pigs (*Sus scrofa*), Barn Owls (*Tyto alba*), Cattle Egrets (*Bubulcus ibis*), and bullfrogs (*Lithobates catesbeianus*) depredate eggs, young, and adult birds, and have negative effects on survival and reproductive success (Engilis and Pratt 1993). Additional threats include avian disease, particularly a paralytic disease caused by botulism, a neurotoxin, which continues to kill Hawaiian waterbirds, especially at altered or man-made wetlands. Also, populations of Hawaiian Ducks on all islands are affected by hybridization with feral Mallards, with Kaua'i Island the only sampled island with a largely genetically pure Hawaiian Duck population (Fowler *et al.* 2008; Wells *et al.* 2019).

In order to better understand Hawaiian waterbird population trends and their relation to wetland management, annual surveys of Hawai'i's endemic waterbirds and wintering waterfowl began in the 1940s, but coverage and methodology have changed over the decades. By the mid-1950s, most large wetlands were surveyed, and in the late-1960s the surveys were adapted to monitor endemic Hawaiian waterbirds (Engilis and Pratt 1993). Survey methods were updated again in 1976 to improve data collection efforts and expanded to include winter and summer surveys. In 1986, simultaneous one-day counts of resident and migratory waterbirds were conducted twice annually during the winter and summer on 6 of the 8 main Hawaiian Islands, which has been the format for surveys ever since. During 2005, increased emphasis was placed on accurately identifying and counting wintering shorebirds, adding another layer of value to the surveys. Although surveys are conducted biannually, analysis of waterbird numbers and trends have been periodic (Engilis and Pratt 1993; Reed *et al.* 2007; USFWS 2011). The last comprehensive analysis of surveys was conducted from 2008 and earlier survey data (Reed *et al.* 2011; USFWS 2011). Data management of survey results lagged from 2007 to 2016, limiting the use of survey data to inform waterbird conservation efforts. Recent efforts by the Hawai'i State Division of Forestry and Wildlife (DOFAW) and U.S. Fish and Wildlife Service (USFWS) to improve data management and utility of waterbird surveys included entering 11 years of data (2008–2016), reconciling survey site names across years, checking data constancy, and other crucial steps for quality assurance.

In this paper we present the results of 31 years of surveys, evaluating trends from 1986–2016 and 2006–2016 using a state-space approach to produce estimates of population changes over time with associated estimates of uncertainty. State-space models account for both annual variation due to biological processes as well as count variation reflec-

tive of sampling error (Humbert *et al.* 2009). We also evaluated the correlation between within-year winter and summer counts to assess the amount of additional information provided by the two biannual counts. Lastly, we evaluated how representative the core wetlands, as identified in the Recovery Plan for Hawaiian Waterbirds (USFWS 2011), are to overall population numbers and trends. Most waterbirds are supported by core wetlands (USFWS 2011), but the degree that they reflect overall trends is unknown.

METHODS

Surveys

Surveys are organized by DOFAW, which collects survey data and archives survey results. The standardized Hawai‘i Biannual Waterbird Survey has been conducted the 3rd Wednesday of January and August each year on 6 of the 8 main Hawaiian Islands since 1986. Counts are conducted on a single day to minimize overcounting waterbirds that may move between wetlands. Observers record all waterbirds seen or heard at each survey location for a minimum of 10 minutes or until all species are counted and recorded. Surveys are designed to be a census of all individuals, but detectability varies by habitat and species, so survey data are considered minimum counts (Elzinga *et al.* 2009). Survey effort, including the number of sites surveyed, varies among years. In addition to species counts, data on weather and wetland conditions, time of survey, and number of observers are also recorded. Survey locations include most lowland and coastal wetlands and aquatic habitats (e.g., reservoirs, golf course ponds, Hawaiian fishponds), as well as some canals, taro lo‘i, aquaculture ponds, and non-wetland areas such as lawns, recreational fields, beaches, and shorelines. Core sites represent wetlands that support a high number of birds, are important for the recovery of waterbirds (USFWS 2011), and are consistently surveyed; whereas, some other wetlands may not be surveyed in all years largely due to access and available surveyor constraints. A survey of all habitats used by endemic waterbirds has not been feasible due to inaccessibility of some sites (e.g., private land) and lack of funding to conduct aerial surveys of remote locations. For example, the survey does not cover many shorelines or upland habitats that could support Hawaiian Stilts (Kawasaki *et al.* 2020), and does not include montane streams commonly used by Hawaiian Ducks. Ni‘ihau Island, which is privately owned, has not been consistently surveyed, so data from this island are not included in the analysis. Because ducks counted on islands other than Kaua‘i Island are likely hybrids (Fowler *et al.* 2008), we only considered duck counts from Kaua‘i Island for the analysis.

Analysis

Because the biannual surveys are simple counts of all individual waterbirds seen without any attempt to correct for imperfect detection, under- and over-counting, and other effects of sampling error, there are no estimates of error for each year's population estimate. To estimate sampling error, we used a state-space analysis of the time series to calculate trends. Specifically, a state-space analysis approach partitions the variation in a time series of counts into sampling error, which reflects variation due to non-biological changes (e.g., observer error, weather conditions), and process variation, which is reflective of biological (true) changes in bird abundance (Camp *et al.* 2016). Such a state-space model can be interpreted as a biologically informed smoother and provides annual population estimates that account for the observed inter-annual noise. However, because the surveys are not conducted in a manner that allows for estimating detectability (Camp *et al.* 2014), the state-spaced estimated survey numbers still represent minimum estimates.

We used the Stan Bayesian modeling language (Carpenter *et al.* 2017) run from an R environment (R Core Team 2019) using the rstan package (Stan Development Team 2018) to model trends as a log-linear model. We used prior distributions of uniform between -5 and 5 for the mean slope, and uniform between 0 and 20 for the standard deviations for normally distributed sampling and process variation, which are diffuse for a log-normal regression on all species. 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 1.05, which is less than the 1.1 threshold below which indicates convergence (Gelman and Rubin 1992).

We used an equivalency testing approach to assess statistical significance of trends (Camp *et al.* 2008), using the posterior distribution of the slope from the 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 0.0089 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 occurs when the slope is within the equivalence region and from which a stable population can be inferred. We categorized the strength of evidence for a trend based on the weight of the posterior distribution falling into that category: > 50% was classified as weak evidence of a trend, > 70% as strong evidence, and > 90% as very strong evidence. An inconclusive result occurs when small sample size and high variation in estimates results in the posterior distribution of the slope providing < 50% weight in all the three outcomes.

Although waterbird surveys are conducted twice a year, the correlation between those surveys had not previously been evaluated. Winter and summer counts

of waterbirds provide two different count estimates within each year, with differences reflecting changes in population sizes, movement among wetlands, and seasonal changes in habitat availability. Summer counts are believed to be higher than winter counts for resident species, as they are post-breeding peak, but could also reflect seasonal differences in distribution (e.g., Hawaiian Ducks breed in high elevation streams in summer, waterbirds may disperse to smaller flooded wetlands in the winter). We estimated the average annual difference between within year winter and summer counts and calculated the correlation between the two counts using a Pearson correlation test. We also evaluated the correlation between core wetlands that are identified in the Recovery Plan for Hawaiian Waterbirds (USFWS 2011) as key to recovery of species, and the many other, typically smaller wetlands that may not be managed for waterbirds. Surveys results were organized by the entire range of the species (global, or state-wide), specific island, and for Figure 1 by Moku (a traditional Hawaiian land division composed of several adjacent ahupuaʻa or watersheds; Hawaii 2020; Fig. 1). Statistical significance was accepted at alpha < 0.05.

Results

In 2016, the most recent year that survey data were available, Kauaʻi Island supported the most endemic waterbirds (mean = 2,922, [95% CI:1,737–5,046]), followed by Oahu Island (mean = 1,195, [400-2,978]), Maui Island (mean = 1,024, [663–1,492]), and the rest of the Hawaiian Islands (mean = 665, [238–1588]; Table 1). Only Kauaʻi Island supported all four endemic waterbird species, followed by Oʻahu Island (excluding Hawaiian Duck), reflecting historical patterns of species distribution and possibly the greater amount of habitat available to waterbirds. The most numerous species was the Hawaiian Stilt (5-yr minimum average population estimate 1,932, [1,552–2,385]), followed by Hawaiian Coot (5-yr minimum average population estimate 1,815, [1,248–2,577]), both

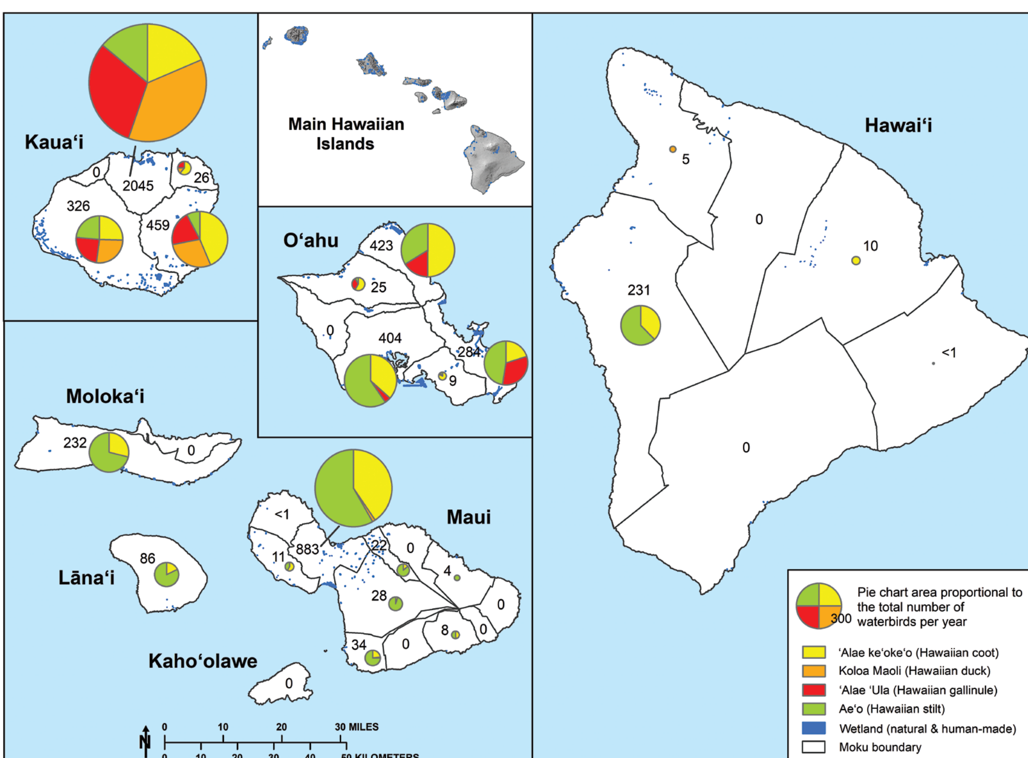

Figure 1. Distribution and average abundance (2014–2016) of Hawaiian waterbirds during the winter survey by island and Moku (traditional Hawaiian land divisions). Each species is represented by a specific color, and the pie charts show the relative proportion that each species contributes to the number of total waterbirds detected per Moku (total number shown as number in each Moku). Wetlands shown are those areas surveyed for waterbirds between 1986 and 2016. Hawaiian Ducks are only shown for Kauaʻi Island, where populations are not hybrids.

Table 1. Survey counts and state-space adjusted counts (SS) for Hawai'i waterbird winter surveys in 2016, the most recent year of count data, and a 5-year average (2012–2016). Counts are organized for each species by island and global (all island) estimates with 95% confidence intervals.

	2016 season only			5-yr average (2012–2016)	
Island	Count	SS mean	SS 95% CI	SS mean	SS 95% CI
Ae'o, Hawaiian Stilt (*Himantopus mexicanus knudseni*)					
Kaua'i	500	494	(182–1,104)	418	(177– 843)
O'ahu	330	485	(294–728)	526	(355–750)
Maui	476	529	(394–722)	602	(443–779)
Moloka'i	161	207	(70–472)	169	(96–280)
Lāna'i	58	63	(40–103)	82	(64–104)
Hawai'i	145	203	(44–613)	159	(124–201)
Global	1,670	1,992	(1,544–2,443)	1,932	(1,552–2,385)
'Alae Ke'oke'o, Hawaiian Coot (*Fulica alai*)					
Kaua'i	585	685	(255–1,572)	680	(270–1451)
O'ahu	152	496	(72–1,604)	520	(242–965)
Maui	545	489	(268–750)	395	(244–602)
Moloka'i	50	81	(39–145)	88	(63–120)
Lāna'i	7	12	(3–39)	29	(7–79)
Hawai'i	76	96	(42–204)	113	(69–176)
Global	1,415	1,834	(1,142–2,659)	1,815	(1,248–2,577)
'Alae 'Ula, Hawaiian Gallinule (*Gallinula galeata sandvicensis*)					
Kaua'i	708	741	(562–1,010)	746	(571–960)
O'ahu	53	213	(34–642)	190	(62–455)
Global	761	942	(608–1,425)	927	(678–1,235)
Koloa Maoli, Hawaiian Duck (*Anas wyvilliana*)					
Kaua'i	936	1,002	(738–1,360)	947	(751–1,185)

of which occurred on Kaua'i, O'ahu, Maui, Moloka'i, Lāna'i, and Hawai'i Islands (Table 1, Fig. 1). The minimum global population estimate for the Hawaiian Gallinule was 927 (5-yr average, [678–1,235]). On Kaua'i Island, the 5-yr minimum population size for Hawaiian Duck was 947 (751–1,185).

Trends

Global long-term (1986-2016) trends indicate increasing population sizes for all four endemic waterbird species, with strong to very strong support for a positive trend ranging from 2.2 to 7.3% annual increases on average (Table 2, Fig. 2). However, island by island trends were more variable, with most species increasing over the 31-yr period, but with weaker evidence. An exception to this was Hawaiian Coots, which had weak negative trends on the Island of Hawai'i, and both Hawaiian Stilts and Hawaiian Coots had indeterminant long-term trends for O'ahu (Table 2).

Short-term (2006–2016) trends were much more variable, with mostly negative or inconclusive trends on all but Kaua'i Island (Table 2). In fact, Kaua'i Island had strong positive trends for both the short-term and long-term trends, indicating that generally increasing global population trends for all four species were heavily influenced by Kaua'i population trends.

Summer versus Winter Counts for Waterbirds

Summer counts were on average similar to winter counts for gallinules and ducks, -36 (95% CI: -52—20) and -36 (95% CI: -69—3), respectively, but higher in summer for stilts and coots (Fig. 3). Hawaiian Stilts had on average 227 (95% CI: 167–287) more individuals counted in summer than winter over a 31-year period, but the difference was highly variable across the time period (Fig. 3). Likewise, Hawaiian Coots had larger numbers

Table 2. State-space average annual population trends of Hawaiian waterbirds for 11-year (2006–2016) and 31-year (1986–2016) periods. For each species and island there are the average population changes for each time period, the 95% confidence interval of that slope, and an equivalency test to determine support for population changes of 25% change over 25 years. Global estimates are for all islands combined. The posterior distribution for an equivalency test of > 0.5 is weak evidence (single arrow), > 0.7 is strong evidence (2 arrows), and > 0.9 is very strong evidence (3 arrows) for decreasing (dec), stable (stbl), or increasing (inc) population. If all values are below 0.5, then the results are indeterminate (ind).

		Equivalency test					Equivalency test			
Island	11-yr trend (95% CI)	dec	stbl	inc		31-yr trend (95% CI)	dec	stbl	inc	
Ae‘o, Hawaiian Stilt (*Himantopus mexicanus knudseni*)										
Kaua‘i	0.117 (-0.297– 0.623)	0.16	0.10	0.74	↑↑	0.085 (-0.048–0.230)	0.05	0.03	0.93	↑↑↑
O‘ahu	-0.071 (-0.329– 0.146)	0.68	0.17	0.15	↓	-0.004 (-0.083–0.073)	0.37	0.30	0.32	ind
Maui	0.002 (-0.209–0.199)	0.39	0.23	0.38	ind	0.024 (-0.065–0.108)	0.16	0.14	0.69	↑
Moloka‘i	0.116 (-0.256–0.508)	0.18	0.10	0.73	↑↑	0.096 (-0.148–0.316)	0.11	0.03	0.86	↑↑
Lāna‘i	-0.035 (-0.205–0.144)	0.55	0.22	0.23	↓	0.109 (-0.128–0.343)	0.15	0.04	0.81	↑↑
Hawai‘i	-0.032 (-0.190–0.155)	0.54	0.29	0.17	↓	0.064 (-0.284–0.394)	0.23	0.04	0.73	↑↑
Global	-0.009 (-0.138–0.115)	0.30	0.46	0.23	ind	0.028 (-0.010–0.058)	0.02	0.09	0.89	↑↑
‘Alae Ke‘oke‘o, Hawaiian Coot (*Fulica alai*)										
Kaua‘i	0.067 (-0.415–0.517)	0.24	0.11	0.64	↑	0.040 (-0.093–0.178)	0.15	0.10	0.76	↑↑
O‘ahu	-0.126 (-0.595–0.301)	0.71	0.10	0.18	↓↓	-0.003 (-0.280–0.243)	0.43	0.09	0.49	ind
Maui	0.057 (-0.199–0.351)	0.18	0.19	0.63	↑	0.056 (-0.124–0.223)	0.19	0.07	0.75	↑↑
Moloka‘i	-0.045 (-0.285–0.196)	0.56	0.18	0.26	↓	0.019 (-0.081–0.116)	0.20	0.16	0.64	↑
Lāna‘i	-0.157 (-0.926–0.623)	0.67	0.06	0.26	↓	0.031 (-0.369–0.409)	0.38	0.05	0.57	↑
Hawai‘i	-0.055 (-0.336–0.216)	0.60	0.19	0.21	↓	-0.019 (-0.212–0.187)	0.53	0.11	0.37	↓
Global	-0.006 (-0.233–0.183)	0.33	0.31	0.36	ind	0.022 (-0.049–0.082)	0.11	0.15	0.75	↑↑
‘Alae ‘Ula, Hawaiian Gallinule (*Gallinula galeata sandvicensis*)										
Kaua‘i	0.161 (0.009–0.306)	0.01	0.02	0.97	↑↑↑	0.094 (-0.046–0.264)	0.07	0.05	0.89	↑↑
O‘ahu	-0.043 (-0.651–0.468)	0.47	0.11	0.41	ind	0.050 (-0.156–0.239)	0.17	0.06	0.77	↑↑
Global	0.101 (-0.083–0.268)	0.06	0.09	0.85	↑↑	0.062 (-0.017–0.132)	0.03	0.03	0.94	↑↑↑
Koloa Maoli, Hawaiian Duck (*Anas wyvilliana*)										
Kaua‘i	0.115 (-0.017–0.248)	0.02	0.04	0.94	↑↑↑	0.073 (-0.019–0.158)	0.03	0.03	0.93	↑↑↑

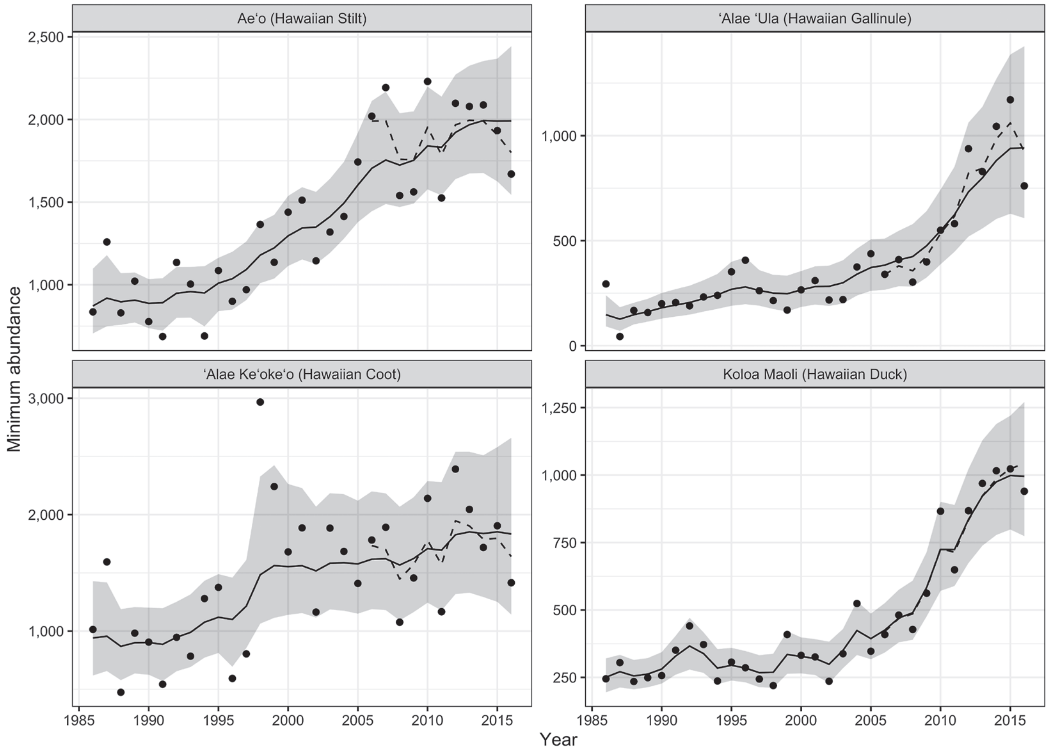

Figure 2. Annual count numbers (black dots) and state-space estimate of minimum population size (black line) and 95% CI (shaded area) for four Hawaiian waterbird species from 1986–2016. Dotted lines represent the state-space average estimates for the short-term (2006–2016) trends. Surveys do not correct for imperfect detectability and represent a minimum population size estimate.

in summer, averaging 358 more individuals (95% CI: 220–496), but there was wide overlap in counts across the years (Fig. 3). Generally, year to year correlation between summer and winter counts was high (statewide) for Hawaiian Stilts ($r = 0.79$, $P < 0.001$), Hawaiian Gallinule (r = 0.90, $P < 0.001$), and Hawaiian Duck (r = 0.86, $P < 0.001$), but not significantly for Hawaiian Coot (r = 0.21, $P = 0.25$).

Core Sites versus all Sites for Waterbirds

On average, between 44% and 65% of total Hawaiian waterbirds counted were detected in core wetlands for a given year: 53.2% (winter) and 44.5% (summer) for Hawaiian Coots; 52.3% (winter) and 56.6% (summer) for Hawaiian Ducks; 46.4% (winter) and 44.8% (summer) for Hawaiian Gallinule; and 65% (winter) and 58.9% (summer) for Hawaiian Stilts. Although the annual waterbird counts include many non-core survey sites, the core sites are highly correlated with the total count for all four species: Hawaiian Coot winter (r = 0.83, $P < 0.001$) and summer (r = 0.51, $P < 0.001$); Hawaiian Duck winter (r = 0.96, $P < 0.001$) and summer (r = 0.94, $P < 0.001$); Hawaiian Gallinule winter (r = 0.99, $P < 0.001$) and summer (r = 0.97, $P < 0.001$); and Hawaiian Stilt winter (r = 0.91, $P < 0.001$) and summer (r = 0.83, $P < 0.001$), and 95% confidence intervals of long-term trends were widely overlapping.

Discussion

Population estimates of Hawaiian waterbirds have increased since the 1980s, although the strength of trends varied by islands. However, short-term tends were mostly negative or indeterminate. Both short (2006–2016) and long (1986–2016) term trends were positive on Kaua'i Island, which may reflect the

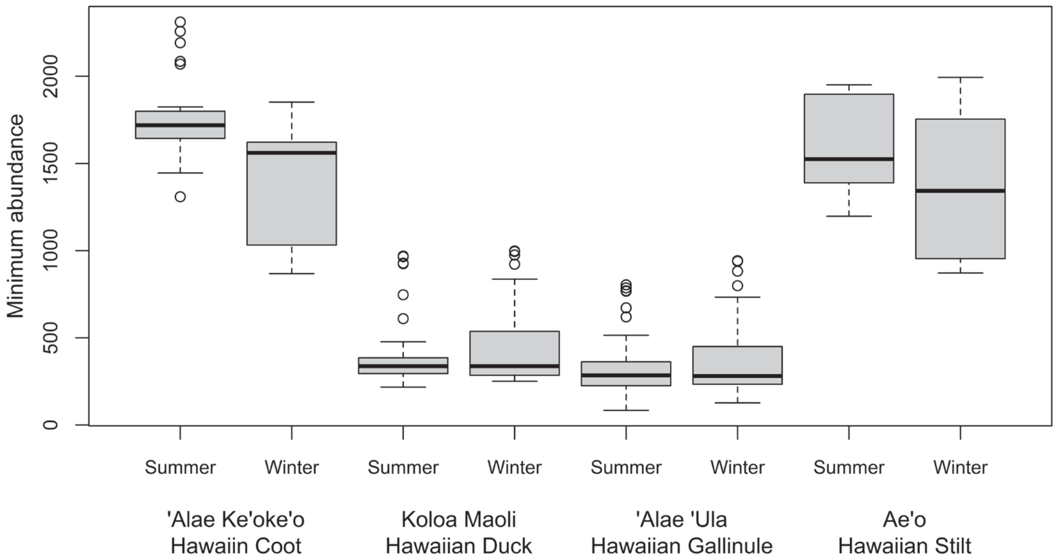

Figure 3. Average abundance of state-space adjusted minimum counts from the intra-annual Summer and Winter bird surveys for the four Hawaiian waterbirds. Hawaiian Stilts and Hawaiian Coots had hundreds more individuals counted in summer compared than winter, on average, while Hawaiian Gallinules and Hawaiian Ducks had, on average, little difference between the two intra-annual counts. Box plots show median value (dark bar), the box indicates the interquartile range, the whiskers indicate the minimum and maximum range of 99.3% of values, and the open circles are outliers.

substantial wetland habitat, restoration, and management on the island, as well as the absence of mongoose, a non-native predator of waterbirds and their nests, which may allow for higher overall population persistence. Kaua'i Island, the oldest of the main Hawaiian Islands, has the most coastal wetland habitat still persisting (Van Rees and Reed 2014), and supports half of all the waterbirds counted across all the islands. There is evidence of substantial movement of waterbirds between Kaua'i Island and neighboring Ni'ihau Island, but the privately owned Ni'ihau Island has not been surveyed since 1999, and it is not known if residential populations of waterbirds occur on the island. O'ahu Island, the second most important island for Hawaiian waterbirds in terms of populations supported, had both short- and long-term negative or indeterminate trends for stilts and coots, with positive long-term but indeterminate short-term trends for gallinules. Overall, O'ahu Island is estimated to have lost 71% of its coastal wetlands since presettlement times (Van Rees and Reed 2014), and continues to experience the most anthropogenic growth of all the Hawaiian Islands, with development continuing to change the amount of wetlands directly through conversion to developed land, or indirectly through retirement of agricultural lands and changes in flood-irrigation and hydrology. Waterbirds on Moloka'i and Lāna'i Islands had weak negative short-term trends, except Hawaiian Stilts had strong positive trends on Moloka'i. Maui Island mostly had weak but positive trends for both short- and long-term periods, except stilts were indeterminate for short-term trends (Table 2). On the Island of Hawai'i, the youngest island with few natural wetlands, all species saw declines in both short- and long-term trends, with the exception of stilts for long-term trends. The amount of wetland habitat suitable for Hawaiian Stilts has changed over time on the Island of Hawai'i, with several human-made aquatic habitats becoming established in the early 2000s including aquaculture and wastewater treatment facilities, but some of these no longer support stilts in the last 5–10 years of the surveys due to multiple reasons including changes in suitability, construction, and active steps to discourage waterbird use in areas near airports.

The establishment, protection, and active management of wetlands for waterbirds is likely the largest contributor to the marked increases from survey estimates in the late 1980s (Fig. 2). However, these increases appear to have plateaued or may even be reversing in the most recent survey periods, which may indicate birds have reached carrying capacity of existing wetlands. Alternatively, the recent declines might indicate other factors counteracting the positive benefits of more managed wetlands. Hawaiian waterbirds experienced other shifts in increasing population size in the past (Reed *et al.* 2011), including following the end of hunting in 1941 (Schwartz and Schwartz 1949), protection under the Endangered Species Act (1967 and 1970), and increased management of wetland habitat, including creation of National Wildlife Refuges and State Management Areas starting in the 1970s. There is a positive relationship between the number of waterbirds each island supports and the amount of coastal wetlands (Van Rees and Reed 2014), although approximately 70% of the remaining wetlands are overgrown with invasive plants and have altered surface and groundwater hydrology (Shallenberger 1977). Considerable management is needed for wetlands to provide sustained long-term quality habitat, such as predator and invasive plant control, and therefore simple measures of hectares of wetlands do not equate to measures of suitable habitat. Recent efforts to restore large tracts of wetlands covered in non-native vegetation, such as Kawainui Marsh on O‘ahu Island, have the potential to make available some of the largest wetlands in Hawai‘i. In addition, a recent installation of a predator-proof fence in Pearl Harbor National Wildlife Refuge, O‘ahu Island, is providing a new approach to protecting waterbirds (Christensen *et al.* 2021). The conservation-reliant nature of Hawaiian waterbirds implies that active management to mitigate threats and habitat loss are key to long-term population persistence.

Population size numbers are an important metric that can be used to help guide the conservation of Hawaiian waterbirds. The Recovery Plan for Hawaiian Waterbirds (USFWS 2011) cites specific population sizes for recovery goals, along with need for long-term wetland habitat management plans. The recovery plan recommends that all species reach population sizes greater than 2,000 for at least 5 consecutive years, show stable or positive trends, and that sufficient wetlands are receiving management and protection to ensure their continued support of waterbirds. Of the four endemic species, Hawaiian Stilts and Hawaiian Coots have minimum estimated population sizes near, but slightly below, the 2,000 individual level, while Hawaiian Gallinule and Hawaiian Ducks have estimates about half the target recovery number. However, several factors affect the accuracy of Hawaiian waterbird counts. There are many areas not surveyed, including Ni‘ihau Island, other private lands, agricultural fields, and small ephemeral wetlands. In addition, stilts can use upland habitat (Kawasaki *et al.* 2020), and Hawaiian Ducks will breed in mountain streams that are not surveyed (Engilis and Pratt 1993). Further, there is the issue of detection probability. Current survey methods do not attempt to estimate detectability, and this can be a substantial problem for accurate counts. For example, the use of call-broadcast survey methods improved the detection of Hawaiian Gallinules by as much as 56% over the current passive survey approach (DesRochers *et al.* 2008). Movement among wetlands by waterbirds can lead to variation in year to year counts within sites and regions, even though the single day count is intended to minimize double counting as a result of inter-wetland movement. The state-space year to year estimates provide biologically informed population estimates that buffer year to year variation in counts, providing more plausible population estimates from the count data. However, the survey counts are still minimum counts and could be substantially undercounting some species such as the gallinule.

Waterbird surveys in Hawai‘i have a long history and have been modified multiple times to improve methodology. Although several improvements over time have increased their reliability and consistency, the

surveys still are conducted as simple area search and count. This sampling approach only provides information on presence and relative abundance. Importantly, the surveys are not conducted in a manner that would allow estimation of detection probability, or statistical consideration of how various factors such as individual observer, weather, wetland size, time, and so on affect detection across surveys. Estimates of detection probability require some form of replicated sampling, such as estimating the detection probability of individuals based on distance (Thomas *et al.* 2010), multiple visits, or a form of double-counter methodology (Royle *et al.* 2005). Importantly, count methodology that allows for statistical analysis can produce unbiased estimates by accounting for factors that influence detectability such as observer skills, survey conditions (e.g., rain, wind), habitat, and changes in detectability over time. Adding estimates of detection probability provides counts with less bias, higher accuracy, and better estimates of statistical precision (such as confidence intervals) that can help improve inferences from the surveys.

Surveying wetlands is difficult, as surveyors are typically restricted to the perimeter of habitat, with vegetation often obscuring large parts of the area of interest. Camp *et al.* (2014) evaluated reliability of different survey methods for Hawaiian waterbirds, including double observer counts, repeated survey sampling, and point-transect and line-transect distance sampling. They found that even two trained observers counting from the same place at the same time could produce average counts with as much as 90% coefficient of variation. The study concluded that a double observer survey method was the most effective approach of those considered, assuming observers are well trained to accurately record covariates. Observers are recruited each year from a diverse assemblage of government agencies, non-governmental organizations, and private volunteers, but recent efforts to standardize protocols and training are expected to greatly increase accuracy of collecting covariates and identifying birds. However, a double-observer approach would add more time and personnel requirements to the survey efforts, which may be difficult to fulfill.

Improving survey methodology would require additional resources, or changes in current surveys. Surveys are conducted biannually and attempt to count all wetlands, a large and logistically challenging effort. With limited resources, waterbird managers would need to balance overall geographic coverage with precision of counts, biannual versus single annual counts, and core wetlands versus many of the small ephemeral wetlands. To help guide future decisions on where to allocate efforts, we assessed the additional information gathered by the two intra-annual counts, and the difference between core wetlands and all wetlands. We found a high correlation between winter and summer counts in three of the four waterbirds, but not for Hawaiian Coots. It may be the large shifts in coot distribution occur as the birds take advantage of seasonally variable wetlands, and these changes in distribution could account for the low correlation. Additionally, there were differences in the winter and summer counts within the same year for stilts and coots, but not gallinules and ducks, which could reflect differences in reproductive output at different times of the year, seasonal movement (such as from Kaua‘i Island to Ni‘ihau Island), and different habitat use. There was also a high degree of correlation between counts at core wetlands and all wetlands, and long-term trends were similar. This indicates that the core wetlands are representative of all available waterbird habitat and that population increases in the core areas are correlated with population increases in the non-core areas. One approach to balance survey precision with limited resources would be to apply more rigorous survey methods at core wetlands, while conducting the simpler search and count methodology at all other wetlands available for surveying. Ultimately, managers need to decide what level of precision in waterbird distribution, abundance, and trends are

needed to adequately manage wetlands and waterbirds for long-term viability.

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